NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
CONFERENCE AND BIOLOGICAL OPINION

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ACRONYM LIST

Abbreviation	Definition
a.i.	Active Ingredient
AChE	Acetylcholinesterase
ACOE	United States Army Corps of Engineers
AGRRA	Atlantic and Gulf Rapid Reef Assessment
AIMS	Avian Information Monitoring System
AIWW	Atlantic Intracoastal Waterway
ALDFG	Abandoned, Lost, or otherwise Discarded Fishing Gear
AMOC	Atlantic Meridional Overturning Circulation
AMOVA	Analysis of Molecular Variance
APA	Administrative Procedure Act
APE	Alkylphenol polyethoxylates
APPS	NOAA Fisheries Authorizations and Permits for Protected Species
ASMFC	Atlantic States Marine Fisheries Commission
ATON	Aid(s) to Navigation
BA	Biological Assessment
BE	Biological Evaluation
BHC	Benzene hexachloride
BHS	Boron hydride sulfide
BMP	Best Management Practice
BOD	Biological Oxygen Demand
BOEM	Bureau of Ocean Energy Management
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRD	Bycatch Reduction Device
BRT	Biological Review Team
BSEE	Bureau of Safety and Environmental Enforcement
BSEP	Brunswick Steam Electric Plant
CADPR	California Department of Pesticide Regulation
CAFO	Concentrated Animal Feeding Operation
CC	California Coastal (Chinook Salmon)
CCC	Central California Coast (coho salmon)
CCE	California Current Ecosystem
CCV	California Central Valley (steelhead)
CDF	Confined Disposal Facility
CDFG	California Department of Fish and Game
CDFW	California Department of Fish and Wildlife
CDL	Cropland Data Layer
CFC	Chlorofluorocarbon
CFGC	California Fish and Game Commission

CFR	Code of Federal Regulations
ChE	Cholinesterase
CI	Confidence Interval
CIDMP	Comprehensive Irrigation District Management Plans
CITES	Convention on International Trade in Endangered Species
CNMI	Commonwealth of the Northern Mariana Islands
COD	Chemical Oxygen Demand
COTS	Crown of Thorn Starfish
CP&L	Carolina Power and Light
CPUE	Catch per Unit Effort
CR	Columbia River (chum)
CRD	Crop Reporting District
CRR	Cohort replacement rates
cSEL	Cumulative Sound Exposure Level
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CV	Central Valley
CVP	Central Valley Project
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
D	Dust
DAM	Destruction or Adverse Modification
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DEIS	Draft Environmental Impact Statement
DEQ	Department of Environmental Quality
DNA	Deoxyribonucleic Acid
DO	Dissolved Oxygen
DOD	Department of Defense
DOE	Department of Energy
DOI	Department of the Interior
DPS	Distinct Population Segment
DTRU	Dry Tortugas Recovery Unit of loggerhead sea turtle
DWH	Deepwater Horizon
EC	Emulsifiable Concentrate
EC50	Effective Concentration (0% of the test organisms show effects)
EEC	Estimated Environmental Concentration
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EGTTR	Eglin Gulf Test and Training Range
EIIS	Ecological Incident Information System

EIS	Environmental Impact Statement
EOG	Electro-olfactogram
EPA	U.S. Environmental Protection Agency
EPR	Eggs per Recruit
EQ	Effective Quiet
ERMA	Environmental Response Management Application
ESA	Endangered Species Act
ESU	Ecologically Significant Unit
EUP	Experimental Use Permit
FAA	Federal Aviation Administration
FCRPS	Federal Columbia River Power System
FEIS	Final Environmental Impact Statement
FEMA	Federal Emergency Management Agency
FERC	Federal Energy Regulatory Commission
FFDCA	United States Federal Food, Drug, and Cosmetic Act
FGB	Flower Gardens Bank
FHWA	Federal Highway Administration
FlC	Flowable Concentrate
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FL	Fork Length
FMP	Fishery Management Plan
FP	Fibropapillomatosis
FP&L	Florida Power and Light Company
FR	Federal Register
FST	F-statistic
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
FWS	Fish and Wildlife Service
G	Granular
GADNR	Georgia Department of Natural Resources
GCRU	Greater Caribbean Recovery Unit of loggerhead sea turtle
GHG	Greenhouse Gas
GIS	Geographic Information System
GIWW	Gulf Intracoastal Waterway
GOM	Gulf of Mexico
GPS	Global Positioning System
GRBO	Gulf of Mexico Regional Biological Opinion
GSI	Gonad Somatic Index
GTR	General Technical Report
HAB	Harmful Algal Bloom
HGMP	Hatchery Genetic Management Plan
HMS	Highly Migratory Species

HUC	Hydrologic Unit Code
IC50	Inhibitory Concentration by 50%
ICTRT	Interior Columbia Technical Recovery Team
ICWW	Intracoastal Waterway
ID	Identification
IDEQ	Idaho Department of Environmental Quality
IDS	Incident Data System
IDWR	Idaho Department of Water Resources
IHN	Infectious Hematopoietic Necrosis
ILWP	Irrigated Lands Waiver Program
INRMP	Integrated Natural Resources Management Plan
IPCC	Intergovernmental Panel on Climate Change
IRED	Interim Registration Eligibility Decision
ISAB	Independent Scientific Advisory Board
ISDA	Idaho State Department of Agriculture
ISED	International Sawfish Encounter Database
ITS	Incidental Take Statement
IUCN	International Union for Conservation of Nature
JAXBO	Jacksonville District's Programmatic Biological Opinion
KG	Kilogram
LAA	A "Likely to Adversely Affect" determination
LC50	Lethal Concentration (50% of the test organisms are killed)
LCR	Lower Columbia River
LLC	Limited Liability Company
LOAEC	Lowest Observable Adverse Effect Concentration
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observed Effect Concentration
LOEL	Lowest Observed Effect Level
LSNFH	Livingston Stone National Fish Hatchery
LWD	Large Woody Debris
MATC	Maximum Acceptable Toxicant Concentration
MCFA	Minor Crop Farmer Alliance
MCR	Middle Columbia River
MHW	Marine Heat Wave
MHWL	Mean High Water Line
MLLW	Mean Lower Low Water
MLW	Mean Low Water
MMHSRP	Marine Mammal Health and Stranding Response Program
MMPA	Marine Mammal Protection Act
MPG	Major Population Group
MRI	Minimum Reapplication Interval
MRID	Master Record Identifier

MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSMA	Monosodium methanearsonate
mtDNA	Mitochondrial DNA
NAD83	North American Datum of 1983
NAS	National Academy of Sciences
NASS	National Agricultural Statistics Services
NAST	National Assessment Synthesis Team
NAWQA	National Water-Quality Assessment
NC	Northern California (steelhead)
NCAP	Northwest Coalition for Alternatives to Pesticides
NCCA	National Coastal Condition Assessment
NCDEQ	North Carolina Department of Environmental Quality
NCDMF	North Carolina Division of Marine Fisheries
NCWRC	North Carolina Wildlife Resources Commission
nDNA	Nuclear DNA
NE	A "No Effect" determination
NEAMAP	Northeast Area Monitoring and Assessment Program
NED	National Economic Development
NEPA	National Environmental Policy Act
NF	North Fork
NGMRU	Northern Gulf of Mexico Recovery Unit of loggerhead sea turtle
NGVD	National Geodetic Vertical Datum
NLAA	A "Not Likely to Adversely Affect" determination
NLCD	National Land Cover Database
NMFS	National Marine Fisheries Service
NOAA	National Ocean and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NOAEL	No Observed Adverse Effect Level
NOEC	No Observed Effect Concentration
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Park Service
NRAS	Natural Resources Assessment Section
NRC	National Research Council of the National Academy of Sciences
NRCS	National Resource Conservation Service
NRDA	Natural Resource Damage Assessment
NRU	Northern Recovery Unit of loggerhead sea turtle
NS	Not Specified
NWFSC	Northwest Fisheries Science Center
OC	Oregon Coast (coho)
OCS	Outer Continental Shelf
OCSLA	Outer Continental Shelf Lands Act

OCT	October
ODFW	Oregon Department of Fish and Wildlife
ODMDS	Ocean Dredged Material Disposal Site
OP	Organophosphate
OPA90	Oil Pollution Act of 1990
OPP	Office of Pesticide Programs
OSRP	Oil Spill Response Plan
PAH	Polycyclic aromatic hydrocarbons
PBDE	Polybrominated Diphenyl Ethers
PBF	Physical and Biological Feature(s)
PC	Pesticide Chemical (pesticide code number used by EPA)
PCB	Polychlorinated Biphenyl(s)
PCDD	Polychlorinated dibenzo-p-dioxin(s)
PCDF	Polychlorinated dibenzofuran(s)
PCE	Primary Constituent Element
PCP	Pentachlorophenal
PCT	Percent Crop Treated
PDC	Project Design Criterion(ia)
PDF	Portable Digital Format
PDO	Pacific Decadal Oscillation
PFAM	Pesticides in Flooded Applications Model (EPA exposure model)
PFC	Perfluorinated Compound
PFRU	Peninsular Florida Recovery Unit of loggerhead sea turtle
PID	Proposed Interim Decision
PIT	Passive Integrated Transponder
PMP	Pesticide Management Plan
POP	Persistent Organic Pollutant
PRD	NMFS Southeast Regional Office Protected Resources Division
PRM	Post-release Mortality
PRZM	Pesticide Root Zone Model
PS	Puget Sound (steelhead)
PSAT	Puget Sound Action Team
PSO	Protected Species Observer
PSP	(Oregon) Pesticide Stewardship Partnership
PTS	Permanent Threshold Shift
PULA	Pesticide Use LimitationArea
PUR	Pesticide Use Reporting
PVA	Population Viability Analysis
PWC	Pesticide Water Calculator
RBDD	Red Bluff Diversion Dam
RCP	Representative Concentration Pathway(s)
RED	Reregistration Eligibility Decision

RKM	River Kilometer
RMS	Root Mean Square
ROV	Remotely Operated Vehicle
RPA	Reasonable and Prudent Alternative(s)
RPM	Reasonable and Prudent Measure(s)
RRT	Regional Response Team
SC	Southern California (steelhead)
SCCC	South-Central California Coast
SCDNR	South Carolina Department of Natural Resources
SCL	Straight Carapace Length
SCTLD	Stony Coral Tissue Loss Disease
SEL	Sound Exposure Level
SHRU	Salmon Habitat Recovery Units
SL	Standard Length
SLCF	Short-lived Climate Forcer(s)
SLN	Special Local Needs FIFRA 24(c) registration
SNP	Single Nucleotide Polymorphism
SONCC	Southern Oregon/Northern California Coasts
SRB	Snake River Basin
SRKW	Southern Resident Killer Whale
SRT	Status Review Team
SSB	Spawning Stock Biomass
SSB/R	Spawning Stock Biomass per Recruit
SSD	Species Sensitivity Distribution
SSP	Shared Socio-economic Pathway(s)
SSRIT	Smalltooth Sawfish Recovery Implementation Team
SSWS	Sea Star Wasting Syndrome
STSSN	Sea Turtle Stranding and Salvage Network
STX	Saxitoxin
SURF	Surface Water Database
SUUM	Summary Use and Usage Matrix (EPA Pesticide Usage Summary)
SWFSC	Southwest Fisheries Science Center
T&C	Term(s) and Condition(s)
TCDD	Tetrachlorodibenzo-p-dioxin
TED	Turtle Excluder Device
TEP	Typical End-use Product
TGAI	Technical Grade Active Ingredient
TKI	Tessenderlo Kerley Incorporated
TL	Total Length
TMDL	Total Maximum Daily Load
TNAP	Temporary Noise Attenuation Pile
TNC	The Nature Conservancy

TRT	Technical Recovery Team
TTS	Temporary Threshold Shift
TWA	Time Weighted Average
U.S.	United States of America
UCR	Upper Columbia River
UDL	Use Data Layer
URL	Uniform Resource Locator
US	United States
USA	United States of America
USACE	U.S. Army Corps of Engineers
USAF	U.S. Air Force
USCG	U.S. Coast Guard
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
USGS	United States Geological Survey
USN	U.S. Navy
USVI	U.S. Virgin Islands
UWR	Upper Willamette River
VOC	Volatile Organic Compound
VSP	Viable Salmonid Population
VVWM	Variable Volume Water Model
WDFW	Washington Department of Fish and Wildlife
WP	Wettable Powder
WQPMT	Water Quality Pesticide Management Team
WRDA	Water Resources Development Act
WSDA	Washington State Department of Agriculture
WSP	Water Soluble Powder
YOY	Young-of-the-year

UNITS OF MEASUREMENTS

Unit	Definition	Unit	Definition
A	acre	lin ft	linear foot/feet
°C	degrees Celsius	m	meter(s)
°F	degrees Fahrenheit	mcy	million cubic yards
cm	centimeter(s)	mgd	million gallons/day
dB	decibel	mi	mile(s)
ft	foot/feet	mi^2	square mile(s)
g	gram(s)	nm^2	square nautical mile(s)
ha	hectare	mm	millimeter(s)
in	Inch(es)	mph	miles per hour
kg	kilogram(s)	nmi	nautical mile(s)
km	kilometer(s)	yd	yard(s)
kt	knot(s)	yd^3	cubic yard(s)
lb	pound(s)		

EXECUTIVE SUMMARY

Key Findings

This conference and biological opinion (opinion) and Incidental Take Statement (ITS) were prepared by the NMFS Office of Protected Resources ESA Interagency Cooperation Division (hereafter referred to as "we" or "us"). This opinion evaluated the effects of the Environmental Protection Agency's (EPA) registration of the pesticides carbaryl and methomyl on Endangered Species Act (ESA)-listed species and their designated critical habitats, as well as those species and habitats that are proposed for listing, under the jurisdiction of the National Marine Fisheries Service (NMFS).

Carbaryl and methomyl are N-methylcarbamate insecticides registered for agricultural and non-agricultural uses. Both pesticides can be applied in a variety of ways and include liquid, granular, and bait forms. Aerial and ground application methods are authorized on the labels (e.g., ground boom, aerial broadcast, and orchard airblast). Studies have shown that both pesticides are highly toxic to fish and aquatic invertebrates. Current application rates and application methods of both pesticides are expected to produce aquatic concentrations through drift and runoff pathways that are likely to harm listed aquatic species, as well as contaminate their designated critical habitats. Species and their prey residing in shallow aquatic habitats proximal to pesticide use sites are expected to be the most at risk.

Analysis and Methods

The assessment approach utilized interagency methods and procedures developed based on the recommendations of the National Academy of Sciences (NAS). This framework relied upon multiple lines of evidence to determine effects to populations, species, and their designated critical habitats. The Assessment Framework in Chapter 3 provides a description of the methodology used throughout this opinion.

When determining the effects of the action (i.e., EPA's registration of pesticides containing carbaryl and methomyl) on listed species, we considered information regarding:

- Toxicity of each chemical to aquatic taxa groups (i.e., fish and invertebrates);
- Specific chemical characteristics of each pesticide (i.e., degradation and sorption);
- Expected environmental concentrations calculated for generic aquatic habitats;
- Authorized pesticide product labels (i.e., application rates and methods);
- Maps showing the spatial overlap of listed species' ranges with pesticide use areas; and
- Species' temporal use of aquatic habitats in proximity to pesticide use areas.

The specific sources of information utilized in our analysis are outlined in Chapter 3. The effects analysis focused around risk hypotheses or statements of anticipated effects to species. We employed a weight-of-evidence approach to determine, for each risk hypothesis, whether the expected risk from pesticide exposure to groups of individuals was high, medium or low. To arrive at that rating for each risk hypothesis, we addressed not only the effect and likelihood of exposure, but also our level of confidence in the risk level. We utilized multiple data sources to evaluate both the likelihood of exposure and the magnitude of effect to groups of individuals occupying similar aquatic habitats. This allowed us to assess the body of evidence that either supported or refuted the risk hypotheses. For each species, all identified risk hypotheses were qualitatively combined into a single determination of risk at the population scale (i.e., the effects of the action) and represented graphically. A similar, yet separate, analysis was conducted for designated critical habitats where risk hypotheses were developed based on potential pesticide effects to physical or biological features of critical habitat. Generally, these included effects to water quality, vegetative cover, and species' prey items. Detailed effects analyses for both species and critical habitats can be found in Chapter 10.

Conclusions in the Draft Opinion

As discussed in Chapter 7 of this opinion, NMFS concurred with most, but not all, of the "not likely to adversely affect" (NLAA) determinations that were made for the 2 pesticides in EPA's Biological Evaluations (BEs). NMFS's subsequent jeopardy and destruction or adverse modification analyses focused on a modified list of species shown in Chapter 0 for which a "likely to adversely affect" (LAA) determination was made (either by EPA, or amended by NMFS). NMFS reviewed the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the action (including the additional conservation measures) and cumulative effects. In doing so, NMFS's conference and biological opinion found that EPA is unable to insure the registration of the uses, as described by product labels, of all pesticide products containing carbaryl is not likely to jeopardize the continued existence of 37 ESA-listed or proposed species, and is not likely to destroy or adversely modify 36 designated or proposed critical habitats within the action area. Likewise, for methomyl, we concluded that EPA is unable to insure that the registration of all pesticide products containing methomyl is not likely to jeopardize the continued existence of 30 ESA-listed or proposed species and is not likely to destroy or adversely modify 29 designated or proposed critical habitats within the action area. The details of our jeopardy and destruction or adverse modification determinations for each species and critical habitat can be found in Attachment 3 of this document. For both active ingredients (a.i.s), the bulk of the jeopardy and destruction or adverse modification of designated critical habitat determinations were for effects of the action to Pacific salmonids and their critical habitat. Also, for both active ingredients, jeopardy and destruction or adverse modification of critical habitat resulting from the effects of the action were determined for the Southern Resident killer whale and its designated critical habitat because salmon are their primary prey. Jeopardy and destruction or adverse modification of critical habitat determinations were also found for some of the other fishes that spend a significant part of their life history in freshwater. NMFS did not find jeopardy or destruction or adverse modification for any of the ESA-listed and proposed invertebrates.

Avoiding Jeopardy and Adverse Modification

Following release of the draft opinion for public comment, EPA and applicants agreed to modify the federal action to adopt the mitigation specified in the draft Reasonable and Prudent Alternative (RPA), and/or to include alternative mitigation that would result in comparable reductions in the transport of the pesticides to ESA-listed species habitats. Therefore, with this modification of the action, this opinion concludes that EPA is able to insure the registration of pesticide products containing carbaryl and methomyl is not likely to jeopardize any ESA-listed species nor cause destruction or adverse modification of designated critical habitats for the species consulted on.

As prescribed by the ESA, our findings of jeopardy to species and destruction or adverse modification of designated critical habitat required the inclusion of RPAs in the draft opinion. The draft RPAs incorporate the best available scientific and commercial information on current agricultural practices and pesticide reduction strategies to reduce pesticide exposure to aquatic species and their habitats. The draft RPAs included a flexible list of chemical-specific alternatives built upon ESA-listed species' life histories and other characteristics. In order to avoid jeopardy and destruction or adverse modification of critical habitat, the draft RPAs would reduce loading of pesticide chemicals into aquatic habitats, incorporate landowners' current conservation efforts, and protect vulnerable aquatic habitats from adverse effects of pesticide exposure.

Prior to finalizing this opinion, EPA and applicants agreed to incorporate conservation measures into the registration action consisting of elements of the draft RPA, specifically mitigation for carbaryl and methomyl exposure and/or alternative mitigation that would result in comparable reductions in the transport of the pesticides to ESA-listed species habitats. Note that NMFS has taken a similar approach in other biological opinions (e.g. that regarding EPA's registration review of three organophosphates; https://doi.org/10.25923/mqyt-xh03). In this biological opinion NMFS, EPA, USDA and the pesticide applicants were able to successfully establish mitigations that would protect ESA-listed species and at the same time offer targeted and flexible mitigation options to pesticide end users. The mitigation approach in the carbaryl and methomyl biological opinion built upon that success by introducing additional mitigation options. The primary mechanisms for addressing these elements include implementation of mitigation to reach targets for drift and runoff reductions as described in the conservation measures sections of this final opinion (Sections 5.2.2 and 5.3.2). The measures will be incorporated through a combination of general label changes and enforceable geographically-specific requirements specified in EPA Endangered Species Protection Program Bulletins (https://www.epa.gov/endangered-species/bulletins-live-two-view-bulletins).

Despite the incorporation of these conservation measures in the action, additional mitigation is required to minimize the impact of incidental taking for all species that are likely to be adversely affected by the action. We prepared an Incidental Take Statement (ITS) with associated Reasonable and Prudent Measures (RPMs) and their implementing Terms and Conditions intended to minimize such take.

Minimizing the Impact of Incidental Take

As prescribed by the ESA, the opinion includes an ITS with RPMs and their implementing Terms and Conditions to minimize the impacts of take of ESA-listed species and to minimize impacts to essential physical or biological features comprising the species designated critical habitats. The RPMs in the ITS were drafted in consultation with applicants and with EPA using the best available information on current agricultural practices and pesticide reduction strategies to minimize incidental take. The RPMs require label changes designed to reduce pesticide loading into aquatic habitats; the development of ESA educational materials to increase awareness of sensitive species in adjacent species habitats; reporting of label compliance monitoring; and clarifications regarding methods of reporting ecological incidents. The ITS and RPMs, and their implementing Terms and Conditions, are presented in Chapter 21 of the opinion.

The initial determinations of jeopardy and destruction or adverse modification of designated or proposed critical habitat in the draft opinion were made by combining the effects of the action with risk modifiers, namely the status of the species, cumulative effects, and environmental baseline. These bodies of information were combined qualitatively, described narratively, and presented graphically as species scorecards (Chapters 16-19). The final conclusions of no jeopardy and no destruction or adverse modification were achieved through a willingness of EPA and applicants to modify the action to incorporate the RPA mitigation identified in NMFS's draft opinion in the action as conservation measures.

1 INTRODUCTION

The Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7(a)(2) of the Act and its implementing regulations require every Federal agency, in consultation with and with the assistance of the Secretary (16 U.S.C. §1532(15)), to insure that any action it authorizes, funds, or carries out, in whole or in part, in the United States or upon the high seas, is not likely to jeopardize the continued existence of any listed species or results in the destruction or adverse modification of critical habitat.

Section 7(a)(4) of the ESA requires federal agencies to confer with the Secretary on any action that is likely to jeopardize the continued existence of proposed species or result in the destruction or adverse modification of proposed critical habitat. If requested by the federal action agency and deemed appropriate, the conference may be conducted in accordance with the procedures for formal consultation in §402.14. An opinion issued at the conclusion of the conference may be adopted as the biological opinion when the species is listed or critical habitat is designated.

Section 7(b)(3) of the ESA requires that, at the conclusion of consultation or conference, NMFS provides an opinion stating whether the federal agency's action is likely to jeopardize ESA-listed or proposed species or destroy or adversely modify their designated or proposed critical habitat. If NMFS determines that the Federal action agency, in this case EPA, cannot insure that the action is not likely to jeopardize ESA-listed or proposed species or destroy or adversely modify designated or proposed critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If an incidental take is expected, section 7(b)(4) requires NMFS to provide an ITS that specifies the amount or extent of incidental taking. Take in the ESA is to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct (16 U.S.C. §1532(19)). The ITS includes RPMs, which are actions necessary or appropriate to minimize impacts of incidental taking and terms and conditions to implement the RPMs.

The action agency for this consultation is the EPA. The EPA requested ESA Section 7(a)(2) consultation from the National Marine Fisheries Service on its registration of the approved uses of pesticide products containing 2 a.i.s pursuant to the Federal Insecticide Fungicide and Rodenticide Act (FIFRA). The 2 a.i.s being reviewed here are carbaryl and methomyl (carbamate insecticides). Applicants to the consultation include the registrants of technical grade carbaryl (Tessenderlo Kerley Incorporated, Bayer Crop Science, and Drexel Chemical Company) and methomyl (Tessenderlo Kerley Incorporated, Sinon Corporation, and Rotam Agrochemical Company).

On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. On November 14, 2022, the Northern District of California issued an order granting the government's request for voluntary remand without vacating the

2019 regulations. The District Court issued a slightly amended order 2 days later on November 16, 2022. As a result, the 2019 regulations remain in effect, and we are applying the 2019 regulations here. For purposes of this consultation and in an abundance of caution, we considered whether the substantive analysis and conclusions articulated in the conference and biological opinion and ITS would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

Consultation in accordance with section 7(a)(2) of the statute (16 U.S.C 1536 (a)(2)), associated implementing regulations (50 C.F.R. §402), and agency policy and guidance (USFWS and NMFS 1998) was conducted by the NMFS OPR ESA Interagency Cooperation Division (hereafter referred to as 'we' or 'us'). We prepared this conference and biological opinion (opinion) and ITS in accordance with section 7(b) of the ESA and implementing regulations at 50 C.F.R. Part §402. This document represents NMFS's opinion on the effects of these actions on 61 species, including 2 proposed species; and 56 critical habitats, including 6 proposed critical habitats.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA; section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). A complete record of this consultation was filed electronically by the NMFS Office of Protected Resources in Silver Spring, Maryland.

2 Consultation History

On March 31, 2021, EPA requested the initiation of ESA section 7(a)(2) consultation on their registration of pesticides products containing carbaryl and methomyl. EPA's BEs assessed risks to all federally-listed threatened and endangered species using EPA Revised Methods (EPA 2020). Once finalized, the current consultation will supersede the NMFS 2009 biological opinion described above in *Background*, Chapter 3.

On January 5, 2022, carbaryl registrants with agricultural products sent a letter to EPA and NMFS proposing potential changes to product labeling to mitigate risk to listed species.

On July 12, 2022, EPA and NMFS met to discuss the next steps of the carbaryl and methomyl consultation and finalize the "Carbamate Applicant Engagement Plan." EPA explained the registration and review status of carbaryl and methomyl, and indicated the action was changing from that described in the biological evaluations; EPA had negotiated mitigation for both chemicals and was also working on specific mitigation to implement the outstanding salmonid opinions. EPA reminded NMFS that EPA had revised Estimated Environmental Concentrations (EECs) for carbaryl (post-BE), to correct issues identified by the registrants in their late comments on EPA's draft BE. However, the revised EECs did not account for recent agreements to modify labels. For methomyl, EPA issued a Proposed Interim Decision in September of 2020 with some mitigation to reduce off-target movement (e.g., mandatory spray drift management language). EPA indicated that further revisions to action may occur with both pesticides given where they are in the registration review process.

On July 14, 2022, NMFS sent EPA an updated applicant engagement plan and proposed scheduling a meeting with applicants to the consultation.

On July 20, 2022, NMFS shared the applicant engagement plan with methomyl applicants and the proposed agenda for a consultation meeting to include EPA, methomyl applicants, and NMFS on August 1, 2022.

On July 25, 2022, NMFS shared the applicant engagement plan with carbaryl applicants and the proposed agenda for a consultation meeting to include EPA, carbaryl applicants, and NMFS on August 4, 2022.

On August 1, 2022, NMFS met with EPA and methomyl applicants to provide a general overview of the consultation process, verify the description of action, and discuss the applicant engagement plan, next-steps, and timeline for completing the consultation.

On August 4, 2022, NMFS met with EPA and carbaryl applicants to provide a general overview of the consultation process, verify the description of action, and discuss the applicant engagement plan, next-steps, and timeline for completing the consultation.

On August 26, 2022, NMFS met with carbaryl applicant Tessenderlo Kerley Incorporated (TKI), to discuss data they planned to submit to NMFS for consideration during the consultation. TKI reviewed proposed changes to agricultural uses and indicated that had provided EPA updated exposure estimates using the PWC model to reflect these changes, and could provide them to NMFS upon request. TKI also summarized their approach for refining PWC in a series of steps at the catchment scale. NMFS communicated their concern that catchment-scale estimates lacked relevance for assessing impacts to individuals.

On August 30, 2022, EPA confirmed in an email that the applicant Corteva had recently divested methomyl, which was purchased by TKI, and that TKI would replace Corteva as an applicant on the methomyl consultation.

On September 7, 2022, NMFS sent a summary of carbaryl uses to EPA and carbaryl applicants indicating that NMFS had made modifications to EPA's master use summary presented in the carbaryl BE by integrating the PID mitigation and mitigation proposed for agricultural labels to reduce risk to listed species (January 5, 2022 letter to EPA). NMFS further described that the document summarizes the primary aspects of the action NMFS will consider to draft its conclusions, and requested that EPA and applicants notify NMFS if the action was mischaracterized in any way.

On September 8, 2022, EPA provided commitment letters from methomyl applicants to change labeling and represent changes to the action.

On September 12, 2022, EPA provided NMFS with updated drift fractions to utilize to adjust methomyl exposure estimates to reflect updated changes to the methomyl action.

On September 12, 2022, EPA communicated that they had reviewed TKI’s EEC and found an error in one of the input parameters for PWC modeling that was likely a typo. TKI acknowledged the error, indicated that they had corrected it, reran the model simulations, and provided updated versions of the PWC output reflecting modifications to the action outlined in the PID and applicant letter of January 5, 2022.

On November 16, 2022, NMFS sent a request to EPA to extend completion of the consultation to June 30, 2023. The correspondence included a revised schedule for applicant engagement in the process. Applicants were transmitted copies of the request. EPA agreed to the extension as outlined in the letter.

On February 16, 2023 NMFS met with EPA, USDA and carbaryl applicants to discuss the authorized use of carbaryl in shrimp ponds in Texas. Subsequent to the meeting NMFS, EPA, USDA and applicants all came to agreement on label changes necessary to reduce impacts to ESA-listed species. This use was discussed separate from the agricultural uses due to its unique profile.

On March 9, 2023, NMFS transmitted a draft conference and biological opinion for EPA to post for a 60-day public comment period. NMFS was interested in receiving public input on the RPAs and other content in the draft opinion. The 60-day comment period started on March 16, 2023 and closed on May 15,2023.

On March 15, 2023, EPA posted the conference and biological opinion on their docket for the public to review.

On March 22, 2023, NMFS met with EPA, USDA, and methomyl applicants to provide an overview of the draft conference and biological opinion. A similar meeting was held with the carbaryl applicants on March 28, 2023.

On May 10, 2023, NMFS met with EPA and the carbaryl applicants to further discuss the draft RPA and RPM mitigation and to receive feedback on the draft opinion.

On May 16, 2023, NMFS received written comments from TKI (dated May 25, 2023) regarding the draft carbaryl and methomyl opinion.

On May 18 and July 26, 2023, NMFS informed EPA that there were several proposed species and critical habitats within the action area. On May 24 and July 31, 2023, EPA requested that NMFS include these proposed species in the consultation process i.e. conduct a conference opinion.

On May 31, 2023, EPA transmitted to NMFS a document containing summaries of the substantive comments that were submitted to their docket in response to NMFS's carbaryl and methomyl draft conference and biological opinion.

On June 13, 2023, NMFS sent a request to EPA to extend completion of the consultation to September 29, 2023. The extension provided NMFS, EPA and the applicants the opportunity to coordinate on critical mitigations contained in the final conference and biological opinion.

On June 15, 2023, EPA agreed with the NMFS's extension request to complete the conference and biological opinion.

On June 26, 2023, NMFS met separately with both the methomyl and carbaryl applicants to receive feedback on the draft opinion including the proposed mitigation. EPA and USDA also attended these meetings.

On July 26, 2023, NMFS received a letter from TKI providing additional comments and information regarding the opinion as a follow-up from the June 26th meeting noted above.

On August 24, 2023, NMFS, along with EPA, met with carbaryl and methomyl applicant TKI to provide clarification of our assessment and effects determination methods, and mitigation requirements in response to the comments submitted on May 15th and July 26th.

On August 31, 2023, TKI sent EPA a letter with additional comments regarding the analysis and conclusions in the draft conference and biological opinion. In their letter, TKI indicated that they would like to further discuss the proposed mitigations.

On September 6, 2023, NMFS requested an additional extension to the consultation for a completion date of November 30, 2023.

On September 7, 2023, EPA emailed NMFS to concur with NMFS's requested extension.

On October 17, 2023, NMFS and EPA received a proposal for additional mitigation options. The proposal was prepared by several companies (Balance EcoSolutions, LLC; Pyxis Regulatory Consulting, Inc.; Applied Analysis Solutions, LLC) on behalf of TKI.

On October 19th and 20th, 2023, NMFS again met separately with both the methomyl and carbaryl applicants. EPA and USDA also attended this meeting. During the meeting the group discussed the additional mitigation options that were submitted on October 17th.

On November 3, 2023, NMFS sent EPA and applicants an updated mitigation proposal applicable to both methomyl and carbaryl. In the email NMFS commented that if EPA as well as the carbaryl and methomyl applicants agree to the updated mitigation proposal, NMFS can revise the biological opinion to determine the action is not likely to jeopardize species nor result in destruction or adverse modification to critical habitat.

Between November 10 and November 20, 2023, EPA and NMFS received email responses from each of the applicants (TKI, Drexel Chemical Company, Albaugh LLC, and Sinon Corporation). These emails conveyed the applicants' acceptance of NMFS's mitigation proposal.

On November 29, 2023, NMFS sent an email requesting an extension from EPA and the applicants to complete the conference and biological opinion to incorporate agreed upon changes to the action. NMFS requested an extension until January 31, 2024.

On November 29, 2023, EPA responded by email agreeing with NMFS's extension request. The applicants also responded on November 29th and 30th agreeing with the extension request.

On December 7, 2023, NMFS sent EPA and applicants a description of the action that had been modified to incorporate, as conservation measures, the label changes necessary for EPA to insure their action is not likely to jeopardize ESA species or destroy or adversely modify critical habitat. These label changes were based on the RPA mitigations from the draft conference and biological opinion and included a number of the additional mitigation options that were proposed by the applicants.

On January 9 and January 18, 2024, EPA submitted to NMFS the final updated label summaries from the description of the action section of methomyl and carbaryl, respectively. EPA conferred with each of the applicants as to the accuracy of the summaries. EPA informed NMFS to include the summaries in the biological opinion. EPA's submission of the summary tables followed the applicants' acceptance of the proposed mitigation and represented EPA's confirmation of the change in the FIFRA action.

3 BACKGROUND

EPA requested consultation from NMFS on its registration pursuant to the FIFRA of the a.i.s carbaryl and methomyl. Pursuant to FIFRA, before a pesticide product may be sold or distributed in the U.S., it must be exempted or registered with a label identifying approved uses by EPA's Office of Pesticide Programs (OPP). Pesticide registration is the process through which EPA examines the ingredients of a pesticide; the site or crop on which it is to be used; the amount, frequency and timing of its use; and storage and disposal practices. Pesticide products (also referred to as "formulated products") may include a.i.s and other ingredients, such as adjuvants and surfactants. EPA authorization of pesticide uses are categorized as FIFRA Sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

On January 30, 2001, the Washington Toxics Coalition, Northwest Coalition for Alternatives to Pesticides, Pacific Coast Federation of Fishermen's Associations, and Institute for Fisheries Resources filed a lawsuit against EPA in the U.S. District Court for the Western District of Washington (see *Wash. Toxics Coalition v. EPA*, Civ. No. C01–132C, 2002 WL 34213031 (W.D.Wash. July 2, 2002), *aff'd*, 413 F.3d 1024 (9th Cir.2005)). This lawsuit alleged that EPA violated section 7(a)(2) of the ESA by failing to consult on the effects to 26 Evolutionarily Significant Units (ESUs) and Distinct Population Segments (DPSs) of listed Pacific salmonids of its continuing approval of 54 pesticide a.i.s. On July 2, 2002, the court ruled that EPA had violated ESA section 7(a)(2) and ordered EPA to initiate interagency consultation and make determinations about effects to the salmonids on all 54 a.i.s by December 2004. Pursuant to this court order, between August 2002 and December 2004, EPA initiated consultations with NMFS on 37 of those pesticides EPA determined "may affect" listed salmonids; the remaining 17 a.i.s were determined to have "no effect" on ESA-listed species or their designated critical habitats.

On January 22, 2004, the court in *Wash. Toxics Coalition v. EPA* (Civ. No. C01–132C), entered an injunction vacating EPA's authorization of certain uses of 54 pesticide a.i.s in certain areas and imposing certain other requirements ("Interim Measures"), until issuance by NMFS of an opinion or other described termination event. The no-spray buffers in the proposed stipulated injunction extend 300 feet from salmon supporting waters for aerial applications and 60 feet for ground applications of these a.i.s. Seventeen of the original 54 a.i.s received "no effect" determinations and thus did not require formal consultation for impacts to listed salmon ESUs.

On November 5, 2007, the Northwest Coalition for Alternatives to Pesticides (NCAP) and others filed a legal complaint in the U.S. District Court for the Western District of Washington, Civ. No. 07 1791, against NMFS for its unreasonable delay in completing the section 7 consultations for EPA's registration of the remaining 37 (of the original 54) pesticide a.i.s.

On July 30, 2008, NMFS entered a settlement agreement with NCAP. In the settlement agreement, NMFS agreed on a schedule for completion of consultation on the effects of each active ingredient to the listed Pacific salmonids, with the final consultation due in early 2013.

On April 20, 2009, NMFS issued an opinion under this schedule for 3 carbamate insecticides: carbaryl, carbofuran, and methomyl. This opinion concluded that EPA was unable to insure that their proposed registration of carbaryl and carbofuran was not likely to jeopardize the continued existence of 22 threatened and endangered Pacific salmonids and likely to destroy or adversely modify designated critical habitat for 20 threatened and endangered salmonids. The opinion further concluded that EPA was unable to insure that their proposed registration of pesticides containing methomyl was not likely to jeopardize the continued existence of 18 threatened and endangered Pacific salmonids and not likely to destroy or adversely modify 16 of their designated critical habitats. NMFS included a RPA that would allow EPA to insure their action would proceed without likely jeopardy and likely destruction or adverse modification. The RPA included no-application buffers, as well as other measures. On March 10, 2011, EPA, on behalf of itself and the Departments of the Interior, Commerce and Agriculture, asked the National Academy of Sciences (“NAS”) to evaluate the differing risk assessment approaches used by these agencies with regard to pesticides and endangered species. Specifically, NAS was asked to evaluate EPA’s and the Services’ (NMFS and FWS) methods for determining risks to ESA-listed species posed by pesticides and to answer questions concerning the identification of the best scientific data, the toxicological effects of pesticides and chemical mixtures, the approaches and assumptions used in various models, the analysis of uncertainty, and the use of geospatial data.

In October 2011, the U.S. District Court for the District of Maryland granted NMFS’s cross-motion for summary judgment and denied plaintiff’s motion for summary judgment, see *Dow AgroSciences, LLC v. NMFS*, 821 F. Supp. 2d 792 (D. Md. 2011) in regards to Dow AgroSciences’ challenge of the 2008 OP biological opinion. The dismissed case was subsequently appealed by plaintiffs to the Fourth Circuit (see *Dow AgroSciences, LLC v. NMFS*, 707 F.3d 462 (4th Cir. 2013)).

On April 30, 2013, the NAS issued a report entitled “Assessing Risks to Endangered and Threatened Species from Pesticides.” In light of the recommendations in the NAS Report, NMFS, FWS, EPA, and the U.S. Department of Agriculture (USDA) developed a common approach to risk assessment for pesticides. The NAS report contained recommendations on scientific and technical issues related to pesticide consultations under the ESA and FIFRA. Since then, the Agencies have worked to implement the recommendations. Joint efforts to date include: collaborative relationship building between EPA, NMFS, FWS and USDA; clarified roles and responsibilities for the EPA, FWS, NMFS and USDA; agency processes designed to improve stakeholder engagement and transparency during review and consultation processes; multiple joint agency workshops resulting in interim approaches to assessing risks to threatened and endangered species from pesticides; a plan and schedule for applying the interim approaches to a set of pesticide compounds; and multiple workshops and meetings with stakeholders to improve transparency as the pesticide consultation process evolves. The Agencies worked collaboratively from April 2013 through January 2017 to develop the shared scientific approaches that reflected the advice provided by the NAS for completing these pesticide consultations. Working together, scientists from the agencies met, analyzed the recommendations, and developed the approaches outlined in the Interagency Interim Approach for implementation of the NAS report: Assessing Risks to Endangered and Threatened Species from Pesticides.

On May 21, 2014, NMFS and NCAP revised the settlement agreement with NMFS on issuing a new opinion on the organophosphates chlorpyrifos, malathion, and diazinon by December 31, 2017. The agreement noted that NMFS, FWS, and EPA were working to develop a common approach to risk assessment in pesticides consultations that would implement the recommendations of the 2013 National Academies of Sciences report. This settlement agreement noted that EPA "intends to reopen its ESA evaluation of the 2 pesticides in the [2009] carbamate opinion for which there are still registered end-use products (carbaryl and methomyl) by preparing, with the assistance of NMFS and FWS, new nationwide BEs that address all NMFS species; and by reinitiating consultation with NMFS as appropriate following the completion of the nationwide evaluations (see *NCAP v. NMFS*, No. 2:07-cv-01791 (W.D. Wash.), Doc. 50, May 21, 2014)."

On March 13 2020, EPA finalized plans to transition from the Interagency Interim Approach and implement revised methods in their BEs to assess the risk of pesticides to listed species (https://www.epa.gov/endangered-species/revised-method-national-level-listed-species-biological-evaluations-conventional).

4 ASSESSMENT FRAMEWORK

Section Contents

4.1 Effects of the Action

Section 7(a)(2) of the ESA requires federal agencies, in consultation with NMFS, to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 C.F.R. §402.02).

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of an ESA-listed species (50 C.F.R. §402.02).

To conduct effects analyses, we follow an ecological risk assessment framework adapted from recommendations of the National Research Council of the National Academies of Sciences (NAS 2013a), which provided a recommended shared approach to assessing the risks to ESA-listed species posed by pesticides that are registered under FIFRA. The NAS report was jointly requested by the EPA, USDA, FWS, and NMFS. The framework divides the pesticide ESA consultation process into 3 steps (Figure 1). Each step builds upon analyses and findings from a previous step. The interagency group worked together to adapt these principles into a transparent, systematic, and rigorous analysis framework based on ecological risk assessment principles. Under this framework, EPA combines Steps 1 and 2 in their BEs and then NMFS conducts Step 3 in our opinions (Figure 1). A "no effect" determination in Step 1 indicates that the stressors of the action will not affect an individual of an ESA-listed species or designated critical habitat. NMFS does not consult on species or habitats that have received a “no effect” determination by the action agency. Although not required, an action agency may request written concurrence from the Services that the action will have no effect on listed species or critical habitat. EPA did not make this request as part of this consultation.

A NLAA determination in Step 2 indicates that the effects of the action on the fitness (survival or reproduction) of an individual of an ESA-listed species or on designated critical habitat is expected to be discountable[1], insignificant[2], or completely beneficial[3] (Endangered Species Consultation Handbook, (USFWS 1998)). Note that if EPA concludes in its Step 2 determination that its action is “not likely to adversely affect” a particular species or habitat, and NMFS concurs, then the consultation process ends at Step 2. If individuals of an ESA-listed species are not likely to be adversely affected, then ESA-listed species and the populations that comprise them are not likely to be adversely affected and no further analysis is needed. Similarly, if a given critical habitat designation is not likely to be adversely affected, then no further analysis is needed of that critical habitat. A LAA determination is made if any adverse effect to any individual of a ESA-listed species or designated critical habitat may occur as a direct or indirect result of the action and the effect is not discountable, insignificant, or beneficial.

Within the Risk Characterization section of the BEs, EPA conducted a weight-of-evidence analysis to determine whether lines of evidence were supported (i.e., an adverse effect was identified). In EPA’s analysis, lines of evidence are used to determine if an individual of an ESA-listed species is adversely affected. The lines of evidence are based on toxicological endpoints such as mortality and reproduction. The lines of evidence are analogous to risk hypotheses. EPA based each line of evidence on either adverse effects to an individual (direct effects) or adverse effects to species’ habitats (indirect effects such as effects on prey). In this manner, a supported line of evidence indicated that an ESA-listed individual’s fitness (its survival or reproduction) would likely be compromised. EPA weighed each line of evidence to determine the risk to individuals and the confidence they had in their conclusion for each line of evidence. Thus, EPA conducted a weight-of-evidence analysis in the BEs. If a line of evidence was supported, EPA made an LAA determination for that species-pesticide combination. If EPA found all lines of evidence to be unsupported, EPA made an NLAA determination.

In Step 3, biological opinion (formal consultation), NMFS evaluates EPA determinations of NLAA and LAA. Where NMFS determines a NLAA determination is warranted on an ESA-listed species, we evaluate whether the anticipated adverse effects to individuals will negatively affect populations and the species they comprise. Using the ecological risk assessment framework, described below, we conducted 2 distinct analyses within this opinion. The first analysis evaluated the risk to populations of ESA-listed species, when identified, and to entire ESA-listed species and provided the jeopardy analysis for each species. The second analysis evaluated the risk to a species’ designated critical habitat and provided the adverse modification

[1] Discountable effects are those extremely unlikely to occur.
[2] Insignificant effects relate to the size of the impact, and are effects a person would not be able to meaningfully measure, detect or evaluate. They should never reach the scale where take occurs.
[3] Beneficial effects are contemporaneous positive effects without any adverse effect to the species.

of designated critical habitat analysis. The analyses were based on the best commercial and scientific data available.

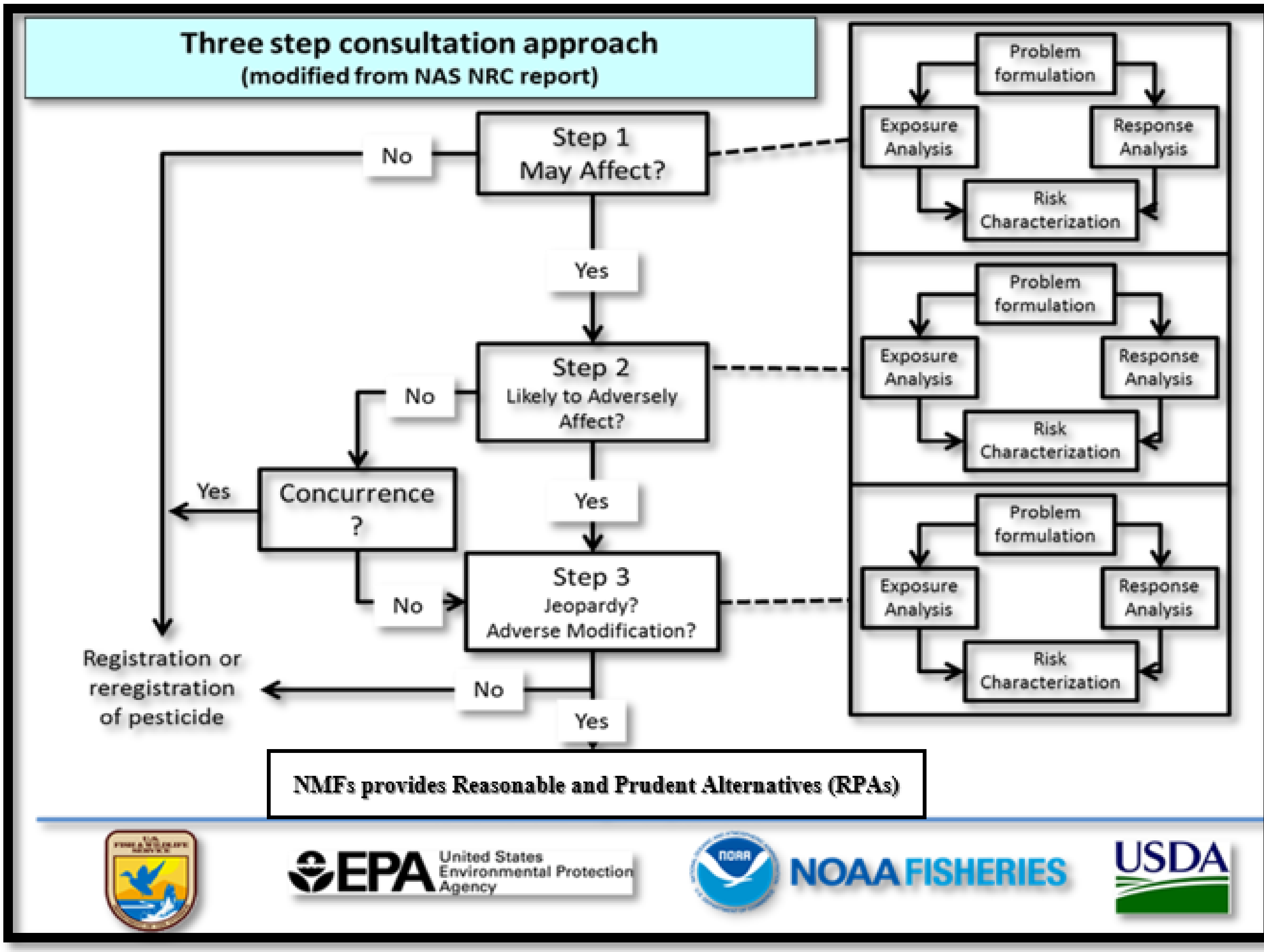

Figure 1. Three step consultation process

4.2 Information used in conference and biological opinion

To comply with our obligation to use the best scientific and commercial data available, we collected information from a variety of sources. This opinion is based on our review and analysis of various information sources, including:

- EPA's BEs
  - Pesticide label information found in Description of the Action section
  - Exposure outputs (estimated environmental concentrations) from EPA's fate and transport modeling
  - Toxicity data found in Response sections
- EPA's ecological risk assessments prepared for Registration Review
- EPA's ECOTOX database; contains published scientific studies and pesticide manufacturer studies

- Geographic locations of label authorized pesticide use sites
    - USDA – National Agricultural Statistics Services (NASS) Census of Agriculture
    - USDA/NASS – Cropland Data Layer
    - USGS – National Land Cover Database
- Published Scientific literature
- Other scientific literature, such as reports of government agencies or non-governmental organizations
- Correspondence (with experts on the subject from EPA and others)
- Available biological and chemical surface water monitoring data and other local, county, and state information
- Pesticide registrant generated data and information
- Pesticide exposure models, i.e., mathematical models that estimate exposure of resources to pesticides
    - Salmonid population models
    - Pesticide exposure models
    - Pesticide Water Calculator
    - AgDRIFT – EPA Models for Pesticide Risk Assessment
- Risk-Plots; NMFS's tool based on R-code that summarizes exposure and toxicity information by use site and is used to determine likelihood of exposure and effect of exposure to groups of individuals and designated critical habitat (see description below).
- Pesticide usage information including Pesticide Use Reports from California Department of Pesticide Regulation and estimated pesticide usage information from surveys conducted by USDA and proprietary survey information summarized by EPA
- Comments, information and data provided by the registrants identified as applicants
- Comments and information submitted by EPA
- Comments received during the public review period
- Pesticide incident reports, monitoring data, and other field data

Collectively, the above information provided the basis for our determinations as to whether the EPA can insure that its authorization of carbaryl and methomyl is not likely to jeopardize the continued existence of threatened and endangered species, and is not likely to result in the destruction or adverse modification of designated critical habitat.

4.3 Problem Formulation

Problem formulation includes conceptual models based on the initial evaluation of the relationships between stressors of the action (pesticides and other identified chemical stressors) and ESA-listed species and their habitats. We consider the toxic mode and mechanism of action of the 2 pesticide a.i.s to provide insight into potential consequences following exposure. Identification of the mode and mechanism of action allows us to identify other chemicals that might co-occur and affect species and their habitats (i.e., identify potential toxic mixtures in the environment).

We utilize the same conceptual models presented in the Step 2 analysis in EPA BEs. A conceptual model example is shown in Figure 2. The model identifies the stressors associated with the actions, the pathways and routes of exposure, the effects to be evaluated, and relationships between exposures and effects. As noted above, the fundamental difference between Step 2, BE, and Step 3, biological opinion, is we evaluate whether the anticipated adverse effects to individuals (described in the BEs) negatively affect populations and the species they comprise. However, we begin our Step 3 analysis by building on the Step 2 analysis. Additionally, we evaluate whether adverse effects to primary biological features (PBFs) reduce designated critical habitat's conservation value.

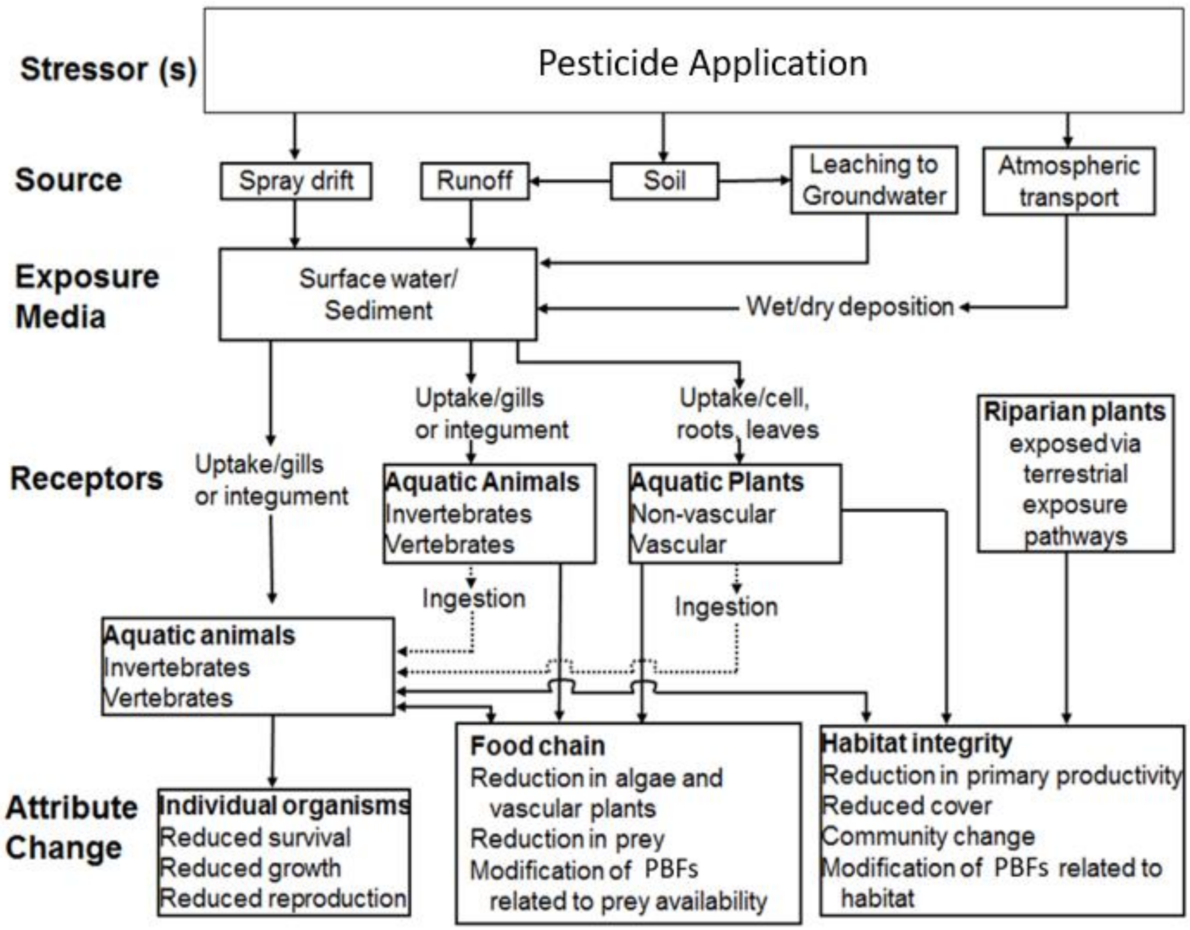

Figure 2. Conceptual model for effects to aquatic organisms

Direct deposition of carbaryl and methomyl onto treated sites, as well as transport via spray drift, runoff and volatilization resulting in atmospheric (including long-range) transport, are depicted in the conceptual models as sources that result in the movement of the pesticides into aquatic and terrestrial habitats. The movement away from the site of application in turn represents exposure pathways for a broad range of biological receptors of concern (non-target organisms) and the potential attribute changes, *i.e.*, effects such as reduced survival, growth and reproduction.

Where EPA determined that individual fitness is likely compromised by the action (lines of evidence were supported), and therefore made an LAA determination, we then determined for the Step 3 analysis if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (assessed using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). Reductions in a population's abundance, reproductive rates, or growth rates (or increased variance in one or more of these rates) based on effects to individuals represents a necessary condition for reductions in a population's viability, which is itself a necessary condition for reductions in a species' viability. Finally, our assessment determines if changes in population viability structured as risk hypotheses are likely to be sufficient to reduce the viability of the species those populations comprise. In this step of our analyses, we consider the Environmental Baseline and Cumulative Effects, and consider the species' pre-action condition, established in the Status of the Species to determine to if the action would reasonably be expected to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species.

For designated critical habitat, we determined if adverse effects (primarily, effects on water quality and prey availability) are likely to be sufficient to appreciably reduce the value of the critical habitat as a whole for the conservation of the species. To determine whether this occurs, we consider the designated critical habitat's pre-action condition, established in the Status of the Listed Resources, as well as Cumulative Effects and the Environmental Baseline.

4.4 Analysis Plan

Our analysis plan applies information from EPA's BEs to develop an assessment plan to conduct Step 3 population level analyses within the risk characterization section of this opinion.

We took the exposure and response information directly from EPA's BEs, as well as information collected via our own scientific reviews and literature analysis. As noted above, we worked closely with EPA in its preparation of this information, and our work builds on this Step 2 analysis. The exposure and response information used in our analysis is described in Chapter 10. We also describe in Chapter 10 species life history information and aggregate the species into groups based on shared life histories and habitat uses. The taxa groupings include: anadromous fish, marine fish, marine invertebrates, cetaceans (whales). Additionally, in Chapter 10, we present the mode and mechanism of toxic action for each pesticide; identify the other stressors of the action such as other chemicals within pesticide formulations; describe a pesticide's chemical and physical properties that influence its persistence in the environment; and identified key assumptions and associated uncertainties of the analytical tools and models used in the effects analyses.

The risk characterization section includes the bulk of our Step 3 analyses where we integrate the exposure and response information developed in EPA's Step 2 BEs. We employed a weight-of-evidence approach to determine for each risk hypothesis whether the risk from the action (without consideration of the species status, the environmental baseline or cumulative effects) was high, medium or low. A risk hypothesis is a statement of anticipated effects to life stage groupings of a species, such as reductions in a population's abundance or productivity following exposure to the stressors of the action. To arrive at that level of risk for each risk hypothesis, we addressed not only effect of exposure and the likelihood of exposure, but also our level of confidence in the risk level. We developed rule-based criteria to provide a systematic approach for assessing the likelihood of exposure and the effect of the exposure. We constructed risk hypotheses for each species grouping and designated critical habitats; an example is shown in Table 1.

Table 1. Example risk hypotheses for species and designated critical habitat

Risk hypotheses for species
Exposure to carbaryl is sufficient to reduce abundance via acute lethality.
Exposure to carbaryl is sufficient to reduce abundance via reduction in prey availability.
Exposure to carbaryl is sufficient to reduce abundance via impacts to growth (direct toxicity).
Exposure to carbaryl is sufficient to reduce productivity via impairments to reproduction.
Exposure to carbaryl is sufficient to reduce abundance and productivity via impairments to ecologically significant behaviors.
Exposure to carbaryl is sufficient to reduce cholinesterase (ChE) activity; the identified mechanism of toxicity
Risk hypotheses for designated critical habitat
1. Exposure to carbaryl is sufficient to reduce the conservation value via reductions in prey in rearing sites.
2. Exposure to carbaryl is sufficient to reduce the conservation value via degradation of water quality in migration, spawning, and rearing sites.

To evaluate risk hypotheses we used Risk-plot graphics, and when available, salmon population modelling. The Risk-plots are a NMFS's analytical tool that overlays toxicity data, i.e., values at which adverse effects are detected, with exposure information, i.e., estimated environmental concentrations (EECs) in differing types of aquatic habitats (referred to as bins in EPA's BEs).

4.4.1 Risk-plot Tool

The Risk-plot summarizes several types of information used in the Risk Characterization section. A Risk-plot displays pesticide exposure estimates (i.e., EECs) and toxicity data (Figure 3). The exposure estimates and the toxicity data are based on information provided by EPA (e.g., the BEs). More details regarding the data can be found later in Chapter 10 and Appendix C. We use the data presented in the Risk-plots to determine whether effect of exposure to carbaryl or methomyl is low, medium, or high for each use. We also use Risk-plots to aid in evaluating the likelihood of exposure for species and critical habitat. The sample Risk-plot below shows data for Lower Columbia River Coho salmon.

A Risk-plot graphic is read by: (1) selecting a use on the left side of the plot; (2) selecting an EEC for the use from the center of the plot; (3) reading up to a toxicity row associated with an endpoint e.g., mortality, to determine the level of effect predicted from the EEC; and (4) looking on the right side of the plot to identify the extent of the species range that overlaps with the use.

Note that the toxicity rows are constructed in different ways, depending on the assessment endpoint and the number of toxicity studies. More details are provided in Chapter 10, but a brief description follows here. For all the rows, 'NE' denotes no response expected at that EEC with the criteria depending on the endpoint. Endpoints for which all of the data fall outside the concentration range of the plot are denoted with a '<' or '>' symbol depending on the data.

For the Enzyme row, several studies were used to select a median inhibition concentration (IC50) and slope for a logistic dose-response curve to represent acetylcholinesterase inhibition. The row denotes various levels of inhibition (i.e., percent reductions in enzyme activity).

The Behavior, Reproduction, and Growth rows include assessment endpoints from all relevant studies presented in EPA's BEs, from the lowest concentration that resulted in an effect to the endpoint on the left to the highest concentration that resulted in an effect on the right, thereby capturing the range of concentrations causing effects to the associated endpoint. The concentrations shown mostly represent reported Lowest Observed Effect Concentrations (LOECs). For these 3 endpoints, there were not enough studies to conduct a species sensitivity distribution. The concentrations eliciting effects for each endpoint typically varied due to a variety of issues including the studies were conducted using different species of animals, different experimental regimes, different aged animals, etc.

For Mortality and Prey, we had a sufficient number of toxicity studies to establish species sensitivity distributions (SSDs), which show the distribution of the various concentrations eliciting death to fish and prey to the same chemical. To insure that our evaluation is sufficiently protective for the mortality and prey we constructed the toxicity row to consider the more sensitive species within the distributions, in this case, the 5th percentile of the fish SSD for Mortality and the 10th percentile of invertebrate SSD for Prey. The resulting values were used as LC50s, and along with a slope, to generate probit dose-response curves. The rows denote various levels of response (i.e., percent mortality in fish or prey).

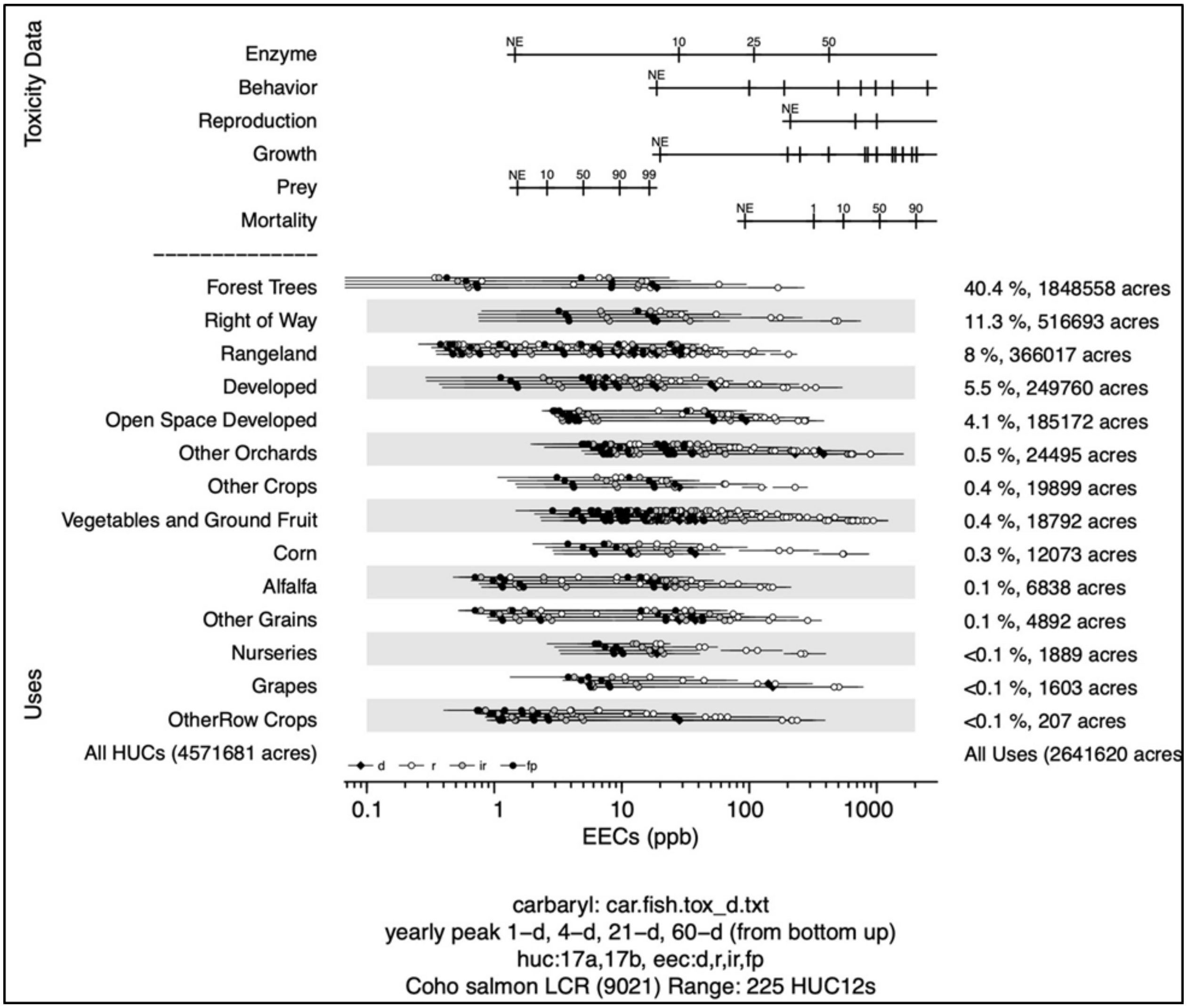

Figure 3. Example Risk-Plot for Carbaryl and Lower Columbia Coho

The bottom 4 lines of the Risk-plot indicate the following: The first line shows the chemical and the text file selected containing the toxicity data shown on the plot. The second line shows the aquatic EEC averaging periods that are being summarized (time weighted averages of 1-day, 1-day, 21-day, and 60-day). The third line provides the HUC-12 region(s)[4] and the EECs being plotted. Four types of EECs are displayed for use in the assessments: (1) aerial deposition into shallow habitat (drift, ‘d’); (2) surface runoff from the edge-of-field (runoff, ‘r’); (3) drift and runoff into EPA’s Index Reservoir (‘ir’); and (4) drift and runoff into EPA’s Farm Pond (‘fp’). For the ‘r’, ‘ir’, and ‘fp’ data points denote the median EEC for the 30 years of data with the 5-95% range of data shown by the error bars. Details on how these modeled EECs were generated and applied as surrogates to represent exposures in aquatic habitats are discussed later (e.g., Chapter 10). The bottom line shows the species name, EPA assigned ID number, and the spatial extent (number of HUC-12s) over which the data is summarized. In this example, data for the

[4] HUC stands for “hydrologic unit code,” and refers to a hierarchical system of geographic units employed by the U.S. Geological Survey. HUC-12 is a subwatershed level area.

entire range for the Columbia River Coho salmon is being aggregated, which consists of 225 HUC-12 regions.

4.4.2 Effect of Exposure

Each use/use site is evaluated to determine whether the effect of exposure is low, medium, or high based on the aquatic habitat bin EECs and the toxicity information. The effect of the exposure metric cannot be interpreted as risk without also considering its companion metric: the likelihood of exposure. Our chemical- and species-specific risk characterizations incorporate and combine these 2 metrics. See section 4.4.6 for a discussion on how confidence in this metric was considered (e.g., how representative the modeled scenario is in estimating species exposure). We took into consideration information such as duration of exposure and species habitat utilization when determining the appropriate time-weighted averages and bins to reference for the different effect endpoints. For example, a 4-day time-weighted average EEC was chosen because the standard acute lethality test with fish requires a 96-hour (4-d) exposure duration. Note that the Risk-plots displayed in the effects analysis chapters may include bins and/or time-weighted averages that were not used for direct comparison to toxicity data. Lower effects thresholds were used to evaluate acute lethality compared to reductions in prey availability as these endpoints equate directly to reductions in survival. In contrast, some reductions in prey may not always translate into fitness level consequences for the species. We apply the following guidelines for assessing the effect of exposure:

When evaluating acute lethality:

- A “none expected” rank is achieved when all relevant EECs are below the calculated 1-in-a-million sensitivity level.
- A “low” rank is achieved when all relevant EECs are below the 1% effect level.
- A “medium” is achieved when any relevant EEC falls between the 1% and the median effect level.
- A “high” is achieved when any relevant EEC exceeds the median effect level for a given toxicity range.

When evaluating reductions in prey abundance:

- A “none expected” rank is achieved when all relevant EECs are below the calculated 1% effect level.
- A “low” rank is achieved when any relevant EECs fall between the 1% and 10% effect level.
- A “medium” is achieved when any relevant EECs fall between the 10% and the median effect level.
- A “high” is achieved when any relevant EECs exceed the median effect level for a given toxicity range.

We apply the following rules when dose-response relationships are not available:

- A “none expected” rank is achieved when all relevant EECs are below all available no effect endpoints (e.g., NOEC). If NOEC values are not available we make none expected calls when concentrations are at least an order of magnitude lower than the lowest LOEC.

- A "low" rank is achieved when any relevant EEC falls between a no effect endpoint and corresponding lowest effect endpoint (e.g., LOEC).
- When EECs exceed the lowest effect endpoints we examine the effects reported at those concentrations to determine whether a low, medium or high characterization is appropriate.

4.4.3 Likelihood of Exposure

The registration of an active ingredient creates a potential for exposure by authorizing application at certain rates, times, and locations (i.e., labeled directions for use). When we assess whether the registration insures that authorized uses are not likely to jeopardize listed or proposed species, we consider the potential that the label allows, and whether or not the labels contain directions that are sufficient to insure species will not likely be jeopardized over the 15-year duration of the action, which corresponds with EPA's required timeframe for pesticide registration review.

The potential for exposure is realized when applications are made directly to aquatic habitats where listed-species are present, or pesticides are transported to these habitats from applications, which occur in proximity. Many pesticides are authorized for use directly adjacent to aquatic habitats, and some are authorized for direct application to aquatic habitats. These situations can be problematic because pesticides are inherently toxic and, therefore, exposure may result in take.

For a given chemical and species, the degree to which risk is anticipated is a function of the extent and frequency of these exposure scenarios over the entire range of the species. The extent and frequency of future usage is driven largely by market forces and the collective choices of individual end-users. Year-to-year variation in the extent and frequency is driven by variables such as: changing pest pressures, emergence of new pests, development of pest resistance, regulatory changes to products, market changes, and the choices of individual end-users. We did examine information on past use ("usage") available to NMFS (e.g., survey information on agricultural uses provided by EPA and use reporting from CalDPR PUR). Appendix D describes important inadequacies and limitations identified with the usage information. Given the degree of uncertainty and speculation associated with these factors, and usage information generally, we determined that in most cases we cannot solely rely on usage information to construct assumptions about the exposure potential. We did, however, incorporate some usage information into our risk assessment (see description below).

The likelihood of exposure assessment allows us to consider the extent of authorized use, species locations and movement, chemical properties, potential for repeated application, as well as the proximity of use sites to known areas of importance to the species. We distinguish between the extent of a use site within areas important to the species (e.g., overlap of species range with a Cropland Data Layer) and the extent and frequency of pesticide applications to that use site (e.g., usage). Over the 15-year duration of the action the former is subject to much less uncertainty, while the latter has substantial uncertainties and limitations (see Appendix D). These factors are first assessed independently of each other, and then combined to arrive at a low, medium, or high likelihood of exposure finding to help guide the risk assessment process. For example, we may find "yes" that multiple applications are authorized, but find "no" for seasonal analysis, indicating that even though pesticides are present, the species are not. In this way individual factors combine and sometimes cancel each other out. We followed a decision key (Figure 4) to help with consistency between species and use sites for this approach. The likelihood of exposure metric cannot be interpreted as risk without also considering its companion metric: the effect of exposure. Only when these 2 metrics are combined do the chemical-specific and species-specific characterizations arise. See section 4.4.6 for a discussion on how confidence in this metric was considered (e.g. how representative the use site spatial data is of actual use locations). The primary factors informing the likelihood of exposure are (Table 2):

1. **Percent overlap of a species' U.S. range with a pesticide's approved uses.** Each use is assigned a category of 1, 2, or 3 depending on the degree of geographic overlap of use acreage with the species' U.S. range acreage (aggregation of HUC-12s that delineate the species range). If EPA has authorized pesticide application for a particular site, that site may receive one or more pesticide applications during the course of the 15-year action. An important consideration in estimating the likelihood of exposure to a species is the extent to which authorized use sites occur in direct proximity to species habitats and/or known occupied areas. However, we do not assume that usage will occur in every authorized use site, nor do we assume that all usage occurs at the same day and time. Instead, we assume that: 1) the pesticide may be applied to any authorized site, and 2) the greater the extent of authorized use sites in the species range, the greater the chance that application may occur in close proximity to species habitat. The distinction, between "will be applied to every" and "may be applied to any," is important in understanding the assumptions of our analysis. The assumption that every use site will receive application is neither realistic nor appropriate.

   Our interpretation of the percent overlap values considered the reality that all registered use sites are not likely to receive application of the pesticide active ingredient, for those that did, applications would not all occur at the same time. We considered the percent overlap value as 1 of 6 factors which qualitatively determines the likelihood of exposure. We assumed that, all else being equal, there is a positive relationship between the amount of land authorized for pesticide application and the chance that a species will be exposed. In recognition of the complexities in this relationship, as well as the numerous other factors influencing the likelihood of exposure, we developed a systematic but qualitative framework to help characterize risk. In this way, the percent overlap serves as a proxy for informing the potential for pesticide application in close proximity to species habitats.

   Acreage of authorized use sites was provided by EPA and is based largely on USDA's Cropland Data Layer; this information is presented on the left Y-axis of the Risk-plot in Figure 2. Species range information comes from NMFS listing documents, recovery plans, and status reviews.

2. **Seasonal analysis based on allowable application timing overlaid with species' timing to determine co-occurrence.** Application timing is based on authorized label restrictions (e.g., language indicating applications are restricted to the pre-emergence period). Species timing of occupancy for aquatic areas is provided in the Status of the

Species section. The co-occurrence addresses whether pesticides are allowed to be applied during species presence at various life stages. We answer "yes" to the question of co-occurrence in cases where the pesticide may legally be applied when a species-life history suggests it may be present.

3. **Persistence of the pesticide based on environmental fate issues.** We evaluated the environmental fate information provided in the BE and EPA ecological risk assessments to determine whether the pesticide is considered persistent. As a rule of thumb, we answered "yes" to persistence if the pesticide has relevant half-life values greater than 100 days. For this purpose, carbaryl and methomyl were both treated as non-persistent.

4. **Number of applications allowed.** We assume that an increase in number of authorized applications increases the likelihood of an exposure and the potential of effect. We reviewed EPA's description of the action, as well as authorized labels, to determine whether multiple applications were allowed on each use site. When answering "yes" or "no", we considered the relative risk of a single application at the maximum allowed rate versus multiple applications at a reduced rate. Some labels do not explicitly state the number of repeat applications authorized, instead labels may specify a maximum single application rate as well as a maximum annual application rate. If, for the majority of labels in a given category (e.g., other grains), the maximum single application rate equals the maximum annual application rate then we answered "no" for this factor. Although it is possible that multiple applications could occur at lower rates, assuming a single application at the maximum rate allows us to capture and assess the potential for risk as authorized by the label.

5. **Proximity analysis, for use sites with less than 1% overlap within a species range.** We evaluated the available spatial data for use sites and species distribution/life history information to determine whether: 1) use sites were aggregated in proximity to sensitive areas (e.g., known spawning streams or nursery areas); or 2) whether up-stream use sites were likely to substantially increase exposure via downstream transport. When evaluating a map, we considered aggregation of use sites within a HUC-12 as "in proximity" when they were within roughly 300 meters of where we anticipate the pesticides would either runoff or drift to those habitats.

   The likelihood that pesticide will be transported to ESA-listed species habitats in concentrations that are harmful to them is inversely related to the distance of the pesticide application to the species habitat. We determined 300 meters to be appropriate for this purpose for the following reasons. First, when considering drift, 300 meters represents an upper limit to which we are able to effectively model transport and evaluate quantitative estimates (e.g., via AgDrift). Although transport beyond 300 will occur, we anticipate the bulk of off-site transport of pesticides from drift and runoff will be constrained within 300m of the application zone in most scenarios. When considering run-off, the potential for transport is highly variable depending on site-specific and chemical-specific factors. Three-hundred meters represents a point at which the potential for transport via sheet or channelized flow can be considered minimal in most cases. Although off-site transport beyond 300 meters following pesticide application is possible, the 300 meter distance is appropriate for focusing attention on applications that pose the greatest risk. Note that for these same reasons, 300 meters was selected for use in our pick-list mitigation approach (see ITS chapter 21). In this approach, we use 300 meters as the distance within which end users must check to see whether restrictions are necessary. The mitigation approach is flexible such that, depending on the application rate and method, restrictions within the 300 meters may or may not be necessary.

For many of the species assessed, we determined sensitive areas by identifying, for example, designated critical habitat within populations that have been identified in recovery plans as "core" or "essential" to the recovery of the species. In cases where we answer yes for this factor, the overall likelihood of exposure characterization is assigned a low, medium, or high on a case-by-case basis depending on the nature of the exposure potential. For example, aggregations in proximity to known spawning sites may represent a higher exposure potential due to the high density of individuals present, as opposed to aggregations to coastal areas used for foraging, where individuals may be more disperse (depending on species).

6. **Duration of species occupancy in aquatic systems.** We review the species life history to determine the approximate duration for residency and migration.

Table 2. Criteria used to determine likelihood of exposure

Factor	**Criteria Description**	**Criteria**
Percent overlap of use site within species HUC-12 watersheds	low overlap = <1 % = category 1 Medium overlap = 1-5 % = category 2 High overlap = >5 % = category 3	category (1;2;3)
Seasonal Analysis (proportion of year life stages are potentially exposed)	Are any species life-stages present in overlapping areas when pesticide application are allowed? (Y/N)	Yes or No
Persistence of pesticide	Is pesticide considered persistent? (Y/N) Pesticide has a relevant half-life greater than 100 days = Y.	Yes or No
Number of applications	Are multiple applications authorized per year? (Y/N)	Yes or No
Proximity Analysis: Use sites proximal to sensitive areas Or Potential for exposure from upstream sources	Are use sites within 300 meters of sensitive areas? (Y/N) Or Are upstream use sites likely to substantially increase exposure via downstream transport? (Y/N)	Yes or No
Time spent occupying aquatic areas	Species residency: Days, months, years <30 days=1 ; 1-6 months(1-2 seasons) = 2; multiple years = 3	category (1;2;3)

	Species migration: Days <7 days =1; 7-21 days =2 ; >21 days = 3	category (1;2;3)

Percent Overlap Category	Seasonal Analysis	Persistence	Multiple Applications	Proximity Analysis	Duration of migration/residency	Likelihood of Exposure
3	yes	yes/no	yes	NA	3	**High**
3	yes	yes	yes	NA	1/2	**High**
2	yes	yes	yes	NA	3	**High**
3	yes	no	yes/no	NA	2	**Medium**
3	yes	no	no	NA	3	**Medium**
2	yes	no	yes/no	NA	3	**Medium**
2	yes	no	yes	NA	2	**Medium**
2	yes/no	no	no	NA	2	**Low**
1	yes/no	yes/no	yes/no	no	1/2/3	**Low**
1	yes/no	yes/no	yes/no	yes	1/2/3	**Low/Med/High***

Figure 4. Likelihood of exposure decision key. *See explanation of proximity analysis, described above.

For each species assessed, NMFS has characterized the "likelihood of exposure" relative to each use site (e.g., corn, wheat) within that species' range. The likelihood of exposure for each use site is characterized as either low, medium or high depending on the combinations of the factors. The decision key was used to help guide decision making, and maintain consistency and transparency across species and chemicals; deviations from the decision key were made on a case by case basis as appropriate and are documented in the effects analysis section. Additionally, note that the combinations provided in this key are not exhaustive of all possible combinations, rather they represent only those combinations which were encountered in this opinion.

4.4.4 Risk Determination for Each Risk Hypothesis

The "likelihood of exposure" and "effect of exposure" evaluations for each use category are all considered together in determining whether the overall risk associated with the risk hypothesis is high, medium, or low. Note, it is important to recognize that these characterizations refer to effects at the species population level of organization; species being evaluated during this phase of the analysis have already received LAA determinations for individual level effects. In this way we consider the combined impact of all uses within the species range or habitat. The 3-by-3 matrix below serves as a conceptual illustration of how likelihood of exposure and effect of exposure combine to create greater or lesser risk. A "high" risk determination is made when we anticipate that adverse effects will impact an extent of the species such that population-level effects may result, depending on the status of the species and environmental baseline. Medium and low risk both indicate that we anticipate some adverse effects but that we do not anticipate these effects capable of scaling to population-level impacts.

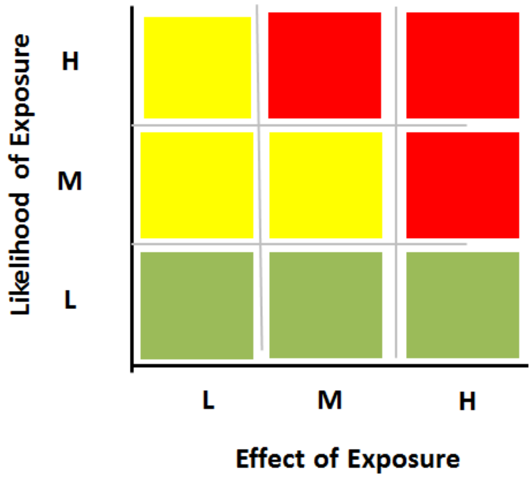

Figure 5. Ranking Risk Hypotheses Based on Uses. Each use is plotted based on Likelihood of Exposure finding and Effect of Exposure finding. L=low (green), M=medium (yellow), H=high (red).

We also considered evidence provided by another important source of information: 2 salmonid population models developed by NMFS to evaluate 4 life history strategies of salmon. This evidence informed our determination of the confidence we had in our assignment of the overall risk from the Effects of the Action to species. In this way, the Risk-plot, and population modeling results are considered together when determining whether a risk hypothesis is supported or not.

Qualitative population-level assessments are conducted by examining the risk hypotheses that directly affect population and species responses to the action. The "likelihood of exposure" and "effect of exposure" evaluations for each use category are all considered together in determining whether the overall risk associated with the risk hypothesis is high, medium, or low. Note, it is important to recognize that these characterizations refer to effects at the species population level of organization; species being evaluated during this phase of the analysis have already received LAA determinations for individual level effects. In this way we consider the combined impact of all active ingredient uses within the species range or habitat. The 3-by-3 matrix above (Figure 5) serves as a conceptual illustration of how the likelihood of exposure and effect of exposure combine to create greater or lesser risk. A "high" risk determination is made when we anticipate that adverse effects will impact a portion of the species such that population-level effects may result, depending on the status of the species and the environmental baseline. Medium and low risk both indicate that we anticipate some adverse effects but that we do not anticipate these effects capable of scaling to population-level impacts. The culmination of all the risk hypotheses determines the overall risk posed to the population or species that is then considered in our jeopardy and destruction/adverse modification analysis.

We also considered important information provided by 2 salmonid population models developed by NMFS to evaluate 4 life history strategies of salmon (for more information, see Appendix A). This evidence informed our determination of the confidence we had in our assignment of the overall risk from the Effects of the Action to species. In this way, the Risk-plot, and population modeling results are considered together when determining whether a risk hypothesis is supported or not.

4.4.5 Salmon Population Models

For certain salmon, we applied peer-reviewed, published population models as a tool to estimate population level responses to the 2 insecticides (see Appendix A). The salmon population modelling results are reported as percent reductions in a population's growth rate. The model results estimating population level effects from mortality to juveniles were used to inform the following risk hypothesis: exposure is sufficient to reduce juvenile abundance via acute lethality. Model results estimating population level effects from effects to juvenile growth from reductions in prey as well as sublethal effects to juveniles were used to inform the following risk hypotheses: exposure is sufficient to reduce juvenile abundance via reduction in prey availability; exposure is sufficient to reduce abundance via impacts to growth (direct toxicity); exposure is sufficient to reduce ChE activity (the identified mechanism of toxicity). Percent changes in the population growth rate were considered significant if they were outside of 1 standard deviation from an unexposed population. A decline in the population growth rate by more than 1 standard deviation was chosen as a cutoff to prevent consequential reductions in population abundance and increases in population variability, which may affect the continued existence of the species. The use of 1 standard deviation reduces the chance of making a type II error, i.e., incorrectly stating that the species is not impacted by the action when it is impacted.

Sufficient data were available to construct population models for 4 Pacific salmon life history strategies. We ran life-history matrix models for ocean-type and stream-type Chinook salmon (*O. tshawytscha*), coho salmon (*O. kisutch*), and sockeye salmon (*O. nerka*). The basic salmonid life history we modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. For specific information on the construction and parameterization of the models see Appendix A. Potential impacts resulting from freshwater exposure to pesticides were integrated into the models as alterations in the first year survival rate. Effects of acute mortality or changes in somatic growth rate were evaluated using independent models discussed below. Population level impacts for both types of models were assessed as changes in the intrinsic population growth rate and quantified as the percent change in population growth rate. Changes that exceeded the variability in the baseline (i.e., 1 standard deviation) were considered to be significant.

Acute toxicity models were constructed that estimated the population-level impacts resulting from sub-yearling exposure to the single a.i.s. The model did not consider multiple exposures, effects to other life stages, or any sublethal or habitat-related effects. We determined population outcomes when different percents of sub-yearlings experienced exposure sufficient to cause lethality to different percents of the individuals exposed (0 to 100% mortality in 5% increments), the range of mortality corresponding to EECs on carbaryl and methomyl Risk-plots.

A somatic growth model was developed explicitly to evaluate the potential for adverse effects to juvenile growth resulting from exposure to the a.i.s (Appendix A). The model links AChE inhibition, feeding behavior, prey availability, and somatic growth of individual salmon to the productivity of salmon populations expressed as a percent change in a population's intrinsic rate of growth (lambda). Subyearling salmon experience size-dependent survival near the end of their first year, linking changes in somatic growth to population productivity. The model scenarios assume annual exposure of sub-yearling juveniles and their prey to the a.i. We integrated 2 avenues of effect to juvenile salmonids' growth from exposure to the 2 a.i.s (Appendix A). The first avenue is a result of direct AChE inhibition on feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors. The second avenue the model addresses is the potential for indirect reductions in juvenile growth resulting from reduction in available prey. Salmon are often food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced somatic growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to AChE inhibitors.

4.4.6 Confidence Ranking for Each Risk Hypothesis

We consider and track the underlying uncertainties when evaluating the risk hypotheses by describing levels of confidence in our effect of exposure, likelihood of exposure, and in our overall risk determination for each risk hypothesis (Table 3). The confidence level is assigned a low, medium, or high level in each of these 3 areas as described below.

Table 3. Example risk hypothesis assessment table illustrating where uncertainties are considered in assessing risk.

Endpoint: Mortality				
Use Category	**Effect of Exposure**	**Confidence**	**Likelihood of Exposure**	**Confidence**
Use Site A	High	(A)	High	(B)
Use Site B	High	(A)	High	(B)
Use Site C	High	(A)	High	(B)
Risk Hypothesis: Exposure to carbaryl is sufficient to reduce abundance via acute lethality.				
Risk	**Confidence**			
High	(C)			

Confidence in the effect of exposure (A)

- *Quality and representativeness of EECs*
  Here we consider how best to interpret the modeled estimates (e.g., runoff values) in the context of estimating the exposure concentrations of these pesticides in actual species habitats. As described earlier, these modeled estimates represent the best available data for anticipated concentrations resulting from applications. However, we do not equate any one modeled scenario directly to any one habitat. Instead, we consider all of the

information available (including monitoring data, habitat type, etc.) to define the most appropriate range of exposure values for comparison against toxicological data. See Chapter 10 for a description of the habitats and the uncertainties associated with exposure estimates.

- *Quality and representativeness of effects data*
  We reviewed the available toxicity information in light of our data quality standards to evaluate the level of confidence in the toxicity information used to determine effects to an ESA-listed species and its habitats. For example, we would ascribe higher confidence for a toxicity endpoint when a robust species sensitivity distribution (SSD) is available and lower confidence when SSDs are not available. Other considerations included: the number of available studies; the distribution of the effects; representativeness of test species; duration of test exposure, etc.

Confidence in the likelihood of exposure (B)

- *Representativeness of CDL/UDL*
  How well does the overlap data (e.g., UDL) represent the specific labels and uses assigned to it? Many of the percent overlap estimates presented in the Risk-plots are based on overlap between species range and Cropland Data Layer (CDL) class groupings (e.g., vegetables and ground fruit). The CDL has over 100 different cultivated classes which were grouped by EPA in order to reduce the likelihood of errors of omission and commission between similar crop categories. CDL groupings were designed to minimize uncertainties, however they also introduce the possibility that overlap percentages include uses for which the active ingredient has not been registered. Spatial data for non-agricultural uses similarly comes with uncertainties and was considered here. We also considered whether or not there is additional evidence, beyond the spatial layers, that registered uses may occur in a species range. Sources of information used to assess this factor include USDA's NASS Census of Agriculture, monitoring data, incident data, and available usage information. Additionally, all the sources of information are snapshots in time and may not be representative of future use over the duration of the action within a species range. While aggregating multiple years of CDLs and multiple CDLs into a UDL addressed some of this uncertainty (e.g., crop rotations), the potential for changes in overlap with time was considered as well.

- *Evidence that future usage will be minimal*
  We evaluated whether there is sufficient evidence that usage is likely to be minimal over the next 15 years. By minimal, we mean that the amount of acres treated or pounds applied are such that even if application is made in proximity to species habitat we would anticipate the exposure level would not result in an appreciable reduction in the reproduction, numbers or distribution of the species or in the value of designated critical habitat for that species. By sufficient evidence, we mean that there is enough certainty in the evidence that we can rely on it as part of an analysis (along with the status of the species and critical habitat, the environmental baseline, and cumulative effects) that ultimately is required by the ESA to insure that the species is not likely to be jeopardized over the 15-year duration of the action. Substantial population-level impacts to species are not only possible via large-scale impacts across the entire species range, but could

occur via smaller scale impacts to essential sub-populations, or essential life stages. Information used to assess this factor include regulatory actions (e.g., state-bans), monitoring data and available usage data. In determining whether the evidence available was sufficient or not, we considered factors such as data quality, reliability, transparency, completeness, and applicability to ESA assessment.

  - Examples of evidence that we found sufficient for this purpose:
    - Mandatory, long-term, geographically specific usage reporting (e.g., CA PUR data). For this information type, we determined there was sufficient evidence that future usage would be minimal if there had been no usage reported in the previous 15 years for the relevant use.
  - Examples of evidence that we found were not sufficient for this purpose:
    - Usage estimates based on survey data (limitations discussed in Appendix D)
  - Water quality monitoring data. In particular, the absence of detections or detections of the pesticide at low concentrations (limitations discussed in Section 10.2.2).

- *Exposure pathway*
  We considered the potential exposure pathways for each species. For example, we have greater confidence in the exposure pathways to salmonid spawning habitat from adjacent use sites than we do for intertidal habitats (e.g., black abalone). Here we also weigh our understanding of species utilization of the particular portions of the range in which there are co-occurrences of use sites and potential habitat. Even in cases with relatively high overlap percentages or acreages it is possible that use sites are not anticipated to contribute significantly to exposure potential.

Confidence associated with overall risk determination (C)

- *Overall confidence from effect of exposure and likelihood of exposure*
  We consider the confidence determinations from the effect of exposure and likelihood of exposure assessments described above.

- *Overall impact of all uses*
  We consider the degree of similar combinations of likelihood of exposure and effect of exposure i.e., the more uses and toxicity endpoints for which there is the same combination of "likelihood of exposure" and "risk of exposure" (e.g., "high/high," ("low/medium"), the more confidence we have in the low/medium/high risk assignment for the associated risk hypothesis.

4.4.7 Effects Analysis Overall Risk

Once we assessed each individual risk hypothesis for its level of risk and confidence, we then translated these values into an assessment of the overall risk posed to the species (low, medium, or high) based on all of the risk hypotheses. To make this conclusion, we plotted the risk hypotheses on a graph based on the risk and confidence determinations for each risk hypothesis. This is illustrated below. For example, if 1 or more risk hypotheses had high risk and high confidence then we determined that the overall risk to the species was high, placing it in the red squares. We also determined the overall risk to the species as "high" if, for any risk hypothesis, one of the variables (level and confidence of risk) was high and the other was medium. If all risk hypotheses landed in the yellow and green squares then the conclusion was determined to be medium risk for the species. If all risk hypotheses landed in the green squares the conclusion was determined to be low risk for the species. For this purpose, the AChE risk hypothesis (e.g., "Exposure to carbaryl is sufficient to reduce ChE activity; the identified mechanism of toxicity") was treated as a supporting risk hypothesis and did not, on its own, lead to a conclusion of high risk.

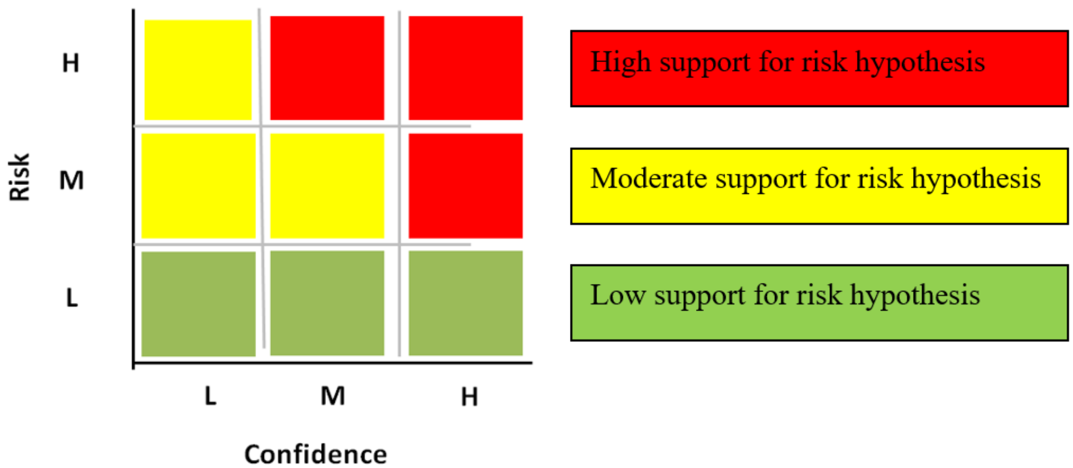

Figure 6. Each individual risk hypothesis is plotted based on its associated risk and confidence.
For each species, the effects analysis concludes with a cumulative risk and confident conclusion considering all risk hypotheses determinations with a summary paragraph as well as a risk bar visual (see example below).

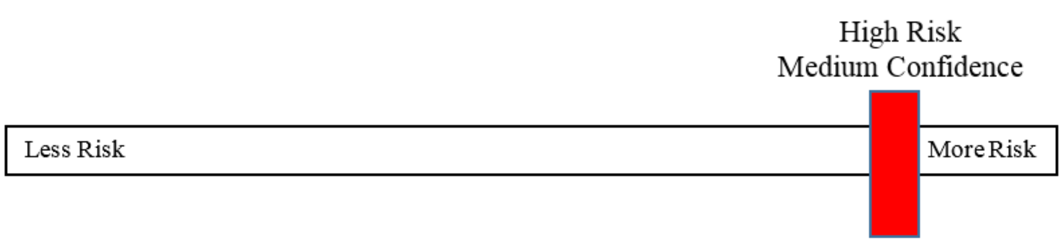

Figure 7. Example risk-bar associated with the stressors of the action

The effects analysis conclusions are then carried forward into the integration and synthesis section to be considered in the context of the Environmental Baseline, Status of the Species, and Cumulative Effects.

4.4.8 Designated Critical Habitat Analyses

Our critical habitat analysis determines whether the action is likely to destroy or adversely modify critical habitat for ESA-listed species by examining potential reductions in the conservation value of the essential features of designated critical habitat. “Destruction or adverse modification” means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat as a whole for the conservation of an ESA-listed species (50 C.F.R. §402.02).

In this section, NMFS evaluates the potential consequences to designated critical habitat from exposure to the stressors of the action. Each risk hypothesis is based on potential impacts to those PBFs essential to the conservation of the species which may require special management considerations or protection (see Appendix B for a description of the PBFs for each of the designated critical habitats). The critical habitat effects analysis concludes with our determination of the risk posed to all the PBFs taken together for a particular species. In the integration and synthesis section we combine the effects analysis with the baseline status of the habitat and the cumulative effects to evaluate the potential consequences to designated critical habitat as a whole.

As described in the preamble to the updated regulations: “Consistent with longstanding practice and guidance, the Services must place impacts to critical habitat into the context of the overall designation to determine if the overall value of the critical habitat is likely to be appreciably reduced. The Services agree that it would not be appropriate to mask the significance of localized effects of the action by only considering the larger scale of the whole designation and not considering the significance of any effects that are occurring at smaller scales (see, e.g., Gifford Pinchot, 378 F.3d at 1075). The revision to the definition does not imply, require, or recommend discounting or ignoring the potential significance of more local impacts. Such local impacts could be significant, for instance, where a smaller affected area of the overall habitat is important in its ability to support the conservation of a species (e.g., a primary breeding site). Thus, the size or proportion of the affected area is not determinative; impacts to a smaller area may in some cases result in a determination of destruction or adverse modification, while impacts to a large geographic area will not always result in such a finding” 84 Fed. Reg. 44976, 44983 (Aug. 27, 2019).

In all of the critical habitat designations that may be exposed to the stressors of this action, water quality and prey availability are key attributes that are either designated as PBFs of the critical habitat, or are relevant to the PBFs. Water quality encompasses a range of typically measured parameters, including dissolved oxygen, temperature, turbidity, and presence of contaminants. Insect and phytoplankton development is critical for the aquatic ecosystem, in particular as prey for juvenile salmonids, as they are essential to juvenile salmonid growth and survival in both freshwater and nearshore marine environments. Here, we use the presence of chemical contaminants as an indicator of degraded water quality. The action could degrade water quality by introducing carbaryl, methomyl and other associated chemicals into designated critical habitats. Therefore, we use the pesticide concentrations likely to adversely affect ESA-listed species, prey (e.g., juvenile fish and invertebrates), or aquatic vegetation as measures of degraded water quality.

Similar to the species assessment we considered risk plots and, where appropriate (e.g., impacts on prey), made effect of exposure and likelihood of exposure determinations. For the likelihood of exposure, we considered a sub-set of the factors, which were used for the species assessment:

1) percent overlap; 2) chemical persistence; and 3) the number of applications allowed. The other 3 factors (application timing, species duration of occupancy, proximity to habitat) were still considered but given less weight.

Also similar to the species effect analysis, we assessed each individual risk hypothesis for its level of risk and confidence, we then translated these values into an assessment of the overall risk posed to the designated critical habitat (low, medium, or high) based on all of the risk hypotheses. Each effects analysis concludes with a summary paragraph as well as a risk bar visual. The effects analysis conclusions are then carried forward into the integration and synthesis section to be considered in the context of the Environmental Baseline and Status.

4.4.9 Integration and Synthesis

In this section, we provide NMFS's opinion regarding whether or not EPA can insure their action, when aggregated with factors analyzed under "environmental baseline," "effects of the action," and "cumulative effects" in the action area, and when viewed against the status of the species or critical habitat as listed or designated, is not likely to jeopardize the continued existence of the species or not likely to result in destruction or adverse modification of critical habitat.

The status of the species, environmental baseline, and cumulative effects are considered in the context within which the action occurs. Additionally, there are several factors within each of these sections, which we anticipate will directly interact with the effects of the action. For example, elevated temperatures have been demonstrated to increase the toxicity of organophosphate pesticides in fish (Mayer 1986; Mayer and Ellersieck 1988; Osterauer and Köhler 2008) and certain mixtures of cholinesterase inhibiting pesticide increase the toxicity to juvenile coho salmon (Laetz et al. 2014).

Once each of the above sections is evaluated, they are depicted graphically on a "scorecard." The influence of each section is represented by an arrow. The magnitude of influence (low or high) is represented by the length of the arrow (short or long). The direction an arrow is pointed indicates the directionality of the section, increasing or decreasing risk. For example, a status of the species arrow pointing towards more risk may indicate that the population dynamics are such that the species would be highly vulnerable to the additional adverse impacts associated with the effects of the action. The level of confidence in each factor is indicated by the thickness of the arrow (high confidence represented by thick arrow, low confidence represented by thin arrow).

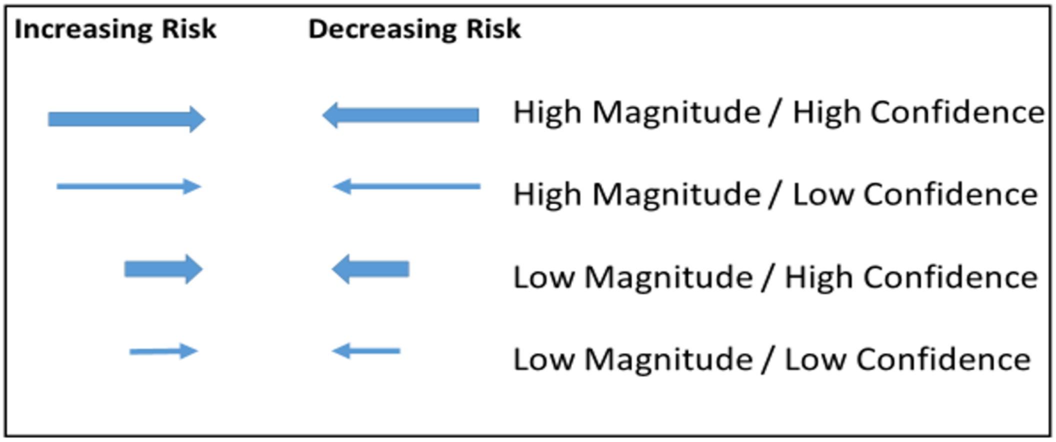

Figure 8. Example of arrows to represent direction, magnitude, and confidence of the status, baseline, and cumulative effects sections used in the species scorecard.

4.4.10 Conclusion

With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on essential habitat features when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

- Reduce appreciably the likelihood of survival and recovery of an ESA-listed or proposed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether EPA has insured their action is not likely to jeopardize the continued existence of such species; or
- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether EPA has insured their action is not likely to destroy or adversely modify designated or proposed critical habitat.

A "scorecard" is generated for each species and designated critical habitat. The effects of the action are characterized as high, medium, or low risk to the species on the top bar ("Effects Analysis") of the scorecard, using the analytical process already described. The scorecard also summarizes the influence of the environmental baseline, cumulative effects, and status of the species, as depicted by the 3 arrows below the Effects Analysis bar. At the bottom of the scorecard, the bar labeled Conclusion shows the overall risk and jeopardy determination (the colored bar beginning with green [less risk] to red [more risk]). A narrative is also presented below the scorecard to identify risk drivers and summarize the overall conclusion. The jeopardy determination and the destruction or adverse modification determination for each species or designated critical habitat is based on best commercial and scientific data available following ecological risk assessment principles.

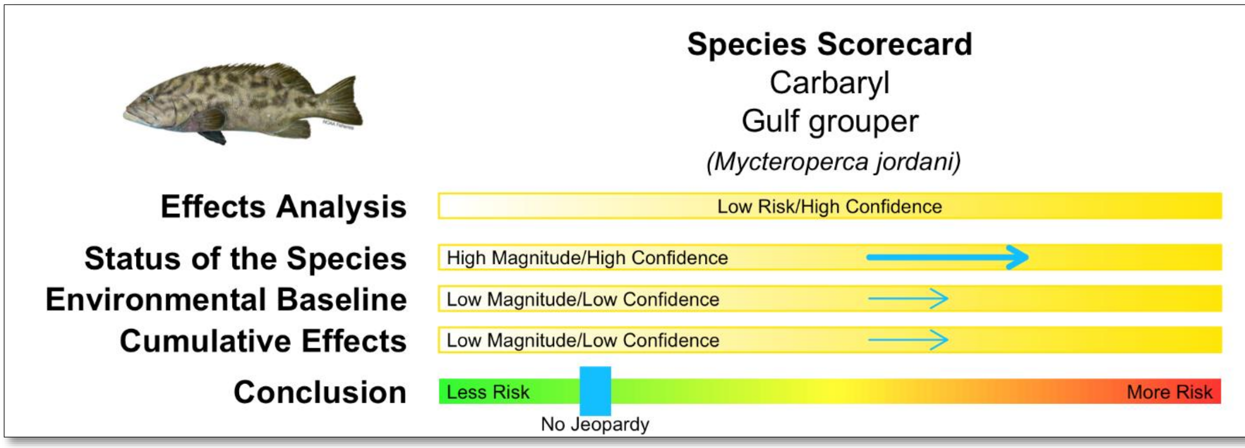

Figure 9. Example Species Integration and Synthesis "scorecard"

If, in completing the last step in the analysis, we determine that EPA is unable to insure the action under consultation is not likely to jeopardize the continued existence of an ESA-listed species or destroy or adversely modify designated or proposed critical habitat, then we must identify reasonable and prudent alternative(s) (RPA[s]) to the action, if any, or indicate that to the best of our knowledge there are no RPAs (See 50 C.F.R. §402.14).

In addition, we include an ITS that specifies the impact of the take, RPMs to minimize the impact of the take, and terms and conditions to implement the RPMs (ESA section 7 (b)(4); 50 C.F.R. §402.14(i)). We also provide discretionary conservation recommendations that may be implemented by action agency (50 C.F.R. §402.14(j)). Finally, we identify the circumstances in which reinitiation of consultation is required (50 C.F.R. §402.16).

5 DESCRIPTION OF THE ACTION

Section Contents

5.1 The Federal Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies.

Under FIFRA, the purpose of the EPA action is to provide pest control that does not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. Under FIFRA, before a pesticide product may be sold or distributed in the U.S., it must be registered with a label identifying OPP-approved uses. Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label(s) (https://www.epa.gov/regulatory-information-topic/regulatory-and-guidance-information-topic-pesticides). EPA authorizations of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) (special local need).

The action for this consultation is EPA's registrations of all pesticides containing carbaryl or methomyl for use as described on product labels.[5] The action includes (1) approved product labels containing carbaryl and methomyl, (2) degradates and metabolites of carbaryl and methomyl, (3) formulations, including other ingredients within formulations, (4) adjuvants, and (5) tank mixtures. EPA is required to reassess each registered pesticide at least every 15 years. A summary of EPA's label changes related to this consultation for carbaryl and methomyl can be reviewed in the attachment folder. See Attachment 4 to review these summaries.

[5] EPA's registrations are two separate actions that we have combined in one opinion. We considered the effects of each of EPA's actions separately and independently. For convenience, we will refer to one action.

EPA's pesticide registration process involves an examination of the ingredients of a pesticide, the site or crop on which it will be used, the application method, amount, frequency and timing, and its storage and disposal practices. Pesticide products may include a.i.s and other ingredients, such as adjuvants, and surfactants (described in greater detail below). The EPA evaluates the pesticide to ensure that it will not have unreasonable adverse effects on humans, the environment, and non-target species. An unreasonable adverse effect on the environment is defined in FIFRA as: "(1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of the pesticide, or (2) a human dietary risk from residues that result from a use of a pesticide in or on any food inconsistent with the standard under section 408 of the United States Federal Food, Drug, and Cosmetic Act (FFDCA) (21 U.S.C. §346a; 7 U.S.C. 136(bb)).

After registering a pesticide, EPA retains discretionary involvement and control over such registration. EPA must periodically review the registration to ensure compliance with FIFRA and other federal laws (7 U.S.C. §136d). A pesticide registration can be canceled whenever "a pesticide or its labeling or other material does not comply with the provisions of FIFRA or, when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment" (7 U.S.C. §136d(b)).

EPA, NMFS, and FWS agreed, on December 12, 2007, that the federal action for EPA's FIFRA registration actions will be defined as the "authorization for use or uses described in labeling of a pesticide product containing a particular pesticide ingredient." EPA must insure that all authorized uses, regardless of whether those uses have occurred historically, are not likely to jeopardize ESA-listed species or result in adverse modification or destruction of designated critical habitat. Thus, NMFS's analysis encompasses the impacts to ESA-listed species of all possible uses authorized by EPA.

Pesticide Labels. For this consultation, EPA's action encompasses all approved product labels containing carbaryl and methomyl, including their degradates, metabolites, and formulations, other ingredients within the formulations, adjuvants, and tank mixtures. The effects of these comprise the stressors of the action. These a.i.s combined are labeled for a variety of uses including applications to croplands and non-crop areas.

Active and Other ingredients. Carbaryl and methomyl are the a.i.s that kill or otherwise affect targeted organisms (listed on the label). Pesticide products that contain these a.i.s also contain other ingredients (referred to as "inerts" or "other" ingredients on the labels). Inert ingredients are ingredients which EPA defines as not "pesticidally" active. The specific identification of the compounds that make up the inert fraction of a pesticide is not required on the label and the inert ingredients in carbaryl and methomyl products have not been identified for NMFS to consider. However, this does not necessarily imply that inert ingredients are non-toxic, non-flammable, or otherwise non-reactive. EPA authorizes the use of chemical adjuvants to make pesticide products more efficacious. An adjuvant aides the operation or improves the effectiveness of a pesticide. Examples include wetting agents, spreaders, emulsifiers, dispersing agents, solvents, solubilizers, stickers, and surfactants. A surfactant is a substance that reduces surface tension of a system, allowing oil-based and water-based substances to mix more readily. A common group of non-ionic surfactants is the alkylphenol polyethoxylates (APEs), which may be used in pesticides or pesticide tank mixes, and also used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disruption effects in aquatic animals.

Formulations. Pesticide products come in a variety of solid and liquid formulations. Examples of formulation types include dusts, dry flowables, emulsifiable concentrates, granulars, solutions, soluble powders, ultra-low volume concentrates, water-soluble bags, powders, and baits. The formulation type can have implications for product efficacy and exposure to humans and other non-target organisms.

Tank Mix. A tank mix is a combination, by the user, of 2 or more pesticide formulations, as well as any adjuvants or surfactants added to the same tank prior to application. Typically, formulations are combined to reduce the number of spray operations or to obtain better pest control as compared to the individual products applied alone. The compatibility section of a label may advise on tank mixes known to be incompatible or provide specific mixing instructions for use with compatible mixes. Labels may also recommend specific tank mixes. Pursuant to FIFRA, EPA has the discretion to prohibit tank mixtures. Applicators are permitted to include any combination of pesticides in a tank mix as long as each pesticide in the mixture is permitted for use on the application site and the label does not explicitly prohibit the mix.

Pesticide Registration. In 2006, EPA commenced a new program called registration review to reevaluate all pesticides on a regular cycle. EPA is required to review each pesticide at least every 15 years to make sure that, as the ability to assess risks to human health and the environment evolves and as policies and practices change, all pesticide products in the marketplace can still be used safely. Registration review includes FIFRA Sections 3, 24(c), and 18. The label on a pesticide package or container is legally enforceable. The label provides information about how to handle and safely use the pesticide product and avoid harm to human health and the environment. Using a pesticide in a manner that is inconsistent with the use directions on the label is a violation of FIFRA and can result in enforcement actions to correct the violations. Pesticide registration is the process through which EPA evaluates product labels; EPA examines the ingredients of a pesticide; the site or crop on which it is to be used; the amount, frequency and timing of its use; and storage and disposal practices. Pesticide products (also referred to as “formulated products”) may include a.i.s and other ingredients, such as adjuvants and surfactants. The eligibility for continued registration may be contingent on label modifications to mitigate risk and can include phase-out and cancellation of uses and pesticide products. Registrants can submit applications for the registration of new products and new uses following registration review of an active ingredient. Several types of products are registered, including the pure (or nearly pure) active ingredient; this product is often referred to as technical grade active ingredient (TGAI), technical, or technical product. This is generally used in manufacturing and testing, and is not applied directly to crops or other use sites. Products that are applied to crops or other use sites (e.g., rights of way, landscaping), either on their own or in conjunction with other products or surfactants in tank mixes, are called typical end-use products (TEPs). Sometimes companies will also register the pesticide in a manufacturing formulation, intended for sale to another registrant who then includes it in a separately registered TEP. Manufacturing formulations are not intended for application directly to use sites. The EPA may also cancel product registrations. EPA typically allows for the use of canceled products, and products that do not reflect registration review label mitigation requirements, until those products have been exhausted. Labels that reflect current EPA mitigation requirements are referred to as "active labels." Products that do not reflect current label requirements are referred to as "existing stocks." EPA's action includes all authorizations for use of pesticide products, including use of existing stocks, and active labels containing carbaryl or methomyl for the duration of the action.

Duration of the Action. EPA is required to reassess registered pesticide a.i.s at least every 15 years. Given EPA's timeframe for pesticide registration reviews, NMFS's evaluation of the action is also 15 years, although NMFS considers any effects that continue beyond the end of the 15 years.

Monitoring and Reporting. The current Federal action does not include any specific provision for monitoring. However, Section 6(a)(2) of FIFRA requires pesticide product registrants to report adverse effects information, such as incident data involving fish and wildlife, to EPA.

5.2 Carbaryl

The following description of carbaryl registrations (the action) is organized into 3 subsections. The first 2 subsections describe the registrant agreements to change product labeling that occurred as part of: 1) the FIFRA registration review process; and, 2) the ESA section 7 consultation process. The final section is an overall summary of all carbaryl uses which includes those changes described in the 2 preceding subsections.

5.2.1 Changes to the Action through FIFRA Registration Review - Carbaryl Proposed Interim Decision (PID) Agreements

Product labels describe where pesticides can be applied (use sites), application methods, and application rates. During the registration review process and subsequent to initiation of this consultation, carbaryl registrants agreed to adopt a number of label changes as part of the proposed Federal action. Changes relevant to this assessment include the following measures to be specified on carbaryl product labels:

Aerial Applications:

- Do not release spray at a height greater than 10 ft above the ground or vegetative canopy, unless a greater application height is necessary for pilot safety.
- Applicators must select nozzle and pressure that deliver medium or coarser droplets in accordance with American Society of Agricultural & Biological Engineers Standard 641 (ASABE S641).
- During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes) must register between 3 and 15 miles per hour.
- Wind speed and direction must be measured on location using a windsock, an anemometer, or an aircraft smoke system.
- Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.

- If the windspeed is 10 miles per hour or less, applicators must use a minimum of ½ swath displacement upwind at the downwind edge of the field. When the windspeed is between 11-15 miles per hour, applicators must use a minimum of ¾ swath displacement upwind at the downwind edge of the field.
- When the windspeed is between 11-15 miles per hour, the boom length must be 65% or less of the wingspan for fixed wing aircraft, and 75% or less of the rotor diameter for helicopters. Otherwise, the boom length must be 75% or less of the wingspan for fixed-wing aircraft, and 90% or less of the rotor diameter for helicopters.
- Do not apply during temperature inversions.

Airblast Applications:

- Sprays must be directed into the canopy.
- During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes), must register between 3 and 10 miles per hour.
- Winds speed and direction must be measured on location using a windsock or anemometer.
- Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.
- User must turn off outward pointing nozzles at row ends and when spraying outer row.
- Do not apply during temperature inversions.

Ground Boom Applications

- During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes), must register between 3 and 15 miles per hour.
- Wind speed and direction must be measured on location using a windsock or anemometer.
- Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.
- Do not release spray at a height greater than 2 feet above the ground or crop canopy.
- Applicators must select nozzle and pressure that deliver medium or coarser droplets in accordance with American Society of Agricultural & Biological Engineers Standard 572 (ASABE S572).
- Do not apply during temperature inversions.

General Outdoor Application Statements for residential uses

- All outdoor spray applications must be limited to spot or crack-and-crevice treatments only, except for the following permitted uses:
  - Applications to soil, lawn, turf, or vegetation;
  - Perimeter band treatments of 6 ft wide or less from the base of a man-made structure to pervious surfaces (e.g., soil, mulch, or lawn);
  - Applications around potential exterior pest entry points into man-made structures such as doorways and windows, when limited to a band not to exceed one inch;

  - Applications to vertical surfaces directly above pervious surfaces, such as bare soil, lawn, turf, mulch or other vegetation, that do not drain into ditches, storm drains, gutters, or surface waters.

Spot Treatment Guidance Statement

- Spot treatments must not exceed 2 square feet in size (for example, 2 ft. by 1 ft. or 4 ft. by 0.5 ft).

Water Protection Statements

- Except for labeled uses to water bodies, including but not limited to, shrimp ponds, paddies, and cranberries, do not spray the product into fish pools, ponds, streams, or lakes. Do not apply directly to sewers or drains, or to any area like a gutter where drainage to sewers, storm drains, water bodies, or aquatic habitat can occur.
- Do not allow the product to enter any drain during or after application.
- Do not apply directly to impervious horizontal surfaces such as sidewalks, driveways, and patios except as a spot or crack-and-crevice treatment.
- Do not apply or irrigate to the point of run-off.

Rain-related statements (except for products that require watering-in)

Do not apply during rain. Do not apply when soil in the area to be treated is saturated (if there is standing water on the field or if water can be squeezed from soil) or if NOAA/National Weather Service predicts a total rainfall of 1 inch or greater over the 48 hours following the day of application, only considering a 48-hour period when, at any point during the 48-hour period, the precipitation potential is 50% or greater. Detailed National Weather Service forecasts for local weather conditions should be obtained on-line at: www.weather.gov or by contacting your local National Weather Service Forecasting Office.

Crack and crevice treatments

- Treat exposed surfaces to cover thoroughly but avoid excess run-off.
- To treat insects harbored in voids and cracks-and-crevices, applications must be made in such a manner to limit dripping and run-off on structural surfaces and plants.

Buffer Zones to Water Bodies

- Ground Application
  - Do not apply within 25 feet of aquatic habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams or ephemeral streams when water is present, wetlands or natural ponds, estuaries, and commercial fish farm ponds)
- Aerial Application
  - Do not apply within 150 feet of aquatic habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams or ephemeral streams when water is present, wetlands or natural ponds, estuaries, and commercial fish farm ponds)

Update Environmental Hazard Statements

- Outdoor, Terrestrial Uses

    - For terrestrial uses: Do not apply directly to water, or to areas where surface water is present or to intertidal areas below the mean high water mark. Do not contaminate water when disposing of equipment washwater or rinsate.
- Outdoor, Residential Consumer Products
    - Liquid concentrate: To protect the environment, do not allow pesticide to enter or run off into storm drains, drainage ditches, gutters or surface waters. Rinsing application equipment over the treated area will help avoid run off to water bodies or drainage systems.
    - Broadcast granular: To protect the environment, do not allow pesticide to enter or run off into storm drains, drainage ditches, gutters or surface waters. Sweeping any product that lands on a driveway, sidewalk, or street, back onto the treated area of the lawn or garden will help to prevent run off to water bodies or drainage systems.
    - Liquid Ready-to-Use: To protect the environment, do not allow pesticide to enter or run off into storm drains, drainage ditches, gutters or surface waters.
- Outdoor, Terrestrial Products Requiring Fish or Aquatic Invertebrate Statements:
    - Drift and runoff may be hazardous to aquatic organisms in water adjacent to treated areas.

5.2.2 Changes to the Action through ESA Section 7 Consultation - Conservation Measures for Listed Species

EPA and carbaryl applicants holding registrations for agricultural use labels have also agreed to modify the action by adopting conservation measures[6] to reduce risk to ESA-listed species. The conservation measures listed below focus on reducing exposure potential to ESA-listed species and their habitats by targeting risk reduction measures that effectively reduce drift and runoff; they include pesticide use restrictions that will be specified on FIFRA labels of all product labels containing carbaryl with agricultural uses. This will be accomplished by incorporating the following pesticide use restrictions into the “Directions for Use” section of the FIFRA labels or on EPA Endangered Species Protection Program Bulletins that serve as enforceable extensions to these labels (https://www.epa.gov/endangered-species/endangered-species-protection-bulletins).

1. **Texas Shrimp Ponds (SLN TX020007)**
   When applying carbaryl to Texas Shrimp Ponds (SLN TX020007), treated water must be at pH $\geq$ 8 for at least 4 days prior to discharge of water from the pond. Pond must be drained and dried before restocking. Do not apply more than 2 gallons of this product per acre per season.

2. **Treatment of Red Scale in California Citrus**

[6] Conservation measures are actions to benefit or promote the recovery of listed species that are included by the Federal agency as an integral part of the action. These actions will be taken by the Federal agency or applicant, and serve to minimize or compensate for, project effects on the species under review. These may include actions taken prior to the initiation of consultation, or actions which the Federal agency or applicant have committed to complete in a biological assessment or similar document.

Product labels authorizing the use of carbaryl in California citrus will include the specifications listed below in Table 5 and the following language in the Directions for Use section of the label: "Do not exceed 5.0 lbs a.i./A for a single application of carbaryl containing products for all combined uses except for California red scale (*Aonidiella aurantii*) outbreaks that cannot be otherwise managed in California. [Product name] may be applied at 12 lbs ai/A in California in rotation with other insecticides only if California red scale (*Aonidiella aurantii*) is the target." Additionally, product labels will specify that:

- This use is prohibited within any Pesticide Use Limitation Area for federally listed threatened or endangered species under NMFS jurisdiction.
- The maximum number of applications for this treatment is 1 application per year.
- The maximum annual application rate for this and all other carbaryl treatments in citrus combined is 20 lbs a.i./A.
- Other applications of carbaryl in citrus will not occur within 14 days of applications for California red scale.

3. General Changes

- Disallow application in Hawaii.
- Remove aerial application method from all uses except APHIS Grasshopper and Mormon Cricket Suppression Program.
- Ground boom applications made with the release height recommended by the manufacturer, but no more than 2 ft above the crop canopy.
- Ground boom applications made using nozzle and pressure that deliver medium or coarser droplets in accordance with American Society of Agricultural and Biological Engineers Standard 572 (ASABE S572).

Asparagus

- Maximum Annual Number of Applications: 4
- Minimum Reapplication Interval: 7 days

Sweet Corn

- Maximum Annual Amount: 8 lbs. a.i./A (active ingredient/acre)
- Maximum Annual Number of Applications: 4

Cucurbit Vegetables

- Maximum Annual Amount: 4 lbs. a.i./A
- Maximum Annual Number of Applications: 4

Fruiting Vegetables

- Maximum Annual Number of Applications: 4

Leafy Vegetables

- Maximum Annual Number of Applications: 4

Peanuts

- Maximum Annual Number of Applications: 4

Prickly Pear Cactus

- Maximum Annual Number of Applications: 4

Sweet Potatoes

- Maximum Annual Amount: 4 lbs a.i./A
- Maximum Annual Number of Applications: 4

Small Fruits and Berries

- Maximum Annual Amount: 8 lbs a.i./A
- Maximum Annual Number of Applications: 4

Citrus Fruits (all sites including CA and FL)

- Maximum Annual Number of Applications: 4

Olives

- Maximum Annual Amount: 10 lbs a.i./A
- Maximum Application Rate: 5 lbs a.i./A

Pome Fruits

- Maximum Annual Amount: 12 lbs. a.i./A
- Maximum Annual Number of Applications: 4

Ornamental Trees and Plants

- Maximum Annual Amount: 4 lbs a.i./A
- Maximum Annual Number of Applications: 4

Turfgrass

(Golf Turf, Sports Fields, Sod Farms, Domestic and Commercial Lawns, Cemeteries, Parks, Campsites, Recreational Areas)

- Maximum Annual Amount: 10 lbs a.i./A
- Maximum Application Rate: 5 lbs a.i./A

4. Geographically-specific restrictions to address potential loading from broadcast applications of carbaryl

As described in the Consultation History, subsequent to NMFS's publication of the public review period for the draft opinion (posted March 15, 2023), EPA and all carbaryl applicants agreed to modify the action further by adopting additional conservation measures to avoid the likelihood of jeopardizing the continued existence of ESA-listed species or the likelihood of resulting in the destruction or adverse modification of critical habitat (50 CFR §402.02). The conservation measures described below were designed to achieve the targeted reductions in pesticide loading into ESA-listed species' habitat, and reflect measures identified in the

RPA presented in NMFS's draft jeopardy opinion for carbaryl and/or alternative mitigation (where applicable) that would result in comparable reductions.

The conservation measures listed below focus on reducing exposure potential to ESA-listed species and their habitats by targeting risk reduction measures that effectively reduce drift and runoff; they include pesticide use restrictions that will be specified on FIFRA labels of all pesticide products containing carbaryl. This will be accomplished by incorporating the following pesticide use restrictions into the "Directions for Use" section of the FIFRA labels or on EPA Endangered Species Protection Program Bulletins that serve as enforceable extensions to these labels (https://www.epa.gov/endangered-species/endangered-species-protection-bulletins). Further details regarding implementation of use restrictions (including details on timing) are provided in EPA's commitment to modify the action and in the ITS (Section 21).

Reducing drift transport into ESA-listed species' habitat

For applications within Pacific Eulachon and Pacific Salmonid Pesticide Use Limitation Areas (PULAs), reduce loading of pesticides into ESA-listed species habitat from airblast applications at rates ≥ l lb carbaryl/A:

- Maintain a functional riparian system > 10 m wide alongside waterways adjacent to treatment area, OR
- Do not apply within 55 feet of aquatic habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams or ephemeral streams when water is present, wetlands or natural ponds, estuaries, and commercial fish farm ponds) when wind is blowing toward the aquatic habitat.

Reducing runoff transport into ESA-listed species habitat

Geographically-specific mitigation will be employed using a point system to address the risk posed by broadcast applications of carbaryl that occur within 300 m of ESA-listed species habitat (within PULAs) listed below. As was described in the public review draft RPA, we have developed a point system (Section 5.4) designed to arrive at sufficient risk reduction measures, which identifies the number of mitigation points needed to avoid jeopardy based on different application scenarios. This point system is further explained at the end of this chapter. The mitigation measures provided are similar to those described in the public review draft opinion's Reasonable and Prudent Alternative as providing sufficient reductions to avoid the likelihood of jeopardizing ESA-listed species or destroying or adversely modifying designated critical habitat. These conservation measures, when incorporated into the action, will similarly achieve the needed reduction in effects to avoid jeopardy or destruction/adverse modification.

Required risk reduction under the point system.

Pesticide labels will implement use restrictions that correspond to the application rate being applied in the field. Applications must comply with use directions on the product label that specify the maximum labeled rate and authorized application methods. This option accounts for the lower risk levels associated with applications that are made below the maximum labeled rate and the use of lower risk application methods. Risk reduction options to achieve the required level of risk reduction and comply with use restrictions are presented at the end of this chapter.

For Pacific Eulachon and Pacific Salmonids

- For application rates <1 lb carbaryl/A: Implement at least 1 runoff reduction measure
- For application rates of 1 to 2 lb carbaryl/A: Implement any combination of runoff reduction measures to achieve at least 50 points
- For application rates of > 2 lb carbaryl/A: Implement any combination of runoff reduction measures to achieve at least 70 points

For Sturgeon and Sawfish

- For application rates <1 lb carbaryl/A: No additional runoff reduction measures required
- For application rates of 1 to 2 lb carbaryl/A: Implement any combination of runoff reduction measures to achieve at least 30 points
- For application rates of > 2 lb carbaryl/A: Implement any combination of runoff reduction measures to achieve at least 50 points

5.2.3 Summary of All Carbaryl Uses

The summary of all carbaryl uses is presented in Appendix E and contains selections of each use that are presumed to represent the highest risk by considering maximum application rates and application methods. To summarize the current carbaryl action, NMFS integrated the mitigation agreed to as part of the PID (Section 5.2.1) as well as the conservation measures discussed above (Section 5.2.2). Based on these considerations, the current single maximum application rate for carbaryl is 12 lbs a.i./A for red scale treatments in California citrus. Carbaryl can be applied at approximately 8 lbs a.i./A to shrimp ponds in Texas and outdoor residential areas. However, the maximum single use rate is limited to ≤ 2 lbs a.i./A in most agricultural crops. The maximum annual use rate for carbaryl on agricultural use sites is 20 lbs a.i./A in citrus, and approximately 33 lbs a.i./A in residential ornamental plantings (up to 8.36 lbs a.i./A with 4 applications).

Carbaryl is an N-methylcarbamate insecticide registered for use on a wide variety of agricultural and non-agricultural uses. EPA’s carbaryl BE (EPA 2021b) indicates there are currently 5 active technical registrants of carbaryl with 61 active product registrations (60 Section 3s and 1 SLN) The formulated products are available for application as liquid sprays, or in bait, granular, and dust forms. Currently, there are at least 7 multi-active ingredient products registered that contain carbaryl. Other a.i.s co-formulated with carbaryl include: captan, malathion, copper sulfate, metaldehyde, and bifenthrin (Table 4). Additionally, carbaryl may be applied as part of a tank mix with other pesticides (*i.e.*, insecticides, miticides and fungicides). In general, carbaryl products can be mixed with other pesticide products and adjuvants unless specifically prohibited on the label(s).

Table 4. Multi-Active Ingredient Products Containing Carbaryl

REGISTRATION #	NAME	PERCENT ACTIVE INGREDIENT	ACTIVE INGREDIENT
4-122	BONIDE A COMPLETE FRUIT TREE SPRAY	11.76	Captan
		6.00	Malathion
		0.30	Carbaryl
4-458	COPPER DRAGON TOMATO & VEGETABLE DUST	7	Basic copper sulfate
		2	Carbaryl
4-474	BONIDE VEGETABLE-FLORAL DUST	13.72	Basic copper sulfate
		1.25	Carbaryl
8119-5	CORRY'S SLUG, SNAIL & INSECT KILLER	5	Carbaryl
		2	Metaldehyde
9198-234	THE ANDERSONS BICARB LAWN INSECT KILLER GRANULES	0.058	Bifenthrin
		2.3	Carbaryl
9198-235	THE ANDERSONS BICARB INSECTICIDE + FERTILIZER	0.058	Bifenthrin
		2.3	Carbaryl
71096-18	GET-A-BUG SNAIL, SLUG & INSECT KILLER	5	Carbaryl
		2	Metaldehyde

5.3 Methomyl

The following description of methomyl registrations (the action) is organized into 3 subsections. The first 2 subsections describe the registrant agreements to change product labeling that occurred as part of: 1) the FIFRA registration review process; and, 2) the ESA Section 7 consultation process. The final section is an overall summary of all methomyl uses which includes those changes described in the 2 preceding subsections.

5.3.1 Changes to the Action through FIFRA Registration Review - Methomyl PID Agreements

Product labels describe where pesticides can be applied (use sites), application methods, and application rates. During the registration review process and subsequent to initiation of this consultation, methomyl registrants agreed to adopt a number of label changes as part of the proposed Federal Action. Additionally, EPA recommended further revisions to spray drift mitigation statements on product labels for consistency with EPA's FIFRA Interim Ecological Mitigation. Changes relevant to this assessment include the following measures to be specified on methomyl product labels:

Spray Drift Mitigation Language

- Aerial Applications:
    - Do not apply within 50 ft of residential areas, including schools, homes, playgrounds, recreational areas, athletic fields, residential lawns, gardens, and other areas where children may be present.
    - Do not release spray at a height greater than 10 ft above the ground or vegetative canopy, unless a greater application height is necessary for pilot safety.
    - Applicators must select nozzle and pressure that deliver medium or coarser droplets in accordance with American Society of Agricultural & Biological Engineers Standard 641 (ASABE S641).
    - During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes) must register between 3 and 10 miles per hour.
    - Wind speed and direction must be measured on location using a windsock, an anemometer, or an aircraft smoke system.
    - Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.
    - Applicators must use a minimum of ½ swath displacement upwind at the downwind edge of the field.
    - The boom length must be 75% or less of the wingspan for fixed-wing aircraft and 90% or less of the rotor diameter for helicopters.
    - Do not apply during temperature inversions.
- Ground Boom Applications:
    - During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes), must register between 3 and 10 miles per hour.
    - Wind speed and direction must be measured on location using a windsock or anemometer.
    - Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.
    - Do not release spray at a height greater than 3 feet above the ground or crop canopy.
    - Applicators must select nozzle and pressure that deliver medium or coarser droplets in accordance with American Society of Agricultural & Biological Engineers Standard 572 (ASABE S572).

    - Do not apply during temperature inversions.
- Airblast Applications:
    - Do not apply within 10 ft of residential areas, including schools, homes, playgrounds, recreational areas, athletic fields, residential lawns, gardens, and other areas where children may be present.
    - Sprays must be directed into the canopy.
    - During application, the Sustained Wind Speed, as defined by the National Weather Service (standard averaging period of 2 minutes), must register between 3 and 10 miles per hour.
    - Winds speed and direction must be measured on location using a windsock or anemometer.
    - Wind speed must be measured at the release height or higher, in an area free from obstructions such as trees, buildings, and farm equipment.
    - User must turn off outward pointing nozzles at row ends and when spraying outer row.
    - Do not apply during temperature inversions.

Runoff Mitigation Language

- Do not apply during rain. Do not apply when soil in the area to be treated is saturated (if there is standing water on the field or if water can be squeezed from soil) or if NOAA/National Weather Service predicts a total rainfall of 1 inch or greater over the 48 hours following the day of application, only considering a 48-hour period when, at any point during the 48-hour period, the precipitation potential is 50% or greater. Detailed National Weather Service forecasts for local weather conditions should be obtained on-line at: www.weather.gov or by contacting your local National Weather Service Forecasting Office.

Application Rate Restriction

- Maximum nationwide annual application rate to be limited to 13 lb a.i./A/year.

5.3.2 Changes to the Action through ESA Section 7 Consultation - Conservation Measures for Listed Species

EPA and methomyl applicants holding registrations for agricultural use labels have also agreed to modify the action by adopting conservation measures to reduce risk to ESA-listed species (50 CFR §402.02). The conservation measures described below were designed to achieve the targeted reductions in pesticide loading into ESA-listed species habitat, and reflect measures identified in the RPA presented in NMFS's draft jeopardy opinion for methomyl and/or alternative mitigation (where applicable) that would result in comparable reductions.

The conservation measures listed below focus on reducing exposure potential to ESA-listed species and their habitats by targeting risk reduction measures that effectively reduce drift and runoff; they include pesticide use restrictions that will be specified on FIFRA labels of all pesticide products containing methomyl. This will be accomplished by incorporating the following pesticide use restrictions into the "Directions for Use" section of the FIFRA labels or on EPA Endangered Species Protection Program Bulletins that serve as enforceable extensions to these labels (https://www.epa.gov/endangered-species/endangered-species-protection-bulletins). Further details regarding implementation of use restrictions (including details on timing) are provided in EPA's commitment to modify the action and in the ITS chapter.

Reducing drift transport into ESA-listed species habitat
For applications within Pacific Salmonid PULAs, reduce loading of pesticides into ESA-listed species habitat from airblast application rates ≥ 0.5 lb methomyl/A:

- Maintain a functional riparian system > 10 m wide alongside waterways adjacent to treatment area, OR
- Do not apply within 55 feet of aquatic habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams or ephemeral streams when water is present, wetlands or natural ponds, estuaries, and commercial fish farm ponds) when wind is blowing toward the aquatic habitat.

Reducing runoff transport into ESA-listed species habitat
Geographically-specific mitigation will be employed using a point system to address the risk posed by broadcast applications of methomyl that occur within 300 m of ESA-listed species habitat (within specified PULAs) listed below. As described in the public review draft RPA, we have developed a point system designed to arrive at sufficient risk reduction measures, which identifies the number of mitigation points needed to avoid jeopardy based on different application scenarios. This point system is further explained at the end of this chapter (Section 5.4). The mitigation measures provided are similar to those described in the public review draft opinion's RPA as providing sufficient reductions to avoid the likelihood of jeopardizing ESA-listed species or destroying or adversely modifying designated critical habitat. These conservation measures, when incorporated into the action, will similarly achieve the needed reduction in effects to avoid jeopardy or destruction/adverse modification.

Required risk reduction under the point system.

Pesticide labels will implement use restrictions that correspond to the application rate being applied in the field. Applications must comply with use directions on the product label that specify the maximum labeled rate and authorized application methods. This option accounts for the lower risk levels associated with applications below the maximum labeled rate and the use of lower risk application methods. Risk reduction options to achieve the required level of risk reduction and comply with use restrictions are presented at the end of this chapter.

For Pacific Salmonids

- For application rates <0.5 lb methomyl/A: Implement any combination of runoff reduction measures to achieve at least 50 points
- For application rates of ≥ 0.5 lb methomyl/A:
    - Implement any combination of runoff reduction measures to achieve at least 60 points
      AND
    - For uses that currently allow > 7 applications per year, limit the number of applications to 7 with a minimum reapplication interval of 4 days.

For Sturgeon

- For application rates <0.5 lb methomyl/A: No additional runoff reduction measures required
- For application rates ≥0.5 lb methomyl/A: Implement at least 1 runoff reduction measure

5.3.3 Summary of All Methomyl Uses

The summary of all methomyl uses is presented in Appendix E and contains selections of each use that is presumed to represent the highest risk by considering maximum application rates and application methods. In general, current single maximum methomyl application rates do not exceed 0.9 lb a.i./A nationwide for flowable formulations; however, a single application rate of 1.5 lb a.i./A is currently permitted for corn and sweet corn use patterns for granular formulation. The maximum annual rate of methomyl that may be applied to certain crop sites is 13 lb a.i./A (e.g. sweet corn). Fly bait labels recommend frequent reapplication (e.g., every 2-5 days; registrations 2724-274 and 7319-6) and do not specify a maximum annual rate.

Methomyl is a N-methylcarbamate insecticide used on a wide variety of agricultural uses including field crops, vegetable crops, and orchard crops. Methomyl is also registered as a fly-bait. There are currently 3 active technical registrants of methomyl with 34 active product registrations (16 Section 3s and 18 SLN), which include formulated products and technical grade methomyl (EPA 2021d). Methomyl can be applied in a liquid, granular (corn only), scatter bait, bait station, or as a brush-on paste. Aerial and ground application methods (including broadcast, soil incorporation, orchard airblast, and chemigation) are allowed. Pesticide product labels for granular products contain a 25-foot (ground) buffer zone adjacent to waterbodies. Additionally, labels for foliar (flowable) applications also contain a 25 or 100-foot buffer zone for ground and aerial applications, respectively. Currently, there are 2 multi-active-ingredient products registered that contain methomyl (Table 5). These are fly bait products co-formulated with cis-9-Tricosene. Methomyl may be applied as part of a tank mix with other pesticides (i.e., insecticides, miticides and fungicides) or adjuvants. In general, a.i.s can be mixed with other products unless specifically prohibited on the label(s).

Table 5. Multi-Active Ingredient Products Containing Methomyl

REGISTRATION #	NAME	PERCENT ACTIVE INGREDIENT	ACTIVE INGREDIENT
2724-274	GOLDEN MALRIN RF-128 FLY KILLER	1	methomyl
		0.049	cis-9-Tricosene
7319-6	LURECTRON SCATTERBAIT	1	methomyl
		0.026	cis-9-Tricosene

5.4 Mitigation Point System Overview

NMFS has proposed a point system to arrive at sufficient runoff reduction measures to reduce risk to non-jeopardy levels. The approach achieves reductions in pesticide loading while allowing maximum flexibility for the grower/applicator. It also rewards landowners who are already implementing reduction measures such as Best Management Practices (BMPs) that reduce loading and improve habitat for ESA-listed species.

We have identified the number of mitigation points needed to avoid jeopardy and adverse modification based on different application scenarios (described above in Sections 5.2.2 and 5.3.2). We have also identified risk reduction options that can be used to achieve the needed mitigation. Each risk reduction measure on the list has a point value based on its efficacy at reducing loading from runoff/drainage. In this approach, applicators look up the point value required based on their location and application parameters. Then, applicators choose which risk reduction measures to implement as long as the required number of points are achieved for the exposure pathway (in this case, runoff/drainage).

Table 6 includes mitigation measures currently available to pesticide applicators, growers, and landowners. EPA will make this list of mitigation options available. This list of mitigation measures may be expanded or modified in the future if there is mutual agreement between EPA and NMFS to do so. NMFS will seek to expand the list of available mitigation options and supports EPA and the applicants providing measures for consideration as well. Any proposals for additional mitigation measures must include written documentation to NMFS describing the mitigation measure, the anticipated efficacy (i.e., load reduction in aquatic habitats), and the proposed number of mitigation points to be awarded for measure implementation. Once agreement is reached, EPA would make the additional mitigation options available to pesticide applicators, growers, and landowners.

The mitigation measures in Table 6 are listed by name only. As with EPA, we have received a number of comments regarding the relationship of the ESA-based mitigation options to the NRCS conservation practice standards. As the agencies work to develop jointly agreed-upon language, NMFS's intent is to be inclusive of conservation practices used in NRCS programs (Table 6 and Table 7). Runoff measures do not necessarily need to be employed on the site of application. The intent is to grant credit for measures that will be employed in a manner that will prevent runoff originating from the application site from entering species aquatic habitats. EPA is currently in discussions with USDA regarding the use of the NRCS standards as part of FIFRA pesticide labeling. Until those discussions are resolved, EPA cannot commit to citing the NRCS standards as directly equivalent to the mitigation options. In the interim, NMFS will consider the descriptions that EPA has developed for many of these mitigation options (see Appendix F). Prior to implementation of this conference and biological opinion, and in accordance with the implementation timelines specified in the ITS, EPA must obtain agreement from NMFS on the descriptions that will define the specifications required for the mitigation measures listed in Table 7.

Table 6. NRCS conservation practice standards relevant to carbaryl and methomyl conservation measures

Mitigation option from conservation measure section	Relevant NRCS Conservation Practice Standard (code #)
Vegetated filter strip	393[a]
Filter strip	393[a]
Contour buffer strips	332[b]
Strip cropping	585
Contour farming	330
Alley cropping	311[c]
Terrace	600[d]
Mulching with natural materials	484
No-till or reduced tillage	392
Grassed waterways, vegetated ditches	412[d]
Field border	386[e]
Riparian forest buffers	391
Retention pond	378

Mitigation option from conservation measure section	Relevant NRCS Conservation Practice Standard (code #)
Water and sediment control basin	638
Constructed wetland	656
Irrigation and drainage tailwater recovery	447
Hedgerow planting	422
Cover cropping	340
Surface roughening	609

a When "Additional Criteria to Reduce Dissolved Contaminants, Suspended Solids and Associated Contaminants in Runoff and Excessive Sediment in Surface Waters" are implemented.
b When "Additional Criteria to Reduce Water Quality Degradation from the Transport of Nutrients Downslope" are implemented.
c When "Additional Criteria to Reduce Surface Water Runoff and Erosion" are implemented.
d When the outlet does not discharge directly to species habitat.
e When "Additional Criteria to reduce Sedimentation Offsite and Protect Water Quality and Excess Nutrients in Surface and Ground Waters" are implemented

How the point system works:

- **Step 1.** Determine whether any mitigation is needed. Is pesticide application to be made within an ESA-listed species PULA (https://www.epa.gov/endangered-species/bulletins-live-two-view-bulletins)? If yes, go to step 2.
- **Step 2.** Determine the number of mitigation points needed for your pesticide application (specified in Sections 5.2.2 and 5.3.2) based on application method and rate applied.
- **Step 3.** Choose mitigation option(s) that provide an equal or greater value of points required. Mitigation options can be added together, based on their point values. Applicable mitigation options (risk reduction measures) are listed in Table 7 below.

Table 7. Risk reduction measures for broadcast applications

Runoff/drainage mitigation option[1]	Points[2]
Vegetated filter strip	
3 meter	15
5 meter	20
10 meter	45
20 meter	60
Inter-row	30
Filter Strip	30
Contour buffer strips	30
Strip cropping	30
Contour farming	20
Alley cropping[3]	20
Terrace	15
Mulching with natural materials	30
No-till or reduced tillage	30

Runoff/drainage mitigation option[1]	**Points[2]**
Grassed Waterways	30
Vegetated ditches	30
Field border	30
Functional riparian system alongside water ways > 10 meters wide	80
Riparian forest buffer 5-10 meters wide	15
Retention pond[4]	55
Water and sediment control basin	55
Constructed wetland	55
Irrigation and drainage tailwater recovery	55
Small Area Applications <0.1A[5]	80
Hedgerow planting	15
Cover cropping	5
Application area has a slope of less than 2%	5
Surface roughening	5
Sediment basin	5
Runoff reduction technology, pesticide stewardship program, etc.	TBD[6]

1 Runoff/drainage measures. Definitions for each of the mitigation options will be finalized in accordance with implementation timelines of this conference and biological opinion.
2 Point values correspond to the effectiveness of each option at reducing pesticide transport via runoff. Efficacy information came from a variety of sources (e.g., Alix et al. (2015); EPA 2023).
3 Alley crop points apply for applications to the tree crops (or vine, etc.) but do not apply for applications to the alley crop itself. The efficacy of this option assumes that the alley crop is functioning as a filter strip for pesticides applied to the trees.
4 Retention ponds include those that may be employed for culturing cranberries or rice.
5 Small area applications are those made to distinct targeted areas, typically using handheld wand and backpack sprayer. Estimated reductions assumed a median field size of 0.278 km2 (Yan and Roy 2016).
6 To be determined. This list of mitigation measures may be expanded or modified in the future if there is mutual agreement between EPA and NMFS to do so.

6 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02). Given EPA's nationwide authorization of these pesticides and anticipated chemical transport following application, the action area includes the entire U.S. and its territories, including all waters in which EPA's action may cause effects to ESA-listed species or designated critical habitat. The action area includes all ESA-listed species and designated critical habitat under NMFS's jurisdiction that occur within the United States and its territorial waters.

7 Species and Critical Habitat Not Likely to be Adversely Affected

Section Contents

7.1 Introduction

Effects of the action are all consequences to listed species or critical habitat that are caused by the action, including the consequences of other activities that are caused by the action. A consequence is caused by the action if it would not occur but for the action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action. A 'No Effect' determination would be the appropriate conclusion when the action agency determines its action will not affect an ESA-listed species or designated critical habitat. No effect determinations do not require ESA section 7 consultation.

NMFS uses 2 criteria to identify the ESA-listed species and critical habitats that are not likely to be adversely affected by the action, as well as the effects of activities that are consequences of the Federal agency's action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the proposed activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that co-occur with a stressor of the action but are not likely to respond to the stressor are also not likely to be adversely affected by the action. We applied these criteria to the ESA-species and designated critical habitats and we summarize our results below.

The probability of an effect on a species or designated critical habitat is a function of exposure intensity and susceptibility of a species to a stressor's effects (i.e., probability of response). An action warrants a NLAA finding when its effects are wholly beneficial, insignificant or discountable. Beneficial effects have an immediate positive effect without any adverse effects to the species or habitat.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when species or critical habitat will be exposed to stressors, but the response will not be detectable outside of normal behaviors. Insignificant is the appropriate effect conclusion when effects are plausible, but will not never rise to the level of take (e.g., impacts to any individual's fitness).

Discountable effects are those that will occur while an ESA-listed species is in the action area, but because of the intensity, magnitude, frequency, duration, or timing of the stressor, the actual exposure is extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did impact an ESA-listed species), but it is very unlikely to occur.

'Likely to adversely affect' is the appropriate conclusion when any effects of the action are not: discountable, insignificant, or wholly beneficial (not NLAA); therefore, adverse effects are possible to ESA-listed species or designated critical habitat as a result of the action. If incidental take is anticipated (e.g., individuals may be harmed or harassed) as a result of the action or the conservation value of a physical and biological feature may be diminished, an LAA determination should be made.

Section 7(a)(4) of the ESA provides for conference opinions that evaluate impacts to species proposed for listing and habitat proposed for designation as critical habitat. In this opinion, NMFS provides advisory recommendations to minimize or avoid adverse effects to 2 proposed species (queen conch [*Alger gigas*] and sunflower sea star [*Pycnopodia helianthoides*]) and the proposed critical habitat for 5 coral species: *Acropora globiceps, Acropora retusa, Acropora speciosa, Euphyllia pardivisa,* and *Isopora crateriformis*. NMFS also considered impacts on the proposed critical habitat for Rices's whale (*Balaenoptera edemi*) and green sea turtle (*Chelonia mydas* [Central North Pacific, Central South Pacific, Central West Pacific, East Pacific, North Atlantic, and South Atlantic DPSs]).

EPA made NLAA and LAA determinations in their 2021 BEs for carbaryl and methomyl. Below, we describe the rationale for the NLAA determinations. NMFS has identified a number of species and critical habitats for which LAA is the appropriate effects determination, as opposed to the NLAA determination made by EPA in the BE. These non-concurrence determinations are identified below. The rationale for making an LAA determination for these species is provided in the Species and Critical Habitat Likely to be Adversely Affected section (Section 8).

7.2 Cetaceans

We concur with the NLAA determinations EPA made for the cetaceans and their designated critical habitats outlined in Table 8. Adverse effects from the action to all but one listed cetacean (as well as their critical habitat) are anticipated to be either insignificant or discountable, as described below.

Effects are discountable in cases where species' habitat use is such that we expect it will be extremely unlikely that pesticides will occur at concentrations that could affect either the species or their designated critical habitat. This is the case for listed cetaceans, and their designated critical habitats, which are found primarily in offshore or circumpolar (*i.e.,* found at high latitudes around the earth's Polar Regions) locations due to the effect of dilution in these large marine environments and, because these areas are expected to be far from any potential application sites, limiting the potential for exposure. The following species received NLAA determinations because effects are discountable: sei whale (*Balaenoptera borealis*), blue whale (*Balaenoptera musculus*), fin whale (*Balaenoptera physalus*), all listed humpback whale DPSs (*Megaptera novaeangliae*), sperm whale (*Physeter macrocephalus*), and bowhead whale (*Balaena mysticetus*). Additionally, the designated critical habitat for the Central America, Western North Pacific, and Mexico humpback whale DPSs received NLAA determinations because effects were discountable.

Effects are insignificant where pesticide exposure is possible but the species or designated critical habitat response to that exposure is undetectable, not measurable, or so minor that they cannot be meaningfully evaluated and never rises to the level of take. This is the case for listed cetaceans that are known to occur in shallow coastal habitats, or designated critical habitat located in these areas. Although exposure is possible, we do not anticipate adverse effects to species, primarily because these pesticides are readily metabolized such that the risk of bioaccumulation and/or biomagnification (via dietary exposure) is low, this is especially relevant for large animals (like whales) for which the body burden required to result in an adverse response in very unlikely to be achieved. For the 5 cetaceans with designated or proposed critical habitat (Table 8), prey availability is the only PBF that may be affected (Appendix B). While impacts to aquatic invertebrate prey could occur, we expect effects to this PBF to be insignificant as they would be localized to areas in proximity of use sites, limited in spatial extent and temporal persistence, and will not meaningfully reduce the overall prey availability to the species. On July 24, 2023, NOAA Fisheries proposed to designate critical habitat in Gulf of Mexico waters between 100 and 400 m depths that are essential to the conservation of the Rice's whale individual growth, reproduction, and development; social behavior; and overall population growth (Proposed Rule 88 FR 47453). The proposed critical habitat designation has been included in this conference and biological opinion for carbaryl and methomyl. The following species (and associated designated or proposed critical habitats) received NLAA determinations because effects are insignificant: Rices's whale (*Balaenoptera edemi*), Beluga whale Cook Inlet DPS (*Delphinapterus leucas*), North Atlantic Right whale (*Eubalaena glacialis*), North Pacific Right whale (*Eubalaena japonica*), and False killer whale Main Hawaiian Islands Insular DPS (*Pseudorca crassidens*).

Table 8. Cetaceans Summary of the NLAA Determinations for Carbaryl and Methomyl

Scientific Name	Common Name	Listing Status[1]	Species Call	Critical Habitat Call[2]
Balaena mysticetus	Bowhead whale	E	NLAA	X
Balaenoptera borealis	Sei whale	E	NLAA	X
Balaenoptera ricei	Rice's whale	E	NLAA	NLAA (P)
Balaenoptera musculus	Blue whale	E	NLAA	X
Balaenoptera physalus	Fin whale	E	NLAA	X
Delphinapterus leucas	Beluga whale (Cook Inlet DPS)	E	NLAA	NLAA
Eubalaena glacialis	North Atlantic Right Whale	E	NLAA	NLAA
Eubalaena japonica	North Pacific Right Whale	E	NLAA	NLAA
Megaptera novaeangliae	Humpback whale	E	NLAA	NLAA
Physeter microcephalus	Sperm whale	E	NLAA	X
Pseudorca crassidens	False killer whale (Main Hawaiian Islands Insular DPS)	E	NLAA	X

[1]*ESA Status: E = Endangered*
[2]*Critical Habitat: P=Proposed, X = Not Applicable (Critical Habitat has not been designated)*

7.3 Cartilaginous Fish

We concur with the NLAA determinations EPA made for the listed sharks in Table 9. Adverse effects from the action to all listed sharks are anticipated to be either insignificant or discountable, as described below.

Effects are discountable for listed sharks, which are found primarily in offshore locations due to the effect of dilution in these large marine environments, and because these areas are expected to range far from any potential application sites, limiting the potential for exposure. The following species (and associated designated critical habitats) received NLAA determinations because effects are discountable: oceanic whitetip shark (*Carcharhinus longimanus*).

Effects are insignificant for listed sharks that are known to occur in shallow coastal habitats. Although exposure is possible, we do not anticipate adverse effects to species, primarily because these pesticides are readily metabolized such that the risk of bioaccumulation and/or biomagnification (via dietary exposure) is low. In regard to prey availability, we expect impacts to be insignificant as affects to prey will be limited spatially, temporally, and to sensitive species (e.g., potentially some invertebrates). Additionally, given the varied diet of these species we do not expect these impacts to meaningfully reduce the overall prey availability. The following species (and associated designated critical habitats) received NLAA determinations because effects are insignificant: scalloped hammerhead shark (*Sphyrna lewini*) Indo-West Pacific DPS, Central and Southwest Atlantic DPS, and Eastern Pacific DPS.

Table 9. Sharks Summary of the NLAA Determinations for Carbaryl and Methomyl

Scientific Name	Common Name	Listing Status[1]	Species Call	Critical Habitat Call[2]
Sphyrna lewini	Scalloped hammerhead shark (Eastern Pacific DPS)	E	NLAA	X
Sphyrna lewini	Scalloped hammerhead shark (Central and Southwest Atlantic DPS)	E	NLAA	X
Sphyrna lewini	Scalloped hammerhead shark (Indo-West Pacific DPS)	T	NLAA	X
Carcharhinus longimanus	Oceanic whitetip shark	T	NLAA	X

[1]*ESA Status: E = Endangered, T = Threatened*
[2]*Critical Habitat: X = Not Applicable, Critical Habitat has not been designated*

7.4 Pinnipeds

We concur with the NLAA determination EPA made for the pinnipeds and designated critical habitats in Table 10. Adverse effects from the action for these listed pinniped species (as well as their designated critical habitat) are anticipated to be discountable or insignificant, as described below.

Effects are discountable in cases for both the bearded seal Beringia DPS and the ringed seal Arctic subspecies, and their associated designated critical habitats. These 2 seal species are strongly associated with ice-covered waters of the Arctic Ocean Basin and southward into adjacent seas. Bearded seals of the Beringia DPS inhabit seasonally ice-covered waters of the Bering, Chukchi, Beaufort, and East Siberian Seas. Their effective habitat is generally restricted to areas where seasonal ice occurs over relatively shallow waters. Designated critical habitat for the bearded seal Beringia DPS comprises an area of marine habitat in the Bering, Chukchi, and Beaufort Seas. Arctic ringed are found in the Arctic Basin and adjacent seas, including the Bering and Labrador Sea. Arctic ringed seals are highly associated with sea ice. Designated critical habitat for the Arctic subspecies of ringed seal (*Phoca hispida hispida*) comprises an area of marine habitat in the Bering, Chukchi, and Beaufort Seas. We anticipate exposures in those environments to be extremely unlikely. The exposure potential for marine-based prey is similarly unlikely such that we anticipate impacts to primary prey resources (a proposed PBF for both species) to be discountable as well.

Effects are insignificant for listed pinnipeds that utilize habitats near use sites approved for carbaryl and methomyl applications. Although exposure is possible, we do not anticipate adverse effects to these species, primarily because these pesticides are readily metabolized such that the risk of bioaccumulation and/or biomagnification (via dietary exposure) in their aquatic habitats is low. Dermal and inhalation exposure are also possible due aerial transport of pesticide droplets and vapors to terrestrial habitats utilized by these pinnipeds. However, we do not expect carbaryl and methomyl will be transported to these species habitats in concentrations sufficient to cause adverse effects from these routes of exposure. For species with designated critical habitat (Table 10), prey availability is the only PBF that may be affected (Appendix B). While impacts to some aquatic invertebrate prey could occur, we expect effects to this PBF to be insignificant as prey impacts would be localized to areas in proximity of use sites, and limited in spatial extent and temporal persistence. Given the varied pinniped diet, these impacts will not meaningfully reduce the overall prey availability to the species. The following species (and associated designated critical habitats) received NLAA determinations because effects are insignificant: Hawaiian monk seal (*Monachus schauinslandi*), Guadalupe fur seal (*Arctocephalus townsendi*) and Steller sea lion (*Eumetopias jubatus*).

Table 10. Pinnipeds Summary of the NLAA Determinations for Carbaryl and Methomyl

Scientific Name	**Common Name**	**Listing Status[1]**	**Species Call**	**Critical Habitat Call[2]**
Erignathus barbatus nauticus	Pacific bearded seal (Beringia DPS)	T	NLAA	NLAA
Phoca (pusa) hispida	Ringed seal (Arctic subspecies)	T	NLAA	NLAA
Eumetopias jubatus	Steller sea lion (Western DPS)	T	NLAA	NLAA
Arctocephalus townsendi	Guadalupe fur seal	T	NLAA	X
Monachus schauinslandi	Hawaiian monk seal	E	NLAA	NLAA

[1]*ESA Status: E = Endangered, T = Threatened*
[2]*Critical Habitat: X = Not Applicable, Critical Habitat has not been designated*

7.5 Sea Turtles

We concur with the NLAA determinations EPA made for the sea turtles and their designated critical habitats outlined in Table 11. Adverse effects from the action for these listed sea turtle species (as well as their proposed designated critical habitat) are anticipated to be discountable or insignificant, as described below.

All of the listed sea turtles use 3 marine habitat zones:

1. supralittoral terrestrial zone - beaches or occasionally estuarine shoreline habitats where egg laying, embryonic development, and hatching occur;
2. open ocean/convergence zones – deep water habitats (depths >200 meters) utilized for ocean juvenile rearing stage and foraging habitat for adults;
3. neritic zone - coastal areas for benthic feeding and migration (0-200 meters depth), the nearshore marine environment used by post-hatchlings moving from beach to convergence zones, by adults and subadults to forage, and as a migration corridor and breeding habitat for adults.

The potential for impacts to sea turtles while occupying beaches and other terrestrial habitats was not assessed by NMFS. NMFS and the USFWS have joint administrative responsibilities of the ESA as it pertains to ESA-listed sea turtles. Under a 1973 Memorandum of understanding, all consultations under section 7(a)(2) of the ESA for activities affecting sea turtles and their habitat in the terrestrial environment shall be the responsibility of USFWS. All consultations under section 7(a)(2) of the ESA for activities affecting sea turtles and their habitat in the marine environment shall be the responsibility of NMFS. USFWS is concurrently consulting with EPA on the effects of carbaryl and methomyl on sea turtles when they are in the terrestrial environment.

The impacts to sea turtles and designated critical habitats in open ocean and convergence zones are discountable. Pesticide transport modeling and consideration of habitat depth suggest that it will be extremely unlikely that pesticides will occur at concentrations that could affect either the species or their designated critical habitat in these deep-water habitats.

Effects are insignificant for listed sea turtles and designated critical habitats in the coastal neritic zone. Sea turtles feed in and migrate through coastal waters in relatively close proximity to pesticide use sites to varying extents. Although dietary exposure is possible, we do not anticipate adverse effects to species, primarily because these pesticides are readily metabolized such that the risk of bioaccumulation and/or biomagnification is low. For species with designated critical habitat (Table 11), prey availability is the only PBF that may be affected (Appendix B). Sea turtles diet and use of the coastal habitats varies among the ESA-listed species. In the coastal neritic zone, green turtles feed primarily on algae and grasses in benthic habitats, hawksbill turtles feed on a variety of aquatic invertebrates and fish associated with coral reef environments and rocky shorelines. Kemp's ridley turtles forage for crabs and other animal prey in muddy or sandy bottom substrates, leatherback turtles feed primarily on soft bodied prey (jelly fish and salps), loggerhead turtles forage on bottom dwelling invertebrates (e.g., mollusks and crabs), and olive ridley turtles have an omnivorous diet that includes algae and benthic invertebrates. While it is possible that some invertebrate prey may be impacted in close proximity to application sites in nearshore habitats within this zone, we do not anticipate that effects will be either widespread or sustained in the marine environment considering the chemical fate characteristics of carbaryl and methomyl and dissipation from mixing associated with tidal exchange and ocean currents. We do not anticipate any meaningful reductions in the overall prey availability to the species.

On July 17, 2023, NOAA Fisheries proposed to designate marine critical habitat from mean high water to 20 m depth and Sargassum habitat in the Gulf of Mexico and Atlantic Ocean to protect access to green sea turtle nesting beaches, migratory corridors, and important feeding and resting areas (Proposed Rule 88 FR 46572). The proposed critical habitat designations for North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central South Pacific, and Central West Pacific DPSs have been included in this conference and biological opinion for carbaryl and methomyl.

The following species received NLAA determinations because effects are insignificant: green sea turtle (*Chelonia mydas* [Central North Pacific, Central South Pacific, Central West Pacific, East Pacific, North Atlantic, and South Atlantic DPSs]), hawksbill sea turtle (*Eretmochelys imbricata)*, Kemp's ridley sea turtle (*Lepidochelys kempii)*, leatherback sea turtle (*Dermochelys coriacea)*, loggerhead sea turtle (*Caretta caretta* [North Pacific Ocean and Northwest Atlantic Ocean DPSs]), and olive Ridley sea turtle (*Lepidochelys olivacea)*. The following designated critical habitats also received NLAA determinations because effects were insignificant: hawksbill sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic Ocean DPS). Finally, the following proposed critical habitat also received NLAA determinations because effects were insignificant: green sea turtle (North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central South Pacific and Central West Pacific DPSs).

Table 11. Sea Turtle Summary of the NLAA Determinations for Carbaryl and Methomyl

Scientific Name	**Common Name**	**Listing Status[1]**	**Species Call**	**Critical Habitat Call[2]**
Chelonia mydas	Green sea turtle, Central North Pacific DPS	T	NLAA	NLAA (P)
Chelonia mydas	Green sea turtle, Central South Pacific DPS	E	NLAA	NLAA (P)
Chelonia mydas	Green sea turtle, Central West Pacific DPS	E	NLAA	NLAA (P)
Chelonia mydas	Green sea turtle, East Pacific DPS	T	NLAA	NLAA (P)

Scientific Name	**Common Name**	**Listing Status[1]**	**Species Call**	**Critical Habitat Call[2]**
Chelonia mydas	Green sea turtle, North Atlantic DPS	T	NLAA	NLAA (P)
Chelonia mydas	Green sea turtle, South Atlantic DPS	T	NLAA	NLAA (P)
Eretmochelys imbricata	Hawksbill sea turtle	E	NLAA	NLAA
Lepidochelys kempii	Kemp's ridley sea turtle	E	NLAA	X
Dermochelys coriacea	Leatherback sea turtle	E	NLAA	NLAA
Caretta caretta	Loggerhead sea turtle, North Pacific Ocean DPS	E	NLAA	X
Caretta caretta	Loggerhead sea turtle, Northwest Atlantic Ocean DPS	T	NLAA	NLAA
Lepidochelys olivacea	Olive ridley sea turtle Mex. Pac. Coast breeding	E	NLAA	X
Lepidochelys olivacea	Olive ridley sea turtle, all other areas	T	NLAA	X

[1]*ESA Status: E = Endangered, T = Threatened*
[2]*Critical Habitat: P= Proposed, X = Not Applicable (Critical Habitat has not been designated)*

7.6 Bony Fish

We concur with the NLAA determination EPA made Gulf grouper (*Mycteroperca jordani*). The Gulf grouper is a foreign ESA-listed species, with only a few records of their occurrence within the action area. These records document strays off the southern extreme of the California coast. Effects are discountable in cases where species habitat use is such that we expect it will be extremely unlikely to occur. This is the case with Gulf grouper whose occurrence within the action area is rare.

EPA made NLAA determinations for a number of fish species: eulachon (*Thaleichthys pacificus*), bocaccio (*Sebastes paucispinis*), yelloweye rockfish (*Sebastes ruberrimus*), and Nassau grouper (*Epinephelus striatus*). NMFS has determined that effects to these species are not wholly beneficial, insignificant or discountable, and thus, we make LAA determinations for these species. EPA made LAA determinations for Ozette Lake sockeye for both carbaryl and methomyl. However, for methomyl, NMFS determined that NLAA is the appropriate conclusion for this species and their designated critical habitat.

In addition, since EPA completed their BE, critical habitat for the Nassau grouper has been designated. NMFS made an effects determination of LAA. The rational for these determinations is provided below.

Exposure estimates were not generated for marine environments. According to the carbaryl and methomyl BEs, exposure of species in the marine/estuarine environment is not reasonably expected to reach concentrations high enough to impact an individual of a species because of dilution and dispersal. However, estimates were derived to represent shallow habitats adjacent to sites of pesticide application. EPA estimates for methomyl in these habitats range from 92 ppb to 2,105 ppb (Bin 2, HUC 17a). For carbaryl, these estimates range from 101ppb to 3,161ppb (Bin 2, HUC 17a). For the purpose of determining whether or not impacts to individuals are insignificant, NMFS finds it appropriate to consider these estimates representing shallow freshwater habitats (Bin 2) when evaluating the potential for adverse effects in shallow marine/estuarine habitats such as enclosed estuaries. According to EPA's methomyl BE, the mortality and sublethal thresholds for estuarine and marine fish are 335ppb and 490ppb, respectively. The mortality threshold for aquatic invertebrates is 3.94ppb. For carbaryl, the mortality and sublethal thresholds for estuarine and marine fish are 1,055ppb and 680ppb, respectively (EPA BE, chapter 2). The mortality threshold for aquatic invertebrates is 1.6ppb. Based on a simple comparison between exposure and response concentrations, we would anticipate that fish species which occupy (or rely on resources dependent on) shallow nearshore habitats are likely to experience adverse effects. Below we discuss the life history and habitat factors that create the potential for exposure for these species. Additional information on exposure and response is further addressed in Chapter 10.

Eulachon

Eulachon are anadromous fish that spend most of their lives at sea but return to freshwater to spawn. Most eulachon production originates in the Columbia River Basin, including the Columbia River, the Cowlitz River the Grays River, the Kalama River, the Lewis River, and the Sandy River (Gustafson 2016). Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters. Larvae and young juveniles become widely distributed in coastal waters, mostly at depths up to 15 meters (Hay and McCarter 2000) but sometimes as deep as 182 meters (Barraclough 1964, as cited in Willson et al. 2006). Adult eulachon are found in coastal and offshore marine habitats. With the exception of some individuals in Alaska, eulachon generally die after spawning (Gustafson 2016). Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001).

Eulachon designated critical habitat consists of 16 areas in the states of Washington, Oregon, and California. The designated areas are a combination of freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km (335 mi) of habitat. The PBFs essential to the conservation of the DPS include:

- Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles.
- Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted.
- Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

Although eulachon spend the majority of their lives in marine habitats, the freshwater phase of their life history is essential. Within freshwater and estuarine environments, eulachon eggs, larva, and adults form dense aggregations within shallow, nearshore habitats. In addition, the PBFs of designated critical habitat include water quality and abundant prey, both of which may be impacted by carbaryl and/or methomyl exposure. Pesticide applications are likely to occur within the watersheds associated with many of the primary eulachon spawning habitats including the Columbia River, the Cowlitz River the Grays River, the Kalama River, the Lewis River, and the Sandy River.

The potential for exposure and adverse effects is such that NMFS does not concur with EPA's determination that the action may affect, but is not likely to adversely affect this species. Therefore, we analyze the effects of the action to eulachon and its designated critical habitat further in this opinion.

Bocaccio and Yelloweye Rockfish

Rockfish are viviparous, meaning the eggs are fertilized internally, the embryonic fish develop within the mother, and the young are released as larvae (Love et al. 2002). Larval rockfish are often observed under free-floating algae, seagrass, and detached kelp (Shaffer et al. 1995; Love et al. 2002), and also occupy the full water column (Weis 2004). Young-of-year juvenile bocaccio occur on shallow rocky reefs and nearshore areas. Young bocaccio associate with macroalgae, especially kelps, and sandy areas that support seagrasses. They form aggregations near the bottom in association with drift algae and throughout the water column in association with canopy-forming kelps (2017 Rockfish Recovery Plan). Unlike bocaccio, juvenile yelloweye rockfish are not typically found in intertidal waters (Love et al. 1991; Studebaker et al. 2009). A few juveniles have been documented in shallow nearshore waters (Love et al. 2002; Palsson et al. 2009), but most settle in habitats along the shallow range of adult habitats in waters greater than 98 feet (30 meters) (Richards 1986; Love et al. 2002)(Yamanaka et al. 2006). Adult yelloweye rockfish remain near the substrate and have relatively small home ranges, while some bocaccio have larger home ranges, move long distances, and spend time suspended in the water column (Demott 1983; Love et al. 2002; Friedwald 2009).

Although these 2 rockfish species spend the majority of their lives in deeper marine habitats, juvenile individuals are found in shallow, nearshore habitat. Species presence in these habitats is less likely for yelloweye than it is for bocaccio. In addition, the PBFs of designated critical habitat include water quality and abundant prey, both of which may be impacted by carbaryl and/or methomyl exposure. Pesticide applications are likely to occur within the watersheds associated with rockfish habitat. Pesticide transport into nearshore marine areas via drift, runoff, and downstream transport create an opportunity for exposure, particularly for juvenile rockfish. The potential for exposure and adverse effects is such that NMFS does not concur with EPA's determination that effects will be insignificant or discountable. Therefore, we analyze the effects of the action to bocaccio and yelloweye rockfish, as well as their designated critical habitats, further in this opinion.

Nassau grouper

Groupers are known as transient aggregate spawners, meaning that they group in large numbers, drawing individuals from a large area to spawn during a specific time of the year for a short period. Fertilized eggs are then transported offshore by ocean currents. After hatching, larvae recruit from oceanic environment to seafloor habitats. Juveniles inhabit macroalgae, coral clumps, and seagrass beds, and are relatively solitary. As they grow, they occupy progressively deeper areas and offshore reefs. When not spawning, adults are most commonly found in waters less than 100 meters deep. Grouper diet changes with age. Juveniles eat plankton, pteropods, amphipods, and copepods. Adults are unspecialized piscivores, bottom-dwelling ambush suction predators (NMFS 2013b).

Spawning aggregation sites are located near significant geomorphological features, such as reef projections (as close as 50 meters to shore) and close to a drop-off into deep water over a wide depth range (6 to 60 meters). Sites are usually several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions. Nassau groupers stay on the spawning site for up to 3 months, spawning at the full moon or between the new and full moons. Spawning occurs within 20 minutes of sunset over the course of several days.
There have been about fifty known spawning sites in insular areas throughout the Caribbean; many of these aggregations no longer form. Current spawning locations are found in Mexico, Bahamas, Belize, Cayman Islands, the Dominican Republic, Cuba, Puerto Rico, the U.S. Virgin Islands, and Florida. Available observation data for spawning are limited, however, observations of spawning aggregations have shown steep declines (Aguilar-Perera 2006; Claro and Lindeman 2003; Sala et al. 2001). Some aggregation sites are comparatively robust and showing signs of increase (Vo et al. 2014; Whaylen et al. 2004). Some Nassau groupers are still observed within the U.S. portion of their range; observations are less common in Florida than in Puerto Rico and the Virgin Islands.

The designation of Nassau grouper critical habitat (89 FR 126) on January 2, 2024 describes several PBFs, one of which is "Nearshore shallow subtidal marine nursery areas with substrate that consists of unconsolidated calcareous medium to very coarse sediments (not fine sand) and shell and coral fragments and may also include cobble, boulders, whole corals and shells, or rubble mounds, to support larval settlement and provide shelter from predators during growth and habitat for prey." Impacts to prey in the nearshore environment may occur as a result of pesticide off-site transport.

Although this grouper species spends the majority of its life in deeper marine habitats, juvenile individuals are found in shallow, nearshore habitat. Pesticide applications are likely to occur within the watersheds associated with these habitats. Pesticide transport into nearshore marine areas via drift, runoff, and downstream transport create an opportunity for exposure, particularly for juveniles. The potential for exposure and adverse effects is such that NMFS does not concur with EPA's determination that effects will be insignificant or discountable. Therefore, we analyze the effects of the action to Nassau grouper and its proposed critical habitat further in this opinion.

Ozette Lake sockeye salmon

EPA made LAA determinations for Ozette Lake sockeye for both carbaryl and methomyl. However, for methomyl, we have determined that NLAA is the appropriate conclusion. EPA's use site spatial data indicate that less than 3 acres of use sites occur within this species' range. Upon closer inspection (and in comparison with the 2019 NLCD), we have determined that these acres are highly unlikely to be treated with methomyl. Additionally, given the size of these use sites and distance from the species habitat (>300 meters) it is extremely unlikely that methomyl could achieve concentrations in the species habitat that could cause any adverse effects to this species or impact any of the PBFs of their designated critical habitat (Appendix B). Therefore, for methomyl, we determined that impacts to the species and its habitat are discountable and the appropriate effects determination is NLAA for Ozette Lake sockeye and its designated critical habitat.

7.7 Marine Invertebrates

Chambered nautilus

We concur with the NLAA determination EPA made for the chambered nautilus (*Nautilus pompilius*). Effects are discountable in cases where species habitat use is such that we expect it will be extremely unlikely that pesticides will occur at concentrations that could affect the species. This is the case with the chambered nautilus.

Chambered nautilus is found in tropical, coastal reef, deep-water habitats of the Indo-Pacific. Within its range, the chambered nautilus has a patchy distribution and is unpredictable in its area of occupancy. The species is considered to be an extreme habitat specialist, physiologically limited by both temperature and depth. It is found in association with steep-sloped forereefs and cannot tolerate temperatures above approximately 25 °C or depths exceeding around 750-800 meters. Although nautilus species have been observed in relatively shallow water (e.g., 5 meters for a different species: *N. macromphalus*), in general, chambered nautilus are considered a deep water species with typical depths between 130 and 700 meters (Dunstan et al. 2011). We anticipate the potential for pesticide exposure in these habitats to be extremely unlikely.

Coral and abalone species

With two exceptions, we do not concur with EPA's NLAA determinations for coral and abalone species or their critical habitats. The two exceptions are for staghorn coral (*Acropora cervicornis*) and elkhorn coral (*Acropora palmata*) designated critical habitat. The PBFs for staghorn and elkhorn coral include substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments. As it is extremely unlikely that carbaryl or methomyl will have any meaningful impacts to the quality or availability of these substrates, these effects are discountable.

The two exceptions aside, based on a simple comparison between exposure and response concentrations, we would anticipate that abalone and coral species that occupy (or rely on resources dependent on) shallow nearshore habitats are likely to experience adverse effects, if exposed. The potential for exposure and adverse effects is such that NMFS does not concur with EPA's determination that effects will be insignificant or discountable.

According to EPA's methomyl BE, the mortality threshold for aquatic invertebrates is 3.94 ppb, this value is used as a proxy (or surrogate) for abalone, as no mollusk data are available. For carbaryl, the mortality and sublethal thresholds for mollusks is 6,600 ppb and 1,000 ppb, respectively. The sublethal response associated with this concentration was decreased fecundity in freshwater snail. Several studies have evaluated the effects of insecticides on coral (see review in (Nalley et al. 2021)). Larval settlement was shown to be sensitive to insecticides (Markey et al. 2007). In this study, carbaryl decreased metamorphosis and the ability of the coral larvae to settle at concentration as low as 3.0 ppb. In an earlier study (Acevedo 1991) carbaryl concentrations of 100 ppm resulted in 70-90% mortality to the swimming planulae stage of corals. No available studies have evaluated the impact of methomyl on coral. However, Nalley et al. (2021) describes coral response endpoints and concentrations for 6 different insecticides. Responses described include: larval survival, settlement, bleaching, and photosynthetic response. Exposure estimates were not generated for marine environments. According to the carbaryl and methomyl BEs, exposure to species in the marine/estuarine environment is not reasonably expected to reach concentrations high enough to impact an individual of a species because of dilution and dispersal. However, estimates were derived to represent shallow habitats adjacent to sites of pesticide application. EPA methomyl estimates in these habitats range from 92 ppb to 2,105 ppb (Bin 2, HUC 17a). For carbaryl, estimates range from 101 ppb to 3,161 ppb (Bin 2, HUC 17a). For the purpose of determining whether or not impacts to individuals are insignificant, NMFS finds it appropriate to consider estimates representing shallow freshwater habitats (Bin 2) when evaluating the potential for adverse effects in shallow marine/estuarine habitats.

Black abalone

Abalone have separate sexes and are broadcast spawners. Female abalone may discharge over 2 million unfertilized eggs per spawning episode and are capable of undergoing multiple episodes each spawning season. As spawning occurs, gametes are dispersed from the gonads of both parents into the sea and fertilization is entirely external. The embryos and larvae that result from this process are exposed to a wide range of physical and biological sources of mortality. Although an average female is capable of producing over 20 million larvae over her lifetime, larval survival to adulthood is estimated at less than 1% (Leighton 2000). Twenty-four hours after fertilization, a free-swimming larva emerges from the fertilized egg and joins with plankton (Leighton 1989; Leighton 2000). After 2–3 weeks in the plankton, the larvae settle to the bottom. One to 3 months after settlement juveniles are fully formed and resemble adults.

The black abalone is found along rocky shorelines and coastal habitats. Black abalone are most commonly observed in the mid to low intertidal, in complex habitats with deep crevices that provide shelter for juvenile recruitment and adult survival. In addition, the PBFs of designated critical habitat include suitable water quality which may be impacted by carbaryl and/or methomyl exposure. Pesticide applications are likely to occur within the watersheds associated with black abalone habitats. Pesticide transport into nearshore marine areas via drift, runoff, and downstream transport create an opportunity for exposure to adults as well as abalone in the embryo and larval stages.

White Abalone

The white abalone historically was found in coastal waters between 5-60 meters deep from Point Conception, California to Punta Abreojos, Baja California, Mexico (Cox 1960, Stierhoff et al. 2012). Prior to the fishery collapse, major concentrations of white abalone occurred between 25-30 meters deep (Stierhoff et al. 2012). Since the fishery collapse, the depth distribution of white abalone has shifted toward deeper depths, as most living individuals are those that were too deep to be fished during the 1960s and 1970s (Lafferty et al. 2004). In surveys conducted at an offshore bank from 2002 – 2010 between depths of 30 to 60 m, white abalone were most abundant and dense at depths of 40-50 meters (Stierhoff et al. 2012). The duration of the larval stage is roughly 1–2 weeks where they drift in the water current. Pesticide transport into marine areas via drift and runoff create an opportunity for exposure to adults as well as abalone in the larval stages.

Coral (12 species in Caribbean and Indo-Pacific)

Coral species are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. Most coral species also contain algal symbionts living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005). Corals employ both sexual and asexual reproductive strategies including brooding and spawning as well as recruitment via fragmentation and reattachment.

The 7 ESA-listed Caribbean coral species are also found in waters off Southern Florida, Puerto Rico, the U.S. Virgin Islands, and throughout the Caribbean. Additionally, several have been observed in the Flower Garden Banks within the Gulf of Mexico. The 5 ESA-listed Indo-Pacific coral species have large scale distributions within the oceanic central and western Pacific Ocean and central Indo-Pacific. Within the United States, species are known to occur in Guam, the Commonwealth of the Northern Mariana Islands, American Samoa, and the Pacific Remote Island Areas. All 12 coral species are currently listed as threatened. However the pillar coral has recently been proposed to be listed as endangered (88 FR 59494). All 7 of the Caribbean coral species have designated critical habitat. The 5 Indo-pacific corals species have proposed critical habitat.

Coral species collectively inhabit a variety of reef types and range from shallow reef flats down to 90 meters in depth. The proximity of these habitats to coastal areas subjects them to impacts from multiple land-based activities including dredging and disposal activities, stormwater run-off, coastal and maritime construction, land development, wastewater and sewage outflow discharges, and point and non-point source pollutant discharges. Pesticide applications are likely to occur within the watersheds adjacent to coral habitats. Pesticide transport into nearshore marine areas via drift, runoff, and downstream transport create an opportunity for exposure to adults as well as early life-stages. In addition, 5 of the ESA-listed coral species have proposed critical habitat that includes as a PBF, "Marine water with levels of anthropogenically-introduced (from humans) chemical contaminants that do not preclude or inhibit any demographic function."

Proposed Marine Invertebrates

EPA did not make effect determinations for the queen conch (*Alger gigas*) or the sunflower sea star (*Pycnopodia helianthoides*) because these species were proposed for listing after EPA completed their BE. However, as noted in Chapter 2, EPA agreed to a conference biological opinion on these species. We determined that LAA was the appropriate determination for these two species based on similar habitat use and anticipated toxicological response as the abalone and coral described above.

Candidate Species

In the BE, EPA made NLAA determinations for several candidate species: cusk (*Brosme brosme*), an additional steelhead DPS (*Oncorhynchus (Salmo) mykiss*), and an additional Chinook ESU (*Oncorhynchus tshawytscha*). NMFS does not consult on candidate species.

8 Status of Species and Critical Habitat Likely to be Adversely Affected

Section Contents

8.1 Introduction

The purpose of this section is to characterize the status of the species that are likely to be adversely affected by the action, and to describe the status, conservation role and function of their respective critical habitats. The species (and designated critical habitat) listed in Table 12 will be carried forward in this opinion for further analysis of the effects of the action and the potential for jeopardy to the species or destruction or adverse modification of critical habitat.

Table 12. Species and Critical Habitat Likely to be Adversely Affected

Species	ESA Status	Critical Habitat Designated?
Atlantic Salmon, Gulf of Maine *Salmo salar*	Endangered	Yes
Chum Salmon, Columbia River *Oncorhynchus keta*	Threatened	Yes
Chum Salmon, Hood Canal summer-run *Oncorhynchus keta*	Threatened	Yes
Chinook Salmon, California Coastal *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Central Valley spring-run *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Lower Columbia River *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Puget Sound *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Sacramento River winter-run *Oncorhynchus tshawytscha*	Endangered	Yes
Chinook Salmon, Snake River fall-run *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Snake River spring/summer run *Oncorhynchus tshawytscha*	Threatened	Yes
Chinook Salmon, Upper Columbia River spring-run *Oncorhynchus tshawytscha*	Endangered	Yes
Chinook Salmon, Upper Willamette River *Oncorhynchus tshawytscha*	Threatened	Yes
Coho Salmon, Central California Coast *Oncorhynchus kisutch*	Endangered	Yes
Coho Salmon, Lower Columbia River *Oncorhynchus kisutch*	Threatened	Yes
Coho Salmon, Oregon Coast *Oncorhynchus kisutch*	Threatened	Yes
Coho Salmon, South Oregon and North Calif. Coast *Oncorhynchus kisutch*	Threatened	Yes
Sockeye Salmon, Ozette Lake (NLAA for methomyl) *Oncorhynchus nerka*	Threatened	Yes
Sockeye Salmon, Snake River *Oncorhynchus nerka*	Endangered	Yes
Steelhead, California Central Valley *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Central California coast *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Lower Columbia River	Threatened	Yes

Species	**ESA Status**	**Critical Habitat Designated?**
Oncorhynchus mykiss		
Steelhead, Middle Columbia River *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Northern California *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Puget Sound *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Snake River Basin *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, South Central California Coast *Oncorhynchus mykiss*	Threatened	Yes
Steelhead, Southern California *Oncorhynchus mykiss*	Endangered	Yes
Steelhead, Upper Columbia River *Oncorhynchus mykiss*	Endangered	Yes
Steelhead, Upper Willamette River *Oncorhynchus mykiss*	Threatened	Yes
Eulachon, Pacific smelt, Southern *Thaleichthys pacificus*	Threatened	Yes
Green sturgeon, Southern *Acipenser medirostris*	Threatened	Yes
Shortnose sturgeon *Acipenser brevirostrum*	Endangered	No
Atlantic sturgeon, Carolina *Acipenser oxyrinchus oxyrinchus*	Endangered	Yes
Atlantic sturgeon, Chesapeake Bay *Acipenser oxyrinchus oxyrinchus*	Endangered	Yes
Atlantic sturgeon, Gulf of Maine *Acipenser oxyrinchus oxyrinchus*	Threatened	Yes
Atlantic sturgeon, New York Bight *Acipenser oxyrinchus oxyrinchus*	Endangered	Yes
Atlantic sturgeon, South Atlantic *Acipenser oxyrinchus oxyrinchus*	Endangered	Yes
Gulf sturgeon *Acipenser oxyrinchus desotoi*	Threatened	Yes
Yelloweye rockfish *Sebastes ruberrimus*	Threatened	Yes
Bocaccio *Sebastes paucispinis*	Endangered	Yes
Nassau grouper *Epinephelus striatus*	Threatened	Yes
Smalltooth sawfish *Pristis pectinata*	Endangered	Yes

Species	ESA Status	Critical Habitat Designated?
Giant Manta Ray *Manta birostris*	Threatened	No
Black abalone *Haliotis cracherodii*	Endangered	Yes
White abalone *Haliotis sorenseni*	Endangered	No
Sunflower Sea Star *Pycnopodia helianthoides*	Proposed Threatened	No
Queen conch *Alger gigas*	Proposed Threatened	No
Staghorn coral *Acropora cervicornis*	Threatened	Yes (NLAA)
Elkhorn coral *Acropora palmata*	Threatened	Yes (NLAA)
Coral, *Acropora globiceps*	Threatened	Proposed
Coral, *Acropora retusa*	Threatened	Proposed
Coral, *Acropora speciosa*	Threatened	Proposed
Coral, *Euphyllia pardivisa*	Threatened	Proposed
Coral, *Isopora crateriformis*	Threatened	Proposed
Coral, Boulder star *Orbicella franksi*	Threatened	Yes (NLAA)
Coral, Lobed star *Orbicella annularis*	Threatened	Yes (NLAA)
Coral, Mountainous star *Orbicella faveolata*	Threatened	Yes (NLAA)
Coral, Pillar *Dendrogyra cylindrus*	Proposed Endangered	Yes (NLAA)
Coral, Rough cactus *Mycetophyllia ferox*	Threatened	Yes (NLAA)
Killer whale, Southern Resident *Orcinus orca*	Endangered	Yes

The evaluation of adverse effects in this opinion begins by summarizing the biology and ecology of those species that are likely to be adversely affected and what is known about their life histories in the action area. The status is determined by the level of risk that the ESA-listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This helps to inform the description of the species' current "reproduction, numbers, or distribution," which is part of the jeopardy determination as described in 50 C.F.R. §402.02. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on this NMFS website: https://www.fisheries.noaa.gov/find-species.

This section also examines the condition of critical habitat throughout the designated area (such as various watersheds and coastal and marine environments that make up the designated area), and discusses the condition and current function of designated critical habitat, including the essential physical and biological features that contribute to that conservation value of the critical habitat.

In assessing the status of the ESA-listed species NMFS made use of the viable salmonid population (VSP) concept and its 4 criteria. NMFS used these criteria to assess salmonids and, where appropriate, non-salmonid species. A VSP is an independent population (a population of which extinction probability is not substantially affected by exchanges of individuals with other populations) with a negligible risk of extinction, over a 100-year period, when threats from random catastrophic events, local environmental variation, demographic variation, and genetic diversity changes are taken into account (McElhany et al. 2000b). The 4 factors defining a viable population are a population's: (1) spatial structure; (2) abundance; (3) annual growth rate, including trends and variability of annual growth rates; and (4) diversity (McElhany et al. 2000b).

A population's tendency to increase in abundance and its variation in annual population growth defines a viable population (McElhany et al. 2000b; Morris and Doak 2002). A negative long-term trend in average annual population growth rate will eventually result in extinction. Further, a weak positive long-term growth rate will increase the risk of extinction as it maintains a small population at low abundances over a longer time frame. A large variation in the growth rates also increases the likelihood of extinction (Lande 1993; Morris and Doak 2002). Thus, in our status reviews of each ESA-listed species, we provide information on population abundance and annual growth rate of extant populations.

The action area for this consultation contains designated critical habitat. Critical habitat is defined as the specific areas within the geographical area occupied by the species, at the time it is listed, on which are found those physical or biological features that are essential to the conservation of the species, and which may require special management considerations or protection. Critical habitat can also include specific areas outside the geographical area occupied by the species at the time it is listed that are determined by the Secretary to be essential for the conservation of the species (ESA of 1973, as amended, section 3(5)(A)).

The primary purpose in evaluating the status of critical habitat is to identify the function of the critical habitat to support the intended conservation role for each species. Such information is important for an adverse modification analysis as it establishes the context for evaluating whether the action results in negative changes in the function and role of the critical habitat for species conservation. NMFS bases its critical habitat analysis on the areas of the critical habitat that are affected by the action and the area's PBFs that are essential to the conservation of a given species, and not on how individuals of the species will respond to changes in habitat quantity and quality.

In evaluating the status of designated critical habitat, we consider the current quantity, quality, and distribution of the physical or biological features that are essential for the conservation of the species. To fully understand the conservation role of these habitats, specific physical and biological habitat features (*e.g.*, water temperature, water quality, forage, natural cover, etc.) were identified for each life stage (see Appendix B).

Note about maps: Each section below contains a map depicting the species range and, when applicable, designated critical habitat. These maps are provided for general reference only and do not necessarily represent the spatial data that was used later in the assessment (e.g., for generating overlap percentages in the effects analysis). For more detailed information on species range and distribution, visit: https://www.fisheries.noaa.gov/species-directory/threatened-endangered.

8.2 Atlantic Salmon, Gulf of Maine DPS

Table 13. Atlantic salmon, Gulf of Maine DPS; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Salmo salar	Atlantic salmon	Gulf of Maine	Endangered	2020	74 FR 29344	2019	74 FR 39903

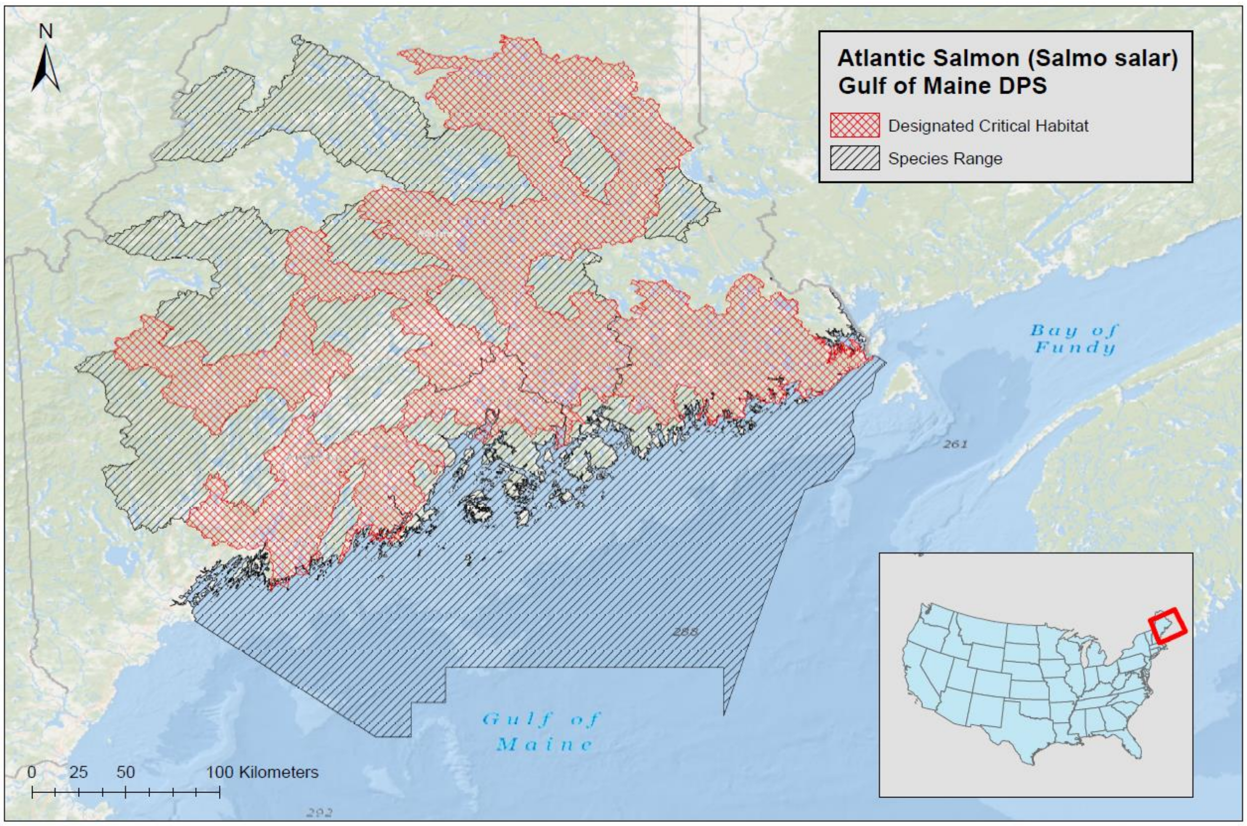

Figure 10. Atlantic salmon range and designated critical habitat

Species Description. Atlantic salmon is an anadromous fish, occupying freshwater streams in North America. There are 3 Atlantic salmon distinct population segments in the United States: Long Island Sound, Central New England, and the Gulf of Maine (Fay et al. 2006). The Gulf of Maine DPS Atlantic salmon are found in watersheds throughout Maine (Figure 10). Adult Atlantic salmon are silver-blue with dark spots. The Gulf of Maine DPS was first listed as endangered by the U. S. Fish and Wildlife Service and NMFS on November 17, 2000 (65 FR 69459). The listing was refined by the Services on June 19, 2009 (74 FR 29344) to include all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment.

Status. Historically, Atlantic salmon occupied U.S. rivers throughout New England, with an estimated 300,000 to 500,000 adults returning annually (Fay et al. 2006). Of the 3 DPSs found in the U.S., native salmon in the Long Island Sound and Central New England DPSs were extirpated in the 1800s. Several rivers within these DPSs are presently stocked with Gulf of Maine DPS salmon. The Gulf of Maine DPS Atlantic salmon was listed as endangered in response to population decline caused by many factors, including overexploitation, degradation of water quality and damming of rivers, all of which remain persistent threats as reported in(Fay et al. 2006). Coastal development poses a threat as well, as artificial light can disrupt and delay fry dispersal (Riley et al. 2013). Climate change may cause changes in prey availability and thermal niches, further threatening Atlantic salmon populations (Mills et al. 2013). The major threats to Atlantic salmon survival and recovery are low marine survival, the direct and indirect effects of dams and road stream crossings, the West Greenland harvest, and climate change.

Life History. Adult Atlantic salmon typically spawn in early November and juveniles spend about 2 years feeding in freshwater until they weigh approximately 2 ounces and are 6 inches in length. Smoltification (the physiological and behavioral changes required for the transition to salt water) usually occurs at age 2 for Gulf of Maine DPS Atlantic salmon. Immediately upon entering marine water, Gulf of Maine DPS Atlantic salmon migrate more than 4,000 km in the open ocean to reach feeding areas in the Davis Strait between Labrador and Greenland. The majority of Gulf of Maine DPS Atlantic salmon (about 90%) spend 2 winters at sea before reaching maturity and returning to their natal rivers, with the remainder spending 1 or 3 winters at sea. At maturity, Gulf of Maine DPS Atlantic salmon typically weigh between 8 to 15 pounds and average thirty inches in length.

Population Dynamics. The following is a discussion of the species' population and its variance over time. This section is broken down into: abundance, population growth rate, genetic diversity, and spatial distribution.

Abundance. The Gulf of Maine DPS of Atlantic salmon remain at critically low abundance with the average 10-year return of naturally reared salmon being below 100 adult spawners in each of the 3 Salmon Habitat Recovery Units (SHRU). This is well below the minimum abundance threshold needed for reclassification from endangered to threatened. The very low population sizes constitutes a significant risk to the resiliency of the species through increasing losses in genetic fitness, loss of adaptive traits, and reduced ability to withstand catastrophic events.

Productivity / Population Growth Rate. Population growth rate of naturally reared fish has improved in recent years to where the 10-year average across SHRUs and within SHRUs has error bounds that encompass 1 (a stable population). Although these growth rates fall within the goals for reclassification, they are overshadowed by the small population sizes.

Genetic Diversity. The Gulf of Maine DPS Atlantic salmon is genetically distinct from other Atlantic salmon populations in Canada, and can be further delineated into stocks by river. The Downeast Coastal stocks include the Dennys, East Machias, Machias, Pleasant and Narraguagus rivers. The hatchery supplementation programs for the Penobscot and Merrymeeting Bays stocks use river-specific broodstock (USASAC 2016).

Distribution. Gulf of Maine DPS Atlantic salmon can be found in at least 8 rivers in Maine: Dennys River, East Machias River, Machias River, Pleasant River, Narraguagus River, Ducktrap River, Sheepscot River, Cove Brook, Penobscot River, Androscoggin River and the Kennebec River.

Designated Critical Habitat. On June 19, 2009, NMFS and the U.S. Fish and Wildlife Service designated critical habitat for Atlantic salmon (74 FR 29300). PBFs considered essential for the conservation of the Gulf of Maine DPS of Atlantic salmon are:

Spawning and Rearing

- Deep, oxygenated pools and cover (e.g., boulders, woody debris, vegetation, etc.), near freshwater spawning sites, necessary to support adult migrants during summer while they await spawning in the fall.
- Freshwater spawning sites that contain clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support spawning activity, egg incubation, and larval development.
- Freshwater spawning and rearing sites with clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support emergence, territorial development and feeding activities of Atlantic salmon fry.
- Freshwater rearing sites with space to accommodate growth and survival of Atlantic parr.
- Freshwater rearing sites with a combination of river, stream, and lake habitats that accommodate parr's ability to occupy many niches and maximize parr production.

Migration

- Freshwater and estuary migratory sites free from physical and biological barriers that delay or prevent access of adult seeking spawning grounds needed to support recovered populations.
- Freshwater and estuary migration sites with pool, lake, and instream habitat that provide cool, oxygenated water and cover items (e.g., boulders, woody debris, and vegetation) to serve as temporary holding and resting areas during upstream migration of adult Atlantic salmon.
- Freshwater and estuary migration sites with abundant, diverse native fish communities to serve as a protective buffer against predation.
- Freshwater and estuary migration sites free from physical and biological barriers that delay or prevent emigration of smolts to the marine environment.
- Freshwater and estuary migration sites with sufficiently cool water temperatures and water flows that coincide with diurnal cues to stimulate smolt migration.

The critical habitat listing document identified a number of activities and associated threats that may affect the PBFs and associated physical and biological features essential to the conservation of Atlantic salmon within the occupied range of the Gulf of Maine DPS. These activities, which include agriculture, forestry, changing land-use and development, hatcheries and stocking, roads and road crossings, mining, dams, dredging, and aquaculture have the potential to reduce the quality and quantity of the PBFs and their associated physical and biological features.

Recovery Goals. See the 2019 Recovery Plan for the Gulf of Maine DPS Atlantic Salmon (USFWS 2018) for complete down listing/delisting criteria for each of their respective recovery goals. The following is the biological criteria for delisting the DPS:

- Abundance: The DPS has a self-sustaining annual escapement of at least 2,000 wild origin adults in each SHRU, for a DPS-wide total of at least 6,000 wild adults.
- Productivity: Each SHRU has a positive mean population growth rate of greater than 1.0 in the 10-year (2-generation) period preceding delisting. In addition, at the time of delisting, the DPS demonstrates self-sustaining persistence, whereby the total wild population in each SHRU has less than a 50% probability of falling below 500 adult wild spawners in the next 15 years based on population viability analysis (PVA) projections.
- Habitat: Sufficient suitable spawning and rearing habitat for the offspring of the 6,000 wild adults is accessible and distributed throughout the designated Atlantic salmon critical habitat, with at least 30,000 accessible and suitable Habitat Units in each SHRU, located according to the known migratory patterns of returning wild adult salmon. This will require both habitat protection and restoration at significant levels.

8.3 Chum salmon, Columbia River ESU

Table 14. Chum salmon, Columbia River ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus keta	Chum Salmon	Columbia River ESU	Threatened	2022	70 FR 37160	2013	70 FR 52630

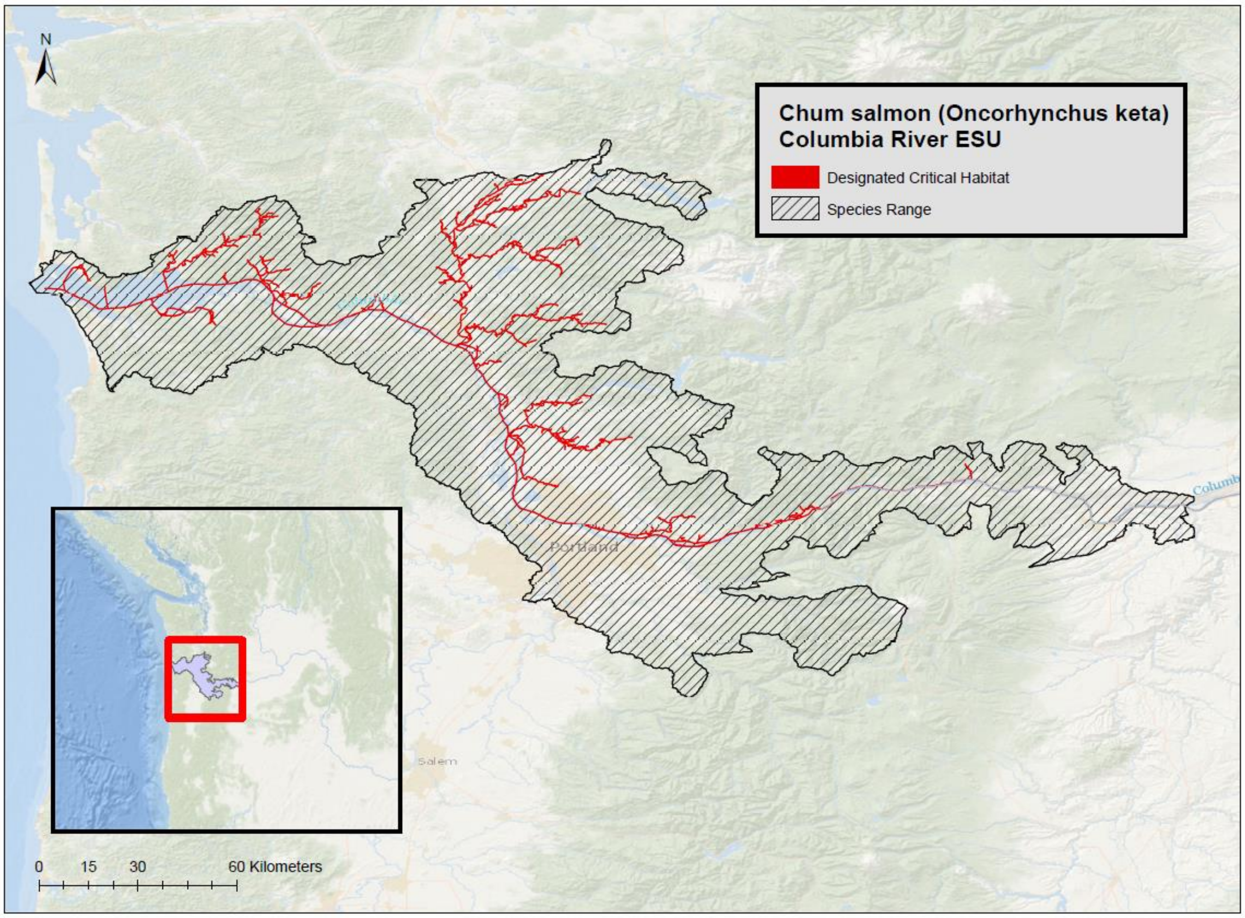

Figure 11. Chum salmon, Columbia River ESU range and designated critical habitat **Species Description.** Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between 8 and 15 pounds, but they can get as large as 45 pounds and 3.6 feet long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning. Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with

black speckles. On March 25, 1999, NMFS listed the Hood Canal Summer-run ESU and the Columbia River ESU of chum salmon as threatened (64 FR 14508). NMFS reaffirmed the status of these 2 ESUs as threatened on June 28, 2005 (70 FR 37160), and in the 2022 Status Update.

Status. For the CR Chum Salmon ESU, some populations have increased in abundance since the last review (2016). However, improvements in a few populations do not warrant a change in the risk category for the ESU as a whole, especially given the uncertainty regarding climatic effects in the near future (Ford 2022; Myers, personal communication, May 11, 2022). The viability of this ESU is relatively unchanged since the last review and therefore remains at moderate to high risk of extinction. The majority of the populations within the Columbia River chum salmon ESU are at high to very high risk, with very low abundances (NWFSC 2015b). These populations are at risk of extirpation due to demographic stochasticity and Allee effects. It is notable that during this most recent review period the 3 populations (Grays River, Washougal, and Lower Gorge demographically independent populations (DIPS) improved markedly in abundance. Improvements in productivity were observed in almost every year during the 2015–19 interval. This is somewhat surprising, given that the majority of chum salmon emigrate to the ocean as subyearlings after only a few weeks, and one would expect the poor ocean conditions to have a strong negative influence on the survival of juveniles (as with many of the other ESUs in this region). In contrast to the 3 DIPs, the remaining populations in this ESU have not exhibited any detectable improvement in status. Abundances for these populations are assumed to be at or near zero, and straying from nearby healthy populations does not seems sufficient to reestablish self-sustaining populations. It may be that the chum salmon life-history strategy of emigrating post-emergence en masse (possibly as a predator swamping mechanism) requires a critical number of spawners to be effective. The potential prospect of poor ocean conditions for the near future may put further pressure on the Columbia River chum salmon ESU (NWFSC 2015b). Freshwater habitat conditions may be negatively influencing spawning and early rearing success in some basins, and contributing to the overall low productivity of the ESU. Columbia River chum salmon were historically abundant and subject to substantial harvest until the 1950s (Johnson et al. 1997). There is no directed harvest of this ESU and the incidental harvest rate has been below 1% for the last 5 years (NWFSC 2015b). Land development, especially in the low gradient reaches that chum salmon prefer, will continue to be a threat to most chum salmon populations due to projected increases in the population of the greater Vancouver-Portland area and the Lower Columbia River overall (Metro 2015). The Columbia River chum salmon ESU remains at a moderate risk of extinction (NWFSC 2015b). Based on the 2022 Status Update, no reclassification for the Columbia River (CR) Chum Salmon ESU is warranted. Therefore, the CR Chum Salmon ESU remains listed as threatened.

Life History. Most chum salmon mature and return to their birth stream to spawn between 3 and 5 years of age, with 60% to 90% of the fish maturing at 4 years of age. Age at maturity appears to follow a latitudinal trend (i.e., greater in the northern portion of the species' range). Chum salmon typically spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to 100 km from the sea. Juveniles out-migrate to seawater almost immediately after emerging from the gravel covered redds (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., coastal cutthroat trout, steelhead, Coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Upon entering marine water, young of year (YOY) chum salmon are nearshore (intertidal zone and shallow subtidal) obligate feeders. For several months they are observed in just several inches of water feeding on small marine invertebrates. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to confuse predators (Miller and Brannon 1982).

Chum salmon spend 2 to 5 years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history compared to other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175 E longitude (Johnson et al. 1997b). North American chum salmon migrate north along the coast in a narrow band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer-run chum, may not make extended coastal migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean (Johnson et al. 1997b).

Table 15. Temporal distribution of Chum salmon, Columbia River ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Population Dynamics

Abundance / Productivity. Chum populations in the Columbia River historically reached hundreds of thousands to a million adults each year (NMFS 2017b). In the past 50 years, the average has been a few thousand a year. The majority of populations in the Columbia River chum ESU remain at high to very high risk, with very low abundances (NWFSC 2015b). Ford (2011) concluded that 14 out of 17 of chum populations in this ESU were either extirpated or nearly extirpated. The very low persistence probabilities or possible extirpations of most chum salmon populations are due to low abundance, productivity, spatial structure, and diversity. Only one population (Grays River) is at low risk, with spawner abundances in the thousands, and demonstrating a recent positive trend. Two other populations (Washougal River and Lower Gorge) are fluctuating, but overall have an increasing numbers of spawners and appear to be relatively stable (NWFSC 2015b).

Genetic Diversity. There are currently 4 hatchery programs in the Lower Columbia River releasing juvenile chum salmon: Grays River Hatchery, Big Creek Hatchery, Lewis River Hatchery, and Washougal Hatchery (NMFS 2017b). Total annual production from these hatcheries has not exceeded 500,000 juvenile fish. All of the hatchery programs in this ESU use integrated stocks developed to supplement natural production. Other populations in this ESU persist at very low abundances and the genetic diversity available would be very low (NWFSC 2015b). Although, hatchery production of Columbia River chum salmon has been limited and hatchery effects on diversity are thought to have been relatively small, diversity has been greatly reduced at the ESU level because of presumed extirpations and low abundance in the remaining populations (fewer than 100 spawners per year for most populations) (LCFRB 2010a; NMFS 2013a).

Distribution. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids. Chum salmon have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean to Monterey Bay, California. Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S. At present, major spawning populations occur as far south as Tillamook Bay on the northern Oregon coast. The Columbia River chum salmon ESU includes all natural-origin chum salmon in the Columbia River and its tributaries in Washington and Oregon. The ESU consists of 3 populations: Grays River, Hardy Creek and Hamilton Creek in Washington State. Chum salmon from 4 artificial propagation programs also contribute to this ESU.

Designated Critical Habitat. NMFS designated critical habitat for the Columbia River chum salmon ESU in 2005 (70 FR 52630). Sixteen of the 19 subbasins reviewed in NMFS's assessment of critical habitat for the CR chum salmon ESU were rated as having a high conservation value. The remaining 3 subbasins were given a medium conservation value. Washington's federal lands were rated as having high conservation value to the species. PBFs considered essential for the conservation of the Columbia River ESU of Chum salmon described in Appendix B.

Limited information exists on the quality of essential habitat characteristics for CR chum salmon. However, the migration PBF has been significantly impacted by dams obstructing adult migration and access to historic spawning locations. Water quality and cover for estuary and rearing PBFs have decreased in quality to the extent that the PBFs are not likely to maintain their intended function to conserve the species.

Recovery Goals. The ESU recovery strategy for Columbia River chum salmon focuses on improving tributary and estuarine habitat conditions, reducing or mitigating hydropower impacts, and reestablishing chum salmon populations where they may have been extirpated (NMFS 2013a). The goal of the strategy is to increase the abundance, productivity, diversity, and spatial structure of chum salmon populations such that the Coast and Cascade chum salmon strata are restored to a high probability of persistence, and the persistence probability of the 2 Gorge populations improves. For details on Columbia River chum salmon ESU recovery goals, including complete down-listing/delisting criteria, see the NMFS 2013 recovery plan (NMFS 2013a).

Recommendations for Future Actions

- Conduct systematic review and analysis of high priority Lower Columbia River mainstem and tributary area habitat needs, identified in NMFS 2013a, and compare needs to what has been accomplished.
- Conduct monitoring to evaluate ship wake stranding frequency and locations where stranding occurs and assess factors contributing to wake stranding such as location, topography, vessel speed, et cetera, to determine best practices to reduce wake stranding mortality.
- Promote riparian plantings of native canopy tree cover species opportunistically in all watersheds.
- Coordinate with EPA in an evaluation of Washington State Water Quality Standards, reflecting Oregon and Idaho consultation outcomes.
- For populations within the below listed major population groups (MPGs), we recommend the following recovery actions over the next 5 years:

Coast MPGs

- Increase the number of projects that reduce sediment load in spawning habitat for Grays/Chinook River chum.
- Promote projects that reduce flashy stream conditions to improve spawning habitat for Grays/Chinook River chum.
- Implement additional habitat improvement projects in the Elochoman River and Abernathy, Mill, and Germany creeks, and their tributaries to augment spawning (chum) and rearing (coho) habitat.

Cascade MPGs

- Identify and implement spawning habitat projects to expand spatial distribution of chum into the Cascade MPG, with priority on the Lewis and Washougal rivers, (Washington Primary populations) and the Cowlitz and Kalama rivers (contributing populations).
- Work with county and city jurisdictions to protect watershed hydrology from long-term development impacts (floodplain development and groundwater withdrawals). Focus these efforts on high growth rate watersheds along the I-5 and I-205 corridors, including the East Fork Lewis River, North Fork Lewis River, Coweeman River, Kalama River, Washougal River, Salmon Creek, and Lower Cowlitz tributaries.

Gorge MPGs

- Continue to work with partners on programs protecting instream and floodplain habitats in key chum spawning areas, such as Duncan Creek and Hamilton Creek, (e.g., evaluate if large wood debris mitigates excess winter stream flows that degrade spawning for Upper Gorge chum).
- Continue to work with partners to identify suitable chum spawning habitat streams and reaches to emplace habitat creation or enhancement projects in order to expand spatial distribution into the gorge strata.
- Implement habitat projects to mitigate excess winter flow to improve spawning habitat for Lower Gorge chum and Upper Gorge chum.

8.4 Chum salmon, Hood Canal summer-run ESU

Table 16. Chum salmon, Hood Canal summer-run ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus keta	Chum salmon	Hood Canal summer-run	Threatened	2022	70 FR 37160	2005	70 FR 52629

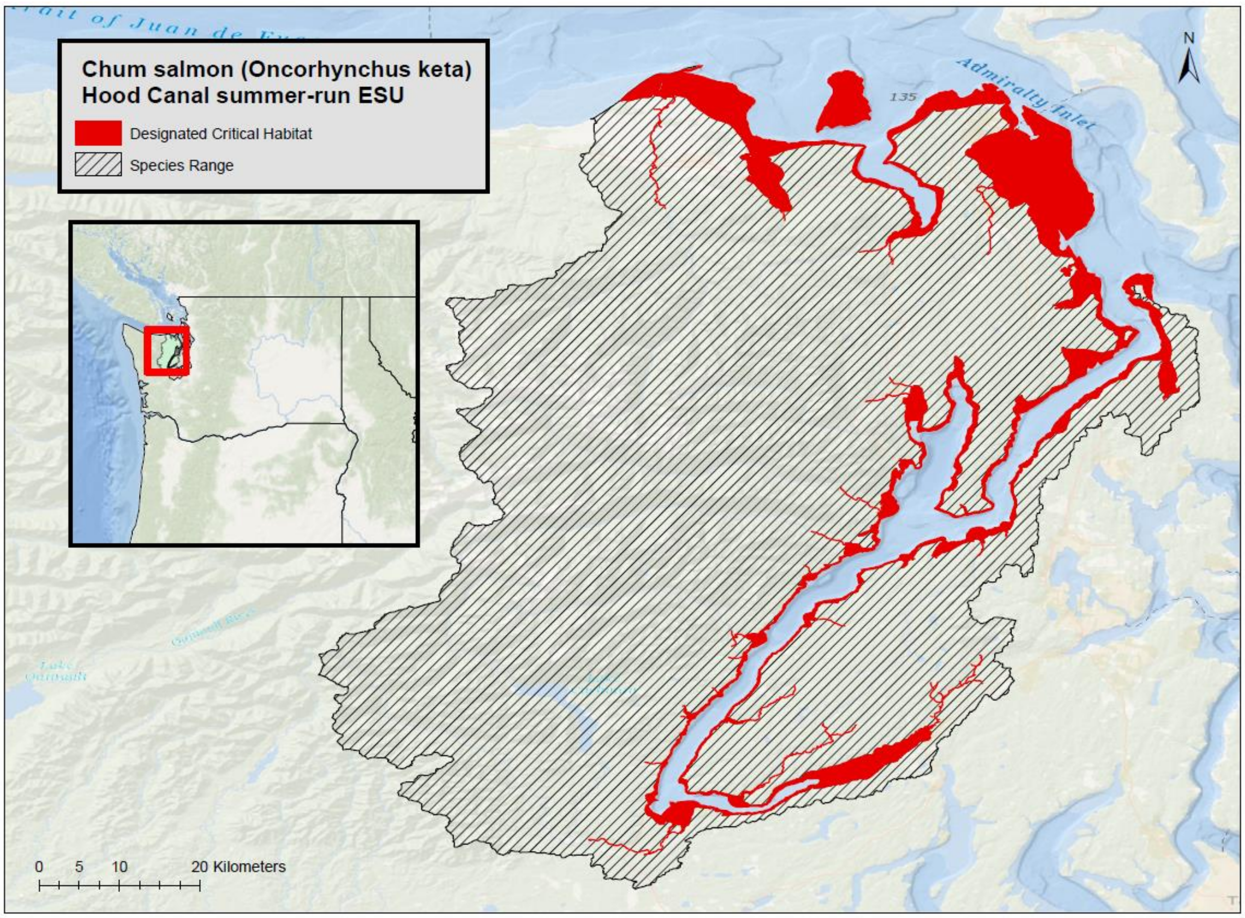

Figure 12. Chum salmon, Hood Canal summer-run ESU range and designated critical habitat

Species Description. Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between 8 and 15 pounds, but they can get as large as 45 pounds and 3.6 feet long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning. Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with black speckles. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids. Chum salmon have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean to Monterey Bay, California. Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S. At present, major spawning populations occur as far south as Tillamook Bay on the northern Oregon coast. On March 25, 1999, NMFS listed the Hood Canal Summer-run ESU and the Columbia River ESU of chum salmon as threatened (64 FR 14508). NMFS reaffirmed the status of these 2 ESUs as threatened on June 28, 2005 (70 FR 37160).

Status. Two previous status reviews (2011 and 2015) indicated some positive signs for the Hood Canal summer-run chum salmon ESU, but now the abundance of returning spawners is declining based on the most recent status update (NMFS WCR 2017). Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; considered a good sign for viability in terms of spatial structure and diversity (Ford 2011). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NWFSC 2015b). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (Ford 2011). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years (NWFSC 2015b). Hatchery supplementation programs have ended (2022). Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time (NWFSC 2015b). Overall, the Hood Canal Summer-run chum salmon ESU remains at a moderate risk of extinction.

Life History. Most chum salmon mature and return to their birth stream to spawn between 3 and 5 years of age, with 60% to 90% of the fish maturing at 4 years of age. Age at maturity appears to follow a latitudinal trend (i.e., greater in the northern portion of the species' range). Chum salmon typically spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to 100 km from the sea. Juveniles out-migrate to seawater almost immediately after emerging from the gravel covered redds ((Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., coastal cutthroat trout, steelhead, Coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Upon entering marine water, YOY chum salmon are nearshore (intertidal zone and shallow subtidal) obligate feeders. For several months they are observed in just several inches of water feeding on small marine invertebrates. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to confuse predators (Miller and Brannon 1982).

Chum salmon spend 2 to 5 years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history compared to other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175 E longitude (Johnson et al. 1997b). North American chum salmon migrate north along the coast in a narrow band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer-run chum, may not make extended coastal migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean (Johnson et al. 1997b).

Table 17. Temporal distribution of Chum salmon, Hood Canal summer-run ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Population Dynamics

Abundance / Productivity. Of the 16 populations that comprise the Hood Canal Summer-run chum ESU, 7 are considered “functionally extinct” (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek and Chimicum). The remaining 9 populations are well distributed throughout the ESU range except for the eastern side of Hood Canal (Johnson et al. 1997b). Two independent major population groups have been identified for this ESU: (1) spawning aggregations from rivers and creeks draining into the Strait of Juan de Fuca, and (2) spawning aggregations within Hood Canal proper (Sands 2009). Criteria for spatial structure/diversity were nearly met for Strait of Juan de Fuca and Hood Canal summer-run chum salmon populations until the recent spawner return abundance downturns that started in 2017. As of 2018, the Quilcene, Dosewallips, Duckabush, Union, and Hamma Hamma River subpopulations met the Technical Recovery Team (TRT) criteria for viability (and also the co-managers’ definition of “robust”). That is, spawning aggregates were present and persistent within 5 of the 6 major ecological diversity groups identified by the PSTRT. Two subpopulations previously considered extirpated (Skokomish and Dewatto Rivers) also rebounded with spawning aggregations despite not having reintroduction projects. An exception to the TRT criteria regarding distance between spawning aggregations is in East Hood Canal (West Kitsap). Spawning abundance in Big Beef Creek has remained consistently low (zero or near-zero for the last 3 years), and the Tahuya River spawning aggregate has not sustained adequate natural production after supplementation efforts ended. NMFS examined average escapements (geometric means) for 5-year intervals and estimated trends over the intervals for all natural spawners and for natural-origin only spawners. For both populations, abundance was relatively high in the 1970s, lowest for the period 1985-1999, and high again for 10 years leading up to 2015 (NWFSC 2015b). The overall trend in spawning abundance is generally stable for the Hood

Canal population (all natural spawners and natural-origin only spawners) and for the Strait of Juan de Fuca population (all natural spawners). Only the Strait of Juan de Fuca population's natural-origin only spawners show a significant positive trend. NMFS determined the abundance trends that appear to be positive occurred during a short time span between 1995-2009, and again recently from 2011 - 2015 is the Juan de Fuca population (NWFSC 2015b). Productivity rates, which were quite low during the 5-year period from 2005-2009 (Ford 2011), increased from 2011-2015 and were greater than replacement rates from 2014-2015 for both major population groups (NWFSC 2015b). Productivity had increased at the time of the last review (NWFSC 2015), but has been down for the last 3 years for the Hood Canal population, and for the last 4 years for the Strait of Juan de Fuca population. Productivity of individual spawning aggregates shows that only 2 of 8 aggregates have viable performance. Spatial structure and diversity viability parameters, as originally determined by the TRT, have improved, and nearly meet the viability criteria for both populations. Despite substantive gains toward meeting viability criteria in the Strait of Juan de Fuca and Hood Canal summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time. Overall, the Hood Canal summer-run chum salmon ESU therefore remains at "moderate" risk of extinction, with viability largely unchanged from the prior review.

Genetic Diversity. There were likely at least 2 ecological diversity groups within the Strait of Juan de Fuca population and at least 4 ecological diversity groups within the Hood Canal population. With the possible exception of the Dungeness River aggregation within the Strait of Juan de Fuca population, Hood Canal ESU summer chum spawning groups exist today that represent each of the ecological diversity groups within the 2 populations (NMFS 2017a). NMFS measured spatial distribution of the Hood Canal chum salmon ESU using the Shannon diversity index (NWFSC 2015b). Higher diversity values indicate a more uniform distribution of the population among spawning sites, which provides greater robustness to the population. Diversity values were generally lower in the 1990s for both independent populations within the ESU, indicating that most of the abundance occurred at a few spawning sites. Although the overall linear trend in diversity appears to be negative, the last 5-year interval shows the highest average value for both populations within the Hood Canal ESU. This results in part from the addition of 1 reintroduced spawning aggregation in the Strait of Juan de Fuca population and 2 reintroduced spawning aggregations in the Hood Canal population (NMFS 2017a).
Distribution. The Hood Canal summer-run chum salmon ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington. This ESU also includes 3 artificial propagation programs: Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery (5 other Hood Canal summer chum hatchery programs were terminated between 2005 and 2010 and are no longer part of the ESU).

Designated Critical Habitat. NMFS designated critical habitat for Hood Canal Summer-run chum salmon in 2005 (70 FR 52630). There are 12 watersheds within the range of this ESU. Three watersheds received a medium rating and 9 received a high rating of conservation value to the ESU (NMFS 2005a). Five nearshore marine areas also received a rating of high conservation value. Habitat areas for the Hood Canal Summer-run chum salmon include 88 mi (142 km) of

stream and 402 mi (647 km) of nearshore marine areas. PBFs considered essential for the conservation of the Hood Canal ESU of Chum salmon described in Appendix B.
The spawning PBF is degraded by excessive fine sediment in the gravel, and the rearing PBF is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Low river flows in several rivers also adversely affect most PBFs. In the estuarine areas, both migration and rearing PBFs of juveniles are impaired by loss of functional floodplain areas necessary for growth and development of juvenile chum salmon. These degraded conditions likely maintain low population abundances across the ESU.

Recovery Goals. The recovery strategy for Hood Canal Summer-run chum salmon focuses on habitat protection and restoration throughout the geographic range of the ESU, including both freshwater habitat and nearshore marine areas within a 1-mile radius of the watersheds' estuaries (NMFS 2007a). The recovery plan includes an ongoing harvest management program to reduce exploitation rates, a hatchery supplementation program, and the reintroduction of naturally spawning summer chum aggregations to several streams where they were historically present. The Hood Canal plan gives first priority to protecting the functioning habitat and major production areas of the ESU's 8 extant stocks, keeping in mind the biological and habitat needs of different life-history stages, and second priority to restoration of degraded areas, where recovery of natural processes appears to be feasible (HCCC 2005). For details on Hood Canal Summer-run chum salmon ESU recovery goals, including complete down-listing/delisting criteria, see the Hood Canal Coordinating Council 2005 recovery plan (HCCC 2005) and the NMFS 2007 supplement to this recovery plan (NMFS 2007a). Both independent populations (Strait of Juan de Fuca, Hood Canal) must have enough fish returning to meet abundance goals, distributed across the ESU to meet spatial structure goals in order to be considered recovered and removed from ESA listing.

8.5 Chinook salmon, California coastal ESU

Table 18. Chinook salmon, California coastal ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook salmon	California Coastal	Threatened	2016	70 FR 37160	2016	70 FR 52488

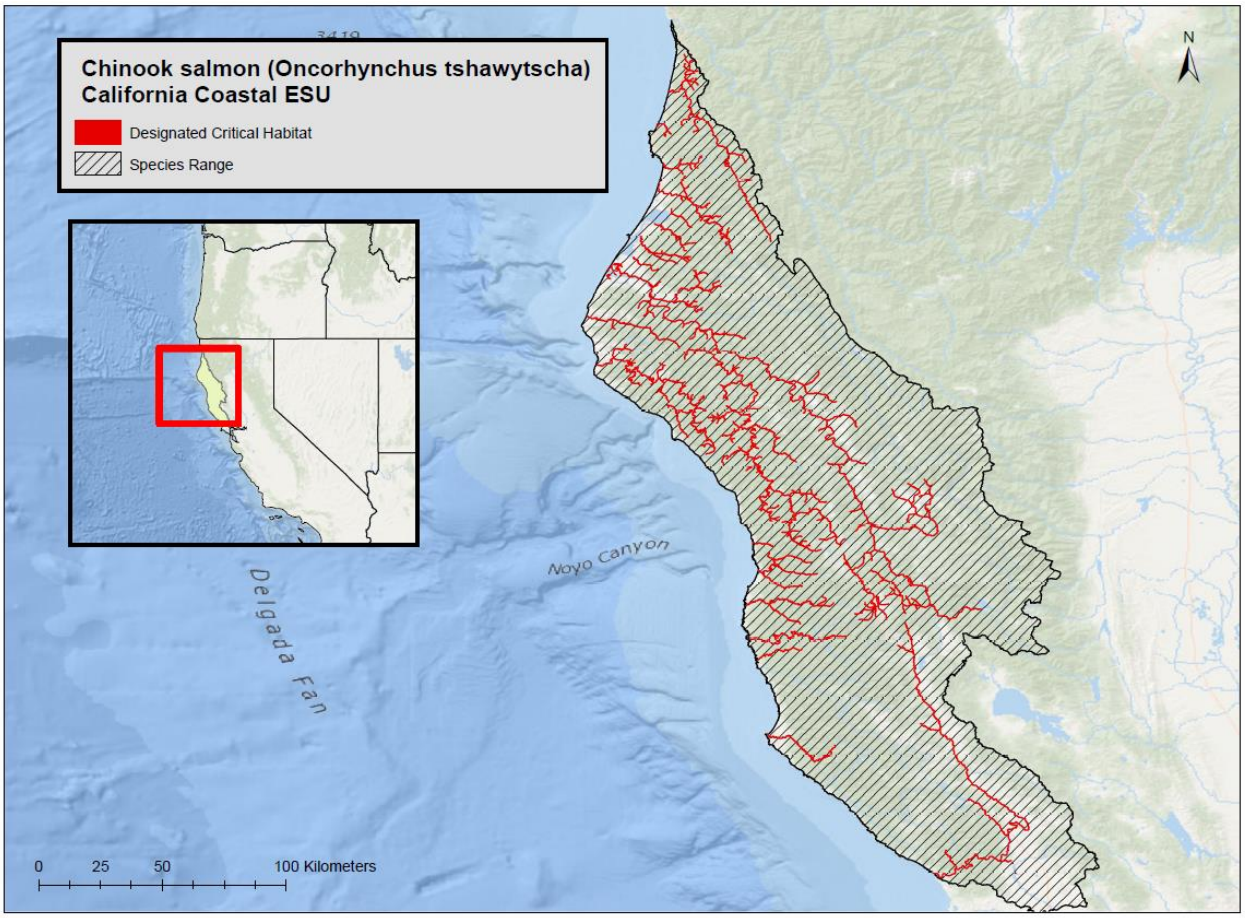

Figure 13. Chinook salmon, California coastal ESU range and designated critical habitat

Species Description. Chinook salmon, also referred to as king salmon, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002a). On September 16, 1999, NMFS listed the California coastal (CC) ESU of Chinook salmon as a threatened species (FR 64 50394). On June 28, 2005, NMFS confirmed the listing of CC Chinook salmon as threatened under the ESA and also added 7 artificially propagated populations from the following hatcheries or programs to the listing. The CC Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River (Humboldt County, CA.) to the Russian River (Sonoma County, CA) (70 FR 37160).

Status. The ESU was historically comprised of 38 populations which included 32 fall-run populations and 6 spring-run populations across 4 Diversity Strata (Spence et al. 2008b). All 6 of the spring-run populations were classified as functionally independent, but are considered extinct (Williams et al. 2011). Good et al. (2005a) cited continued evidence of low population sizes relative to historical abundance, mixed trends in the few available time series of abundance indices available, and low abundance and extirpation of populations in the southern part of the ESU. In addition, the apparent loss of the spring-run life history type throughout the entire ESU as a significant diversity concern. The 2016 recovery plan determined that the 4 threats of greatest concern to the ESU are channel modification, roads and railroads, logging and wood harvesting, and both water diversion and impoundments and severe weather patterns.

Life History. California coastal Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt et al. 2005). The different populations vary in run timing depending on latitude and hydrological differences between watersheds. Entry of California coastal Chinook salmon into the Russian River depends on increased flow from fall storms, usually in November to January. Juveniles of this ESU migrate downstream from April through June and may reside in the estuary for an extended period before entering the ocean.

The length of time required for embryo incubation and emergence from the gravel is dependent on water temperature. For maximum embryo survival, water temperatures reportedly must be between 41°F and 55.4°F and oxygen saturation levels must be close to maximum. Under those conditions, embryos hatch in 40 to 60 days and remain in the gravel as alevins (the life stage between hatching and egg sack absorption) for another 4 to 6 weeks before emerging as fry. Juveniles may reside in freshwater for 12 to 16 months, but some migrate to the ocean as young-of-the- year in the winter or spring months within 8 months of hatching.

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982b; MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 19. Temporal distribution of Chinook salmon, California coastal ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
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<tr><td>Rearing and migration (juveniles)</td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Population Dynamics

Abundance. Comparison of historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many basins (Bennet 2005; Good et al. 2005b; NMFS 2008a); only the Russian River currently has a run of any significance (Bjorkstedt et al. 2005). The 2000 to 2007 median observed (at Mirabel Dam) Russian River Chinook salmon run size is 2,991 with a maximum of 6,103 (2003) and a minimum of 1,125 (2008) adults (Cook 2008; Sonoma County Water Agency (SCWA) 2008).

Productivity / Population Growth Rate. The available data, a mixture of short-term (6-year or less) population estimates or expanded redd estimates and longer-term partial population estimates and spawner/red indexes, provide no indication that any of the independent populations (likely to persist in isolation) are approaching viability targets. Overall, there is a lack of compelling evidence to suggest that the status of these populations has improved or deteriorated appreciably since the previous status review back in 2011 (Williams et al. 2011).

Genetic Diversity. At the ESU level, the loss of the spring-run life history type represents a significant loss of diversity within the ESU, as has been noted in previous status reviews (Good et al. 2005b; Williams et al. 2011). Concern remains about the extremely low numbers of Chinook salmon in most populations of the North-Central Coast and Central Coast strata, which diminishes connectivity across the ESU. However, the fact that Chinook salmon have regularly been reported in the Ten Mile, Noyo, Big, Navarro, and Garcia rivers represents a significant improvement in our understanding of the status of these populations in watersheds where they were thought to have been extirpated. These observations suggest that spatial gaps between extant populations are not as extensive as previously believed.

Distribution. The California Coastal Chinook ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River to the Russian River, California (64 FR 50394; September 16, 1999). Seven artificial propagation programs are considered to be part of the ESU: The Humboldt Fish Action Council (Freshwater Creek), Yager Creek, Redwood Creek, Hollow Tree, Van Arsdale Fish Station, Mattole Salmon Group, and Mad River Hatchery fall-run Chinook hatchery programs. These artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the ESU (NMFS 2005a).

Designated Critical Habitat. NMFS designated critical habitat for the California coastal Chinook salmon on September 2, 2005 (70 FR 52488). It includes multiple CALWATER hydrological units north from Redwood Creek and south to Russian River. The total area of critical habitat includes 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. PBFs considered essential for the conservation of the California coastal ESU of Chinook salmon described in Appendix B.

There are 45 occupied CALWATER Hydrologic Subarea watersheds within the freshwater and estuarine range of this ESU. Eight watersheds received a low rating, 10 received a medium rating, and 27 received a high rating of conservation value to the ESU (70 FR 52488). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating. Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The current condition of PBFs of the California coastal Chinook salmon critical habitat indicates that PBFs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU.

Recovery Goals. Recovery goals, objectives and criteria for the CC Chinook salmon are fully outlined in the 2016 Recovery Plan(NMFS 2016d). Recovery plan objectives are to: 1. Reduce the present or threatened destruction, modification, or curtailment of habitat or range; 2. Ameliorate utilization for commercial, recreational, scientific, or educational purposes; 3. Abate disease and predation; 4. Establish the adequacy of existing regulatory mechanisms for protecting CC Chinook salmon now and into the future (i.e., post-delisting); 5. Address other natural or manmade factors affecting the continued existence of CC Chinook salmon; and 6. Ensure the status of CC Chinook salmon is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

8.6 Chinook salmon, Central Valley spring-run ESU

Table 20. Chinook salmon, Central Valley spring-run ESU; overview table

Species	Common Name	Distinct Population Segments (DPS)	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook Salmon	Central Valley Spring-run	Threatened	2016	1999 64 FR 50394 2014 79 FR 20802	2014	2005 70 FR 52630

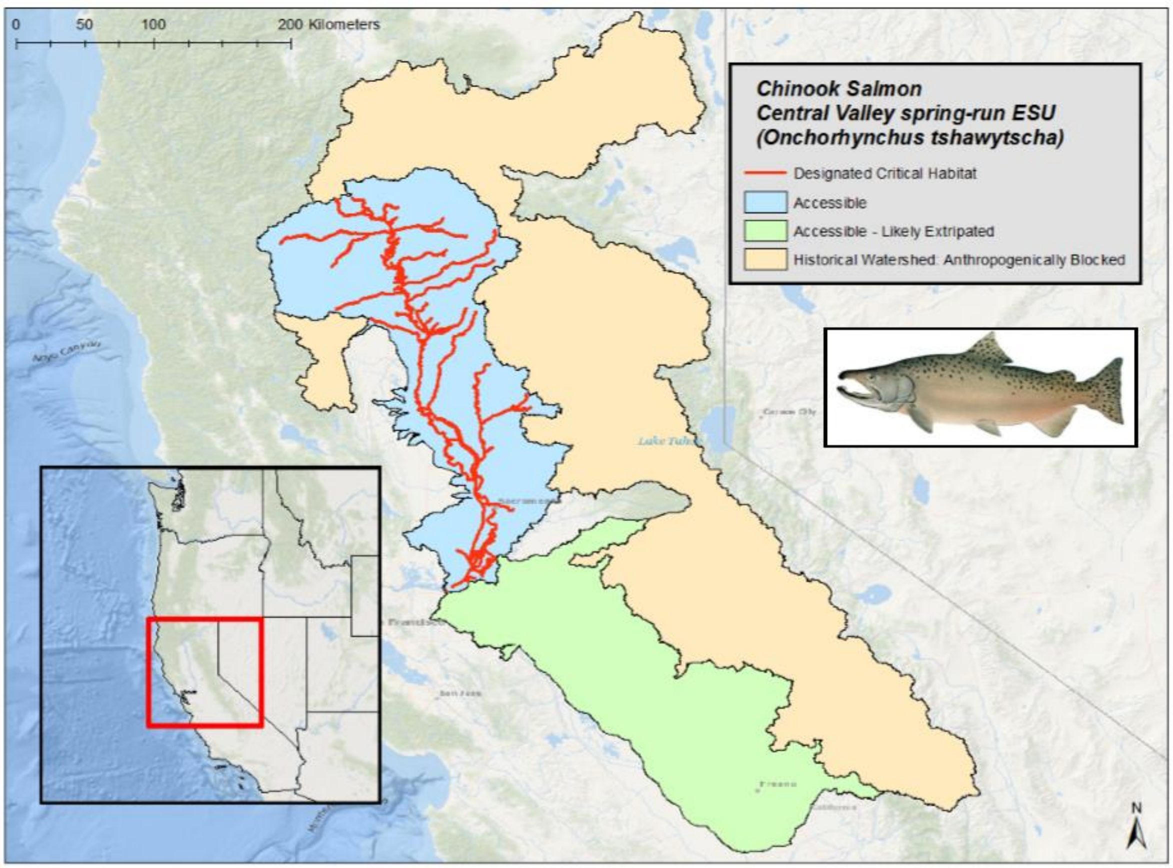

Figure 14. Chinook salmon, Central Valley spring-run ESU range and designated critical habitat

Species Description. On September 16, 1999, NMFS listed the Central Valley ESU of spring-run Chinook salmon as a threatened species (FR 64 50394). Historically, spring-run Chinook salmon occurred in the headwaters of all major river systems in the Central Valley where natural barriers to migration were absent. The only known streams that currently support self-sustaining populations of non-hybridized spring-run Chinook salmon in the Central Valley are Mill, Deer and Butte creeks. Each of these populations is small and isolated (NMFS 2014d).

Status. Although spring-run Chinook salmon were probably the most abundant salmonid in the Central Valley, this ESU has suffered the most severe declines of any of the 4 Chinook salmon runs in the Sacramento River Basin (Fisher 1994). The ESU is currently limited to independent populations in Mill, Deer, and Butte creeks, persistent and presumably dependent populations in the Feather and Yuba rivers and in Big Chico, Antelope, and Battle creeks, and a few ephemeral or dependent populations in the Northwestern California region (e.g., Beegum, Clear, and Thomes creeks). The Central Valley spring-run Chinook salmon ESU is currently faced with 3 primary threats: (1) loss of most historic spawning habitat; (2) degradation of the remaining habitat; and (3) genetic introgression with the Feather River fish hatchery spring-run Chinook salmon strays. The potential effects of climate change are likely to adversely affect spring-run Chinook salmon and their recovery (NMFS 2014d).

Life History. Adult Central Valley spring-run Chinook salmon leave the ocean to begin their upstream migration in late January and early February, and enter the Sacramento River between March and September, primarily in May and June (Moyle 2002a; Yoshiyama et al. 1998). Spring-run Chinook salmon generally enter rivers as sexually immature fish and must hold in freshwater for up to several months before spawning. While maturing, adults hold in deep pools with cold water. Spawning normally occurs between mid- August and early October, peaking in September (Moyle 2002a).

The length of time required for embryo incubation and emergence from the gravel is dependent on water temperature. For maximum embryo survival, water temperatures reportedly must be between 41°F and 55.4°F and oxygen saturation levels must be close to maximum. Under those conditions, embryos hatch in 40 to 60 days and remain in the gravel as alevins (the life stage between hatching and egg sack absorption) for another 4 to 6 weeks before emerging as fry. Spring-run fry emerge from the gravel from November to March (Moyle 2002a). Juveniles may reside in freshwater for 12 to 16 months, but some migrate to the ocean as young-of-the- year in the winter or spring months within 8 months of hatching.

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982b; MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 21. Temporal distribution of Chinook salmon, Central Valley spring-run ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
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<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. The Central Valley as a whole is estimated to have supported spring-run Chinook salmon runs as large as 600,000 returning spawners between the late 1880s and 1940s. The only known streams that currently support self-sustaining populations of nonhybridized spring-run Chinook salmon in the Central Valley are Mill, Deer and Butte creeks. Abundance and trend estimates for these streams as well as streams supporting dependent populations are provided in Table 22 (NMFS 2014d).

Table 22. Viability metrics for Central Valley spring-run ESU Chinook salmon populations.

Population	N	Ŝ	10-year trend 2005-2015 (95% CI)	Recent Decline (%)
Antelope Creek	8.0	2.7	-0.375 (-0.706, -0.045)	87.8
Battle Creek	1836	612	0.176 (0.033, 0.319)	9.0
Big Chico Creek	0.0	0.0	-0.358 (-0.880, 0.165)	60.7
Butte Creek	20169	6723	0.353 (-0.061, 0.768)	15.7
Clear Creek	822	274	0.010 (-0.311, 0.330)	63.3
Cottonwood Creek	4	1.3	-0.343 (-0.672, -0.013)	87.5
Deer Creek	2272	757.3	-0.089 (-0.337, 0.159)	83.8
Feather River Fish Hatchery	10808	3602.7	0.082 (-0.015, 0.179)	17.1
Mill Creek	2091.0	697.0	-0.049 (-0.183, 0.086)	58.0
Sacramento River[a]	-	-	-	-
Yuba River	6515	2170.7	0.67 (-0.138, 0.272)	9.0

N: Total population size (N) is estimated as the sum of estimated run sizes over the most recent 3 years for Core 1 populations (bold) and Core 2 populations.
Ŝ: The mean population size (Ŝ) is the average of the estimated run sizes for the most recent 3 years (2012 to 2014).
Population growth/decline rate (10 year trend) is estimated from the slope of log-transformed estimated run size.
The catastrophic metric (recent decline) is the largest year-to-year decline in total population size (N) over the most recent 10 such ratios.
[a] *Beginning in 2009, estimates of spawning escapement of Upper Sacramento River spring chinook were no longer monitored.*

Productivity / Population Growth Rate. Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation. The majority of Central Valley (CV) spring-run Chinook salmon are found to return as 3-year olds, therefore looking at returns every 3 years is used as an estimate of the CRR. In the past the CRR has fluctuated between just over 1.0 to just under 0.5, and in the recent years with high returns (2012 and 2013), CRR jumped to 3.84 and 8.68 respectively. CRR for 2014 was 1.85, and the CRR for 2015 with very low returns was a record low of 0.14. Low returns in 2015 were further decreased due to high temperatures and most of the CV spring-run Chinook salmon tributaries experienced some pre-spawn mortality. Butte Creek experienced the highest prespawn mortality in 2015, resulting in a carcass survey CRR of only 0.02.

Genetic Diversity. Threats to the genetic integrity of spring-run Chinook salmon was identified as a serious concern to the species when it was listed in 1999 (FR 64 50394; Myers et al. 1998a). Three main factors compromised the genetic integrity of spring-run Chinook salmon: (1) the lack of reproductive isolation following dam construction throughout the Central Valley resulting in introgression with fall-run Chinook salmon in the wild; (2) within basin and inter-basin mixing between spring and fall broodstock for artificial propagation, resulting in introgression in hatcheries; and (3) releasing hatchery-produced juvenile Chinook salmon in the San Francisco estuary, which contributes to the straying of returning adults throughout the Central Valley (NMFS 2014d).

Distribution. The Central Valley Technical Recovery Team delineated 18 or 19 historic independent populations of CV spring-run Chinook salmon, and a number of smaller dependent populations, that are distributed among 4 diversity groups (southern Cascades, northern Sierra, southern Sierra, and Coast Range) (Lindley et al. 2004). Of these independent populations, only 3 are extant (Mill, Deer, and Butte creeks) and they represent only the northern Sierra Nevada diversity group. Of the dependent populations, CV spring-run Chinook salmon are found in Battle, Clear, Cottonwood, Antelope, Big Chico, and Yuba creeks, as well as the Sacramento and Feather rivers and a number of tributaries of the San Joaquin River including Mokelumne, Stanislaus, and Tuolumne rivers. The 2005 listing determination concluded that the Feather River Fish Hatchery spring-run Chinook salmon production should be included in the Central Valley spring-run Chinook salmon ESU (79 FR 20802; NMFS 2016a).

Designated Critical Habitat NMFS published a final rule designating critical habitat for Central Valley spring-run Chinook on September 2, 2005 (70 FR 52488). The designated critical habitat includes 1,853 km (1,158 mi) of streams and 655 km^2 (254 km^2) of estuarine habitat. PBFs considered essential for the conservation of the Central Valley spring-run ESU of Chinook salmon are described in Appendix B.

The current condition of PBFs of the CV Spring-run Chinook salmon critical habitat indicates that PBFs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. Spawning and rearing PBFs are degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds which maintained cool and clean water throughout the summer. The rearing PBF is degraded by floodplain habitat being disconnected from the mainstem of larger rivers throughout the Sacramento River watershed, thereby reducing effective foraging. The migration PBF is degraded by lack of natural cover along the migration corridors. Juvenile migration is obstructed by water diversions along Sacramento River and by 2 large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Recovery Goals. Recovery goals, objectives and criteria for the Central Valley spring-run Chinook are fully outlined in the 2014 Recovery Plan (NMFS 2014d). The ESU delisting criteria for the spring-run Chinook are: 1) 1 population in the Northwestern California Diversity Group at low risk of extinction; 2) 2 populations in the Basalt and Porous Lava Diversity Group at low risk of extinction; 3) 4 populations in the Northern Sierra Diversity Group at low risk of extinction; 4) 2 populations in the Southern Sierra Diversity Group at low risk of extinction; and 5) Maintain multiple populations at moderate risk of extinction.

8.7 Chinook salmon, Lower Columbia River ESU

Table 23. Chinook salmon, Lower Columbia River ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook Salmon	Lower Columbia River ESU	Threatened	2022	70 FR 37160	2013	70 FR 52630

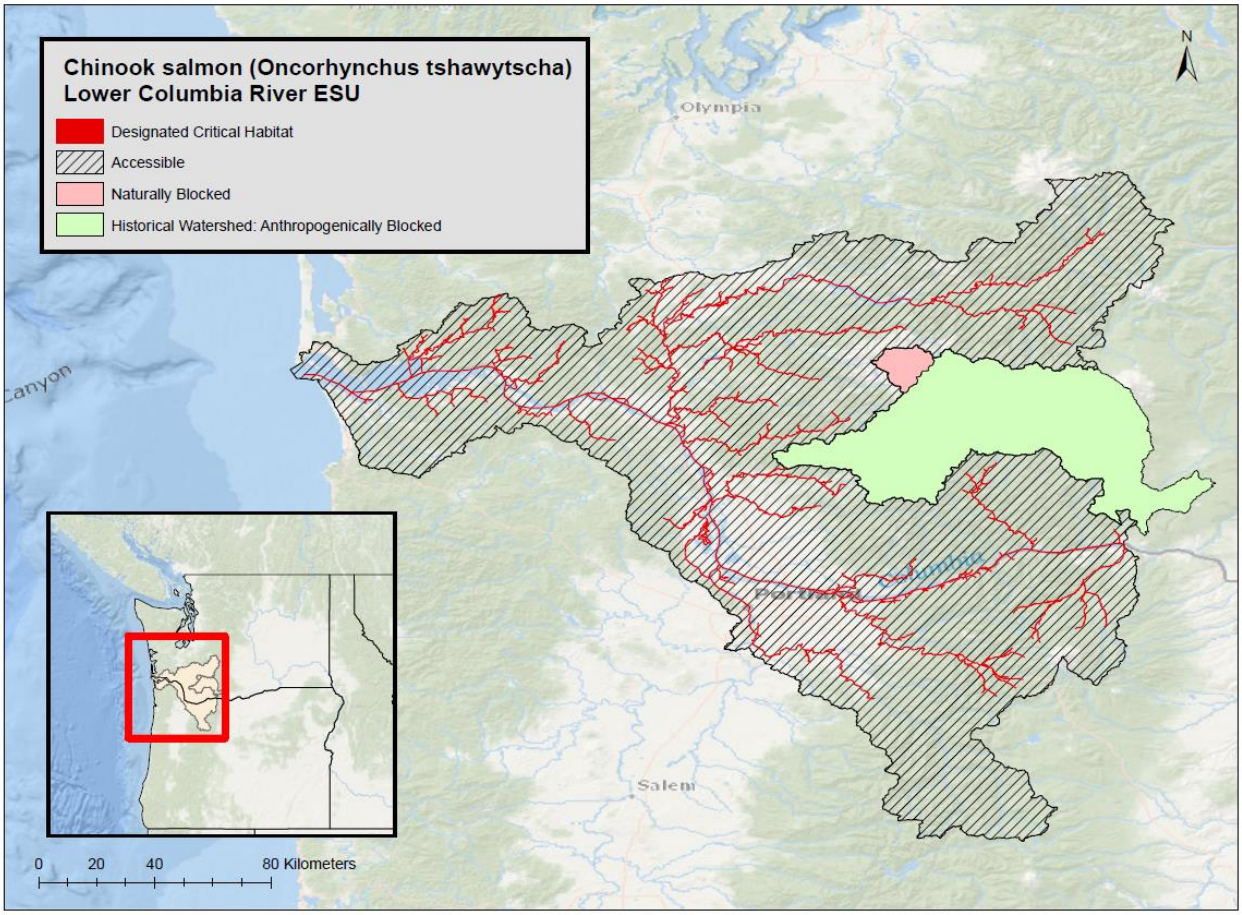


Figure 15. Chinook salmon, Lower Columbia River ESU range and designated critical habitat

Species Description. Chinook salmon, also referred to as king salmon in California, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002a).

On March 24, 1999, NMFS listed the Lower Columbia River ESU of Chinook salmon as a threatened species (64 FR 14308). The listing was revisited and confirmed as threatened in 2005 (70 FR 37160). The Lower Columbia River Chinook salmon ESU reaffirmed in the 2022 Status Update, includes all naturally-spawned populations of fall-run and spring-run Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Oregon and Washington, east of the Hood River and the White Salmon River and any such fish originating from the Willamette River and its tributaries below Willamette Falls. Twenty artificial propagation programs are included in the ESU (70 FR 37160). The Northwest Fisheries Science Center's review (Ford 2022) found that no new information has become available that would justify a change in the delineation of the LCR Chinook salmon ESU.

Status. The 2022 update determined that no reclassification for the LCR Chinook salmon ESU is warranted. Therefore, the LCR Chinook salmon ESU remains listed as threatened. Although many of the populations in this ESU are at high risk, it is important to note the poor ocean and freshwater conditions existed during the 2015-2019 period and despite these conditions the status of a number of populations improved, some remarkably so (Grays River, Lower Cowlitz River, and Kalama River fall runs). Overall, the viability of the Lower Columbia River Chinook Salmon ESU has increased somewhat since the last 5-year review, although the ESU remains at moderate risk of extinction (Ford 2022). Populations of Lower Columbia River Chinook salmon have declined substantially from historical levels. Out of the 32 populations that make up this ESU, only the late-fall run Sandy River run is considered viable. Of the 7 spring-run DIPs in this MPG, there are abundance estimates for the Upper Cowlitz/Cispus Rivers (2 DIPs combined), Kalama River, North Fork Lewis River, and Sandy River populations. Of these, only the Sandy River population appears to be sustaining natural-origin abundance at near-recovery levels. The most-recent 5-year geomean abundance for the Sandy River was 3,359, which represents an 89% increase over 2010–14. Most populations (26 out of 32) have a very low probability of persistence over the next 100 years and some are extirpated or nearly so. Five of the 6 strata fall significantly short of the recovery plan criteria for viability. Low abundance, poor productivity, losses of spatial structure, and reduced diversity all contribute to the very low persistence probability for most Lower Columbia River Chinook salmon populations. All of the spring-run populations except Sandy River exhibited a recent uniform decline, possibly related to climatic and oceanic conditions (Tables 28 and 29, Figure 52). Elsewhere in this MPG natural-origin abundances for spring-run Chinook salmon were very low, with negative trends. For the Upper Cowlitz/Cispus Rivers, Kalama River, and North Fork Lewis River populations, hatchery returns currently constitute the vast majority of fish returning to the river. Hatchery contribution to naturally-spawning fish remains high for a number of populations, and it is likely that many returning unmarked adults are the progeny of hatchery origin parents, especially where large hatchery programs operate. Continued land development and habitat degradation in combination with the potential effects of climate change will present a continuing strong negative influence into the foreseeable future. Based on the 2022 Status Update, no reclassification for the LCR Chinook Salmon ESU is warranted. Therefore, the LCR Chinook Salmon ESU remain listed as threatened.

Life History. Lower Columbia River Chinook salmon display 3 run types including early fall-runs, late fall-runs, and spring-runs. Presently, the fall-run is the predominant life history type.

Spring-run Chinook salmon were numerous historically. Fall-run Chinook salmon enter fresh water typically in August through October. Early fall-run spawn within a few weeks in large river mainstems. The late fall-run enters in immature conditions, has a delayed entry to spawning grounds, and resides in the river for a longer time between river entry and spawning. Spring-run Chinook salmon enter fresh water in March through June to spawn in upstream tributaries in August and September.

Offspring of fall-run spawning may migrate as fry to the ocean soon after yolk absorption (*i.e.*, ocean-type), at 30–45 mm in length (Healey 1991). In the Lower Columbia River system, however, the majority of fall-run Chinook salmon fry migrate either at 60-150 days post-hatching in the late summer or autumn of their first year. Offspring of fall-run spawning may also include a third group of yearling juveniles that remain in fresh water for their entire first year before emigrating. The spring-run Chinook salmon migrates to the sea as yearlings (stream-type) typically in spring. However, the natural timing of LCR spring-run Chinook salmon emigration is obscured by hatchery releases (Myers et al. 2006). Once at sea, the ocean-type LCR Chinook salmon tend to migrate along the coast, while stream-type LCR Chinook salmon appear to move far off the coast into the central North Pacific Ocean (Healey 1991; Myers et al. 2006). Adults return to tributaries in the lower Columbia River predominantly as 3- and 4-year-olds for fall-run fish and 4- and 5-year-olds for spring-run fish.

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982b; MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 24. Temporal distribution of Chinook salmon, Lower Columbia River ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td colspan="10">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Emergence (alevin to fry phases</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Populations of Lower Columbia River Chinook salmon have declined substantially from historical levels. Many of the ESU's populations are believed to have very low abundance of natural-origin spawners (100 adult spawners or fewer), which increases genetic and demographic risks. Other populations have higher total abundance, but several of these also have high proportions of hatchery-origin spawners.

Productivity / Population Growth Rate. Trend indicators for most populations are negative. The majority of populations for which data are available have a long-term trend of <1; indicating the population is in decline (Bennet 2005; Good et al. 2005b). Only the Sandy River population appears to be sustaining natural-origin abundance at near-recovery levels. The most-recent 5-year geomean abundance for the Sandy River was 3,359, which represents an 89% increase over 2010–14. The late-fall run of the Sandy River seems to be gone (NMFS 2022a).

Genetic Diversity. The genetic diversity of all populations (except the late fall-run Chinook salmon) has been eroded by large hatchery influences and periodically by low effective population sizes. The near loss of the spring-run life history type remains an important concern for maintaining diversity within the ESU.

Distribution. The basin wide spatial structure has remained generally intact. However, the loss of about 35% of historic habitat has affected distribution within several Columbia River subbasins. Currently, only 1 population appears self-sustaining (Good et al. 2005b).

Designated Critical Habitat. NMFS designated critical habitat for LCR Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins. PBFs considered essential for the conservation of Chinook salmon, Lower Columbia River ESU are described in Appendix B.

Timber harvest, agriculture, and urbanization have degraded spawning and rearing PBFs by reducing floodplain connectivity and water quality, and by removing natural cover in several rivers. Hydropower development projects have reduced the timing and magnitude of water flows, thereby altering the water quantity needed to form and maintain physical habitat conditions and support juvenile growth and mobility. Adult and juvenile migration PBFs are affected by several dams along the migration route.

Recovery Goals. NMFS has developed the following delisting criteria for the Lower Columbia River Chinook salmon ESU (NMFS 2013a) . For a complete description of the ESU recovery goals, including complete down-listing/delisting criteria, see the 2013 recovery plan.
All strata that historically existed have a high probability of persistence or have a probability of persistence consistent with their historical condition. High probability of stratum persistence is defined as:

- At least 2 populations in the stratum have at least a 95% probability of persistence over a 100-year time frame (i.e., 2 populations with a score of 3.0 or higher based on the Technical Recovery Team's (TRT) scoring system).
- Other populations in the stratum have persistence probabilities consistent with a high probability of stratum persistence (i.e., the average of all stratum population scores is 2.25 or higher, based on the TRT's scoring system).
- Populations targeted for a high probability of persistence are distributed in a way that minimizes risk from catastrophic events, maintains migratory connections among populations, and protects within-stratum diversity.

- A probability of persistence consistent with historical condition refers to the concept that strata that historically were small or had complex population structures may not have met Criteria A through C, above, but could still be considered sufficiently viable if they provide a contribution to overall ESU viability similar to their historical contribution.

Recommendations for Future Actions. The 2022 Status Update recommended for all populations and all MPGs that comprise the LCR Chinook salmon, future recovery actions over the next 5 years include:

- Conduct systematic review and analysis of high priority Lower Columbia River mainstem and tributary area habitat needs, identified in NMFS 2013a, and compare needs to what has been accomplished.
- Conduct monitoring to evaluate ship wake stranding frequency and locations where stranding occurs and assess factors contributing to wake stranding such as location, topography, vessel speed, et cetera, to determine best practices to reduce wake stranding mortality.
- Promote riparian plantings of native canopy tree cover species opportunistically in all watersheds.
- Coordinate with EPA in an evaluation of Washington State Water Quality Standards, reflecting Oregon and Idaho consultation outcomes.
- Increase the number of habitat projects that target fall Chinook salmon spawning (Big Creek, Elochoman/Skamokawa, Clatskanie River, Mill/Abernathy/Germany Creek, Toutle River, and Hood River).

8.8 Chinook salmon, Puget Sound ESU

Table 25. Chinook salmon, Puget Sound ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook salmon	Puget Sound ESU	Threatened	2022	70 FR 37160	2007	70 FR 52630

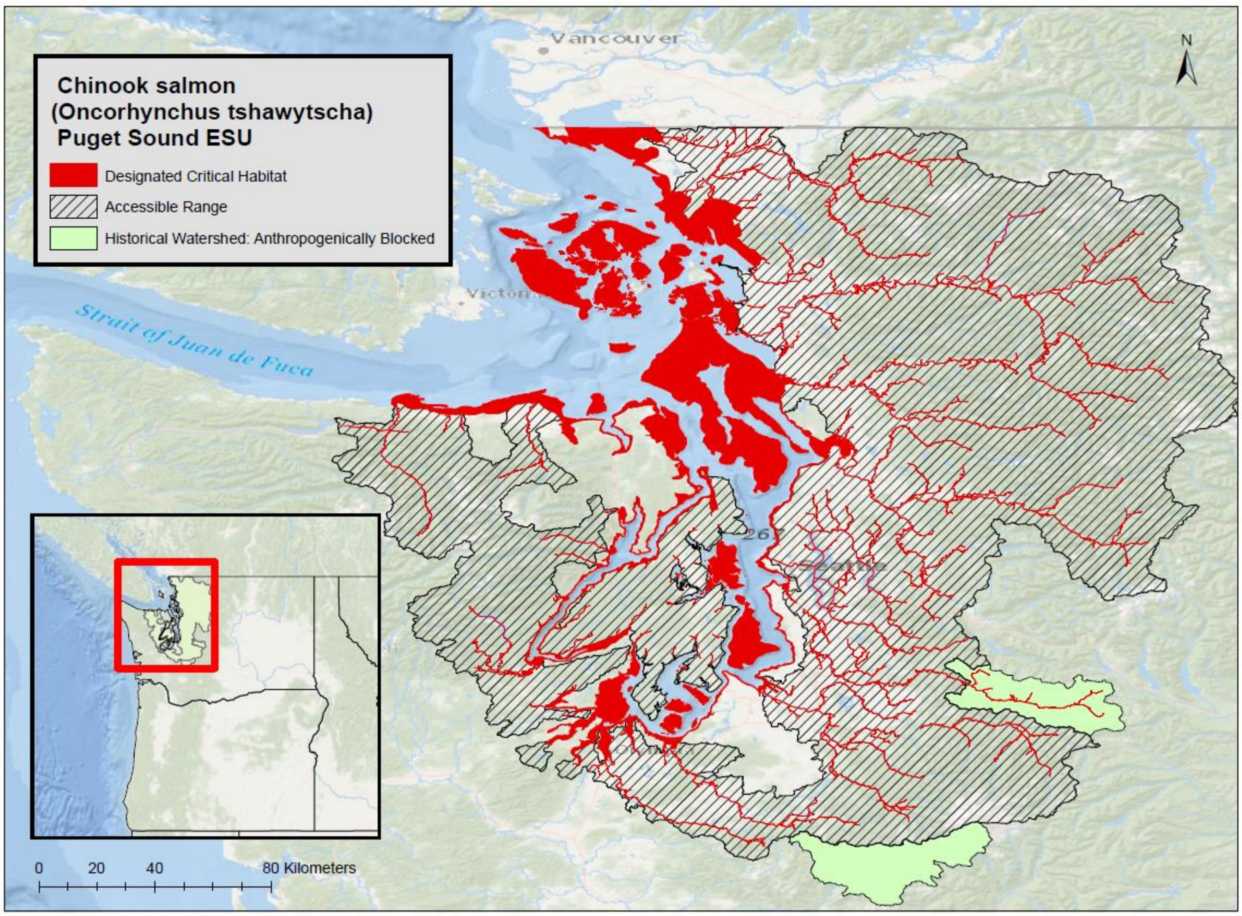


Figure 16. Chinook salmon, Puget Sound ESU range and designated critical habitat

Species Description. Chinook salmon, also referred to as king salmon in California, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002a). On March 24, 1999, NMFS listed the Puget Sound ESU of Chinook salmon as a threatened species (64 FR 14308). The listing was revisited and confirmed as “threatened” in 2005 (70 FR 37160). The Puget Sound ESU includes naturally spawned Chinook salmon originating from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia. Twenty-six artificial propagation programs are included as part of the ESU.

Status. All Puget Sound Chinook salmon populations are well below escapement abundance levels identified as required for recovery to low extinction risk in the recovery plan. In addition, most populations are consistently below the productivity goals identified in the recovery plan as necessary for recovery. Although trends vary for individual populations across the ESU, most populations have declined in total natural origin recruit abundance since the last status review; and natural origin recruit escapement trends since 1995 are mostly stable. A few populations have reached goals but not consistently during the past 10 years (2018 Washington State of the Salmon Report). While some have met their high productivity goals, but never their low (minimum) productivity goals, none of the Puget Sound populations of Chinook salmon could be considered exceeding their abundance recovery goals. Several of the risk factors identified in an early status review (Good et al. 2005b) are still present, including high fractions of hatchery fish in many populations and widespread loss and degradation of habitat. Although this ESU's total abundance is greatly reduced from historic levels, recent abundance levels do not indicate that the ESU is at immediate risk of extinction. This ESU remains relatively well distributed over 22 populations in 5 geographic areas across the Puget Sound. Although current trends are concerning, the available information indicates that this ESU remains at moderate risk of extinction.

Life History. Puget Sound Chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) Chinook salmon spawners (Healey 1991). Juvenile Chinook salmon within the Puget Sound generally exhibit an "ocean-type" life history. However, substantial variation occurs with regard to juvenile residence time in freshwater and estuarine environments. Hayman(Hayman et al. 1996) described 3 juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20% to 60% of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer et al. 2005). Juveniles in tributaries to Lake Washington exhibit both a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (<1 meter) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor et al. 2006). Puget Sound Chinook salmon also has several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing (Beamer et al. 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature Chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan et al. 2004).

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1981; MacFarlane and Norton 2002; Sommer et al. 2001a). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991;

MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 26. Temporal distribution of Chinook salmon, Puget Sound ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td colspan="8">Present</td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="6">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td colspan="6">Present</td></tr>
<tr><td>Emergence (alevin to fry phase)</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Estimates of the historic abundance range from 1,700 to 51,000 Puget Sound Chinook salmon spawners per population. During the period from 1996 to 2001, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 adult spawners. Thus, the historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005b). Generally, many populations experienced increases in total abundance during the years 2000–08, and more recently in 2015–17, but general declines during 2009–14, and a downturn again in the 2 most-recent years, 2017–18 (Figure 90). Abundance across the Puget Sound Chinook salmon ESU has generally increased since the last status review, with only 2 of the 22 populations (Cascade River and North and South Fork Stillaguamish Rivers) showing a negative percentage change in the 5-year geometric mean natural-origin spawner abundances since the prior status review (Table 50). Fifteen of the remaining 20 populations with positive percentage changes since the prior status review have relatively low natural spawning abundances (<1,000 fish), so some of these increases represent small changes in total abundance. Given lack of high confidence in survey techniques, particularly with small populations, there remains substantial uncertainty in detecting trends in small populations.

Table 27. Past Puget Sound Chinook salmon population structure, abundances, and hatchery contributions (Good et al. 2005).

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions (%)
Nooksack-North Fork	26,000	1,538	91
Nooksack-South Fork	13,000	338	40
Lower Skagit	22,000	2,527	0.2
Upper Skagit	35,000	9,489	2
Upper Cascade	1,700	274	0.3
Lower Sauk	7,800	601	0

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions (%)
Upper Sauk	4,200	324	0
Suiattle	830	365	0
Stillaguamish-North Fork	24,000	1,154	40
Stillaguamish-South Fork	20,000	270	Unknown
Skykomish	51,000	4,262	40
Snoqualmie	33,000	2,067	16
Sammamish	Unknown	Unknown	Unknown
Cedar	Unknown	327	Unknown
Duwamish/Green			
Green	Unknown	8,884	83
White	Unknown	844	Unknown
Puyallup	33,000	1,653	Unknown
Nisqually	18,000	1,195	Unknown
Skokomish	Unknown	1,392	Unknown
Mid Hood Canal Rivers			
Dosewallips	4,700	48	Unknown
Duckabush	Unknown	43	Unknown
Hamma Hamma	Unknown	196	Unknown
Mid Hood Canal	Unknown	311	Unknown
Dungeness	8,100	222	Unknown
Elwha	Unknown	688	Unknown

Productivity / Population Growth Rate. Fifteen-year trends in log natural-origin spawner abundance were computed over 2 time periods (1990–2005 and 2004–19) for each Puget Sound Chinook salmon population. Trends were negative for 4 of the populations in the earlier period, and for 16 of the 22 populations in the later period. Thus, there is a general decline in natural-origin spawner abundance across all MPGs in the most-recent 15 years. Upper Sauk and Suiattle Rivers (Whidbey Basin MPG), Nisqually River (Central/South Sound MPG), and Mid-Hood Canal (Hood Canal MPG) are the only populations with positive trends, though Mid-Hood Canal has an extremely low population size. Further, no change in trend between the 2 time periods was detected in South Fork Nooksack River (Strait of Georgia MPG) or Green and Nisqually Rivers (Central/South MPG). The average trend in population growth rate across the ESU for 1990–2005 was a positive 3%. The average trends for the MPGs are: Strait of Georgia, up 3%; Whidbey Basin, up 4%; Central/South Sound, up 4%; Hood Canal, 3%; and Strait of Juan de Fuca, up 1%. However the average trend across the ESU declined between 2004 –2019 at minus 2%. The average trends for the MPGs are: Strait of Georgia, minus 2%; Whidbey Basin, minus 2%; Central/South Sound, minus 2%; Hood Canal, minus 2%; and Strait of Juan de Fuca, minus 8%. The previous status review (NWFSC 2015) concluded that there were widespread negative trends for the total ESU, despite variable escapements and trends for individual populations. The addition of the data to 2019 now shows even more substantially either flat or negative trends for the entire ESU in natural-origin Chinook salmon spawner population abundances. Across the Puget Sound Chinook salmon ESU, 10 of 22 Puget Sound populations show natural productivity below replacement in nearly all years since the mid-1980s. However, the median overall long-term trend in abundance is close to 1 for most populations that have a lambda exceeding 1, indicating that most of these populations are barely replacing themselves. In recent years, only 5 populations have had productivities above zero. These are Lower and Upper Skagit, Lower and Upper Sauk, and Suiattle Rivers in the Whidbey Basin MPG. This is consistent with, and continues the decline reported in, the 2015 status review (NWFSC 2015).

Genetic Diversity / Spatial Distribution. The Northwest Fisheries Science Center estimated the diversity index for 5 year time intervals over the 25 year time span of the available data. In general, a higher diversity value indicates a healthier distribution of salmon among the streams and rivers in the ESU. Current estimates of diversity show a decline over the past 25 years, indicating a decline of salmon in some areas and increases in others. Salmon returns to the Whidbey Region increased in abundance while returns to other regions declined. In aggregate, the diversity of the ESU as a whole has been declining over the last 25 years and there is a concern that some populations are no longer distinct (NMFS 2022a).

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes 1,683 km of stream channels, 41 square km of lakes, and 3,512 km of nearshore marine habitat. PBFs considered essential for the conservation of Chinook salmon, Puget Sound ESU are described in Appendix B.

Forestry practices have heavily impacted migration, spawning, and rearing PBFs in the upper watersheds of most river systems within critical habitat designated for the Puget Sound Chinook salmon. Degraded PBFs include reduced conditions of substrate supporting spawning, incubation and larval development caused by siltation of gravel; and degraded rearing habitat by removal of cover and reduction in channel complexity. Urbanization and agriculture in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have reduced channel function and connectivity, reduced available floodplain habitat, and affected water quality. Thus, these areas have degraded spawning, rearing, and migration PBFs. Hydroelectric development and flood control also obstruct Puget Sound Chinook salmon migration in several basins. The most functional PBFs are found in northwest Puget Sound: the Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlaps with critical habitat designated for the Puget Sound Chinook salmon. However, estuary PBFs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels.

Recovery Goals. The ESU-wide delisting and recovery criteria (PSTRT, 2002) provide flexibility in meeting the requirements of the Endangered Species Act, and preserve options for Puget Sound Chinook in the future. The recommendations by the TRT describe the biological characteristics that would constitute a viable ESU for Puget Sound Chinook. The ESU would have a high likelihood of persistence if:

- All populations improve in status and at least some achieve a low risk status.
- At least 2-4 viable Chinook populations are present in each of the 5 regions.
- Each region has 1 or more viable populations from each major diversity group that was historically present within that region.
- Freshwater tributary habitats in Puget Sound are providing sufficient function for ESU persistence. Ecological functioning occurs even in those habitats that do not currently support any of the 22 identified Chinook populations, since they affect nearshore processes and may provide future habitat options.
- The production of Chinook salmon in Puget Sound tributaries is consistent with ESU recovery objectives, and contributes to the health of the overall ecosystem in the region.
- None of the 22 remaining Chinook populations go extinct, and the direct and indirect effects of habitat, harvest and hatchery management actions are consistent with ESU recovery.

8.9 Chinook salmon, Sacramento River winter-run ESU

Table 28. Chinook salmon, Sacramento winter-run ESU; overview table

Species	Common Name	Distinct Population Segments (DPS)	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook Salmon	Sacramento River winter-run	Endangered	2016	1990 54 FR 32085 1994 59 FR 440	2014	1993 58 FR 33212

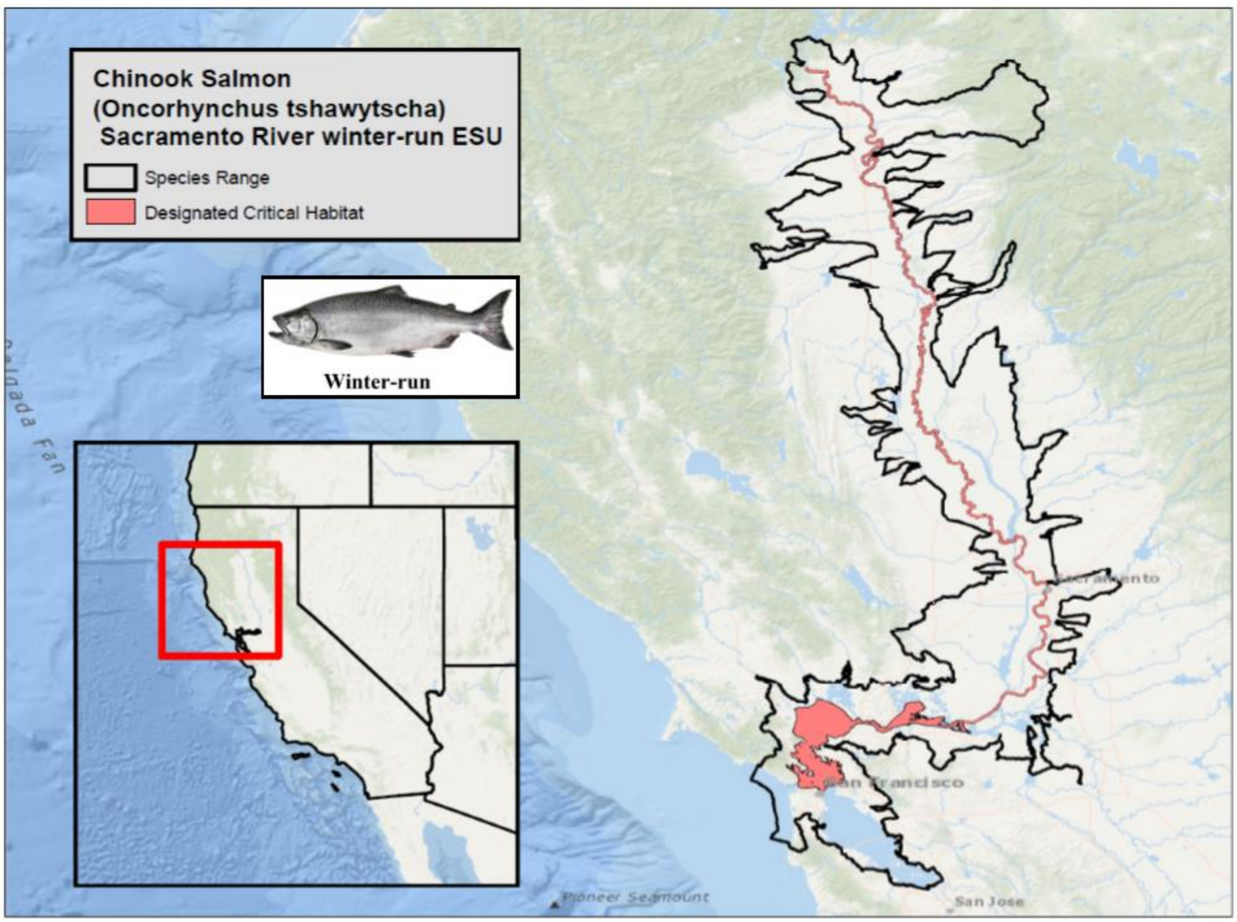


Figure 17. Chinook salmon, Sacramento winter-run ESU range and designated critical habitat

Species Description. Chinook salmon, also referred to as king salmon in California, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002a). On January 4, 1994, NMFS listed the Sacramento River winter-run ESU of Chinook salmon as Endangered (59 FR 440). The Sacramento River winter-run Chinook salmon ESU includes winter-run Chinook salmon spawning naturally in the Sacramento River and its tributaries, as well as winter-run Chinook salmon that are part of the conservation hatchery program at the Livingston Stone National Fish Hatchery (LSNFH). Winter-run Chinook salmon originally spawned in the upper Sacramento River system (Little Sacramento, Pit, McCloud and Fall rivers) and in Battle Creek (Yoshiyama et al. 1998; Yoshiyama et al. 2001). Currently, winter-run Chinook salmon spawning habitat is likely limited to the reach of the Sacramento River extending from Keswick Dam downstream to the Red Bluff Diversion Dam.

Status. The Sacramento River winter-run Chinook salmon ESU is composed of just 1 small population that is currently under severe stress caused by 1 of California's worst droughts on record. Over the last 10 years of available data (2003-2013), the abundance of spawning winter-run Chinook adults ranged from a low of 738 in 2011 to a high of 17,197 in 2007, with an average of 6,298. The population subsists in large part due to agency-managed cold water releases from Shasta Reservoir during the summer and artificial propagation from Livingston Stone National Fish Hatchery's winter-run Chinook salmon conservation program. Winter-run Chinook salmon are dependent on sufficient cold water storage in Shasta Reservoir, and it has long been recognized that a prolonged drought could have devastating impacts, possibly leading to the species' extinction. The probability of extended droughts is increasing as the effects of climate change continue (NMFS 2014d). In addition to the drought, another important threat to winter-run Chinook salmon is a lack of suitable rearing habitat in the Sacramento River and Delta to allow for sufficient juvenile growth and survival (NMFS 2016g).

Life History. Winter-run Chinook salmon are unique because they spawn during summer months when air temperatures usually approach their yearly maximum. As a result, winter-run Chinook salmon require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in summer. Adult winter-run Chinook salmon immigration and holding (upstream spawning migration) through the Delta and into the lower Sacramento River occurs from December through July, with a peak during the period extending from January through April (Fish and Service 1995). Winter-run Chinook salmon are sexually immature when upstream migration begins, and they must hold for several months in suitable habitat prior to spawning. Spawning occurs between late-April and mid-August, with a peak in June and July as reported by California Department of Fish and Wildlife (CDFW) annual escapement surveys (2000-2006).

Winter-run Chinook salmon embryo incubation in the Sacramento River can extend into October (Vogel et al. 1988). Winter-run Chinook salmon fry rearing in the upper Sacramento River exhibit peak abundance during September, with fry and juvenile emigration past Red Bluff Diversion Dam (RBDD) primarily occurring from July through November (Poytress and Carrillo 2010; Poytress and Carrillo 2011; Poytress and Carrillo 2012). Emigration of winter-run Chinook salmon juveniles past Knights Landing, located approximately 155.5 river miles downstream of the RBDD, reportedly occurs between November and March, peaking in December, with some emigration continuing through May in some years (Snider and Titus 2000).

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982a; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 29. Temporal distribution of Chinook salmon, Sacramento winter-run ESU in freshwater habitats

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="6">Present</td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td colspan="6">Present</td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
</table>

Population Dynamics

Abundance. Over the last 10 years of available data (2003-2013), the abundance of spawning winter-run Chinook adults ranged from a low of 738 in 2011 to a high of 17,197 in 2007, with an average of 6,298 (Figure 18).

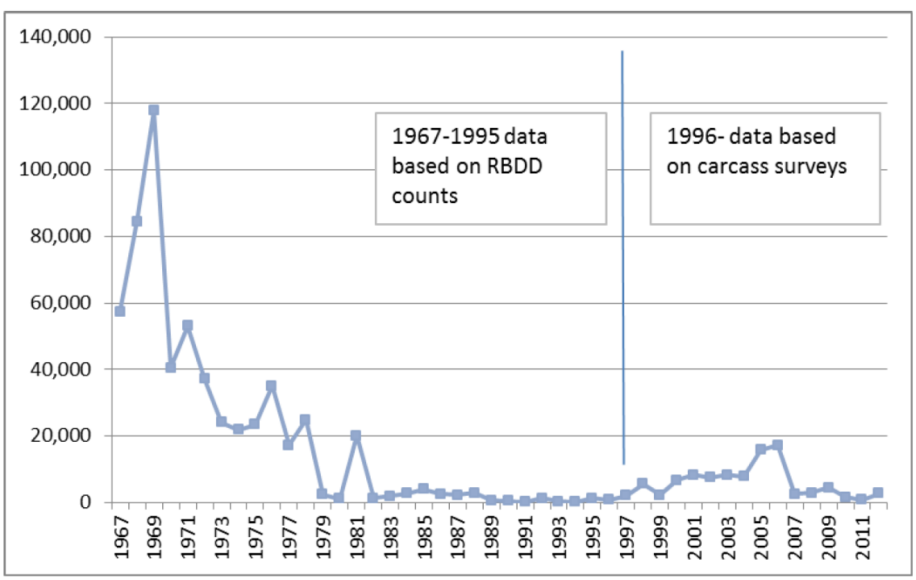


Figure 18. Estimated Sacramento River winter-run Chinook salmon run size (1967-2012)

Productivity / Population Growth Rate. The population declined from an escapement of near 100,000 in the late 1960s to fewer than 200 in the early 1990s (Good et al. 2005a). More recent population estimates of 8,218 (2004), 15,730 (2005), and 17,153 (2006) show a 3-year average of 13,700 returning winter-run Chinook salmon (CDFW Website 2007). However, the run size decreased to 2,542 in 2007 and 2,850 in 2008. Monitoring data indicated that approximately 5.6% of winter-run Chinook salmon eggs spawned in the Sacramento River in 2014 survived to the fry life stage (3 to nearly 10 times lower than in previous years). The ongoing drought has made 2015 another challenging year for winter-run Chinook salmon (NMFS 2016g).

Genetic Diversity. The rising proportion of hatchery fish among returning adults threatens to increase the risk of extinction. Lindley et al. (2007) recommend that in order to maintain a low risk of genetic introgression with hatchery fish, no more than 5% of the naturally-spawning population should be composed of hatchery fish. Since 2001, hatchery origin winter-run Chinook salmon have made up more than 5% of the run, and in 2005 the contribution of hatchery fish exceeded 18% (Lindley et al. 2007).

Distribution. The range of winter-run Chinook salmon has been greatly reduced by Keswick and Shasta dams on the Sacramento River and by hydroelectric development on Battle Creek. Currently, winter-run Chinook salmon spawning is limited to the main-stem Sacramento River between Keswick Dam (River Mile [RM] 302) and the RBDD (RM 243) where the naturally-spawning population is artificially maintained by cool water releases from the dams. Within the Sacramento River, the spatial distribution of spawners is largely governed by water year type and the ability of the Central Valley Project to manage water temperatures (NMFS 2014d).

Designated Critical Habitat. NMFS designated critical habitat for the Sacramento winter-run Chinook on June 16, 1993 (58 FR 33212). It includes: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters**.** Physical and biological features that are essential for the conservation of Sacramento winter-run Chinook salmon, based on the best available information, include (1) access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River; (2) the availability of clean gravel for spawning substrate; (3) adequate river flows for successful spawning, incubation of eggs, fry development and emergence, and downstream transport of juveniles; (4) water temperatures between 42.5 and 57.5 °F (5.8 and 14.1 degrees Celsius (°C)) for successful spawning, egg incubation, and fry development; (5) habitat and adequate prey free of contaminants; (6) riparian habitat that provides for successful juvenile development and survival; and (7) access of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean (58 FR 33212).

The current condition of PBFs for the Sacramento River Winter-run Chinook salmon indicates that they are not currently functioning or are degraded. Their conditions are likely to maintain low population abundances across the ESU. Spawning and rearing PBFs are especially degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds where water maintains lower temperatures. The rearing PBF is further degraded by floodplain habitat disconnected from the mainstems of larger rivers throughout the Sacramento

River watershed. The migration PBF is also degraded by the lack of natural cover along the migration corridors. Rearing and migration PBFs are further affected by pollutants entering the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Juvenile migration is obstructed by water diversions along Sacramento River and by 2 large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Recovery Goals. Recovery goals, objectives and criteria for the Sacramento River winter-run Chinook are fully outlined in the 2014 Recovery Plan (NMFS 2014d). In order to achieve the downlisting criteria, the species would need to be composed of 2 populations – 1 viable and 1 at moderate extinction risk. Having a second population would improve the species' viability, particularly through increased spatial structure and abundance, but further improvement would be needed to reach the goal of recovery. To delist winter-run Chinook salmon, 3 viable populations are needed. Thus, the downlisting criteria represent an initial key step along the path to recovering winter-run Chinook salmon.

8.10 Chinook salmon, Snake River fall-run

Table 30. Chinook salmon, Snake River fall-run ESU; overview table

Species	Common Name	Distinct Population Segments (DPS)	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook Salmon	Snake River fall-run	Threatened	2022	2005 70 FR 37160 2014 79 FR 20802	2017	1993 58 FR 68543

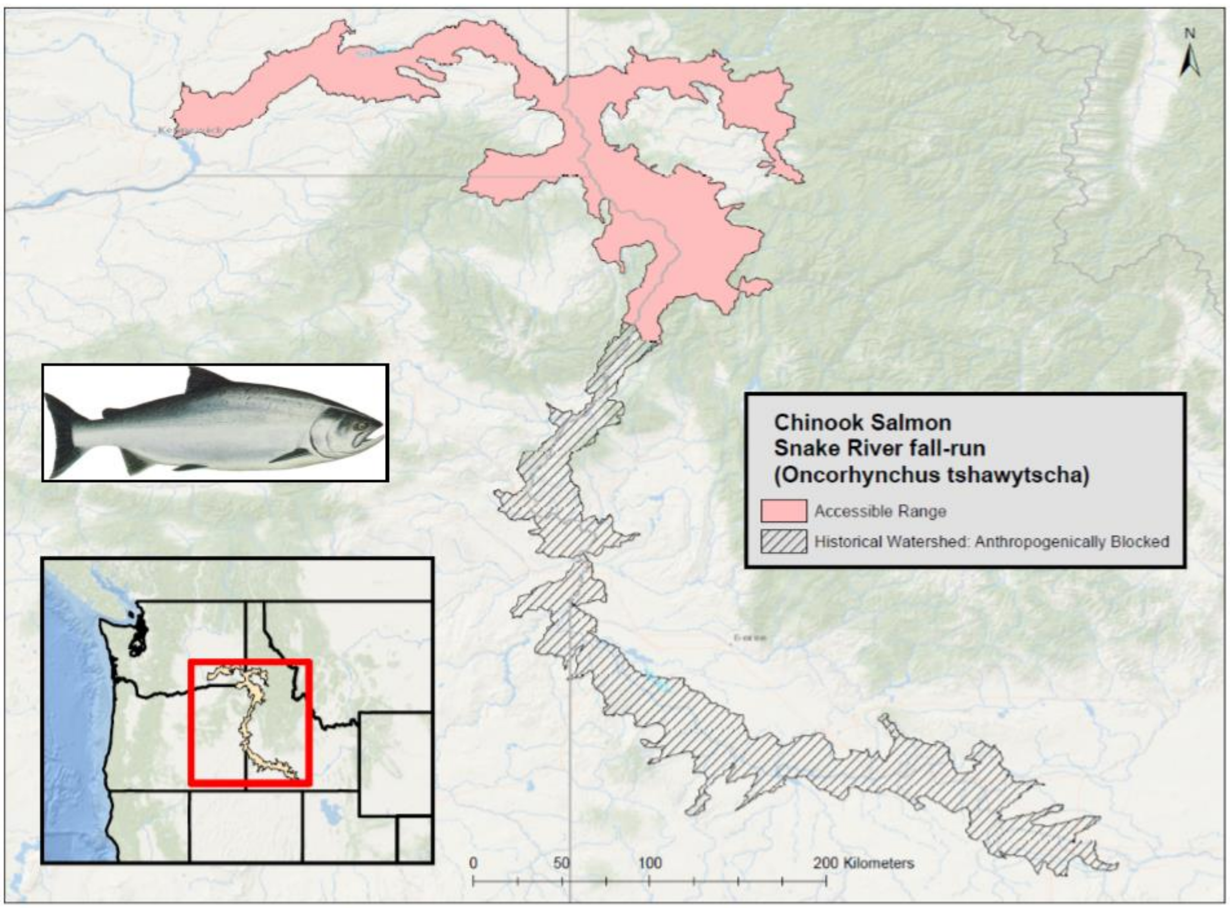


Figure 19. Chinook salmon, Snake River fall-run ESU range and designated critical habitat

Species Description. Chinook salmon are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002c). NMFS first listed Snake River fall Chinook salmon as a threatened species under the ESA on April 22, 1992 (57 FR 14658). NMFS reaffirmed the listing status in June 28, 2005 (70 FR 37160), and reaffirmed the status again in its 2014 (79 FR 20802). Snake River fall Chinook salmon historically spawned throughout the 600-mile reach of the mainstem Snake River from its mouth upstream to Shoshone Falls, a 212-foot high natural barrier near Twin Falls, Idaho (RM 614.7). The Northwest Fisheries Science Center's review (Ford 2022) found that no new information has become available that would justify a change in the delineation of the SR fall-run Chinook Salmon ESU. The listed ESU currently includes all natural-origin fall-run Chinook salmon originating from the mainstem Snake River below Hells Canyon Dam (the lowest of 3 impassable dams that form the Hells Canyon Complex) and from the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins. The listed ESU also includes fall-run Chinook salmon from 4 artificial propagation programs (NMFS 2011a; NMFS 2015b).

Status. As late as the late 1800s, approximately 408,500 to 536,180 fall Chinook salmon are believed to have returned annually to the Snake River. The run began to decline in the late 1800s and then continued to decline through the early and mid-1900s as a result of overfishing and other human activities, including the construction of major dams. Overall, the status of Snake River fall-run Chinook salmon has clearly improved compared to the time of listing. The single extant population in the ESU is currently meeting the criteria for a rating of "viable" developed by the ICTRT, but the ESU as a whole is not meeting the recovery goals described in the recovery plan for the species, which require the single population to be "highly viable with high certainty" and/or will require reintroduction of a viable population above the Hells Canyon Complex (NMFS 2017b). The Snake River fall-run Chinook salmon ESU therefore is considered to be at a moderate-to-low risk of extinction, with viability largely unchanged from the prior review. Supplementation and other measures since listing led to large increases in natural-origin returns, gradually at first and then, in 2013, adult spawner abundance reached over 20,000 fish (Figure 27, Table 15). From 2012–15, natural-origin returns were over 10,000 adults. Spawner abundance has declined since 2016 to 4,998 adult natural-origin spawners in 2019. In 2018, natural-origin spawner abundance was 4,916, a quarter of the return in 2013. This appears as a high negative percent change in the 5-year geometric mean but, when looking at the trend in longer time frames, across more than 1 brood cycle, it shows an increase in the 10-year geometric mean relative to the last status review, and a near-zero population change for the 15-year trend in abundance. The geometric mean natural adult abundance for the most recent 10 years (2010–19) is 9,034 (0.15 SE), higher than the 10-year geomean reported in the most recent status review (6,418, 0.19 SE, 2005–14; NWFSC 2015). While the population has not been able to maintain the higher returns it achieved in 2010 and 2013–15, it has maintained at or above the ICTRT defined Minimum Abundance Threshold (3,000) during climate challenges in the ocean and rivers. While the number of natural-origin fall Chinook salmon has been high, substantial uncertainty remains about the status of the species' productivity and diversity. Threats posed by straying out-of-ESU hatchery fish have declined due to improved management. Still, large reaches of historical habitat remain blocked and inundated, and the mainstem Snake and Columbia River hydropower system, while less of a constraint than in the past, continues to cause juvenile and adult losses. The number of hatchery-origin fall Chinook salmon on the spawning grounds continues to threaten natural-origin fish productivity and genetic diversity. Further, the combined and relative effects of the different threats across the life cycle ─ including threats from climate change ─ remain poorly understood (NMFS 2011a; NMFS 2015b). The overall low risk rating for the ESU remains unchanged from that reported in the 2016 5-year status review. As also reported in the 2016 5-year status review, while the extinction risk status of the ESU has improved since it was listed in 1992, and while the ESU is at low risk, it is still not meeting its recovery goals (NMFS 2016a, 2017). The implementation of sound management actions to address hydropower, habitat, hatcheries, harvest, and predation remain essential to the recovery of this ESU. The ESA Snake River Fall Chinook Salmon Recovery Plan (NMFS 2017) will be the primary guide for identifying future actions to target and address limiting factors and threats for this ESU.

Life History. Snake River fall-run Chinook return to the Columbia River in August and September, pass Bonneville Dam from mid-August to the end of September, and enter the Snake River between early September and mid-October (DART 2013). Once they reach the Snake River, fall Chinook salmon generally travel to 1 of 5 major spawning areas and spawn from late October through early December (Connor et al. 2014).

Upon emergence from the gravel, most young fall Chinook salmon move to shoreline riverine habitat (recovery plan). Some fall Chinook salmon smolts sustain active migration after passing Lower Granite Dam and enter the ocean as subyearlings, whereas some delay seaward migration and enter the ocean as yearlings (Connor et al. 2005; McMichael et al. 2008; NMFS 2015b). Snake River fall Chinook salmon can be present in the estuary as juveniles in winter, as fry from March to May, and as fingerlings throughout the summer and fall (Fresh et al. 2005; Roegner et al. 2012; Teel et al. 2014).

Once in the Northern California Current, dispersal patterns differ for yearlings and subyearlings. Subyearlings migrate more slowly, are found closer to shore in shallower water, and do not disperse as far north as yearlings (Fisher et al. 2014; Sharma and Quinn 2012; Trudel et al. 2009; Tucker et al. 2011). Snake River basin fall Chinook salmon spend 1 to 4 years in the Pacific Ocean, depending on gender and age at the time of ocean entry (Connor et al. 2005).

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982a; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 31. Temporal distribution of Chinook salmon, Snake River fall-run ESU in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)								Present				
Spawning										Present		
Incubation (eggs)	Present									Present		
Emergence (alevin to fry phases	Present											Present
Rearing and migration (juveniles)	Present											

Population Dynamics

Abundance. The naturally spawning fall Chinook salmon in the lower Snake River have included both returns originating from naturally spawning parents and from returning hatchery releases. The geometric mean natural origin adult abundance for the most recent 10 years (2010–19) is 9,034 (0.15 SE) (NMFS WCR 2022), higher than the 10-year geomean reported in the most recent status review (6,418, 0.19 SE, 2005–14; NWFSC 2015).

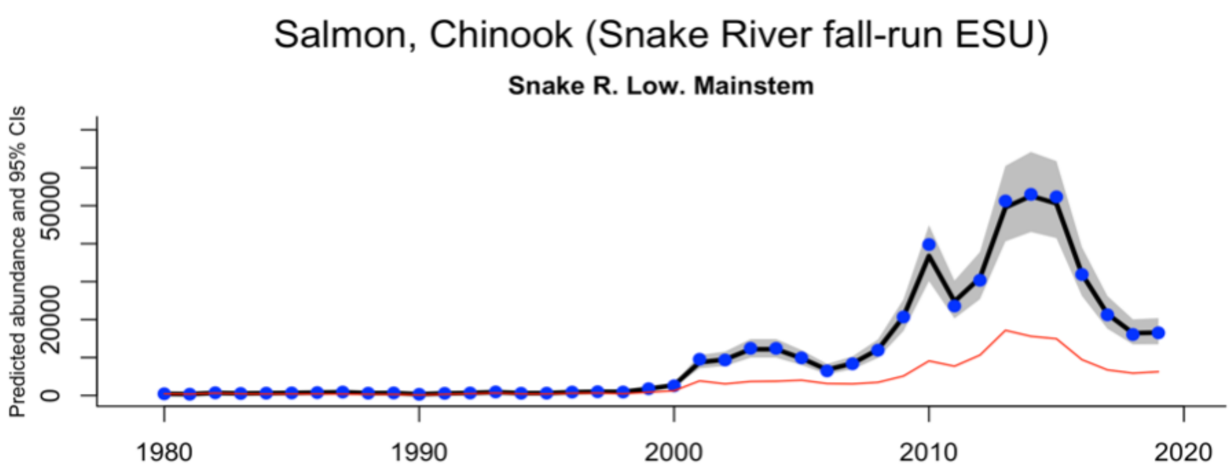


Figure 20. Smoothed trend in estimated total abundance (thick black line, with 95% confidence interval in gray) and natural (thin red line) population spawning abundance. In portions of a time series where a population has no annual estimates but smoothed spawning abundance is estimated from correlations with other populations, the smoothed estimate is shown in light gray. Points show the annual raw spawning abundance estimates. For some trends, the smoothed estimate may be influenced by earlier data points not included in the plot (from NMFS 2022a).

Productivity / Population Growth Rate. Productivity, as seen in broodyear returns-per-spawner, has been below replacement (1:1) in recent years, and a longer-term, 20-year geometric mean raw productivity is 0.63 (Figure 28)—likely an underestimate of intrinsic productivity. While below-replacement returns are concerning, the long-term (15-year) abundance trend is stable and the population remains well above the minimum abundance threshold set by the ICTRT.

Genetic Diversity. Genetic samples from the aggregate population in recent years indicate that composite genetic diversity is being maintained and that the Snake River Fall Chinook hatchery stock is similar to the natural component of the population, an indication that the actions taken to reduce the potential introgression of out-of-basin hatchery strays has been effective. Overall, the current genetic diversity of the population represents a change from historical conditions and, applying the Interior Columbia Technical Recovery Team (ICTRT) guidelines, the rating for this metric is moderate risk (NMFS 2015b).

Distribution. The extant Lower Snake River Fall Chinook salmon population consists of a spatially complex set of 5 historical major spawning areas (Cooney et al. 2007), each of which consists of a set of relatively discrete spawning patches of varying size. The primary Major spawning area (MaSA) in the extant Lower Mainstem Snake River population is the 96-km Upper Mainstem Snake River Reach, extending upriver from the confluence of the Salmon River to the Hells Canyon Dam site, where the canyon walls narrow and strongly confine the river bed. A second mainstem Snake River MaSA, the Lower Mainstem Snake River Reach, extends 69 km downstream from the Salmon River confluence to the upper end of the contemporary Lower Granite Dam pool. The lower mainstem reaches of 2 major tributaries to the mainstem Snake River, the Grande Ronde and the Clearwater Rivers, were also identified by the ICTRT as MaSAs. Both of these river systems currently support fall Chinook salmon spawning in the lower reaches. In addition, there is some historical evidence for production of late spawning Chinook salmon in spatially isolated reaches in upriver tributaries to each of these systems (NMFS 2015b).

Designated Critical Habitat. NMFS designated critical habitat for SR Fall-run Chinook salmon on December 28, 1993 (58 FR 68543). PBFs considered essential for the conservation of Chinook salmon, Snake River fall-run ESU are shown in Table 32.

Table 32. Essential features of critical habitats designated for SR spring/summer-run Chinook salmon, SR fall-run Chinook salmon, SR sockeye salmon, SONC coho salmon, and corresponding species life history events.

Essential Features Site	Essential Features Site Attribute	Species Life History Event
Spawning and juvenile rearing areas	Access (sockeye) Cover/shelter Food (juvenile rearing) Riparian vegetation Space (Chinook, coho) Spawning gravel Water quality Water temp (sockeye) Water quantity	Adult spawning Embryo incubation Alevin growth and development Fry emergence from gravel Fry/parr/smolt growth and development
Adult and juvenile migration corridors	Cover/shelter Food (juvenile) Riparian vegetation Safe passage Space Substrate Water quality Water quantity Water temperature Water velocity	Adult sexual maturation Adult upstream migration and holding Kelt (steelhead) seaward migration Fry/parr/smolt growth, development, and seaward migration
Areas for growth and development to adulthood	Ocean areas – not identified	Nearshore juvenile rearing Subadult rearing Adult growth and sexual maturation Adult spawning migration

The major degraded PBFs within critical habitat designated for SR Fall-run Chinook salmon include: (1) safe passage for juvenile migration which is reduced by the presence of the Snake and Columbia River hydropower system within the lower mainstem; (2) rearing habitat water quality altered by influx of contaminants and changing seasonal temperature regimes caused by water flow management; and (3) spawning/rearing habitat PBF attributes (spawning areas with gravel, water quality, cover/shelter, riparian vegetation, and space to support egg incubation and larval growth and development) that are reduced in quantity (80% loss) and quality due to the mainstem lower Snake River hydropower system.

Water quality impairments in the designated critical habitat are common within the range of this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine sediments from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary; traveling along with contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Recovery Goals. Recovery goals, objectives and criteria for the Snake River fall-run Chinook are fully outlined in the 2017 Recovery Plan (NMFS 2017c). ESA recovery goals should support conservation of natural fish and the ecosystems upon which they depend. Thus, the ESA recovery goal for Snake River fall Chinook salmon is that: the ecosystems upon which Snake River fall Chinook salmon depend are conserved such that the ESU is self-sustaining in the wild and no longer needs ESA protection.

8.11 Chinook salmon, Snake River spring/summer-run ESU

Table 33. Chinook salmon, Snake River spring/summer-run ESU; overview table

Species	Common Name	Distinct Population Segments (DPS)	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook Salmon	Snake River Spring and Summer run	Threatened	2022	2005 70 FR 37160 2014 79 FR 20802	2017	1999 64 FR 57399

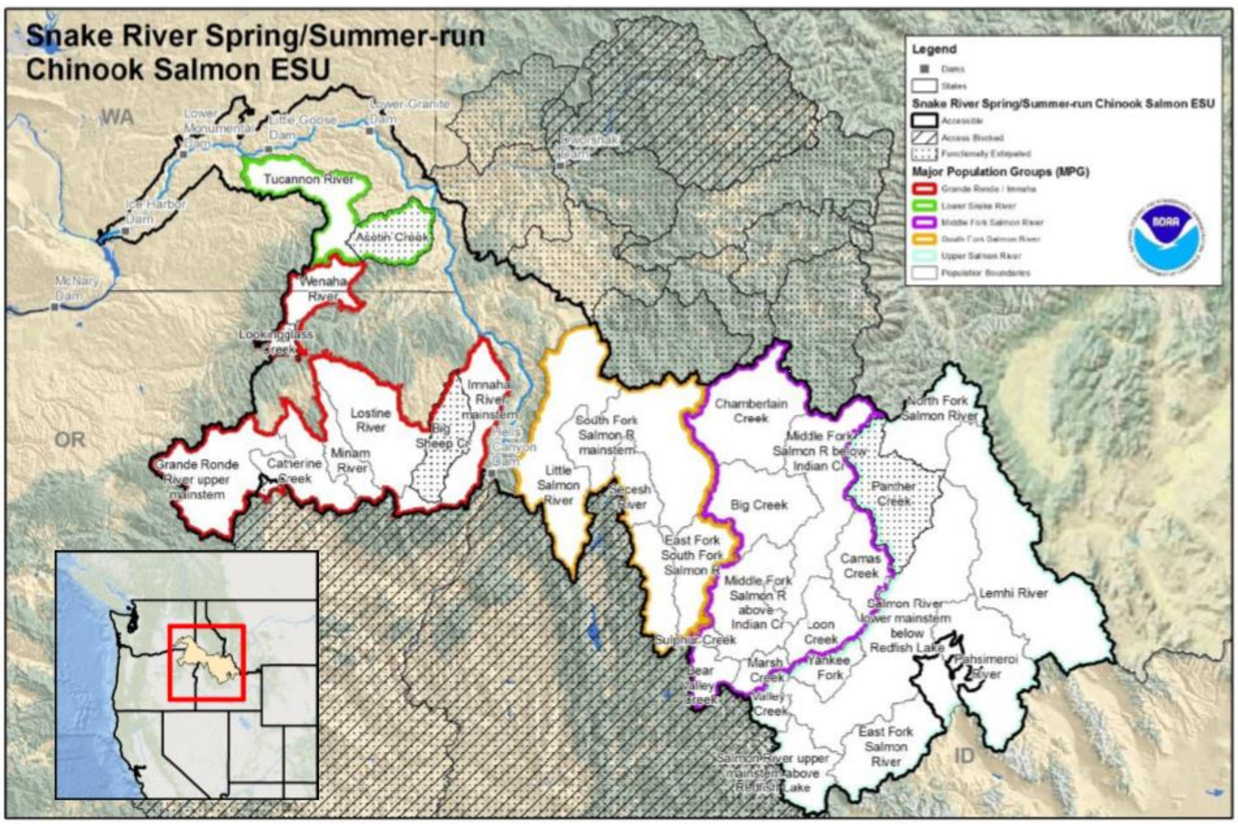


Figure 21. Chinook salmon, Snake River spring/summer-run ESU range and designated critical habitat

Species Description. Chinook salmon are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002c). The 2022 Status Update found no new information that would justify a change in the delineation of the

Snake River spring/summer Chinook salmon ESU (Ford 2022). The Snake River spring/summer-run Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon, Grande Ronde, Imnaha, and Salmon River sub-basins, as well as in 15 artificial propagation programs (NMFS 2022a).

Status. The historical run of Chinook in the Snake River likely exceeded 1 million adult spawners annually in the late 1800s, by the 1950s the run had declined to near 100,000 adults per year. The adult counts fluctuated throughout the 1980s but then declined further, reaching a low of 2,200 adult spawners in 1995. Currently, the majority of extant populations in the Snake River spring/summer Chinook salmon ESU remain at high overall risk of extinction, with a low probability of persistence within 100 years. Factors cited in the 1991 status review as contributing to the species' decline since the late 1800s include overfishing, irrigation diversions, logging, mining, grazing, obstacles to migration, hydropower development, and questionable management practices and decisions (Matthews and Waples 1991). In addition, new threats ─ such as those posed by toxic contamination, increased predation by non-native species, and effects due to climate change ─ are emerging (NMFS 2016a).

Life History. Adult spring-run Chinook salmon destined for the Snake River return to the Columbia River from the ocean in early spring and pass Bonneville Dam beginning in early March and ending May 31st. Snake River summer-run Chinook salmon return to the Columbia River from June through July. Adults from both runs hold in deep pools in the mainstem Columbia and Snake Rivers and the lower ends of the spawning tributaries until late summer, when they migrate into the higher elevation spawning reaches. Generally, Snake River spring-run Chinook salmon spawn in mid- through late August. Snake River summer-run Chinook salmon spawn approximately 1 month later than spring-run fish and tend to spawn lower in the tributary drainages, although their spawning areas often overlap with those of spring-run spawners

The eggs that Snake River spring and summer Chinook salmon deposit in late summer and early fall incubate over the following winter, and hatch in late winter and early spring. Juveniles rear through the summer, overwinter, and typically migrate to sea in the spring of their second year of life, although some juveniles may spend an additional year in freshwater. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Most yearling fish are thought to spend relatively little time in the estuary compared to sub-yearling ocean-type fish, however there is considerable variation in residence times in different habitats and in the timing of estuarine and ocean entry among individual fish (Holsman et al. 2012; McElhany et al. 2000a). Snake River spring/summer-run Chinook salmon range over a large area in the northeast Pacific Ocean, including coastal areas off Washington, British Columbia, and southeast Alaska, the continental shelf off central British Columbia, and the Gulf of Alaska (NMFS 2016e). Most of the fish spend 2 or 3 years in the ocean before returning to tributary spawning grounds primarily as 4- and 5-year-old fish. A small fraction of the fish spend only 1 year in the ocean and return as 3-year-old "jacks," heavily predominated by males (Good et al. 2005a).

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982a; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 34. Temporal distribution of Chinook salmon, Snake River spring/summer-run ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance / Productivity

Spring/summer-run Chinook salmon ESU populations are summarized in 5-year increments. The most recent 5-year geometric mean abundance estimates for 26 of the 27 populations are lower than the corresponding estimates for the previous 5-year period by varying degrees; the estimate for the 27th population was a slight increase from a very low abundance in the prior 5-year period. Data show a consistent and marked pattern of declining population size, with the recent 5-year abundance levels for the 27 populations declining by an average of 55%. Medium-term (15-year) population trends in total spawner abundance were positive over the period 1990–2005 for all of the population natural-origin abundance series, and are all declining over the more recent time interval (2004–19). The consistent and sharp declines for all populations in the ESU are concerning, as the abundances for some populations are approaching similar levels to those of the early 1990s when the ESU was listed.

Lower Snake River Major Population Group (MPG): Abundance and productivity remain the major concern for the Tucannon River population. Natural spawning abundance (10-year geometric mean) has decreased and remains well below the minimum abundance threshold for the single extant population in this MPG. Poor natural productivity continues to be a major concern.

Grande Ronde/Imnaha MPG: The Wenaha River, Lostine/Wallowa River and Minam River populations showed substantial decreases in natural abundance relative to the previous ICTRT review, and each remains below their respective minimum abundance thresholds. The Catherine Creek and Upper Grande Ronde populations each remain in a critically depressed state. Geometric mean productivity estimates remain relatively low for all populations in the MPG.
South Fork Salmon River MPG: Natural spawning abundance (10-year geometric mean) estimates decreased for the populations with available data series. Viability ratings based on the combined estimates of abundance and productivity remain at high risk, although the

survival/capacity gaps relative to moderate and low risk viability curves are smaller than for other ESU populations.

Middle Fork Salmon River MPG: Natural-origin abundance and productivity remains extremely low for populations within this MPG.

Upper Salmon River MPG: Abundance and productivity estimates for populations within this MPG remain at very low levels relative to viability objectives and all are in decline.
Genetic Diversity / Spatial Structure

Lower Snake River MPG: The integrated spatial structure/diversity risk rating for the Lower Snake River MPG is moderate.

Grande Ronde/Imnaha MPG: The Upper Grande Ronde population is rated at high risk for spatial structure and diversity while the remaining populations are rated at moderate.

South Fork Salmon River MPG: Spatial structure/diversity risks are currently rated moderate for the South Fork Mainstem population (relatively high proportion of hatchery spawners) and low for the Secesh River and East Fork South Fork populations.

Middle Fork Salmon River MPG: Spatial structure/diversity risk ratings for Middle Fork Salmon River MPG populations are generally moderate. This primarily is driven by moderate ratings for genetic structure assigned by the ICTRT because of uncertainty arising from the lack of direct genetic samples from within the component populations.

Upper Salmon River MPG: Spatial structure/diversity risk ratings vary considerably across the Upper Salmon River MPG. Four of the 8 populations are rated at low or moderate risk for overall spatial structure and diversity and could achieve viable status with improvements in average abundance/productivity. The high spatial structure/diversity risk rating for the Lemhi population is driven by a substantial loss of access to tributary spawning/rearing habitats and the associated reduction in life-history diversity. High risk ratings for Pahsimeroi River, East Fork Salmon River, and Yankee Fork Salmon River are driven by a combination of habitat loss and diversity concerns related to low natural abundance combined with chronically high proportions of hatchery spawners in natural areas.

Distribution. The Snake River spring/summer Chinook salmon ESU includes all naturally spawned populations of spring/summer Chinook salmon in the mainstem Snake River and the Tucannon River, Grand Ronde River, Imnaha River, and Salmon River subbasins. The ESU is broken into 5 major population groups (MPG). Together, the MPGs contain 28 extant independent naturally spawning populations, 3 functionally extirpated populations, and 1 extirpated population. The Upper Salmon River MPG contains 8 extant populations and 1 extirpated population. The Middle Fork Salmon River MPG contains 9 extant populations. The South Fork Salmon River MPG contains 4 extant populations. The Grande Ronde/Imnaha Rivers MPG contains 6 extant populations, with 2 functionally extirpated populations. The Lower Snake River MPG contains 1 extant population and 1 functionally extirpated population. The South

Fork and Middle Fork Salmon Rivers and Grand Ronde currently support most of the natural spring/summer Chinook salmon production in the Snake River drainage (NMFS 2016e).

Designated Critical Habitat. Critical habitat for Snake River spring/summer Chinook salmon was designated on December 28, 1993 (58 FR 68543) and revised slightly on October 25, 1999 (64 FR 57399). PBFs considered essential for the conservation of Chinook salmon, Snake River spring/summer-run ESU are shown in Table 32.

Spawning and juvenile rearing PBFs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access that has resulted from altered flow regimes from hydroelectric dams. According to the ICBTRT, the Panther Creek population was extirpated because of legacy and modern mining-related pollutants creating a chemical barrier to fish passage (Chapman and Julius 2005).

Presence of cool water that is relatively free of contaminants is particularly important for the spring/summer run life history as adults hold over the summer and juveniles may rear for a whole year in the river. Water quality impairments are common in the range of the critical habitat designated for this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine bottom substrate from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Recovery Goals. Recovery goals, scenarios and criteria for the Snake River spring and summer-run Chinook salmon are fully outlined in the recovery plan issued in 2017 (NMFS 2017d). The status levels targeted for populations within an ESU or DPS are referred to collectively as the "recovery scenario" for the ESU or DPS. NMFS has incorporated the viability criteria into viable recovery scenarios for each Snake River spring/summer Chinook salmon and steelhead MPG. The criteria should be met for an MPG to be considered Viable, or low (5% or less) risk of extinction, and thus contribute to the larger objective of ESU or DPS viability. These criteria are:

- At least one-half the populations historically present (minimum of 2 populations) should meet viability criteria (5% or less risk of extinction over 100 years).
- At least 1 population should be highly viable (less than 1% risk of extinction).
- Viable populations within an MPG should include some populations classified as "Very Large'" or "Large," and "Intermediate" reflecting proportions historically present.
- All major life history strategies historically present should be represented among the populations that meet viability criteria.
- Remaining populations within an MPG should be maintained (25% or less risk of extinction) with sufficient abundance, productivity, spatial structure, and diversity to provide for ecological functions and to preserve options for ESU or DPS recovery.
- For MPGs with only 1 population, this population must be highly viable (less than 1% risk of extinction).

8.12 Chinook salmon, Upper Columbia River spring-run ESU

Table 35. Chinook salmon, Upper Columbia River spring-run ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook salmon	Upper Columbia River spring-run ESU	Endangered	2022	70 FR 37160	2007	70 FR 52630

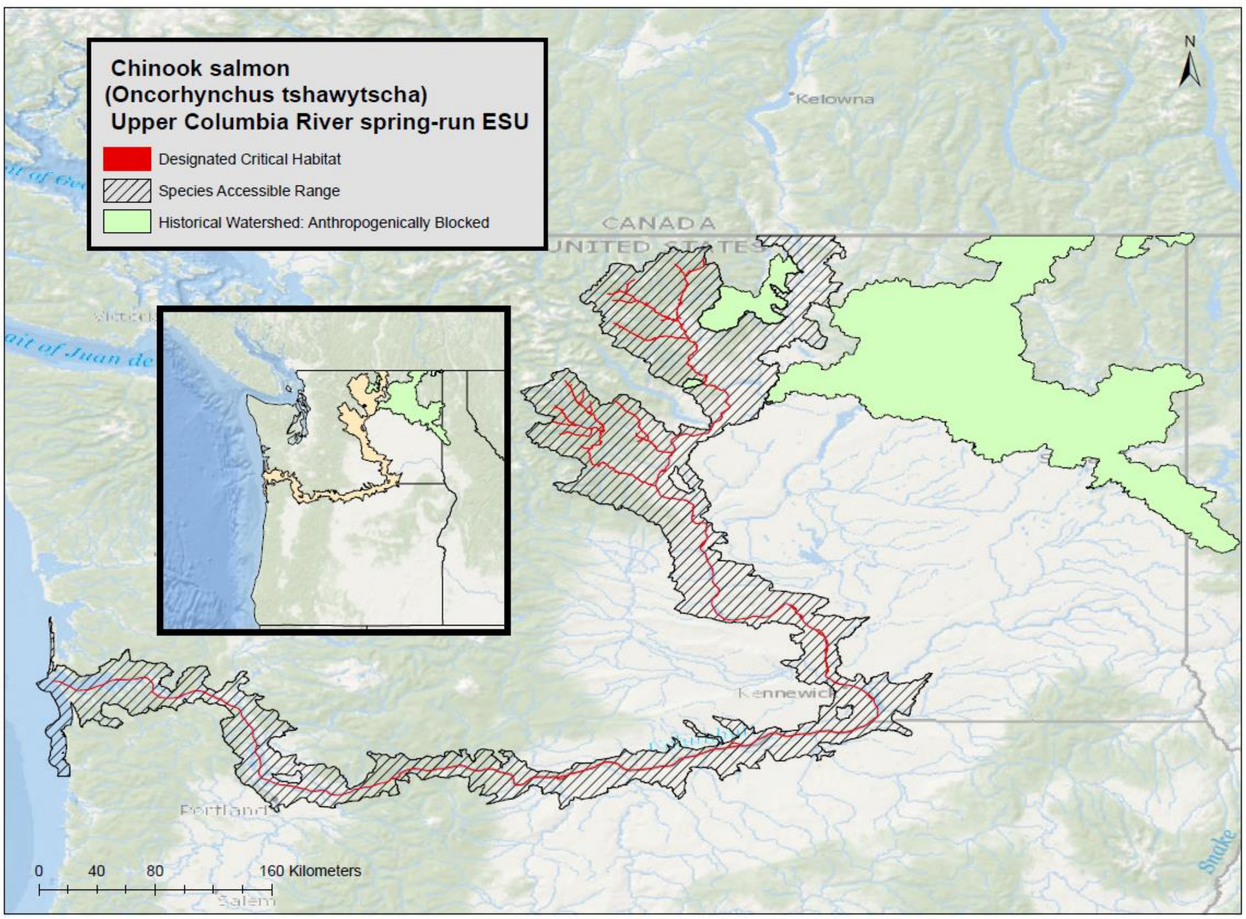


Figure 22. Chinook salmon, Upper Columbia River spring-run ESU range and designated critical habitat

Species Description. Chinook salmon are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002c). Upper

Columbia River spring-run Chinook salmon ESU was listed as an endangered species under the ESA on March 24, 1999 (64 FR 14308). NMFS reaffirmed the listing on June 28, 2005 (70 FR 37160). This ESU includes naturally spawned spring-run Chinook salmon originating from Columbia River tributaries upstream of the Rock Island Dam and downstream of Chief Joseph Dam (excluding the Okanogan River subbasin). Also, spring-run Chinook salmon from 6 artificial propagation programs. The Northwest Fisheries Science Center's review (Ford 2022) found that no new information had become available that would justify a change in the delineation of the UCR spring-run Chinook salmon ESU.

Status. The Upper Columbia spring Chinook ESU includes 3 extant populations (Wenatchee, Entiat, and Methow), as well as 1 extinct population in the Okanogan subbasin (ICBTRT 2003). All 3 populations continued to be rated at low risk for spatial structure but at high risk for diversity criteria. Large-scale supplementation efforts in the Methow and Wenatchee Rivers are ongoing, intended to counter short-term demographic risks given current average survival levels and the associated year-to-year variability. Under the current recovery plan, habitat protection and restoration actions are being implemented that are directed at key limiting factors. Although the status of the ESU has improved relative to measures available at the time of listing, all 3 populations remain at high risk (NWFSC 2015). The Northwest Fisheries Science Center's review of updated information (Ford 2022) does not indicate a change in the biological risk category for this species since the time of the last 5-year review (NWFSC 2015). Analysis of the ESA section 4(a)(1) factors indicates that the collective risk to the UCR Spring-run Chinook salmon's persistence has not changed significantly since our previous 5-year review for the UCR spring-run Chinook salmon ESU.

Life History. Adult spring Chinook in the Upper Columbia Basin begin returning from the ocean in the early spring, with the run into the Columbia River peaking in mid-May. Spring Chinook enter the Upper Columbia tributaries from April through July. After migration, they hold in freshwater tributaries until spawning occurs in the late summer, peaking in mid to late August. Juvenile spring Chinook spend a year in freshwater before migrating to salt water in the spring of their second year of life. Most Upper Columbia spring Chinook return as adults after 2 or 3 years in the ocean. Some precocious males, or jacks, return after 1 winter at sea. A few other males mature sexually in freshwater without migrating to the sea. However, 4 and 5 year old fish that have spent 2 and 3 years at sea, respectively, dominate the run. Fecundity ranges from 4,200 to 5,900 eggs, depending on the age and size of the female.

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982a; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 36. Temporal distribution of Chinook salmon, Upper Columbia River spring-run ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td colspan="7">Present</td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. For all populations, average abundance over the recent 10-year period is below the average abundance thresholds that the ICTRT identifies as a minimum for low risk (ICTRT 2008a; ICTRT 2008b; ICTRT 2008c). All 3 populations in the UCR spring-run Chinook salmon ESU remain at high overall risk. Natural origin abundance has decreased over the levels reported in the prior review for all populations in this ESU, in many cases sharply. The abundance data for the entire ESU show a downward trend over the last 5 years, with the recent 5-year abundance levels for all 3 populations declining by an average of 48%. The consistent and sharp declines for all populations in the ESU are concerning. Relatively low ocean survivals in recent years were a major factor in recent abundance patterns.

Productivity / Population Growth Rate. Using the updated data series for this review, the short-term (5-year) trend in wild spawners has been strongly negative for all 3 extant populations. Longer-term (15-year) trends are also negative for all 3 populations, although the 95% confidence intervals in each case include 0. In general, both total and natural-origin escapements for all 3 populations increased sharply from 1999 through 2002 and have shown substantial year-to-year variations in the years following, with peaks around 2001 and 2010 and declines after 2010. Average natural-origin returns remain well below ICTRT minimum threshold levels. The smolt to adult ratio estimates for the range of brood years 2002–15. Over the period of record, the geometric mean SAR for the Entiat and Methow River populations (~3%) represents a low, but reasonable marine survival, with the Wenatchee River SAR of ~1.5% being on the low end, as 2% is roughly a replacement rate (ICTRT 2008a; ICTRT 2008b; ICTRT 2008c).

Genetic Diversity. The ICTRT characterizes the diversity risk to all Upper Columbia River (UCR) Spring-run Chinook populations as “high”. The high risk is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project in 1939-1943.

Distribution. Spring Chinook currently spawn and rear in the upper main Wenatchee River upstream from the mouth of the Chiwawa River, overlapping with summer Chinook in that area (Peven et al. 1994). The primary spawning areas of spring Chinook in the Wenatchee subbasin include Nason Creek and the Chiwawa, Little Wenatchee, and White rivers. (Hamstreet and Carie 2003) described the current spawning distribution for spring Chinook in the Entiat subbasin as the Entiat River (river mile 16.2 to 28.9) and the Mad River (river mile 32 1.5-5.0). Spring Chinook of the Methow population currently spawn in the mainstem Methow River and the Twisp, Chewuch, and 5 Lost drainages (Humling and Snow 2005; Scribner et al. 1993). A few also spawn in Gold, Wolf, 6 and Early Winters creeks.

Designated Critical Habitat. NMFS designated critical habitat for Upper Columbia River Spring-run Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to Chief Joseph Dam and several tributary subbasins. PBFs considered essential for the conservation of Chinook salmon, Upper Columbia River spring-run ESU are described in Appendix B.

Spawning and rearing PBFs are somewhat degraded in tributary systems by urbanization in lower reaches, grazing in the middle reaches, and irrigation and diversion in the major upper drainages. These activities have resulted in excess erosion of fine sediment and silt that smother spawning gravel; reduction in flow quantity necessary for successful incubation, formation of physical rearing conditions, and juvenile mobility. Moreover siltation further affects critical habitat by reducing water quality through contaminated agricultural runoff; and removing natural cover. Adult and juvenile migration PBFs are heavily degraded by Columbia River Federal dam projects and a number of mid-Columbia River Public Utility District dam projects also obstruct the migration corridor.

Recovery Goals. Recovery goals, objectives and detailed criteria for the Central Valley spring-run Chinook are fully outlined in the 2016 Recovery Plan (NMFS 2007b). The general recovery objectives are:

- Increase the abundance of naturally produced spring Chinook spawners within each population in the Upper Columbia ESU to levels considered viable.
- Productivity 21 Increase the productivity (spawner:spawner ratios and smolts/redds) of naturally produced spring Chinook within each population to levels that result in low risk of extinction.
- Restore the distribution of naturally produced spring Chinook to previously occupied areas (where practical) and allow natural patterns of genetic and phenotypic diversity to be expressed.

8.13 Chinook salmon, Upper Willamette River ESU

Table 37. Chinook salmon, Upper Willamette River ESU; overview table

Species	Common Name	Distinct Population Segment	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook salmon	Upper Willamette River ESU	Threatened	2022	70 FR 37160	2011	70 FR 52630

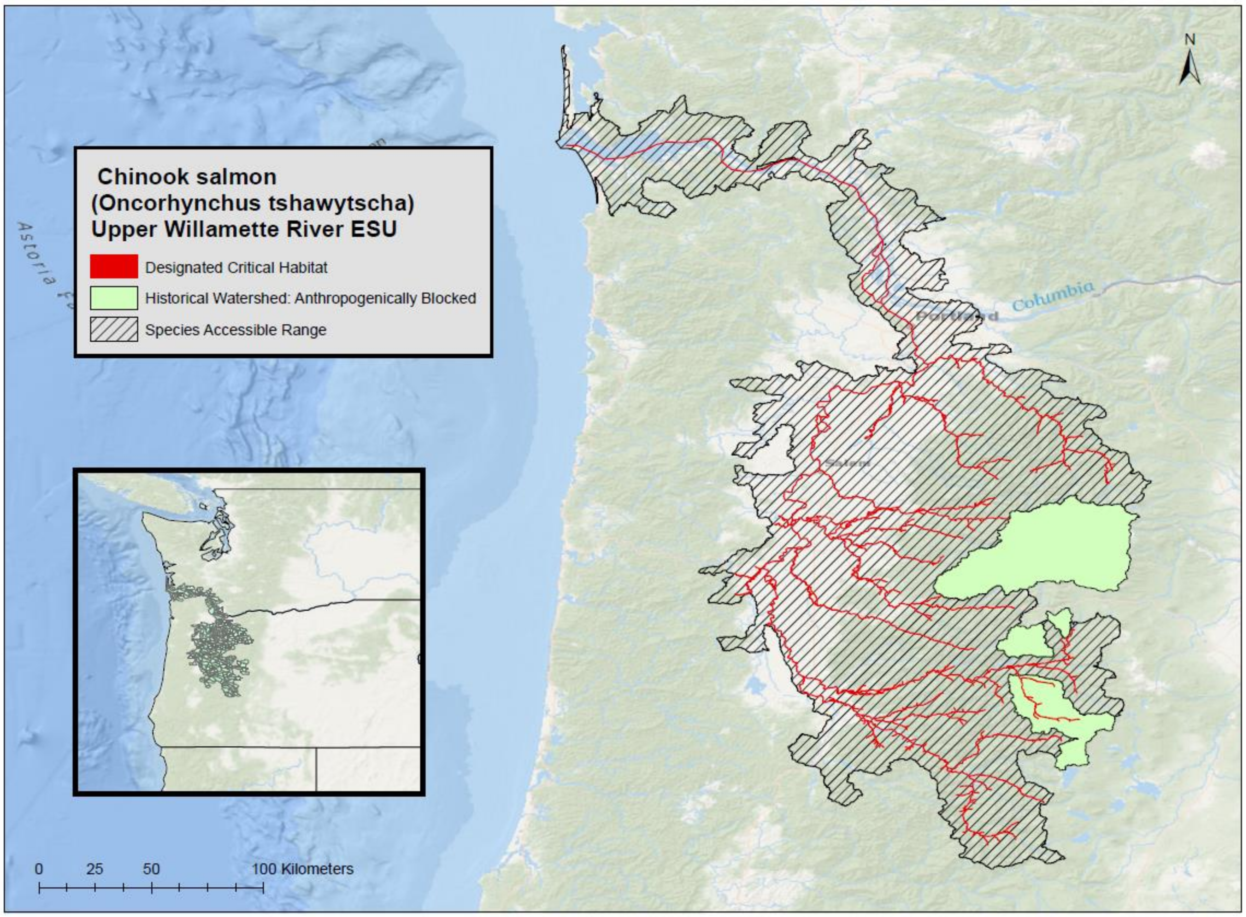


Figure 23**. Chinook salmon, Upper Willamette River ESU range and designated critical habitat**

Species Description. Chinook salmon are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002c). Upper Willamette River Chinook salmon ESU was listed as a threatened species under the ESA on March 24, 1999 (64 FR 14308). NMFS reaffirmed the listing on June 28, 2005 (70 FR 37160). This ESU includes naturally spawned spring-run Chinook salmon originating from the

Clackamas River and from the Willamette River and its tributaries above Willamette Falls. Also, spring-run Chinook salmon from 6 artificial propagation programs.

Status. The Upper Willamette River Chinook ESU is considered to be extremely depressed, likely numbering less than 10,000 adult spawners compared to a historical abundance estimate of 300,000 (Myers et al. 2003). There are 7 demographically independent populations of spring-run Chinook salmon in the Upper Willamette River (UWR) Chinook salmon ESU: Clackamas, Molalla, North Santiam, South Santiam, Calapooia, McKenzie, and the Middle Fork Willamette (Myers et al. 2006). Currently, significant natural production occurs in only the Clackamas and McKenzie populations (McElhany et al. 2007a). Juvenile spring Chinook produced by hatchery programs are released throughout many of the subbasins and adult Chinook returns to the ESU are typically 80-90% hatchery origin fish. Access to historical spawning and rearing areas is restricted by large dams in the 4 historically most productive tributaries, and in the absence of effective passage programs will continue to be confined to more lowland reaches where land development, water temperatures, and water quality may be limiting. Pre-spawning mortality levels are generally high in the lower tributary reaches where water temperatures and fish densities are generally the highest.

Life history. Upper Willamette River Chinook salmon exhibit an earlier time of entry into the Columbia River than other spring-run Chinook salmon ESUs (Myers et al. 1998b). Adults appear in the lower Willamette River in February, but the majority of the run ascends Willamette Falls in April and May, with a peak in mid- to late May. However, present-day salmon ascend the Willamette Falls via a fish ladder. Consequently, the migration of spring Chinook salmon over Willamette Falls extends into July and August (overlapping with the beginning of the introduced fall-run of Chinook salmon).

The adults hold in deep pools over summer and spawn in late fall or early winter when winter storms augment river flows. Fry may emerge from February to March and sometimes as late as June (Myers et al. 2006). Juvenile migration varies with 3 distinct juvenile emigration “runs”: fry migration in late winter and early spring; sub-yearling (0 yr +) migration in fall to early winter; and yearlings (1 yr +) migrating in late winter to spring. Sub-yearlings and yearlings rear in the mainstem Willamette River where they also use floodplain wetlands in the lower Willamette River during the winter-spring floodplain inundation period.

Juvenile Chinook salmon are nearshore obligate feeders foraging in shallow areas with protective cover, such as eelgrass in tidally influenced sandy beaches and other vegetated zones (Healey et al. 1991). Invertebrates including cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982a; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Table 38. Temporal distribution of Chinook salmon, Upper Willamette River ESU in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)					Present							
Spawning								Present				
Incubation (eggs)									Present			
Emergence (alevin to fry phases)	Present										Present	
Rearing and migration (juveniles)	Present											

Population Dynamics

Abundance. The UWR Chinook ESU is considered to be extremely depressed, likely numbering less than 10,000 adult spawners compared to a historical abundance estimate of 300,000 (Myers et al. 2003). Abundance levels for all but 1 of the 7 DIPs in this ESU remain well below their recovery goals. The Clackamas River DIP currently exceeds its abundance recovery goal and its pHOS goal (<10% hatchery-origin fish). Alternatively, the Calapooia River may be functionally extinct, and the Molalla River remains critically low (there is considerable uncertainty in the level of natural production in the Molalla River). Abundances in the North and South Santiam Rivers have declined since the last review, with natural-origin abundances in the low hundreds of fish. The Middle Fork Willamette River is at a very low abundance, even with the inclusion of natural-origin spring-run Chinook salmon spawning in Fall Creek. While returns to Fall Creek Dam number in the low hundreds, prespawn mortality rates are very high in the basin; however, the Fall Creek program does provide valuable information on juvenile fish passage through operational drawdown. With the exception of the Clackamas River, the proportions of natural-origin spawners in the remainder of the ESU are well below those identified in the recovery goals. While the Clackamas River appears to be able to sustain above recovery goal abundances, even during relatively poor ocean and freshwater conditions, the remainder of the ESU is well short of its recovery goal.

Productivity / Population Growth Rate. The spring Chinook salmon population in the McKenzie River is the only remaining self-sustaining naturally reproducing independent population. Within the recent review period, the average natural-origin abundance in the McKenzie River has increased by 13%, to a 5-year geomean of 1,664. This improvement in abundance marks a reversal of long-term declines. Still, the long-term trend in abundance (2015–19) is –2%. The McKenzie River has been a bellwether for natural production in the upper Willamette River basin, with the majority of historical spawning habitat still accessible. Natural-origin spawners represent the majority of spawners, 57%, especially in the upper reaches (NAI 2020). The other natural-origin populations in this ESU have very low current abundances, and long- and short-term population trends are negative. Cohorts returning from 2015–19 were strongly influenced by warmer-than-normal and less-productive ocean conditions, in addition to warmer- and drier-than-normal freshwater conditions. The 5-year average abundance geomean for 2015–19 was 6,916 natural-origin (unmarked) adults, a 31% decrease from the previous period. While there was a substantial downward trend in total and natural-origin spring-run abundance at Willamette Falls, there were some indications of improving abundance in 2019 and 2020. Improvements in abundance corresponded with improved ocean and freshwater conditions, as well as changes in pinniped predation. Over the last 15 years, the long-term trend for natural-origin returns was –4%, suggesting an overall decline in those populations above Willamette Falls.

Genetic Diversity. Access of fall-run Chinook salmon to the upper Willamette River and the mixing of hatchery stocks within the ESU have threatened the genetic integrity and diversity of the species. Much of the genetic diversity that existed between populations has been homogenized (Myers et al. 2006). While hatchery populations were most similar to natural Chinook salmon from the same basin, they tended to present greater allelic richness. It is not clear whether this is due to the small effective population size of naturally spawning populations, or the legacy of interhatchery transfers between basins (Johnson and Friesen 2014).

Distribution. Radio-tagging results from 2014 suggest that few fish strayed into west-side tributaries (no detections) and relatively fewer fish were unaccounted for between Willamette Falls and the tributaries (12.9% of clipped fish and 5.3% of unclipped fish) (Jepson et al. 2015). In contrast to most of the other populations in this ESU, McKenzie River Chinook salmon have access to much of their historical spawning habitat, although access to historically high quality habitat above Cougar Dam (South Fork McKenzie River) is still limited by poor downstream juvenile passage. Similarly, natural-origin returns to the Clackamas River have remained flat, despite adults having access to much of their historical spawning habitat. Although returning adults have access to most of the Calapooia and Molalla basin, habitat conditions are such that the productivity of these systems is very low. Natural-origin spawners in the Middle Fork Willamette River in the last 10 years consisted solely of adults returning to Fall Creek. While these fish contribute to the Demographically Independent Populations (DIP) and ESU, at best the contribution will be minor. Finally, improvements were noted in the North and South Santiam DIPs. The increase in abundance in both DIPs was in contrast to the other DIPs and the counts at Willamette Falls. While spring-run Chinook salmon in the South Santiam DIP have access to some of their historical spawning habitat, natural origin spawners in the North Santiam are still confined to below Detroit Dam and subject to relatively high prespawning mortality rates (NWFSC 2015).

Designated Critical Habitat. NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River as well as specific stream reaches in a number of subbasins. PBFs considered essential for the conservation of Chinook salmon, Upper Willamette River ESU are described in Appendix B.
The current condition of PBFs of the UWR Chinook salmon critical habitat indicates that migration and rearing PBFs are not currently functioning or are degraded. These conditions impact their ability to serve their intended role for species conservation. The migration PBF is degraded by dams altering migration timing and water management altering the water quantity necessary for mobility and survival. Migration, rearing, and estuary PBFs are also degraded by loss of riparian vegetation and instream cover. Pollutants such as petroleum products, fertilizers, pesticides, and fine sediment enter the stream through runoff, point source discharge, drift during application, and non-point discharge where agricultural and urban development occurs. Degraded water quality in the lower Willamette River where important floodplain rearing habitat is present affects the ability of this habitat to sustain its role to conserve the species.

Recovery Goals. Recovery goals, objectives and detailed criteria for the Upper Willamette River Chinook are fully outlined in the 2011 Recovery Plan (NMFS 2011b). The 2011 recovery plan outlines 5 potential scenario options for meeting the viability criteria for recovery. Of the 5 scenarios, scenario 1 reportedly represented the most balanced approach given limitations in some populations. The approach in this Plan to achieve ESU delisting of UWR Chinook salmon is to: recover the McKenzie (core and genetic legacy population) and the Clackamas populations to an extinction risk status of very low risk (beyond minimal viability thresholds); to recover the North Santiam and Middle Fork Willamette populations (core populations) to an extinction risk status of low risk; to recover the South Santiam population to moderate risk; and to improve the status of the remaining populations from very high risk to high risk.

8.14 Coho salmon, Central California Coast ESU

Table 39. Coho salmon, central California coast ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus kisutch	Coho salmon	Central California Coast	Endangered	2023	2005 70 FR 37160	2012	1999 64 FR 24049

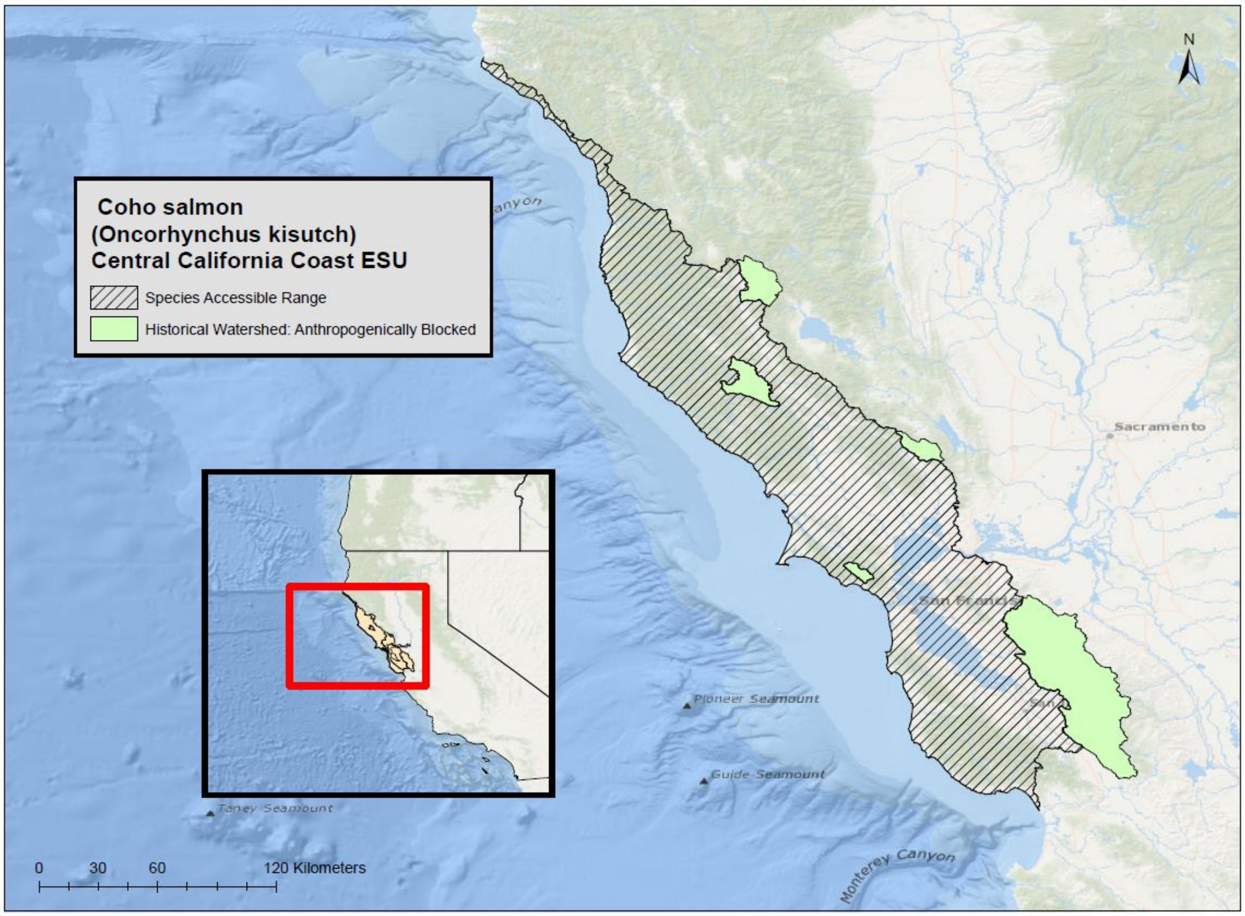


Figure 24. Coho salmon, central California coast ESU range

Species Description. Coho salmon are an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn). Adult coho salmon are typically about 2 feet long and 8 pounds. Coho have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Central California coast coho salmon ESU was listed as threatened under the ESA on October 31, 1996 (64 FR 56138). NMFS re-classified the ESU as endangered on June 28, 2005 (70 FR 37160). This ESU includes naturally spawned coho salmon originating from rivers south of Punta Gorda, California to and including Aptos Creek, as well as such coho salmon originating from tributaries to San Francisco Bay. Also, coho salmon from 3 artificial propagation programs are included in this ESU.

Status. The low survival of juveniles in freshwater, in combination with poor ocean conditions, has led to the precipitous declines of Central California Coast (CCC) coho salmon populations. Most independent CCC coho salmon populations remain at critically low levels, with those in the southern Santa Cruz Mountains strata likely extirpated. Data suggests some populations show a slight positive trend in annual escapement, but the improvement is not statistically significant. Overall, all CCC coho salmon populations remain, at best, a slight fraction of their recovery target levels, and, aside from the Santa Cruz Mountains strata, the continued extirpation of dependent populations continues to threaten the ESU's future survival and recovery. The evaluation of current habitat conditions and ongoing and future threats led to the conclusion that summer and winter rearing survival are very low due to impaired instream habitats. These impairments were due to a lack of complexity formed by instream wood, high sediment loads, lack of refugia habitats during winter, low summer flows and high instream temperatures. Additionally, populations throughout the ESU, but particularly at the southern end of the range, are likely to be significantly impacted by climate change in the future (NMFS 2012b).

Life History. Central California Coast coho salmon typically enter freshwater from November through January, and spawn into February or early March (Moyle 2002a). The upstream migration towards spawning areas coincides with large increases in stream flow (Hassler 1987). Coho salmon often are not able to enter freshwater until heavy rains have caused breaching of sand bars that form at the mouths of many coastal California streams. Spawning occurs in streams with direct flow to the ocean, or in large river tributaries (Moyle 2002c). Female coho salmon choose a site to spawn at the head of a riffle, just downstream of a pool where water flow changes from slow to turbulent, and where medium to small size gravel is abundant (Moyle 2002c).

Eggs incubate in redds from November through April, and hatch into "alevins" after a period of 35-50 days (Shapovalov and Taft 1954b). The period of incubation is inversely related to water temperature. Alevins remain in the gravel for 2 to 10 weeks then emerge into the water column as young juveniles, known as "fry". Juveniles, or fry, form schools in shallow water along the undercut banks of the stream to avoid predation. The juveniles feed heavily during this time, and as they grow they set up individual territories. Juveniles are voracious feeders, ingesting any organism that moves or drifts over their holding area. The juvenile's diet is mainly aquatic insect larvae and terrestrial insects, but small fish are taken when available (Moyle 2002a).

After 1 year in freshwater juvenile coho salmon undergo physiological transformation into "smolts" for outmigration to the ocean. Smolts may spend time residing in the estuarine habitat prior to ocean entry, to allow for the transition to the saline environment. After entering the ocean, the immature salmon initially remain in the nearshore waters close to their natal stream. They gradually move northward, generally staying over the continental shelf (Brown et al. 1994). After approximately 2 years at sea, adult coho salmon move slowly homeward. Adults begin their freshwater migration upstream after heavy fall or winter rains breach the sandbars at the mouths of coastal streams (Sandercock 1991) and/or flows are sufficient to reach upstream spawning areas.

Table 40. Temporal distribution of Coho salmon, central California coast ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Spawning</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Limited information exists on the abundance of coho salmon within the CCC coho salmon ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (Good et al. 2005b). This escapement declined to about 99,000 by the 1960s with approximately 56,000 (56%) originating from streams within the CCC coho salmon ESU. The estimated number of coho salmon produced within the ESU in 2011 was between 2,000 and 3,000 wild adults (Gallagher et al. 2010).

Productivity / Population Growth Rate. Within the Lost Coast – Navarro Point stratum, current population sizes range from 4% to 12% of proposed recovery targets, with 2 populations (Albion River and Big River, respectively) at or below their high-risk depensation thresholds. Most independent populations show positive but non-significant population trends. Dependent populations within the stratum have declined significantly since 2011. Similar results were obtained immediately south within the Navarro Point – Gualala Point stratum, where 2 of the 3 largest independent populations, the Navarro and Garcia rivers, have averaged 257 and 46 adult returns, respectively, during the 2005 – 2011 years (both populations are at or below their high-risk depensation threshold). Data from the 3 dependent populations within the stratum (Brush, Greenwood and Elk creeks) suggest little to no adult coho salmon escapement since 2011. In the Russian River and Lagunitas Creek watersheds, which are the 2 largest within the Central Coast strata, recent coho salmon population trends suggest limited improvement, although both populations remain well below recovery targets. Likewise, most dependent populations within the strata remain at very low levels, although excess broodstock adults from the Russian River and Olema Creek were recently stocked into Salmon Creek and the subsequent capture of juvenile fish indicates successful reproduction occurred. Finally, recent sampling within Pescadero Creek and San Lorenzo River, the only 2 independent populations within the Santa Cruz Mountains strata, suggest coho salmon have likely been extirpated within both basins. A bright spot appears to be the recent improvement in abundance and spatial distribution noted within the strata's dependent populations; Scott Creek experienced the largest coho salmon run in a decade during 2014/15, and researchers' recently detected juvenile coho salmon within 4 dependent watersheds where they were previously thought to be extirpated (San Vincente, Waddell, Soquel and Laguna creeks).

Genetic Diversity. Hatchery raised smolts have been released infrequently but occasionally in large numbers in rivers throughout the ESU (Bjorkstedt et al. 2005). Releases have included transfer of stocks within California and between California and other Pacific states as well as smolts raised from eggs collected from native stocks. However, genetic studies show little homogenization of populations, *i.e.*, transfer of stocks between basins have had little effect on the geographic genetic structure of CCC coho salmon (Sonoma County Water Agency (SCWA) 2002). The CCC coho salmon likely has considerable diversity in local adaptations given that the ESU spans a large latitudinal diversity in geology and ecoregions, and includes both coastal and inland river basins.

Distribution. The TRT identified 11 “functionally independent”, 1 “potentially independent” and 64 “dependent” populations in the CCC coho salmon ESU (Bjorkstedt *et al.*, 2005 with modifications described in Spence *et al.* 2008). The 75 populations were grouped into 5 Diversity Strata. ESU spatial structure has been substantially modified due to lack of viable source populations and loss of dependent populations. One of the 2 historically independent populations in the Santa Cruz mountains (*i.e.*, South of the Golden Gate Bridge) is extirpated (Good et al. 2005b; Spence et al. 2008a). Coho salmon are considered effectively extirpated from the San Francisco Bay (NMFS 2001; Spence et al. 2008a). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC coho salmon (NOAA 2013). The Russian River population, once the largest and most dominant source population in the ESU, is now at high risk of extinction because of low abundance and failed productivity (Spence et al. 2008a). The Lost Coast to Navarro Point to the north contains the majority of coho salmon remaining in the ESU.

Designated Critical Habitat. Critical habitat for the CCC coho salmon ESU was designated on May 5, 1999 (64 FR 24049). It encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda and the San Lorenzo River (inclusive) in California. Critical habitat for this species also includes 2 streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. PBFs considered essential for the conservation of Coho salmon, central California coast ESU are:

Within the range of both ESUs, the species’ life cycle can be separated into 5 essential habitat types:

- Juvenile summer and winter rearing areas;
- juvenile migration corridors;
- areas for growth and development to adulthood;
- adult migration corridors; and
- spawning areas.

Essential features of coho critical habitat include adequate

- substrate,
- water quality,
- water quantity,
- water temperature,
- water velocity,
- cover/shelter,

- food,
- riparian vegetation,
- space, and
- safe passage conditions.

NMFS (2008a) evaluated the condition of each habitat attribute in terms of its current condition relative to its role and function in the conservation of the species. The assessment of habitat for this species showed a distinct trend of increasing degradation in quality and quantity of all PBFs as the habitat progresses south through the species range, with the area from the Lost Coast to the Navarro Point supporting most of the more favorable habitats and the Santa Cruz Mountains supporting the least. However, all populations are generally degraded regarding spawning and incubation substrate, and juvenile rearing habitat. Elevated water temperatures occur in many streams across the entire ESU.

Recovery Goals See the 2012 Recovery Plan for complete down listing/delisting criteria for each of the following recovery goals (NMFS 2012b):

- Prevent extinction by protecting existing populations and their habitats;
- Maintain current distribution of coho salmon and restore their distribution to previously occupied areas essential to their recovery;
- Increase abundance of coho salmon to viable population levels, including the expression of all life history forms and strategies;
- Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta populations;
- Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally;
- Ensure all factors that led to the listing of the species have been ameliorated; and
- Develop and maintain a program of monitoring, research, and evaluation that advances understanding of the complex array of factors associated with coho salmon survival and recovery and which allows for adaptively managing our approach to recovery over time.

8.15 Coho salmon, Lower Columbia River ESU

Table 41. Coho salmon, lower Columbia River ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus kisutch	Coho salmon	Lower Columbia River	Threatened	2022	2005 70 FR 37160	2013	81 FR 9251

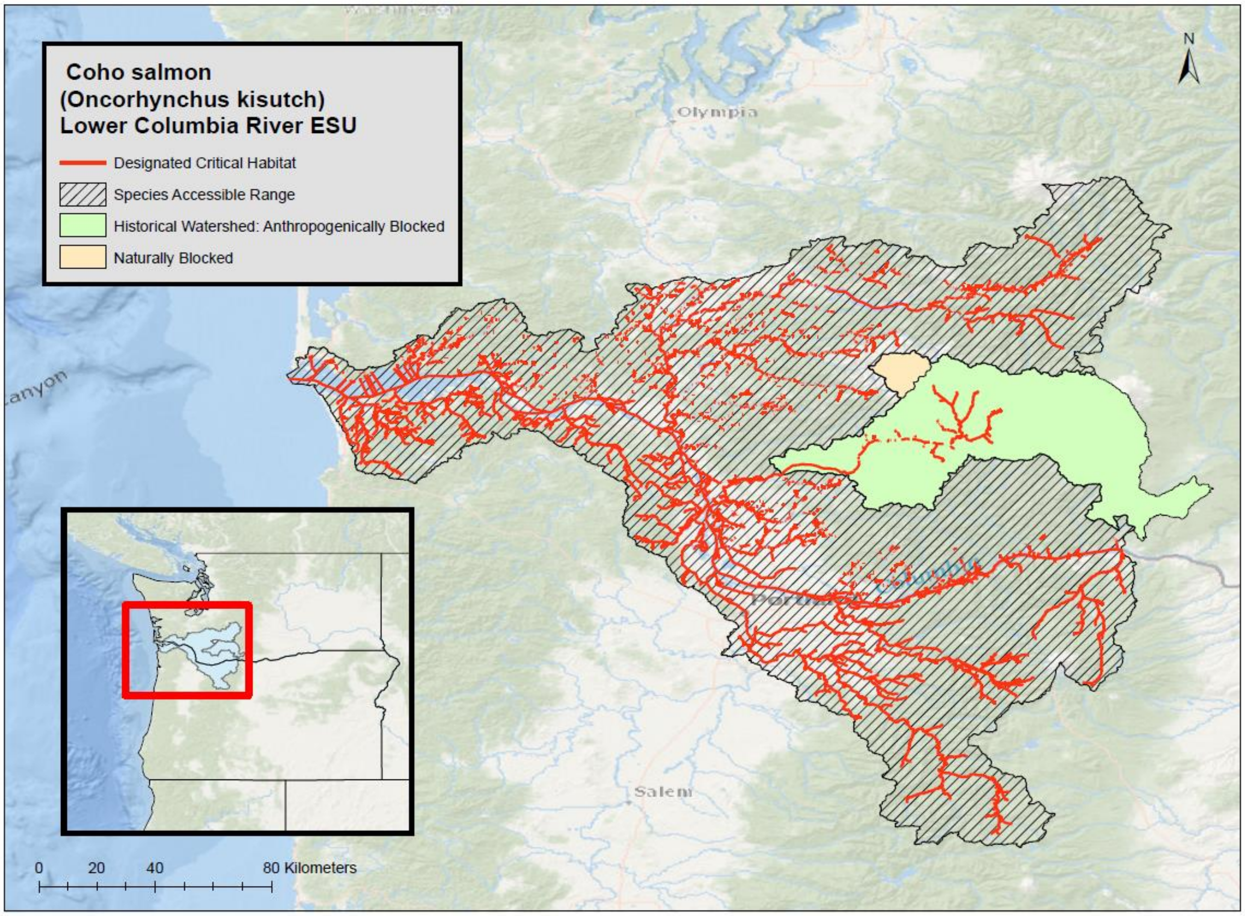


Figure 25**. Coho salmon, lower Columbia River ESU range and designated critical habitat**

Species Description. Coho salmon are an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn). Adult coho salmon are typically about 2 feet long and 8 pounds. Coho have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Lower Columbia River coho salmon ESU was listed as threatened under the ESA on June 28, 2005 (70 FR 37160). (Ford 2022) found that no new information had become available that would justify a change in the delineation of the LCR Coho Salmon ESU. This ESU includes naturally spawned coho salmon originating from the Columbia River and its tributaries downstream from the Big White Salmon and Hood Rivers (inclusive) and any such fish originating from the Willamette River and its tributaries below Willamette Falls. Myers et al. (2006) identified 3 MPGs (Coastal, Cascade, and Gorge), containing a total of 24 demographically independent populations (DIPs), in the Lower Columbia River coho salmon ESU. Also, coho salmon from 21 artificial propagation programs are included in this ESU.

Status. For the individual populations within the LCR Coho Salmon ESU, overall abundance trends for the ESU are generally negative. Natural spawner and total abundances have decreased in almost all DIPs. The risk of extinction spans the full range from low to very high. Overall, the LCR Coho Salmon ESU's risk of extinction was found to have increased since the previous review period. The NWFSC determined the ESU remains at moderate risk of extinction (Ford 2022). Abundances are still at low levels and the majority of the DIPs remain at moderate or high risk. For the lower Columbia River region, land development and increasing human population pressures will likely continue to degrade habitat, especially in lowland areas. Based on the 2022 Status Update, no reclassification for the LCR Coho Salmon ESU is warranted. Therefore, the LCR Coho Salmon ESU remain listed as threatened.

Life History. Lower Columbia River coho salmon are typically categorized into early- and late-returning stocks. Early-returning (Type S) adult coho salmon enter the Columbia River in mid-August and begin entering tributaries in early September, with peak spawning from mid-October to early November. Late-returning (Type N) coho salmon pass through the lower Columbia from late September through December and enter tributaries from October through January. Most spawning occurs from November to January, but some occurs as late as March (LCFRB 2010b).

Coho salmon typically spawn in small to medium, low- to-moderate elevation streams from valley bottoms to stream headwaters. Coho salmon construct redds in gravel and small cobble substrate in pool tailouts, riffles, and glides, with sufficient flow depth for spawning activity (NMFS 2013b). Eggs incubate over late fall and winter for about 45 to 140 days, depending on water temperature, with longer incubation in colder water. Fry may thus emerge from early spring to early summer (ODFW 2010). Juveniles typically rear in freshwater for more than a year. After emergence, coho salmon fry move to shallow, low-velocity rearing areas, primarily along the stream edges and inside channels. Juvenile coho salmon favor pool habitat and often congregate in quiet backwaters, side channels, and small creeks with riparian cover and woody debris. Side-channel rearing areas are particularly critical for overwinter survival, which is a key regulator of freshwater productivity (LCFRB 2010b).

Most juvenile coho salmon migrate seaward as smolts in April to June, typically during their second year. Salmon that have stream-type life histories, such as coho, typically do not linger for extended periods in the Columbia River estuary, but the estuary is a critical habitat used for feeding during the physiological adjustment to salt water. Juvenile coho salmon are present in the Columbia River estuary from March to August. Columbia River coho salmon typically range throughout the nearshore ocean over the continental shelf off of the Oregon and Washington coasts. Early-returning (Type S) coho salmon are typically found in ocean waters south of the Columbia River mouth. Late-returning (Type N) coho salmon are typically found in ocean waters north of the Columbia River mouth. Most coho salmon sexually mature at age 3, except for a small percentage of males (called “jacks”) who return to natal waters at age 2, after only 5 to 7 months in the ocean (LCFRB 2010b).

Table 42. Temporal distribution of Coho salmon, lower Columbia River ESU in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)	Present								Present			
Spawning	Present									Present		
Incubation (eggs)	Present									Present		
Emergence (alevin to fry phases)			Present									
Rearing and migration (juveniles)	Present											

Population Dynamics

Abundance. In contrast to the previous status review update, which occurred at a time of near-record returns for several populations, the ESU’s abundance has declined during the last 5 years. Only 6 of the 23 populations for which we have data appear to be above their recovery goals. This includes the Youngs Bay and Big Creek DIPs, which have very low recovery goals, and the Tilton River and Salmon Creek DIPs, which were not assigned goals but have relatively high abundances. Of the remaining DIPs in the ESU, 3 are at 50–99% of their recovery goals, 7 are at 10–50% of their recovery goals, and 7 are at <10% of their recovery goals (this includes the Lower Gorge DIP, for which there are no data, but it is assumed that the abundance is low). Hatchery production has been relatively stable, and the proportion of hatchery-origin fish on the spawning grounds has increased for some populations and decreased for others. The transition from segregated hatchery programs to integrated local broodstock programs should reduce the risks from domestication and non-native introgression. Spatial structure has improved incrementally, with improved passage programs at several major dams.

Overall abundance trends for the Lower Columbia River coho salmon ESU are generally negative. Natural spawner and total abundances have decreased in almost all DIPs, and Coastal and Gorge MPG populations are all at low levels, with significant numbers of hatchery-origin coho salmon on the spawning grounds. Improvements in spatial structure and diversity have been slight, and overshadowed by declines in abundance and productivity. In light of the poor ocean and freshwater conditions that occurred during much of this recent review period, it should be noted that some of the populations exhibited resilience and only experienced relatively small declines in abundance. Some populations were exhibiting positive productivity trends during the last year of review, representing the return of the progeny from the 2016 adult return. For individual populations, the risk of extinction spans the full range, from “low” to “very high.” Overall, the Lower Columbia River coho salmon ESU remains at “moderate” risk, and viability is largely unchanged from the prior status review.

Both short- and long-term trends for almost all coho salmon populations in the Coastal MPG were negative during the 2015–19 review period for 6 of the 7 Coastal MPG populations that were analyzed. Only the Mill/Abernathy/Germany DIP abundance was stable, with a 5-year geomean of 685. Average natural-origin abundances were in the hundreds of fish, with the exception of the Youngs Bay and likely Big Creek DIPs. Given the propensity of coho salmon to spawn in smaller tributaries and the year-long freshwater residence of juveniles, the poor freshwater conditions during this period likely affected coho salmon in the Coastal MPG more than in the larger rivers of the Cascade MPG.

Cascade MPG experienced a marked decline in abundance following the "boom" year of 2014. The 5-year geometric means for these populations were in the high hundreds to low thousands, with the exception of the Kalama River and Washougal River DIPs. Population trends were strongly negative, with the exception of the small Kalama River and Salmon Creek populations. The Salmon Creek DIP experienced a slight decline in 5-year geometric abundance (4% decline), but maintains a relatively high absolute abundance for a relatively small basin with a 5-year geomean of 1,546. While recent returns of unmarked fish to the Clackamas River have shown a marked decline since the 2014–15 record return year, when 10,670 spawners were counted, the 21% decline is one of the smallest in the MPG. The long-term (15-year) trend for this population is slightly positive, and the current 5-year geomean of 2,889 is still the largest abundance in the ESU. The 6 populations in the Cowlitz River basin account for the majority of naturally spawning coho salmon in the MPG, with the Lower Cowlitz River late coho salmon DIP 5-year geomean of 2,622. In the Cowlitz River basin, those coho salmon populations that relied on dam passage programs (Upper Cowlitz/Cispus Rivers and Tilton River) exhibited a greater decline relative to those populations located below the high-head dams (Lower Cowlitz River, North and South Fork Toutle rivers, and Coweeman River).
Natural-origin abundances in the Gorge MPG are low; the 2 populations available (Hood River, and Washington Upper Gorge Tributaries/White Salmon River) both had geomeans of less than 50 (Table 32). Hatchery-origin fish contribute a large proportion of the total number of spawners, most notably in the Hood River. The trend was strongly negative in the Hood River and slightly positive in the White Salmon River. With the exception of the Hood and White Salmon Rivers, much of the spawning habitat is in small independent tributaries to the Columbia River and, in many cases, the accessibility is relatively poor.

Productivity / Population Growth Rate. Both the long- and short-term trend, and lambda for the natural origin (late-run) portion of the Clackamas River coho salmon are negative but with large confidence intervals (Good et al. 2005b). The short-term trend for the Sandy River population is close to 1, indicating a relatively stable population during the years 1990 to 2002 (Good et al. 2005b). The long-term trend (1977 to 2002) for this same population shows that the population has been decreasing (trend=0.54); there is a 43% probability that the median population growth rate (lambda) was less than 1. Both short- and long-term trends for almost all coho salmon populations were negative during the 2015–19 review period for 6 of the 7 Coastal MPG populations that were analyzed (Table 32). Improvements in diversity and spatial structure noted in the 2022 Status Update have been slight and overshadowed by declines in abundances and productivity. In light of the poor ocean and freshwater conditions that occurred during much of this recent review period, it should be noted that some of the populations exhibited resilience and only experienced relatively small declines in abundance and even positive productivity trends during the last year of review (2019).

Genetic Diversity. The spatial structure of some populations is constrained by migration barriers (such as tributary dams) and development in lowland areas. Low abundance, past stock transfers, other legacy hatchery effects, and ongoing hatchery straying may have reduced genetic diversity within and among coho salmon populations (LCFRB 2010a, ODFW 2010). It is likely that hatchery effects have also decreased population productivity. Hatchery releases have remained relatively steady at 10–17 million since the 2005 BRT report, with approximately 14 million coho salmon juveniles released in 2019. Many of the populations in the ESU contain a substantial number of hatchery-origin spawners. Production has been shifted into localized areas (e.g., Youngs Bay, Big Creek, and Deep Creek) in order to reduce the influence of hatchery fish in other nearby populations.

Distribution. The Lower Columbia River coho salmon ESU historically consisted of a total of 24 independent populations. Because NMFS had not yet listed the ESU in 2003 when the WLC TRT designated core and genetic legacy populations for other ESUs, there are no such designations for Lower Columbia River coho salmon. However, the Clackamas and Sandy subbasins contain the only populations in the ESU that have clear records of continuous natural spawning (McElhany et al. 2007b).

Designated Critical Habitat. Critical habitat for the lower Columbia River coho salmon ESU was designated on February 24, 2016 (81 FR 9252). PBFs considered essential for the conservation of Coho salmon, lower Columbia River ESU are described in Appendix B.

Reduced complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migrating continues to be a concern for all 4 lower Columbia River listed species. Loss of habitat from conversion to agricultural or urbanized uses continues to be a particular concern throughout the lower Columbia River region, especially the loss of habitat complexity in the lower tributary/mainstem Columbia River interface, and concomitant changes in water temperature (LCFRB 2010b; NMFS 2013b; ODFW 2010). Toxic contamination through the production, use, and disposal of numerous chemicals from multiple sources including industrial, agricultural, medical and pharmaceutical, and common household uses that enter the Columbia River in wastewater treatment plant effluent, stormwater runoff, and nonpoint source pollution is a growing concern (Morace 2012).

Recovery Goals. NMFS has developed the following delisting criteria for the Lower Columbia River coho salmon ESU:

- All strata that historically existed have a high probability of persistence or have a probability of persistence consistent with their historical condition.

For populations within the below listed MPGs, NMFS recommend the following recovery actions over the next 5 years:

Coast MPGs

- Implement projects that increase the amount of side channel/pool rearing habitat for Grays/Chinook River coho.
- Promote projects that reduce flashy stream conditions to improve spawning habitat for Grays/Chinook River coho.
- Implement projects to increase summer and winter rearing habitat complexity for Mill/Abernathy/Germany Creek coho.

- Implement additional habitat improvement projects in the Elochoman River and Abernathy, Mill, and Germany creeks, and their tributaries to augment spawning (chum) and rearing (coho) habitat.

Cascade MPGs

- Reestablish and improve passage on multiple rivers to benefit multiple populations from the Cascade MPGs, such as the North Fork Lewis River (NF Lewis River spring Chinook, NF Lewis River winter steelhead, NF Lewis River coho), and Cowlitz River (Upper Cowlitz River spring Chinook, Upper Cowlitz River fall Chinook, Upper Cowlitz River coho, Upper Cowlitz River winter steelhead).
- Work with county and city jurisdictions to protect watershed hydrology from long-term development impacts (floodplain development and groundwater withdrawals). Focus these efforts on high growth rate watersheds along the I-5 and I-205 corridors, including the East Fork Lewis River, North Fork Lewis River, Coweeman River, Kalama River, Washougal River, Salmon Creek, and Lower Cowlitz tributaries.

Gorge MPGs

- Pacific salmon and steelhead recovery partners are encouraged to develop and implement a long-term management strategy to reduce pinniped predation on Pacific salmon and steelhead in the Columbia River basin by removing, reducing, and-or minimizing the use of manmade haul outs used by pinnipeds in select areas (e.g., river mouths/migratory pinch points).
- Pacific salmon and steelhead recovery partners are encouraged to expand, develop, and implement monitoring efforts in the Columbia River basin, to identify pinniped predation interactions in select areas (e.g., river mouths/migratory pinch points) and quantitatively assess predation impacts by pinnipeds on Pacific salmon and steelhead stocks.

8.16 Coho salmon, Oregon Coast ESU

Table 43. Coho salmon, Oregon coast ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus kisutch	Coho salmon	Oregon Coast	Threatened	2022	2011 76 FR 35755	2016	2008 73 FR 7816

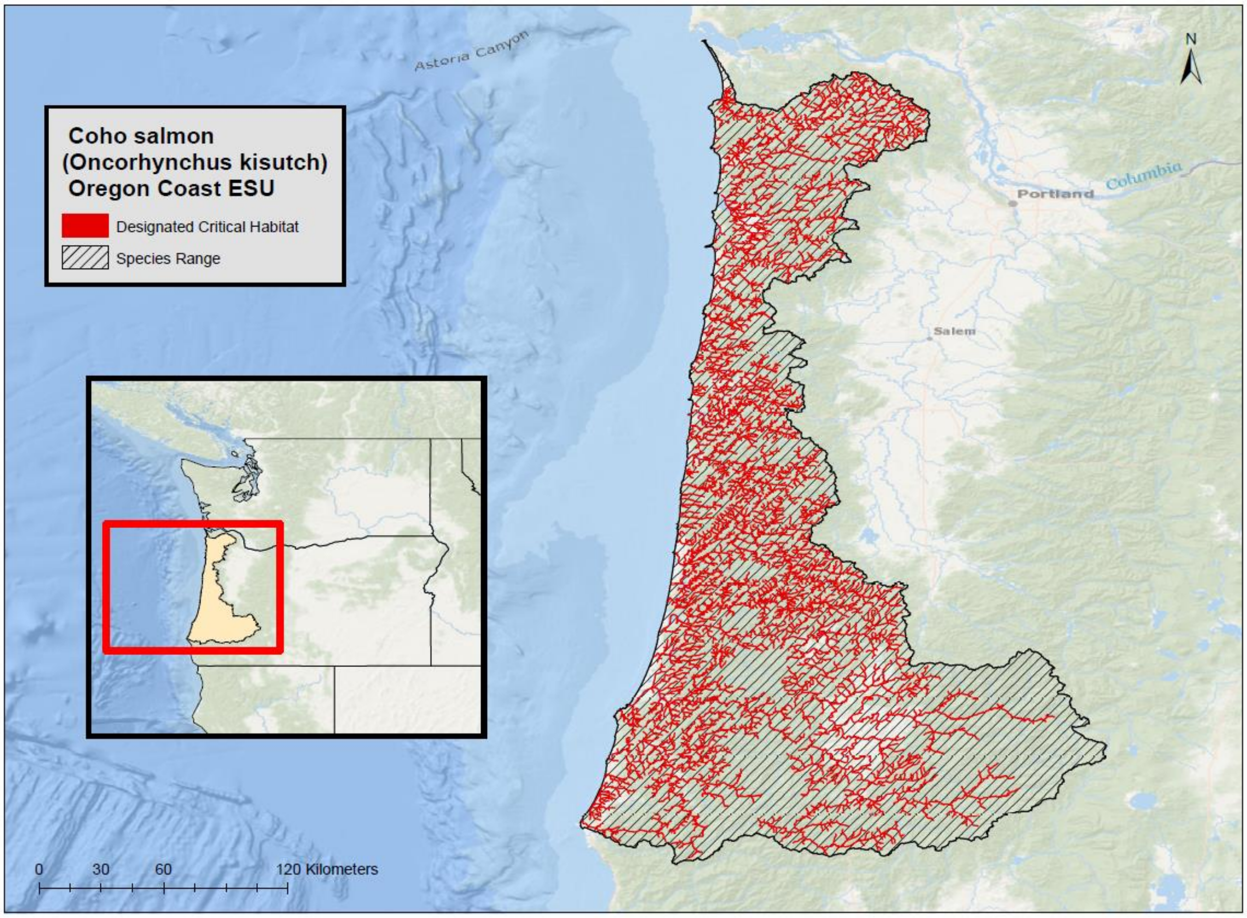


Figure 26**. Coho salmon, Oregon coast ESU range and designated critical habitat**

Species Description. Coho salmon are an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn). Adult coho salmon are typically about 2 feet long and 8 pounds. Coho have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Oregon coast coho salmon ESU was listed as threatened under the ESA on August 10, 1998 (63 FR 42587). The 2022 Status Update found no new information that would justify a change in the delineation of the OC coho salmon ESU (Ford 2022). The listing was revisited and confirmed as threatened on June 20, 2011 (76 FR 35755). This ESU consists of 5 strata and includes naturally spawned coho salmon originating from coastal rivers south of the Columbia River and north of Cape Blanco, and also coho salmon from 1 artificial propagation program: Cow Creek Hatchery Program. The strata include: the North Coast, the Mid Coast, the Lakes, the Umpqua, and the Mid-South Coast. These strata in turn are made up of several independent populations and each must meet viability standards.

Status. Findings by the NWFSC (2015a) and ODFW (2016) show many positive improvements to Oregon Coast coho salmon in recent years, including positive long-term abundance trends and escapement. Results from the NWFSC recent review show that while Oregon Coast coho salmon spawner abundance varies by time and population, the total abundance of spawners within the ESU has been generally increasing since 1999, with total abundance exceeding 280,000 spawners in 3 of the last 5 years. Overall, the NWFSC (2015a) found that increases in Oregon Coast coho salmon ESU scores for persistence and sustainability clearly indicate that the biological status of the ESU is improving, due in large part to management decisions (reduced harvest and hatchery releases). It determined, however, that Oregon Coast coho salmon abundance remains strongly correlated with marine survival rates. After considering the biological viability of the OC coho salmon ESU during the 2015-2020 downturn in ocean conditions and low marine productivity with the current status of its ESA section 4(a)(1) factors, the 2022 assessment concludes that the risk to the species' persistence has improved since the 2016 5-year review. However, further implementation of sound management actions, habitat restoration and protection efforts must continue to improve population and species viability.

Life History. The anadromous life cycle of coho salmon begins in their home stream where they emerge from eggs as 'alevins' (a larval life stage dependent on food stored in a yolk sac). These very small fish require cool, slow moving freshwater streams with quiet areas such as backwater pools, beaver ponds, and side channels (Reeves et al. 1989) to survive and grow through summer and winter seasons. Current production of coho salmon smolts in the Oregon Coast coho salmon ESU is particularly limited by the availability of complex stream habitat that provides the shelter for overwintering juveniles during periods when flows are high, water temperatures are low, and food availability is limited (ODFW 2007).

The Oregon Coast coho salmon follow a yearling-type life history strategy, with most juvenile coho salmon migrating to the ocean as smolts in the spring, typically from as late as March into June. Coho salmon smolts outmigrating from freshwater reaches may feed and grow in lower mainstem and estuarine habitats for a period of days or weeks before entering the nearshore ocean environment. The areas can serve as acclimation areas, allowing coho salmon juveniles to adapt to saltwater. Research shows that substantial numbers of coho fry may also emigrate downstream from natal streams into tidally influenced lower river wetlands and estuarine habitat (Bass 2010; Chapman 1962; Koski 2009).

Oregon Coast coho salmon tend to make relatively short ocean migrations. Coho from this ESU are present in the ocean from northern California to southern British Columbia, and even fish from a given population can be widely dispersed in the coastal ocean, but the bulk of the ocean harvest of coho salmon from this ESU are found off the Oregon coast. The majority of coho salmon adults return to spawn as 3–year-old fish, having spent about 18 months in freshwater and 18 months in saltwater (Sandercock 1991). The primary exceptions to this pattern are ''jacks,'' sexually mature males that return to freshwater to spawn after only 5 to 7 months in the ocean.

Table 44. Temporal distribution of Coho salmon, Oregon coast ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics - Updated Biological Risk Summary (from the 2022 Status Review). The ODFW's 12-year assessment of the Oregon Coast Coho Conservation Plan (ODFW 2021) highlights favorable improvements for OC coho salmon overall, consistent with the Ford (2022) assessment. It notes the strong role that ocean conditions play on adult returns to the ESU, including recent low abundances associated with strong marine heat waves.

The latest ESU scores for persistence (high certainty of ESU persistence) and sustainability (low to moderate certainty of ESU sustainability) also demonstrate that the biological status of the ESU has decreased slightly since the 2016 review (high certainty of persistence, moderate certainty of sustainability), which covered a period of favorable ocean conditions and high marine survival rates. However, current ESU scores have improved relative to the 2012 assessment (moderate certainty of persistence, low to moderate certainty of sustainability). This improvement occurred despite similar or better abundances and marine survival rates during the earlier period, suggesting continued benefits due to management decisions to reduce both harvest and hatchery releases.

Despite these somewhat optimistic results for OC coho salmon, it is unclear what the future will bring. A recent assessment of the vulnerability of ESA-listed salmonid species to climate change indicated that OC coho salmon had high overall vulnerability, high biological sensitivity and climate exposure, but only moderate adaptive capacity (Crozier et al. 2019). Because young coho spend a full year in freshwater before ocean entry, the juvenile freshwater stage was considered to be highly vulnerable. The ESU also scored high in sensitivity at the marine stage due to expected changes due to ocean acidification. These results are consistent with the climate change assessment by Wainwright and Weitkamp (2013), which indicated OC coho salmon will likely be negatively affected by climate change at all stages of the life cycle. Overall, the OC coho salmon ESU is at moderate-to-low risk of extinction, with viability largely unchanged from the prior review.

Abundance. The spawner abundance within the Oregon Coast coho salmon ESU varies by time and population. The large populations (abundances >6,000 spawners since 2015) include Nehalem, Tillamook Bay, Alsea, Siuslaw, Lower Umpqua, Coos, and Coquille. The total abundance of spawners within the ESU generally increased between 1999 and 2014, before dropping in 2015 and remaining low. Five-year geometric mean natural raw spawner abundances increased from 17–7,228 per population in the 1990–94 time period to 189–23,741 for the 2010–14 time period, the highest in the time series. Populations decreased during the most recent period (2015–19), to 67–6,740. All populations exhibited a substantial decrease in the geometric mean abundance between the previous 5-year period (2010–14) and the current one (2015–19), ranging from –55% (Siletz) to –75%.

Productivity / Population Growth Rate.
Patterns of natural spawner abundances, including short- and long-term trends for dependent populations in the North and Mid Coasts, were similar to larger independent populations. Short-term trends declined by –59% and –51%, respectively, between the 2 5-year periods (2010–14 and 2015–19; Table 65). Long-term trends were slightly positive (0.02 and 0.05, respectively) during the 2004–19 period, although the 15-year trend confidence intervals included zero. Spawner-to-spawner ratios show the same cycle of positive and negative periods displayed by independent populations. Given that small populations are more likely to "wink out" than large populations due to stochastic processes, these patterns suggest small dependent populations on the Oregon Coast were not unduly impacted by unfavorable ocean conditions and respond much like their larger neighbors. This synchrony suggests the overriding importance of marine survival to recruitment and escapement of Oregon Coast coho salmon (NWFSC 2015a).

Genetic Diversity. While the 2008 biological review team status review concluded that there was low certainty that ESU-level genetic diversity was sufficient for long-term sustainability in the ESU (Wainwright et al. 2008), the recent NWFSC review suggests this is an unlikely outcome. The observed upward trends in abundance and productivity and downward trends in hatchery influence make decreases in genetic or life history diversity or loss of dependent populations in recent years unlikely (NWFSC 2015a).

Distribution. The geographic setting for the Oregon Coast coho salmon ESU includes the Pacific Ocean and the freshwater habitat (rivers, streams, and lakes) along the Oregon Coast from the Necanicum River near Seaside on the north to the Sixes River near Port Orford on the south. The Oregon/Northern California Coasts Technical Recovery Team identified 56 historical populations that function collectively to form the Oregon Coast coho salmon ESU. The team classified 21 of the populations as independent because they occur in basins with sufficient historical habitat to have persisted through several hundred years of normal variations in marine and freshwater conditions (NMFS 2016f).

Designated Critical Habitat. NMFS designated critical habitat for Oregon Coast coho salmon on February 11, 2008 (73 FR 7816). PBFs considered essential for the conservation of Coho salmon, Oregon coast ESU are described in Appendix B.

Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation, and larval development.

The spawning PBF has been impacted in many watersheds from the inclusion of fine sediment into spawning gravel from timber harvest and forestry related activities, agriculture, and grazing. These activities have also diminished the channels' rearing and overwintering capacity by reducing the amount of large woody debris in stream channels, removing riparian vegetation, disconnecting floodplains from stream channels, and changing the quantity and dynamics of stream flows. The rearing PBF has been degraded by elevated water temperatures in 29 of the 80 HUC 5 watersheds; rearing PBF within the Nehalem, North Umpqua, and the inland watersheds of the Umpqua subbasins have elevated stream temperatures. Water quality is impacted by contaminants from agriculture and urban areas in low lying areas in the Umpqua subbasins, and in coastal watersheds within the Siletz/Yaquina, Siltcoos, and Coos subbasins. Reductions in water quality have been observed in 12 watersheds due to contaminants and excessive nutrition. The migration PBF has been impacted throughout the ESU by culverts and road crossings that restrict passage. As described above the PBFs vary widely throughout the critical habitat area designated for OC coho salmon, with many watersheds heavily impacted with low quality PBFs while habitat in other coho salmon bearing watersheds having sufficient quality for supporting the conservation purpose of designated critical habitat. While marine survival is important for Oregon Coast coho salmon (Falcy and Suring 2018), so is high-quality freshwater habitat for juvenile rearing, adult spawning, and egg incubation. These are also habitats that humans control and influence through their actions across the landscape. Of particular importance to Oregon Coast coho salmon BRTs and TRTs (Wainwright et al. 2008, Stout et al. 2012) are the need for American beaver (*Castor canadensis*) within the ESU. Beavers are a keystone species that has wide-ranging impacts on stream ecosystems, because their dams create pools that serve as high-quality habitat for a number of plant, invertebrate, and vertebrate species, including juvenile coho salmon. However, widespread historical removal of beavers has resulted in beaver populations that are a small fraction of their historical abundance (Pollock et al. 2003, 2015). Loss of high-quality beaver-associated habitat has been identified as limiting the production of Oregon Coast coho salmon (see review in Stout et al. 2012).

Recovery Goals. See the 2016 Recovery Plan for detailed descriptions of the recovery goals and delisting criteria (NMFS 2016f). In the simplest terms, NMFS will remove the Oregon Coast coho salmon from federal protection under the ESA when we determine that:

- The species has achieved a biological status consistent with recovery—the best available information indicates it has sufficient abundance, population growth rate, population spatial structure, and diversity to indicate it has met the biological recovery goals.
- Factors that led to ESA listing have been reduced or eliminated to the point where federal protection under the ESA is no longer needed, and there is reasonable certainty that the relevant regulatory mechanisms are adequate to protect Oregon Coast coho salmon sustainability.

8.17 Coho salmon, Southern Oregon/Northern California Coast ESU

Table 45. Coho salmon, Southern Oregon/Northern California ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus kisutch	Coho salmon	Southern Oregon / Northern California Coast	Threatened	2016	2005 70 FR 37160	2014	1999 64 FR 24049

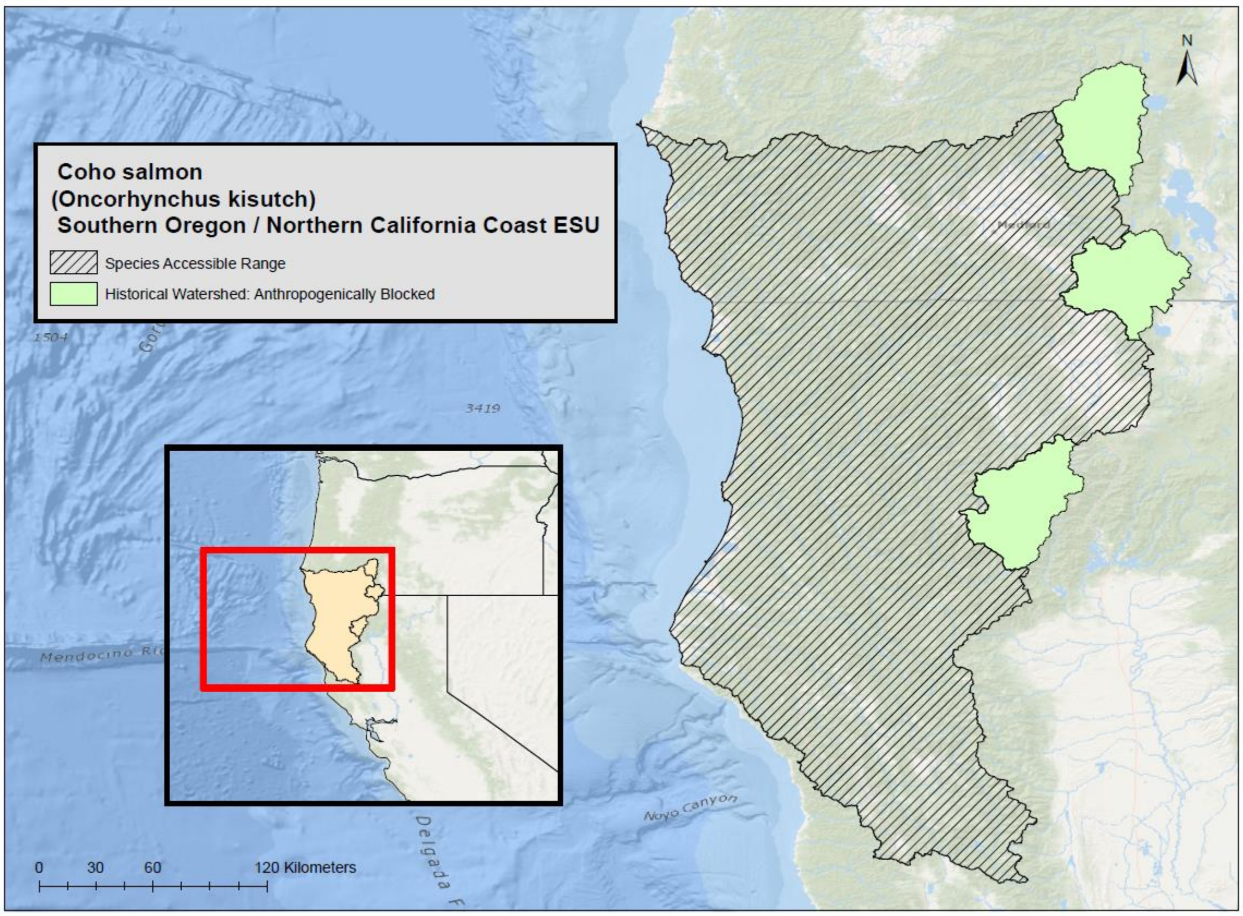


Figure 27**. Coho salmon, Southern Oregon/Northern California Coast ESU range and designated critical habitat**

Species Description. Coho salmon are an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn). Adult coho salmon are typically about 2 feet long and 8 pounds. Coho have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Southern Oregon / Northern California Coast (SONCC) coho salmon ESU was listed as threatened under the ESA on May 6, 1997 (62 FR 24588). The listing was revisited and confirmed as threatened on June 28, 2005 (70 FR 37160). This ESU includes naturally spawned coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California. Also, coho salmon from 3 artificial propagation programs.

Status. Though population-level estimates of abundance for most independent populations are lacking, the best available data indicate that none of the 7 diversity strata appears to support a single viable population as defined by the SONCC coho salmon technical recovery team's viability criteria (low extinction risk; Williams et al. (2008b)). Further, 24 out of 31 independent populations are at high risk of extinction and 6 are at moderate risk of extinction. Based on the above discussion of the population viability parameters, and qualitative viability criteria presented in Williams et al. (2008b), NMFS concludes that the SONCC coho salmon ESU is currently not viable and is at high risk of extinction. The primary causes of the decline are likely long-standing human-caused conditions (e.g., harvest and habitat degradation), which exacerbated the impacts of adverse environmental conditions (e.g., drought and poor ocean conditions) (60 FR 38011; July 25, 1995).

Life History. Coho salmon is an anadromous fish species that generally exhibits a relatively simple 3-year life cycle. Adults typically begin their freshwater spawning migration in the late summer and fall, spawn by mid-winter, and then die. The run and spawning times vary between and within populations. Depending on river temperatures, eggs incubate in ''redds'' (gravel nests excavated by spawning females) for 1.5 to 4 months before hatching as ''alevins'' (a larval life stage dependent on food stored in a yolk sac). Once most of the yolk sac is absorbed, the 30 to 35 millimeter fish (then termed "fry") begin emerging from the gravel in search of shallow stream margins for foraging and safety (Council 2004). Coho salmon fry typically transition to the juvenile stage by about mid-June when they are about 50 to 60 mm, and both stages are collectively referred to as "young of the year." Juveniles develop vertical dark bands or "parr marks", and begin partitioning available instream habitat through aggressive agonistic interactions with other juvenile fish (Quinn 2005). Juveniles rear in fresh water for up to 15 months, then migrate to the ocean as ''smolts'' in the spring. Coho salmon typically spend 2 growing seasons in the ocean before returning to their natal stream to spawn as 3 year-olds. Some precocious males, called ''jacks,'' return to spawn after only 6 months at sea (NMFS 2014c).

Table 46. Temporal distribution of Coho salmon, Southern Oregon/Northern California ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Incubation (eggs)</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Population-level estimates of abundance for most independent populations are lacking. The best available data indicate that none of the 7 diversity strata appears to support a single viable population (1 at low risk of extinction) as defined by in the viability criteria. In fact, most of the 30 independent populations in the ESU are at high risk of extinction for abundance because they are below or likely below their depensation threshold (NMFS 2014c).
Productivity / Population Growth Rate. Available data show that the 95% confidence intervals for the slope of the population growth rate regression line include zero for many populations, indicating that whether the slope is negative or positive cannot be determined. However, there is 95% confidence that the slope of the regression line is negative, indicating a decreasing trend, for Mill Creek in the Smith River and Freshwater Creek in Humboldt Bay Tributaries. In contrast, there is 95% confidence that the slope of the regression line is positive, indicating an increasing trend, at Gold Ray Dam in the Upper Rogue River(NMFS 2014c).

Genetic Diversity. The primary factors affecting the genetic and life-history diversity of SONCC coho salmon appear to be low population abundance and the influence of hatcheries and out-of-basin introductions. The ESU's current genetic variability and variation in life-history likely contribute significantly to long-term risk of extinction. Given the recent trends in abundance across the ESU, the genetic and life-history diversity of populations is likely very low and is inadequate to contribute to a viable ESU (NMFS 2014c).

Distribution. The SONCC Coho Salmon ESU includes all naturally spawned populations of coho salmon in coastal streams between Cape Blanco, Oregon and Punta Gorda, California, as well as coho salmon produced by 3 artificial propagation programs: Cole Rivers Hatchery, Trinity River Hatchery, and Iron Gate Hatchery. The ESU is comprised of 40 populations within 7 diversity strata. Recent information for SONCC coho salmon indicates that their distribution within the ESU has been reduced and fragmented, as evidenced by an increasing number of previously occupied streams from which they are now absent. However, extant populations can still be found in all major river basins within the ESU (70 FR 37160; June 28, 2005).

Designated Critical Habitat. NMFS designated critical habitat for the SONCC coho salmon on May 5, 1999 (64 FR 24049). PBFs considered essential for the conservation of Coho salmon, Southern Oregon/Northern California ESU are shown in Appendix B.

Critical habitat designated for the SONCC coho salmon is generally of good quality in northern coastal streams. Spawning PBF has been degraded throughout the ESU by logging activities that have increased fines in spawning gravel. Rearing PBF has been considerably degraded in many inland watersheds from the loss of riparian vegetation resulting in unsuitably high water temperatures. Rearing and juvenile migration PBFs have been reduced from the disconnection of floodplains and off-channel habitat in low gradient reaches of streams, consequently reducing winter rearing capacity.

Recovery Goals. See the 2014 recovery plan for complete down listing/delisting criteria for this ESU (NMFS 2014c).

8.18 Sockeye salmon, Ozette Lake ESU

Table 47. Sockeye salmon, Ozette Lake ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus nerka	Sockeye salmon	Ozette Lake	Threatened	2022	2005 70 FR 37160	2009	2005 70 FR 52630

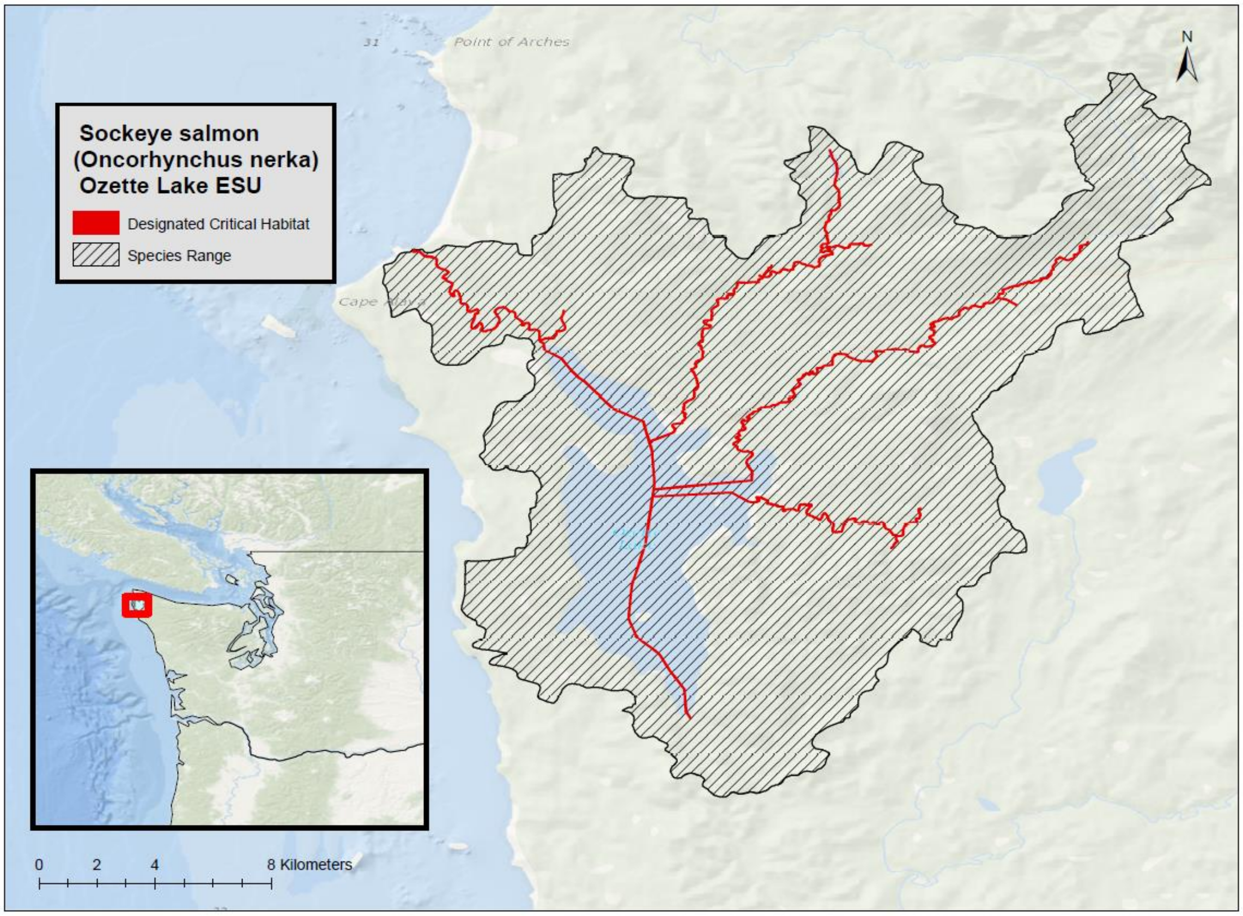


Figure 28**. Sockeye salmon, Ozette Lake ESU range and designated critical habitat**

Species Description. The sockeye salmon is an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn), although some sockeye spend their entire lives (about 5 years) in freshwater. Adult sockeye salmon are about 3 feet long and 8 pounds. Sockeyes are bluish black with silver sides when they are in the ocean, and they turn bright red with a green head when they are spawning. On March 25, 1999, NMFS listed the Ozette Lake sockeye salmon ESU as threatened (64 FR 14528) and reaffirmed the ESU's status as threatened on June 28, 2005 (70 FR 37160). The 2022 Status Update found no new information that would justify a change in the delineation of the Ozette Lake Sockeye ESU (Ford 2022).This ESU includes naturally spawned sockeye salmon originating from the Ozette River and Ozette Lake and its tributaries. However, since the 2016 update, NMFS combined the Umbrella Creek Hatchery Program and Big River Hatchery Program, which are included in the ESU, into 1 program called the Umbrella Creek/Big River Hatchery Program. This integrated program uses broodstock from Umbrella Creek that were derived from natural-origin fish from Lake Ozette and releases fish into the Umbrella Creek and Big River subwatersheds (85 FR 81822, December 17, 2020).

Status. NMFS listed the Ozette Lake sockeye salmon ESU because of habitat loss and degradation from the combined effects of logging, road building, predation, invasive plant species, and overharvest. Ozette Lake sockeye salmon have not been commercially harvested since 1982 and only minimally harvested by the Makah Tribe since 1982 (0 to 84 fish per year); there is no known marine fishing of this ESU. Overall abundance is substantially below historical levels, and whether the decrease in abundance is a result of fewer spawning aggregations, lower abundances in each aggregation, or a combination of both factors is unknown. Regardless, this ESU’s viability has not improved, and the ESU would likely have a low resilience to additional perturbations. However, recovery potential for the Ozette Lake sockeye salmon ESU is good, particularly because of protections afforded it based on the lake’s location within a national park (NMFS 2009d).

Life History. Most sockeye salmon exhibit a lake-type life history (i.e., they spawn and rear in or near lakes), though some exhibit a river-type life history. Spawning generally occurs in late summer and fall, but timing can vary greatly among populations. In lakes, sockeye salmon commonly spawn along “beaches” where underground seepage provides fresh oxygenated water. Females spawn in 3 to 5 redds (nests) over a couple of days. Incubation period is a function of water temperature and generally lasts 100-200 days (Burgner 1991). Sockeye salmon spawn once, generally in late summer and fall, and then die (semelparity).

Sockeye salmon fry primarily rear in lakes; river-emerged and stream-emerged fry migrate into lakes to rear. In the early fry stage from spring to early summer, juveniles forage exclusively in the warmer littoral (i.e., shoreline) zone where they depend mostly on fly larvae and pupae, copepods, and water fleas. Sub-yearling sockeye salmon move from the littoral habitat to a pelagic (i.e., open water) existence where they feed on larger zooplankton; however, flies may still make up a substantial portion of their diet. From 1 to 3 years after emergence, juvenile sockeye salmon generally rear in lakes, though some river-spawned sockeye may migrate to sea in their first year. Juvenile sockeye salmon feeding behaviors change as they transition through life stages after emergence to the time of smoltification. Distribution in lakes and prey preference is a dynamic process that changes daily and yearly depending on many factors including water temperature, prey abundance, presence of predators and competitors, and size of the juvenile. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye populations (lower than 52ºN latitude) and as late as early July in northern populations (62ºN latitude) (Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending 1 to 4 years at sea. The diet of adult salmon consists of amphipods, copepods, squid and other fish.

Table 48. Temporal distribution of Sockeye salmon, Ozette Lake ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td>Present</td><td></td><td></td><td colspan="9">Present</td></tr>
<tr><td>Spawning</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Incubation (eggs)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Based on an evolving understanding of both the status of the VSP parameters and the uncertainty in the status of the Ozette Lake sockeye beach spawning aggregates, there appears to be an increase in biological risk for Ozette Lake sockeye (NMFS 2022a).

Abundance. The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 spawners (Blum 1988). Kemmerich (Kemmerich 1945), reported a decline in the run size since the 1920s weir counts and Makah Fisheries Management (Makah Fisheries Management 2000) concluded a substantial decline in the Tribal catch of Ozette Lake sockeye salmon occurred at the beginning of the 1950s. Whether decrease in abundance compared to historic estimates is a result of fewer spawning aggregations, lower abundances at each aggregation, or both, is unknown (Good et al. 2005b).

For the period from 1977 to 2019, the estimated natural spawners ranged from 438 to 12,829, well below the 31,250–121,000 viable population range set in the Ozette Lake sockeye salmon recovery plan (NMFS 2009b). There remains little evidence of a strong trend in the raw or smoothed abundance series—over the full range of years, or more recently—since the last status review (NWFSC 2015). However, the geometric mean of abundance from 2015 to 2019 was higher than the previous 5-year geometric mean, and the trend over the last 15 years has been positive. Based on an evolving understanding of both the status and the uncertainty in the status of the Ozette Lake sockeye salmon beach-spawning aggregates, we believe the biological risk for Ozette Lake sockeye salmon has increased somewhat compared to prior reviews, largely due to a clearer understanding of the poor condition of the beach-spawning aggregates. For the last 4 decades, the abundance of Ozette Lake sockeye natural adult spawners ranged from 438 to 12,829, well below the lower viability threshold of 31,250 – 121,000 viable population range established in the 2009 NMFS Technical Memorandum (Rawson et al. 2009) and the 2009 recovery plan. The VSP criteria for Abundance remain unmet.

Productivity / Population Growth Rate. The Ozette Lake sockeye salmon ESU is composed of one historical population (Currens et al. 2009) with multiple spawning aggregations and 2 populations from the Umbrella Creek/Big River Sockeye Hatchery Program. Historically, at least 4 lake beaches were used for spawning; today only 2 beach spawning locations, Allen's and Olsen's Beaches, are used. Additionally, spawning occurs in the 2 tributaries of the hatchery programs (NWFSC 2015b). The historical abundance of Ozette Lake sockeye salmon is poorly documented, but it may have been as high as 50,000 individuals (Blum 1988). Declines began to be reported in the 1920s. For the period from 1977 to 2011 the estimated annual number of natural spawners ranged from 699 to 5,313, well below the 31,250 – 121,000 viable population range proposed in the Lake Ozette sockeye recovery plan (Haggerty et al. 2009). There is some evidence of the dominant 4-year age of return in the abundance series, with the 1980 brood cycle line surpassing the other lines in the late 80s and maintaining this higher level for most 4-year cycles since. Estimated productivity, calculated as the abundance in year *t* divided by the abundance in year (*t* – 4), has shifted between negative and positive values with a suggestive 10–20-year cycle. There are sufficient data to determine that the total Ozette Lake abundance is well below the desired lower bound, although the population has increased since the last review and over the past 15 years. Over the last few decades, productivity for the total Ozette Lake population has exhibited a 10–20-year cyclical pattern alternating between negative and positive values. Average rates over the last 5- and 15-year periods have been slightly positive, although we may be entering a negative phase.

As stated above, over the last few decades, estimated productivity for the total Ozette Lake population has alternated between positive and negative periods. While estimated average productivity over the most recent 5- and 15-year periods has been positive, based on historical cyclical patterns in productivity, Ozette Sockeye may now be entering another negative phase. Previous downturns have generally lasted 4 to 8 years (Ford 2022). This cyclical pattern in productivity makes it difficult to interpret historical trends and predict future trends in productivity and may increase risk due to the potential for sustained periods of negative productivity. Because of the cyclical pattern of positive and negative productivity over the years, there is no evidence of sustained increases in productivity of the population, much less the necessary VSP “productivity criterion” growth rate of greater than 1 until the ESU achieves a viable abundance. The VSP criteria for Productivity remain unmet.

Genetic Diversity. For the Ozette Lake sockeye salmon ESU, it appears that the Umbrella Creek hatchery program has successfully introduced a tributary spawning aggregate. This has increased the spatial and possibly genetic structure of the population while maintaining a genetic reservoir initially established with beach-spawning fish. The addition of the tributary aggregate may have increased or stabilized overall abundance, although this is not yet confirmed by the abundance trends. However, Ozette Lake sockeye have a relatively low genetic diversity compared to other sockeye salmon populations examined in Washington State (Crewson et al. 2001). Genetic differences do occur among age cohorts. However, because different age groups do not reproduce together, the population may be more vulnerable to significant reductions in population structure due to catastrophic events or unfavorable conditions affecting a single year class. Finally, actions identified in the Ozette Lake Sockeye Salmon Hatchery and Genetics Management Plan are being implemented, but the tributary hatchery reintroduction program will not reduce genetic diversity in the natural beach spawning aggregation because there is very little straying of hatchery-origin fish to beach spawning areas (NOAA 2016a). The estimated fraction of hatchery-origin fish returning to Ozette Lake has averaged only 6% in recent years (2000–18). However, the large contribution of the hatchery-supplemented tributary aggregations to the population as a whole allows for larger total hatchery fractions when Umbrella Creek hatchery fraction is high. For example, in 2012, over half (52%) of the estimated 5,152 fish returning to Umbrella Creek were designated as hatchery-origin.

Distribution. The Ozette Lake sockeye salmon ESU includes all naturally spawned aggregations of sockeye salmon in Lake Ozette and streams and tributaries flowing into Lake Ozette, Washington. The ESU also includes fish originating from the Umbrella Creek/Big River Sockeye Hatchery Program. It once appeared that the Umbrella Creek Hatchery program had successfully introduced a tributary spawning aggregate, thereby increasing the spatial and possibly genetic structure of the population, while maintaining a genetic reservoir initially established with beach spawning fish. However, there is accumulating evidence of a sustained reduction in abundance and distribution of beach spawners, aggravating the conditions originally identified by the PSTRT that “the limited distribution of Lake Ozette sockeye spawners [at that time] put the ESU at high risk.” Critical gaps in our knowledge of the beach spawning aggregates prevent any quantitative assessment of abundance or trends for these beach spawning population aggregates, which are considered critical for recovery of the single population Ozette Lake Sockeye Salmon ESU. The VSP criteria for Spatial Structure remain unmet.

Designated Critical Habitat. NMFS designated critical habitat for Ozette Lake sockeye salmon on September 2, 2005 (70 FR 52630). It encompasses areas within the Hoh/Quillayute subbasin, Ozette Lake, and the Ozette Lake watershed. PBFs considered essential for the conservation of Sockeye salmon, Ozette Lake ESU are described in Appendix B.

Spawning habitat has been affected by loss of tributary spawning areas and exposure of much of the available beach spawning habitat due to low water levels in summer. Further, native and non-native vegetation as well as sediment have reduced the quantity and suitability of beaches for spawning. The rearing PBF is degraded by excessive predation and competition with introduced non-native species, and by loss of tributary rearing habitat. Migration habitat may be adversely affected by high water temperatures and low water flows in summer which causes a thermal block to migration (La Riviere 1991).

Recovery Goals. Recovery goals, objectives and criteria for Ozette Lake sockeye salmon are fully outlined in the 2009 recovery plan (NMFS 2009b). The proposed viability criteria indicates a goal of 31,250 to 121,000 spawners.

8.19 Sockeye salmon, Snake River ESU

Table 49. Sockeye salmon, Snake River ESU; overview table

Species	Common Name	ESU	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus nerka	Sockeye salmon	Snake River	Endangered	2022	2005 70 FR 37160	2015	1993 58 FR 68543

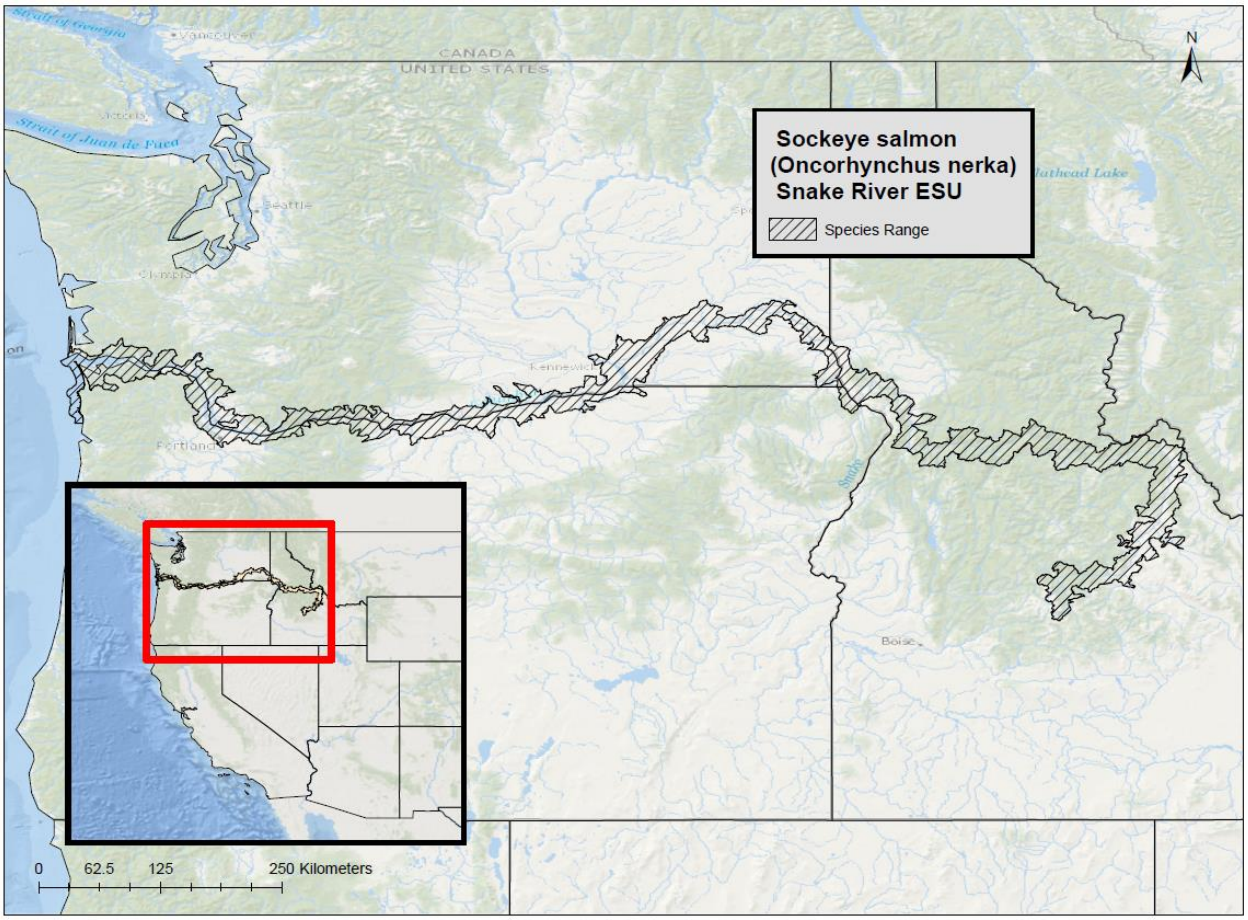


Figure 29. Sockeye salmon, Snake River ESU range and designated critical habitat

Species Description. The sockeye salmon is an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn), although some sockeye spend their entire lives (about 5 years) in freshwater. Adult sockeye salmon are about 3 feet long and 8 pounds. Sockeyes are bluish black with silver sides when they are in the ocean, and they turn bright red with a green head when they are spawning. On November 20, 1991 NMFS listed the Snake River sockeye salmon ESU as endangered (70 FR 37160) and reaffirmed the ESU's status as endangered on June 28, 2005 (70 FR 37160). This ESU includes naturally spawned anadromous and residual sockeye salmon originating from the Snake River basin, as well as sockeye salmon from the Redfish Lake Captive Broodstock Program and the Snake River Sockeye Salmon

Hatchery Program (USOFR 2005a, 2020, 85 FR 81822). Since 2016 update, NMFS added a new smolt production program to the ESU because the Redfish Lake Captive Broodstock Program currently produces the eggs used in the new smolt production program. Therefore, the smolts produced for this new hatchery program are a category 1a (Jones 2015) and should be included in the SR sockeye salmon ESU. We therefore listed this program under Idaho Department of Fish and Game's program name, the "Snake River Sockeye Salmon Hatchery Program."

Status. The Snake River sockeye salmon ESU includes only 1 population comprised of all anadromous and residual sockeye salmon from the Snake River Basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. Historical evidence indicates that the Snake River sockeye once had a range of life history patterns, with spawning populations present in several of the small lakes in the Sawtooth Basin (NMFS 2011a). NMFS listed the Snake River sockeye salmon ESU because of habitat loss and degradation from the combined effects of damming and hydropower development, overexploitation, fisheries management practices, and poor ocean conditions. Recent effects of climate change, such as reduced stream flows and increased water temperatures, are limiting Snake River ESU productivity (NMFS 2016j). Adults produced through the captive propagation program currently support the entire ESU. This ESU is still at extremely high risk across all 4 basic risk measures (abundance, productivity, spatial structure, and diversity) and would likely have a very low resilience to additional perturbations. Habitat improvement projects have slightly decreased the risk to the species, but habitat concerns and water temperature issues remain. Overall, although the status of the Snake River sockeye salmon ESU appears to be improving, there is no indication that the biological risk category has changed (NWFSC 2015b).

Life History. Most sockeye salmon exhibit a lake-type life history (i.e., they spawn and rear in or near lakes), though some exhibit a river-type life history. Spawning generally occurs in late summer and fall, but timing can vary greatly among populations. In lakes, sockeye salmon commonly spawn along "beaches" where underground seepage provides fresh oxygenated water. Females spawn in 3 to 5 redds (nests) over a couple of days. Incubation period is a function of water temperature and generally lasts 100-200 days (Burgner 1991). Sockeye salmon spawn once, generally in late summer and fall, and then die (semelparity).

Sockeye salmon fry primarily rear in lakes; river-emerged and stream-emerged fry migrate into lakes to rear. In the early fry stage from spring to early summer, juveniles forage exclusively in the warmer littoral (i.e., shoreline) zone where they depend mostly on fly larvae and pupae, copepods, and water fleas. Sub-yearling sockeye salmon move from the littoral habitat to a pelagic (i.e., open water) existence where they feed on larger zooplankton; however, flies may still make up a substantial portion of their diet. From 1 to 3 years after emergence, juvenile sockeye salmon generally rear in lakes, though some river-spawned sockeye may migrate to sea in their first year. Juvenile sockeye salmon feeding behaviors change as they transition through life stages after emergence to the time of smoltification. Distribution in lakes and prey preference is a dynamic process that changes daily and yearly depending on many factors including water temperature, prey abundance, presence of predators and competitors, and size of the juvenile. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye populations (lower than 52ºN latitude) and as late as early July in northern populations (62ºN latitude)

(Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending 1 to 4 years at sea. The diet of adult salmon consists of amphipods, copepods, squid and other fish.

Table 50. Temporal distribution of Sockeye salmon, Snake River ESU in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="12">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td></tr>
<tr><td>Incubation (eggs)</td><td>Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">Present</td></tr>
<tr><td>Emergence (alevin to fry phases</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance / Productivity. For the Snake River ESU, the only extant population at the time of listing occurred in Redfish Lake. Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 spawners (Bjornn et al. 1968). In 1985, 1986, and 1987, 11, 29, and 16 returning adult sockeye, respectively, were counted at the Redfish Lake weir. Since 1987, only 18 natural-origin sockeye salmon have returned to the Stanley Basin. The first adult returns from the captive broodstock program returned to the Stanley Basin in 1999. From 1999 through 2005, 345 captive brood adults that had migrated to the ocean returned to the Stanley Basin, and returns increased to over 600 in 2008 and more than 700 returning adults in 2009. Annual adult releases during 2011-2014 averaged over 1,200 (NWFSC 2015b). Adult returns of sockeye salmon crashed in 2015, and natural returns have remained low. The low returns of fish collected at the Redfish Lake and Sawtooth Hatchery weirs have limited anadromous releases into Redfish Lake to 311 anadromous hatchery fish in 2016. No natural anadromous fish have been released since 2014, as they are required to be spawned in the captive broodstock program under NMFS Section 10 Permit 1454. Captive adult releases have continued to support spawning in Redfish Lake. Smolt-to-adult return rates suggest that volitional spawning within Redfish Lake appears to be important to the success of the Snake River sockeye salmon captive broodstock-based hatchery program (Kozfkay et al. 2019).While increased abundance of hatchery-reared Snake River sockeye salmon has reduced the risk of loss, levels of naturally-produced sockeye salmon returns have remained extremely low (Ford 2011; NWFSC 2015b). Substantial increases in survival rates across life history stages must occur to re-establish sustainable natural production (Hebdon et al. 2004; Keefer et al. 2008). In terms of natural production, the Snake River sockeye salmon ESU remains at "extremely high risk," although there has been substantial progress on the first phase of the proposed recovery approach—developing a hatchery-based program to amplify and conserve the stock to facilitate reintroductions. Current climate change modeling supports the "extremely high risk" rating with the potential for extirpation in the near future (Crozier et al. 2020). The viability of the Snake River sockeye salmon ESU therefore has likely declined since the time of the prior review, and the extinction risk category remains "high."

Genetic Diversity. For the Snake River ESU, the Sawtooth Hatchery is focusing on genetic conservation (NMFS 2016b). An overrepresentation of genes from the anadromous population in

Redfish Lake exists, but inbreeding is low, which is a sign of a successful captive broodstock program (Kalinowski et al. 2012).

Distribution. The Snake River sockeye salmon ESU includes only 1 population comprised of all anadromous and residual sockeye salmon from the Snake River Basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. At present, anadromous returns are dominated by production from the captive spawning component. The ongoing reintroduction program is still in the phase of building sufficient returns to allow for large-scale reintroduction into Redfish Lake, the initial target for restoring natural production (NMFS 2015). Initial releases of adult returns directly into Redfish Lake have been observed spawning in multiple locations along the lake shore, as well as in Fishhook Creek (NMFS 2015). There is some evidence of very low levels of early timed returns in some recent years from outmigrating, naturally produced Alturas Lake smolts. At this stage of the recovery efforts, the ESU remains rated at "high risk" for both spatial structure and diversity.

Designated Critical Habitat. NMFS designated critical habitat for Snake River sockeye salmon on December 28, 1993 (58 FR 68543). The critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to salmon of this ESU (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Specific PBFs are shown in Appendix B.

Recovery Goals. See the 2015 recovery plan (NMFS 2015d) for the Snake River sockeye salmon ESU for complete down-listing/delisting criteria for recovery goals for the species. Broadly, recovery plan goals emphasize restoring historical lake populations and improving water quality and quantity in lakes and migration corridors.

8.20 Steelhead, California Central Valley DPS

Table 51. Steelhead, California Central Valley DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	California Central Valley	Threatened	2016	2006 71 FR 834	2014	2005 70 FR 52488

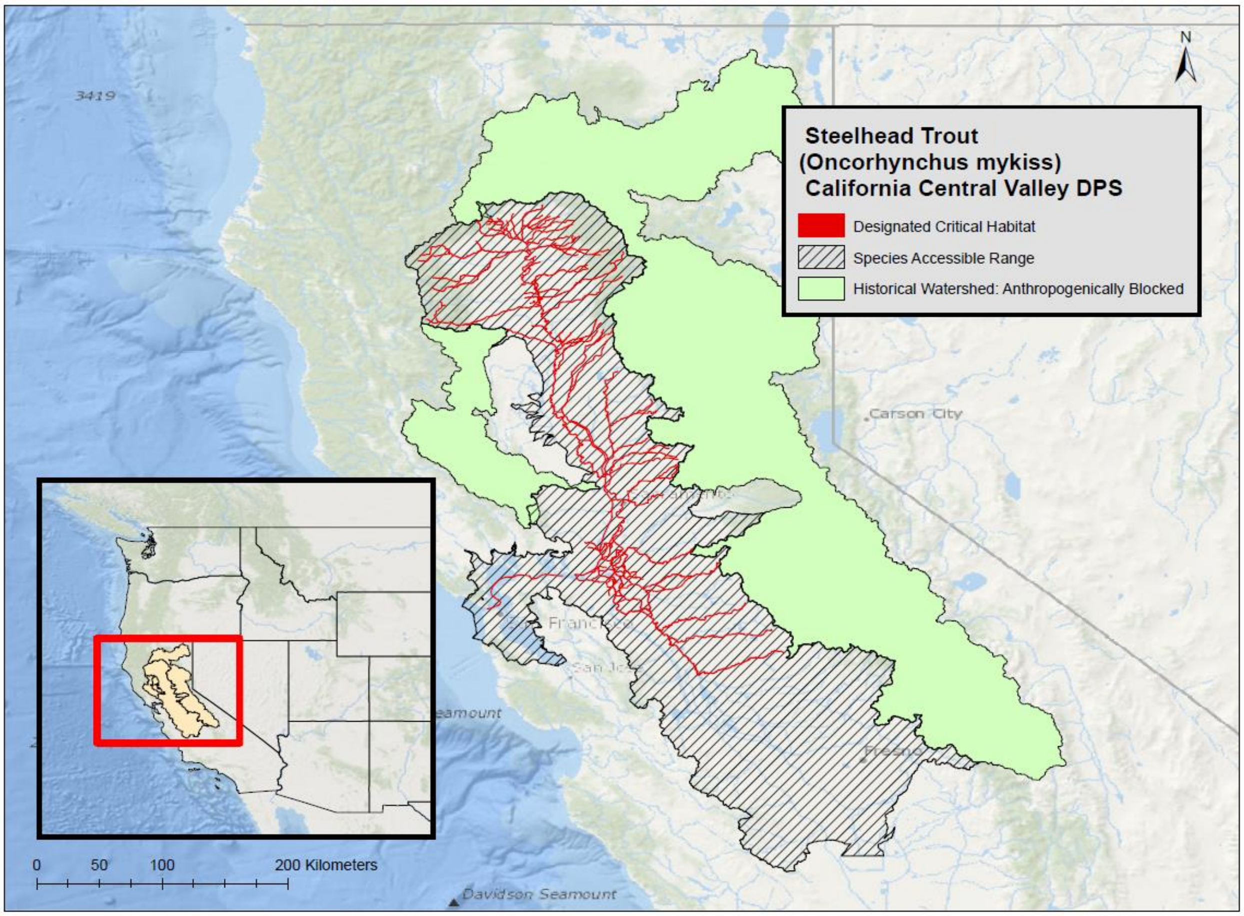


Figure 30**. Steelhead, California Central Valley DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On March 19, 1998 NMFS listed the California Central Valley (CCV) DPS of steelhead as threatened (63 FR 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from the Sacramento and San Joaquin Rivers and their tributaries; excludes such fish originating from San Francisco and San Pablo Bays and their tributaries. This DPS includes steelhead from 2 artificial propagation programs.

Status. Many watersheds in the Central Valley are experiencing decreased abundance of CCV steelhead. Dam removal and habitat restoration efforts in Clear Creek appear to be benefiting CCV steelhead as recent increases in non-clipped (wild) abundance have been observed. Despite the positive trend in Clear Creek, all other concerns raised in the previous status review remain, including low adult abundances, loss and degradation of a large percentage of the historic spawning and rearing habitat, and domination of smolt production by hatchery fish. Many other planned restoration and reintroduction efforts have yet to be implemented or completed, or are focused on Chinook salmon, and have yet to yield demonstrable improvements in habitat, let alone documented increases in naturally produced steelhead. There are indications that natural production of steelhead continues to decline and is now at a very low level. Their continued low numbers in most hatcheries, domination by hatchery fish, and relatively sparse monitoring makes the continued existence of naturally reproduced steelhead a concern. CCV steelhead is likely to become endangered within the foreseeable future throughout all or a significant portion of its range.

Life History. Central Valley steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems. The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (Gallagher 2000). Following deposition of fertilized eggs in the redd, they are covered with loose gravel. The eggs hatch in 3 to 4 weeks at 50°F to 59°F, and fry emerge from the gravel 4 to 6 weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or 2 of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002c). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

Steelhead typically migrate to marine waters after spending 2 years in fresh water. They reside in marine waters for typically 2 or 3 years prior to returning to their natal stream to spawn as 4- or 5-year olds. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002c). Currently, Central Valley steelhead are considered “ocean-maturing” (also known as winter) steelhead, although summer steelhead may have been present prior to construction of large dams. Ocean maturing steelhead enter fresh water with well-developed gonads and spawn shortly after river entry. Central Valley steelhead enter fresh water from August through April. They hold until flows are high enough in tributaries to enter for spawning (Moyle 2002c). Steelhead adults typically spawn from December through April, with peaks from January through March in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961a; McEwan 2001).

Table 52. Temporal distribution of Steelhead, California Central Valley DPS in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)	Present						Present					
Spawning	Present											Present
Incubation (eggs)	Present											Present
Emergence (alevin to fry phases)	Present											
Rearing and migration (juveniles)	Present											

Population Dynamics

Abundance. Historic CCV steelhead run size may have approached 1-2 million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.* (1961b) estimated an average of 20,540 adult steelhead in the Sacramento River, upstream of the Feather River, through the 1960s. Steelhead were counted at the RBDD up until 1993. Counts at the dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s. An estimated total annual run size for the entire Sacramento-San Joaquin system was no more than 10,000 adults during the early 1990s (McEwan and Jackson 1996; McEwan 2001). Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of CCV steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005b).

Productivity / Population Growth Rate. CCV steelhead lack annual monitoring data for calculating trends and lambda. However, the RBDD counts and redd counts up to 1993 and later sporadic data show that the DPS has had a significant long-term downward trend in abundance (NMFS 2009a).

Genetic Diversity / Distribution. The CCV steelhead distribution ranged over a wide variety of environmental conditions and likely contained biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006). Thus, the loss of populations and reduction in abundances have reduced the large diversity that existed within the DPS. The genetic diversity of the majority of CCV steelhead spawning runs is also compromised by hatchery-origin fish. Designated Critical Habitat. NMFS designated critical habitat for CCV steelhead on September 2, 2005 (70 FR 52488). PBFs considered essential for the conservation of Steelhead, California Central Valley DPS are described in Appendix B.

Recovery Goals. See the 2014 recovery plan (NMFS 2014d) for the California Central Valley steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. The delisting criteria for this DPS are:

- One population in the Northwestern California Diversity Group at low risk of extinction

- Two populations in the Basalt and Porous Lava Flow Diversity Group at low risk of extinction
- Four populations in the Northern Sierra Diversity Group at low risk of extinction
- Two populations in the Southern Sierra Diversity Group at low risk of extinction
- Maintain multiple populations at moderate risk of extinction

The current condition of CCV steelhead critical habitat is degraded, and does not provide the conservation value necessary for species recovery. In addition, the Sacramento-San Joaquin River Delta, as part of CCV steelhead designated critical habitat, provides very little function necessary for juvenile CCV steelhead rearing and physiological transition to salt water.

The spawning PBF is subject to variations in flows and temperatures, particularly over the summer months. Some complex, productive habitats with floodplains remain in the system and flood bypasses (*i.e.*, Yolo and Sutter bypasses). However, the rearing PBF is degraded by the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system and which typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Stream channels commonly have elevated temperatures.

The current conditions of migration corridors are substantially degraded. Both migration and rearing PBFs are affected by dense urbanization and agriculture along the mainstems and in the Delta which contribute to reduced water quality by introducing several contaminants. In the Sacramento River, the migration corridor for both juveniles and adults is obstructed by the RBDD gates which are down from May 15 through September 15. The migration PBF is also obstructed by complex channel configuration making it more difficult for CCV steelhead to migrate successfully to the western Delta and the ocean. In addition, the state and federal government pumps and associated fish facilities change flows in the Delta which impede and obstruct a functioning migration corridor that enhances migration. The estuarine PBF, which is present in the Delta, is affected by contaminants from agricultural and urban runoff and release of wastewater treatment plants effluent.

8.21 Steelhead, Central California Coast DPS

Table 53. Steelhead, Central California Coast DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Central California Coast	Threatened	2016	2006 71 FR 834	2016	2005 70 FR 52488

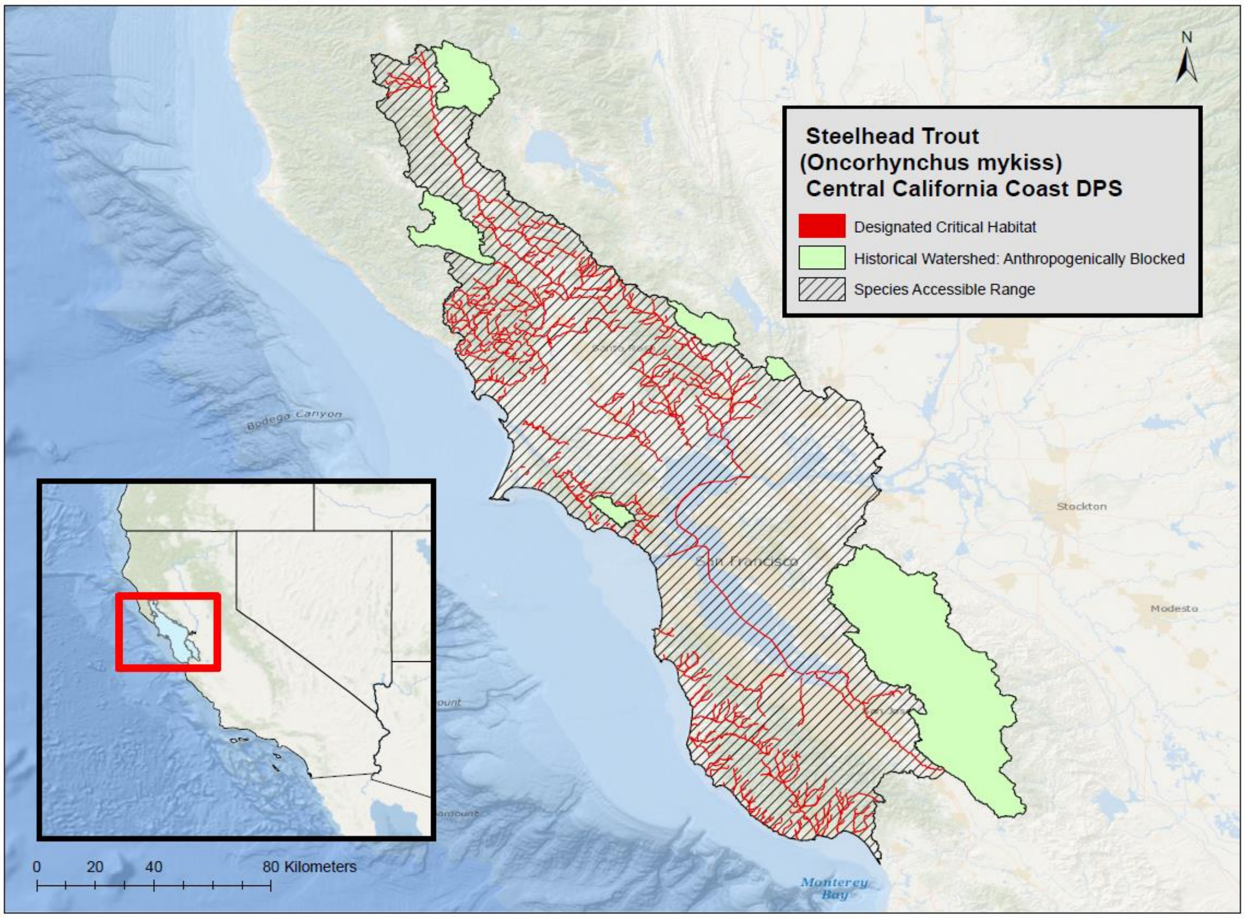


Figure 31. Steelhead, Central California Coast DPS range and designated critical habitat

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On August 18, 1997 NMFS listed the Central California Coast (CCC) DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). This DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Russian

River to Aptos Creek, Santa Cruz County, California (inclusive). It also includes the drainages of San Francisco and San Pablo Bays.

Status. The CCC steelhead consisted of 9 historic functionally independent populations and 23 potentially independent populations (Bjorkstedt et al. 2005). Of the historic functionally independent populations, at least 2 are extirpated while most of the remaining are nearly extirpated. Current runs in the basins that originally contained the 2 largest steelhead populations for CCC steelhead, the San Lorenzo and the Russian Rivers, both have been estimated at less than 15% of their abundances just 30 years earlier (Good et al. 2005b). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of CCC steelhead (NOAA 2013). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby et al. 1996; NMFS 2008a).

Life History. The DPS is entirely composed of winter-run fish, as are those DPSs to the south. Adults return to the Russian River and migrate upstream from December – April, and smolts emigrate between March – May) (Hayes et al. 2004; Shapovalov and Taft 1954a). Most spawning takes place from January through April. While age at smoltification typically ranges for 1 to 4 years, recent studies indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age 2 (Sogard et al. 2009). Survival in fresh water reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard et al. 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard et al. 2009). Greater movement of juveniles in fresh water has been observed in winter and spring versus summer and fall time periods. Smaller individuals are more likely to be observed to exceed 0.3 mm per day, and are highest in winter through spring, potentially due to higher water flow rates and greater food availability (Boughton et al. 2007; Sogard et al. 2009).

Table 54. Temporal distribution of Steelhead, Central California Coast DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Spawning</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td colspan="5">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Historically, the entire CCC steelhead DPS may have consisted of an average runs size of 94,000 adults in the early 1960s (Good et al. 2005b). Information on current CCC steelhead populations consists of anecdotal, sporadic surveys that are limited to only smaller portions of watersheds. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good et al. 2005b).

Productivity / Population Growth Rate. Though the information for individual populations is limited, available information strongly suggests that no population is viable. Long-term population sustainability is extremely low for the southern populations in the Santa Cruz mountains and in the San Francisco Bay (NMFS 2008a). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good et al. 2005b). The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners; it may be able to be sustained over the long-term but hatchery management has eroded the population's genetic diversity (Bjorkstedt et al. 2005; NMFS 2008a). Data on abundance trends do not exist for the DPS as a whole or for individual watersheds. Thus, it is not possible to calculate long-term trends or lambda.
Genetic Diversity / Distribution. This DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Russian River to Aptos Creek, Santa Cruz County, California (inclusive). It also includes the drainages of San Francisco and San Pablo Bays.

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes the Russian River watershed, coastal watersheds in Marin County, streams within the San Francisco Bay, and coastal watersheds in the Santa Cruz Mountains down to Apos Creek. PBFs considered essential for the conservation of Steelhead, Central California Coast DPS are described in Appendix B.

Streams throughout the critical habitat have reduced quality of spawning PBFs; sediment fines in spawning gravel have reduced the ability of the substrate attribute to provide well oxygenated and clean water to eggs and alevins. High proportions of fines in bottom substrate also reduce forage by limiting the production of aquatic stream insects adapted to running water. Elevated water temperatures and impaired water quality have further reduced the quality, quantity and function of the rearing PBF within most streams. These impacts have diminished the ability of designated critical habitat to conserve the CCC steelhead.

Recovery Goals. See the 2016 recovery plan (NMFS 2016c) for the Central California Coast steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. Recovery plan objectives are to:

- Reduce the present or threatened destruction, modification, or curtailment of habitat or range;
- Ameliorate utilization for commercial, recreational, scientific, or educational purposes;
- Abate disease and predation;
- Establish the adequacy of existing regulatory mechanisms for protecting CCC steelhead now and into the future (i.e., post-delisting);
- Address other natural or manmade factors affecting the continued existence of CCC steelhead;
- Ensure CCC steelhead status is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

8.22 Steelhead, Lower Columbia River DPS

Table 55. Steelhead, Lower Columbia River DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Lower Columbia River	Threatened	2022	2006 71 FR 834	2013	2005 70 FR 52630

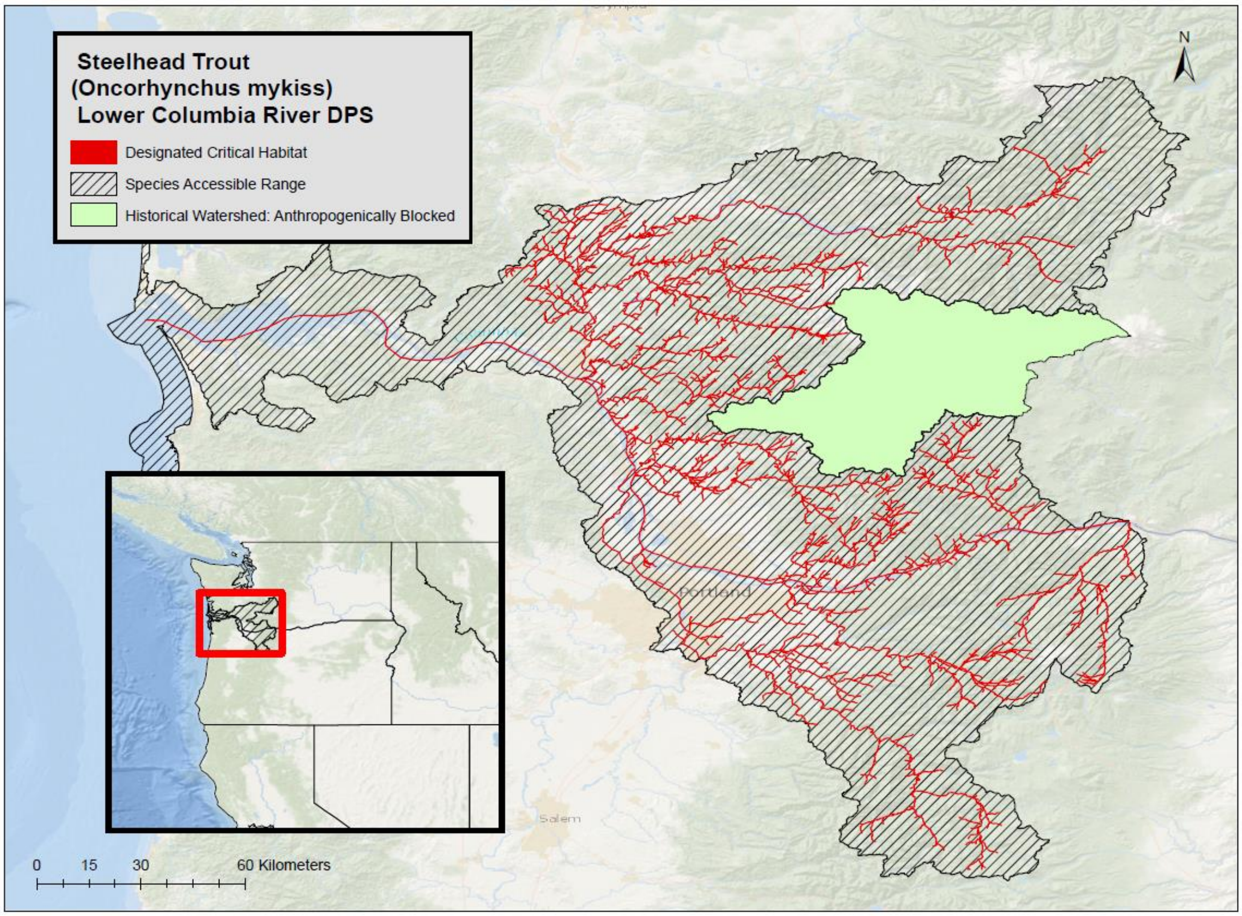


Figure 32. Steelhead, Lower Columbia River DPS range and designated critical habitat

Species Description. On March 19, 1998 NMFS listed the Lower Columbia River (LCR) DPS of steelhead as threatened (63 FR 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). This DPS reaffirmed in the 2022 Status Update, includes naturally spawned anadromous *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from rivers between the Cowlitz and Wind Rivers (inclusive) and the Willamette and Hood Rivers (inclusive); excludes such fish originating from the upper Willamette River basin above Willamette Falls. This DPS includes steelhead from 7 artificial propagation programs. Since 2016, NMFS (1) added the recently initiated Upper Cowlitz Wild Program because the source for these fish is local, natural-origin fish from the Upper Cowlitz River, which is included in the DPS; (2) added the recently initiated Tilton River Wild Program because the source for these fish is local, natural-origin fish from the Tilton River; and (3) removed ODFW stock numbers from the names of the Clackamas Hatchery Late Winter-run Program, Sandy Hatchery Late Winter-run Program, and Hood River Winter-run Program (85 FR 81822, December 17, 2020).

Status. Overall, the viability trend for the Lower Columbia River Steelhead DPS remains unchanged since the previous (2015) review. While a number of DIPs exhibited increases in their 5-year geometric mean, others still remain depressed, and neither the winter nor summer-run MPGs are near viability in the Gorge. Given these concerns, the Lower Columbia River Steelhead DPS is at a moderate risk of extinction (Ford 2022). The LCR steelhead had 17 historically independent winter steelhead populations and 6 independent summer steelhead populations (McElhany et al. 2003; Myers et al. 2006). All historic LCR steelhead populations are considered extant. However, spatial structure within the historically independent populations, especially on the Washington side, has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydropower development. The majority of winter-run steelhead populations in this DPS continue to persist at low abundances (NWFSC 2015b). Hatchery interactions remain a concern in select basins, but the overall situation is somewhat improved compared to prior reviews. Summer-run steelhead DIPs were similarly stable, but at low abundance levels. Habitat degradation continues to be a concern for most populations. Even with modest improvements in the status of several winter-run populations, none of the populations appear to be at fully viable status, and similarly none of the MPGs meet the criteria for viability. The DPS therefore continues to be at moderate risk (NWFSC 2015b). Based on the 2022 Status Update, no reclassification for the LCR Steelhead DPS is warranted. Therefore, the LCR Steelhead DPS remain listed as threatened.

Life History. The LCR steelhead DPS includes both summer- and winter-run stocks. Summer-run steelhead return sexually immature to the Columbia River from May to November, and spend several months in fresh water prior to spawning. Winter-run steelhead enter fresh water from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer-run steelhead tend to spawn at higher elevations than the winter-run. The majority of juvenile LCR steelhead remain for 2 years in freshwater environments before ocean entry in spring. Both winter- and summer-run adults normally return after 2 years in the marine environment.

Table 56. Temporal distribution of Steelhead, Lower Columbia River DPS in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present
Spawning			Present	Present	Present	Present						
Incubation (eggs)			Present	Present	Present	Present	Present					
Emergence (alevin to fry phases					Present	Present	Present	Present				
Rearing and migration (juveniles)	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present

Population Dynamics

Abundance. All LCR steelhead populations declined from 1980 to 2000, with sharp declines beginning in 1995. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeded 20,000 spawners. During the 1990s, abundance dropped to 1,000 to 2,000 spawners. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 spawners for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas. Many of the long-and short-term trends in abundance of individual populations are negative.

Productivity / Population Growth Rate. There is a difference in population stability between winter- and summer-run LCR steelhead. The winter-run steelhead in the Cascade region has the highest likelihood of being sustained as it includes a few populations with moderate abundance and positive short-term population growth rates (Good et al. 2005b; McElhany et al. 2007a). For most winter-run populations in this MPG, the trend within the 2015–19 period is strongly negative as expressed in annual productivity estimates. There is some concern that this downward trend may be indicative of something more systemic than short-term freshwater or oceanic conditions. The Gorge summer-run steelhead is at the highest risk over the long-term as the Hood River population is at high risk of being lost (McElhany et al. 2007a). Wind River and Hood River are the 2 DIPs in the summer run of this MPG. Hood River summer-run steelhead monitoring has been problematic since the removal of Powerdale Dam. Adult abundance in the Wind River has declined since the last review and is trending downward (Table 36, Figure 68). Recent 5-year abundance for Wind River summer-run, a designated natural steelhead gene bank, is 627, a 13% decline from the 2010–14 average (Table 36, Figure 68). The long-term (2005–19) abundance trend for the Wind River is a 2% annual decline (Table 37). Given the presence of only 2 summer-run DIPs in this MPG and the recent downward trend, the overall status of the MPG is uncertain.

Genetic Diversity / Distribution. This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from rivers between the Cowlitz and Wind Rivers (inclusive) and the Willamette and Hood Rivers (inclusive); excludes such fish originating from the upper Willamette River basin above Willamette Falls. This DPS includes steelhead from 7 artificial propagation programs. The WLC TRT identified 23 historical independent populations of Lower Columbia River steelhead: 17 winter-run populations and 6 summer-run populations, within the Cascade and Gorge ecozones.

Designated Critical Habitat. Critical habitat was designated for the LCR steelhead on September 2, 2005 (70 FR 52488). PBFs considered essential for the conservation of Steelhead, Lower Columbia River DPS are described in Appendix B.

Critical habitat is affected by reduced quality of rearing and juvenile migration PBFs within the lower portion and alluvial valleys of many watersheds; contaminants from agriculture affect both water quality and food production in these reaches of tributaries and in the mainstem Columbia River. Several dams affect adult migration PBF by obstructing the migration corridor. Watersheds which consist of a large proportion of federal lands such as is the case with the

Sandy River watershed, have relatively healthy riparian corridors that support attributes of the rearing PBF such as cover, forage, and suitable water quality.

Recovery Goals. NMFS has developed the following delisting criteria for the Lower Columbia River steelhead DPS:

- All strata that historically existed have a high probability of persistence or have a probability of persistence consistent with their historical condition
- The threats criteria described in the recovery plan have been met.

Recommendations for Future Actions

- The 2022 review of the listing factors and the Northwest Fisheries Science Center's biological viability assessment, NMFS identified many recommended actions to improve factors influencing the status of the Lower Columbia River Steelhead DPS. Below are actions that provide the greatest opportunity to improve the VSP parameters, and advance their recovery.
- For all populations and all MPGs that comprise the LCR steelhead, recommended future recovery actions over the next 5 years include:
- Conduct systematic review and analysis of high priority Lower Columbia River mainstem and tributary area habitat needs, identified in NMFS 2013a, and compare needs to what has been accomplished.
- Conduct monitoring to evaluate ship wake stranding frequency and locations where stranding occurs and assess factors contributing to wake stranding such as location, topography, vessel speed, et cetera, to determine best practices to reduce wake stranding mortality.
- Promote riparian plantings of native canopy tree cover species opportunistically in all watersheds.
- Coordinate with EPA in an evaluation of Washington State Water Quality Standards, reflecting Oregon and Idaho consultation outcomes.

For populations within the below listed MPGs, NMFS recommend the following recovery actions over the next 5 years:
Cascade MPGs

- Reestablish and improve passage on multiple rivers to benefit multiple populations from the Cascade MPGs, such as the North Fork Lewis River (NF Lewis River spring Chinook, NF Lewis River winter steelhead, NF Lewis River coho), and Cowlitz River (Upper Cowlitz River spring Chinook, Upper Cowlitz River fall Chinook, Upper Cowlitz River coho, Upper Cowlitz River winter steelhead).
- Work with county and city jurisdictions to protect watershed hydrology from long-term development impacts (floodplain development and groundwater withdrawals). Focus these efforts on high growth rate watersheds along the I-5 and I-205 corridors, including the East Fork Lewis River, North Fork Lewis River, Coweeman River, Kalama River, Washougal River, Salmon Creek, and Lower Cowlitz tributaries.

Gorge MPGs

- Continue to work with partners on programs protecting instream and floodplain habitats in key chum spawning areas, such as Duncan Creek and Hamilton Creek, (e.g., evaluate if large wood debris mitigates excess winter stream flows that degrade spawning for Upper Gorge chum).

- Continue to work with partners to identify suitable chum spawning habitat streams and reaches to emplace habitat creation or enhancement projects in order to expand spatial distribution into the gorge strata.
- Improve understanding of key factors limiting recovery by evaluating summer-run Gorge steelhead losses between Bonneville Dam and Shipherd Falls.
- Implement the EPA 2021 Columbia River Cold Water Refuges Plan, for example in Woodard Creek, to benefit Upper Gorge (Wind River and White Salmon rivers) LCR fall Chinook salmon, Lower Gorge (Woodard Creek) winter steelhead, Upper Gorge (Wind River) steelhead, and Wind River summer steelhead.
- Increase channel complexity to improve juvenile rearing habitat for Wind River summer steelhead.

8.23 Steelhead, Middle Columbia River DPS

Table 57. Steelhead, Middle Columbia River DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Middle Columbia River	Threatened	2022	2006 71 FR 834	2009	2005 70 FR 52630

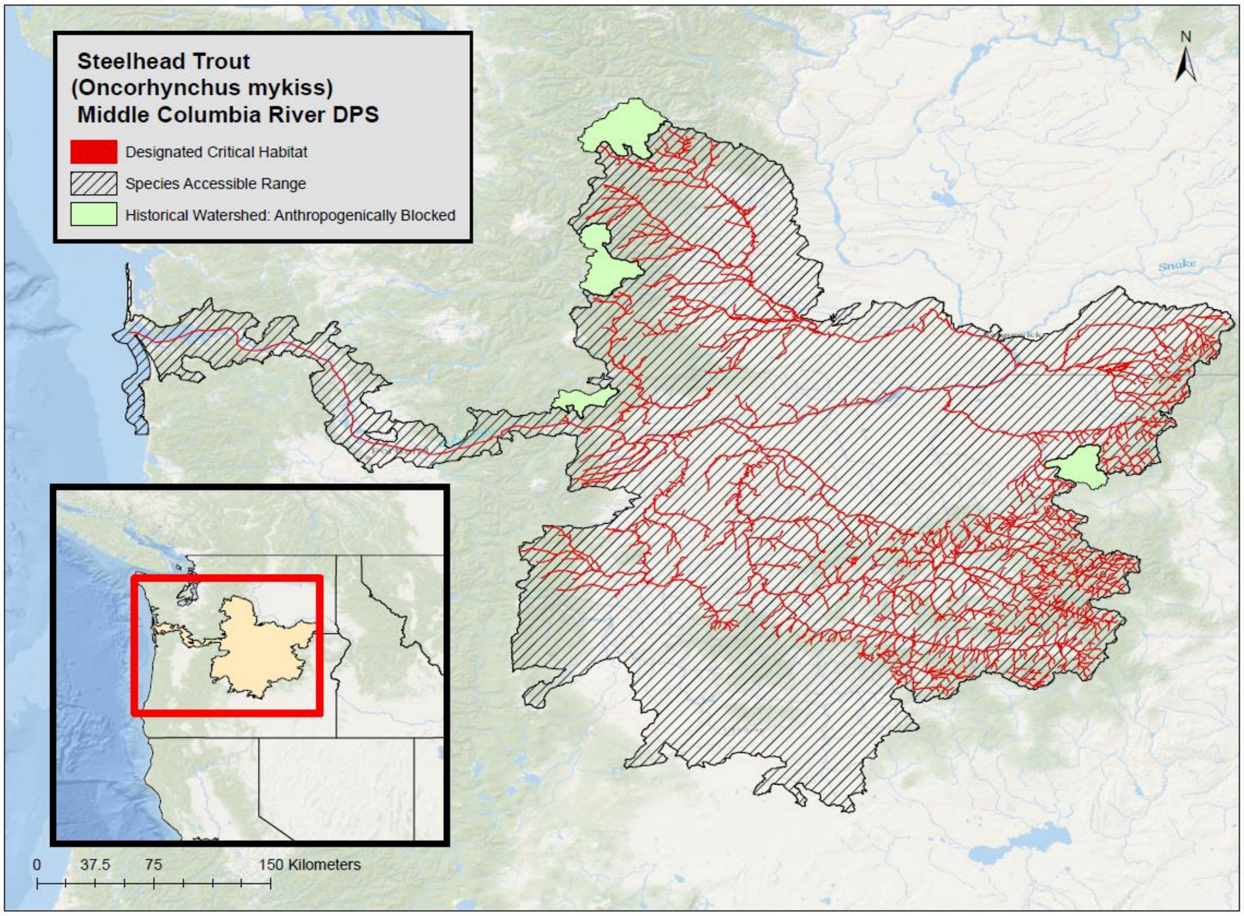


Figure 33**. Steelhead, Middle Columbia River DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On March 25, 1999 NMFS listed the Middle Columbia River (MCR) DPS of steelhead as threatened (64 FR 14517) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). The MCR steelhead DPS includes all naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from the Columbia River and its tributaries upstream of the

Wind and Hood Rivers (exclusive) to and including the Yakima River; and excludes such fish originating from the Snake River basin. This DPS does include steelhead from 4 artificial propagation programs: the Touchet River Endemic Program; Yakima River Kelt Reconditioning Program (in Satus Creek, Toppenish Creek, Naches River, and Upper Yakima River); Umatilla River Program; and the Deschutes River Program. This DPS does not include steelhead that are designated as part of an experimental population (79 FR 20802; Figure 2). For recovery planning and development of recovery criteria, the ICTRT identified independent populations within the MCR steelhead DPS and grouped them into genetically similar major population groups (MPGs) (ICTRT 2003). The DPS is composed of 4 MPGs: Cascades Eastern Slope Tributaries, John Day River, Yakima River, and Walla Walla and Umatilla Rivers.

Status. Based on the information identified in the 2022 Status Update, it was determined that no reclassification for the MCR steelhead DPS is appropriate, and therefore the MCR steelhead DPS should remain listed as threatened.

ESU/DPS delineation: The Northwest Fisheries Science Center's review (Ford 2022) found that except for removal of the Yakima River Kelt Reconditioning Program from the DPS, no new information has become available that would justify a change in the delineation of the MCR steelhead DPS.

The ICTRT identified 16 extant populations in 4 major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and 1 unaffiliated independent population (Rock Creek) (ICTRT 2003). There are 2 extinct populations in the Cascades Eastern Slope major population group: the White Salmon River and the Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on geographical proximity, topography, distance, ecological similarities or differences. Using criteria for abundance and productivity, the ICTRT modeled a gaps analysis for each of the 4 MPGs in this DPS under 3 different ocean conditions and a base hydro condition (most recent 20-year survival rate). The results showed that none of the MPGs would be able to achieve a 5% or less risk of extinction over 100 years without recovery actions. It is important to consider that significant gaps in factors affecting spatial structure and diversity also contribute to the risk of extinction for these fish.

Life History. MCR steelhead populations are mostly of the summer-run type. Adult steelhead enter fresh water from June through August. The only exceptions are populations of inland winter-run steelhead which occur in the Klickitat River and Fifteenmile Creek (Busby et al. 1996). The majority of juveniles smolt and outmigrate as 2-year olds. Most of the rivers in this region produce about equal or higher numbers of adults having spent 1 year in the ocean as adults having spent 2 years. However, summer-run steelhead in Klickitat River have a life cycle more like LCR steelhead whereby the majority of returning adults have spent 2 years in the ocean (Busby et al. 1996). Adults may hold in the river up to a year before spawning.

Table 58. Temporal distribution of Steelhead, Middle Columbia River DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="12">Present</td></tr>
<tr><td>Spawning</td><td colspan="7">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td colspan="7">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td colspan="5">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby et al. 1996). The 5-year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good et al. 2005b). The 5-year average for these basins is 298 and 1,492 spawners, respectively (Good et al. 2005b). Total escapement and natural-origin escapements declined relative to the prior 5-year review for all 5 of the John Day MPG populations. Only 2 of the 5 populations in this group had a positive 15-year trend in natural-origin abundance, driven largely by peak returns in the early 2000s, despite the strong declines over the most recent 5-year period. Total spawning escapements have decreased in the most recent brood cycle for all 3 populations in the Umatilla/Walla Walla MPG as well. The 15-year trend in natural-origin abundance was positive for the Umatilla River population and slightly negative for Touchet River, though the trends are shallow. Population productivity was cyclical, with most populations following a similar pattern of growth and decline.

Productivity / Population Growth Rate. Good et al. (2005b) calculated that the median estimate of long-term trend over 12 indicator data sets was –2.1% per year (–6.9 to 2.9), with 11 of the 12 being negative. Long-term annual population growth rates (λ) were also negative (Good et al. 2005b). The median long-term λ was 0.98, assuming that hatchery spawners do not contribute to production, and 0.97 assuming that both hatchery- and natural-origin spawners contribute equally.

Distribution. The MCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from above the Wind River, Washington, and the Hood River, Oregon (exclusive), upstream to, and including, the Yakima River, Washington, excluding steelhead from the Snake River Basin. Steelhead from the Snake River basin (described later in this section) are excluded from this DPS. Seven artificial propagation programs are part of this DPS. They include: the Touchet River Endemic, Yakima River Kelt Reconditioning Program (in Satus Creek, Toppenish Creek, Naches River, and Upper Yakima River), Umatilla River, and the Deschutes River steelhead hatchery programs. These artificially propagated populations are considered no more divergent relative to the local natural populations than would be expected between closely related natural populations within the DPS. According to the ICBTRT (ICTRT 2003), this DPS is composed of 16 populations in 4 major population groups (Cascade Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River), and 1 unaffiliated population (Rock Creek). Updated information on spawner and juvenile rearing distribution does not support a change in spatial structure status for Middle Columbia River steelhead DPS populations, though the newly re-established run in the White Salmon River and the developing time series of population data from the Klickitat River and Rock Creek do warrant consideration in the DPS recovery plan. Viability indicators for within-population diversity have changed for some populations, although in most cases the changes have not been sufficient to shift composite risk ratings for a particular population.

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). PBFs considered essential for the conservation of Steelhead, Middle Columbia River DPS are described in Appendix B.

The current condition of critical habitat designated for the MCR steelhead is moderately degraded. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. Reduced quality of the rearing PBFs has diminished its contribution to the conservation value necessary for the recovery of the species. Several dams affect adult migration PBF by obstructing the migration corridor.

Recovery Goals. See the 2009 recovery plan for the Middle Columbia River steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. There has been functionally no change in the viability ratings for the component populations, and the Middle Columbia River steelhead DPS does not currently meet the viability criteria described in the Middle Columbia River steelhead recovery plan. In addition, several of the factors cited by the 2005 BRT remain as concerns or key uncertainties. While recent (5-year) returns are declining across all populations, the declines are from relatively high returns in the previous 5-to-10 year interval, so the longer-term risk metrics that are meant to buffer against short-period changes in abundance and productivity remain unchanged. Natural-origin spawning estimates are highly variable relative to minimum abundance thresholds across the populations in the DPS. Two of the 4 MPGs in this DPS include at least 1 population rated at “low” or “very low” risk for abundance and productivity, while the other 2 MPGs remain in the “moderate” to “high” risk range. Updated information indicates that stray levels into the John Day River populations have decreased in recent years. Out-of-basin hatchery stray proportions, although reduced, remain high in spawning reaches within the Deschutes River basin and the Umatilla, Walla Walla, and Touchet River populations. Overall, the Middle Columbia River steelhead DPS remains at “moderate” risk of extinction, with viability unchanged from the prior review.

Significant habitat restoration and protection actions at the Federal, state, and local levels have been implemented to improve degraded habitat conditions and resolve fish passage issues described in the 2009 Middle Columbia River Steelhead Recovery Plan. While these efforts have been substantial and are expected to benefit the survival and productivity of the targeted populations, the 2022 Status Update found no evidence demonstrating that improvements in habitat conditions have led to improvements in population viability.

8.24 Steelhead, Northern California DPS

Table 59. Steelhead, Northern California DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Northern California	Threatened	2016	2006 71 FR 834	2016	2005 70 FR 52488

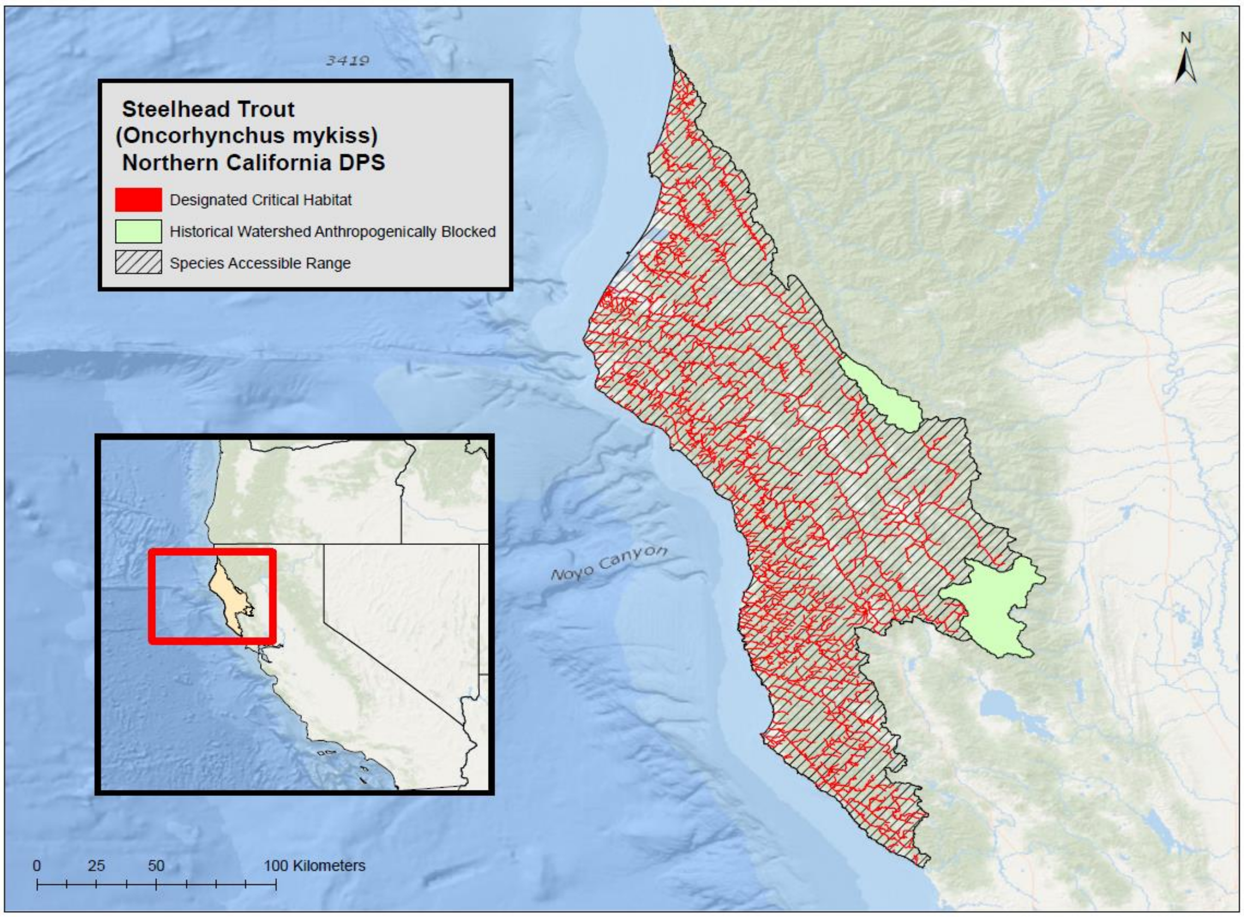


Figure 34. Steelhead, Northern California DPS range and designated critical habitat

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On June 7, 2000 NMFS listed the Northern California (NC) DPS of steelhead as threatened (65 FR 36074) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers in California coastal river basins from Redwood Creek to and including the Gualala River.

Status. The available data for winter-run populations— predominantly in the North Coastal, North-Central Coastal, and Central Coastal strata— indicate that all populations are well below viability targets, most being between 5% and 13% of these goals. For the 2 Mendocino Coast populations with the longest time series, Pudding Creek and Noyo River, the 13-year trends have been negative and neutral, respectively (Williams et al. 2016). However, the short-term (6-year) trend has been generally positive for all independent populations in the North-Central Coastal and Central Coastal strata, including the Noyo River and Pudding Creek. Data from Van Arsdale Station likewise suggests that, although the long-term trend has been negative, run sizes of natural-origin steelhead have stabilized or are increasing. Thus, we have no strong evidence to indicate conditions for winter-run populations in the DPS have worsened appreciably since the last status review (Williams et al. 2016). Summer-run populations continue to be of significant concern because of how few populations currently exist. The Middle Fork Eel River population has remained remarkably stable for nearly 5 decades and is closer to its viability target than any other population in the DPS. Although the time series is short, the Van Duzen River appears to be supporting a population numbering in the low hundreds. However, the Redwood Creek and Mattole River populations appear small, and little is known about other populations including the Mad River and other tributaries of the Eel River (i.e., Larabee Creek, North Fork Eel, and South Fork Eel). Most populations for which there are population estimates available remain well below viability targets; however, the short-term increases observed for many populations, despite the occurrence of a prolonged drought in northern California, suggests this DPS is not at immediate risk of extinction.

Life History. This DPS includes both winter- and summer –run steelhead. In the Mad and Eel Rivers, immature steelhead may return to fresh water as "half-pounders" after spending only 2 to 4 months in the ocean. Generally, a half-pounder will overwinter in fresh water and return to the ocean in the following spring.

Juvenile out-migration appears more closely associated with size than age but generally, throughout their range in California, juveniles spend 2 years in fresh water (Busby et al. 1996). Smolts range from 14-21 cm in length. Juvenile steelhead may migrate to rear in lagoons throughout the year with a peak in the late spring/early summer and in the late fall/early winter period (Shapovalov and Taft 1954a; Zedonis 1992).

Steelhead spend anywhere from 1 to 5 years in salt water, however, 2–3 years are most common (Busby et al. 1996). Ocean distribution is not well known but coded wire tag recoveries indicate that most NC steelhead migrate north and south along the continental shelf (Barnhart 1986).

Table 60. Temporal distribution of Steelhead, Northern California DPS in freshwater habitats

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">Present</td></tr>
<tr><td>Spawning</td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td>Present</td></tr>
<tr><td>Incubation (eggs)</td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. Northern California steelhead historic functionally independent populations and their abundances and hatchery contributions are provided in Table 61.

Table 61. Northern California steelhead historic and recent abundance

Population	Historical Abundance	Past Spawner Abundance	Hatchery Abundance Contributions (%)
Mad River (S)	6,000	162-384	2
MF Eel River (S)	Unknown	384-1,246	0
NF Eel River (S)	Unknown	Extirpated	N/A
Mattole River (S)	Unknown	9-30*	Unknown
Redwood Creek (S)	Unknown	6*	Unknown
Van Duzen (W)	10,000	Unknown	Unknown
Mad River (W)	6,000	Unknown	Unknown
SF Eel River (W)	34,000	2743-20,657	Unknown
Mattole River (W)	12,000	Unknown	Unknown
Redwood Creek (W)	10,000	Unknown	Unknown
Humboldt Bay (W)	3,000	Unknown	Unknown
Freshwater Creek (W)		25-32	
Ten Mile River (W)	9,000	Unknown	Unknown
Noyo River (W)	8,000	186-364*	Unknown
Big River (W)	12,000	Unknown	Unknown
Navarro River (W)	16,000	Unknown	Unknown
Garcia River (W)	4,000	Unknown	Unknown
Gualala River (W)	16,000	Unknown	Unknown
Total	198,000	Unknown	

**From Spence et al. (2008). Redwood Creek abundance is the mean count over 4 generations. Mattole River abundances from surveys conducted between 1996 and 2005. Noyo River abundances from surveys conducted since 2000.*

Population	Historical Abundance	Past Spawner Abundance	Hatchery Abundance Contributions (%)
Summer –run steelhead is noted with a (S) and winter-run steelhead with a (W)			

Productivity / Population Growth Rate. When first listed, Good *et al.* (2005b) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good et al. 2005b). According to the NMFS 2016 status review, the available data for winter-run populations— predominately in the North Coastal, North-Central Coastal, and Central Coastal strata— indicate that all populations are well below viability targets, most being between 5% and 13% of these goals. Data from Van Arsdale Station likewise suggests that, although the long-term trend has been negative, run sizes of natural-origin steelhead have stabilized or are increasing (Spence 2016). Thus, we have no strong evidence to indicate conditions for winter-run populations in the DPS have worsened appreciably since the last status review (NMFS 2011). Summer-run populations continue to be of significant concern because of how few populations currently exist. The Middle Fork Eel River population has remained remarkably stable for nearly 5 decades and is closer to its viability target than any other population in the DPS (Spence 2016). Reduction of summer-run steelhead populations has significantly reduced current DPS diversity compared to historic conditions. Of the 10 summer-run steelhead populations, only 4 are extant. Of these, only the Middle Fork Eel River population is at moderate risk of extinction, the remaining 3 are at high risk (Spence et al. 2008a). Hatchery influence has likely been limited.

Genetic Diversity / Distribution. Artificial propagation was identified as negatively affecting wild stocks of salmonids through interactions with non-native fish, introductions of disease, genetic changes, competition for space and food resources, straying and mating with native populations, loss of local genetic adaptations, mortality associated with capture for broodstock and palliating the destruction of habitat and concealing problems facing wild stocks.

Designated Critical Habitat. NMFS designated critical habitat for NC steelhead on September 2, 2005 (70 FR 52488). PBFs considered essential for the conservation of Steelhead, Northern California DPS are described in Appendix B.

The current condition of critical habitat designated for the NC steelhead is moderately degraded. Nevertheless, it does provide some conservation value necessary for species recovery. Within portions of its range, especially the interior Eel River, rearing PBF quality is affected by elevated temperatures by removal of riparian vegetation. Spawning PBF attributes such as the quality of substrate supporting spawning, incubation, and larval development have been generally degraded throughout designated critical habitat by silt and sediment fines in the spawning gravel. Bridges and culverts further restrict access to tributaries in many watersheds, especially in watersheds with forest road construction, thereby reducing the function of adult migration PBF.

Recovery Goals. See the 2016 recovery plan for the Northern California steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2016d).

8.25 Steelhead, Puget Sound DPS

Table 62. Steelhead, Puget Sound DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Puget Sound	Threatened	2022	2007 72 FR 26722	2019	2016 81 FR 9251

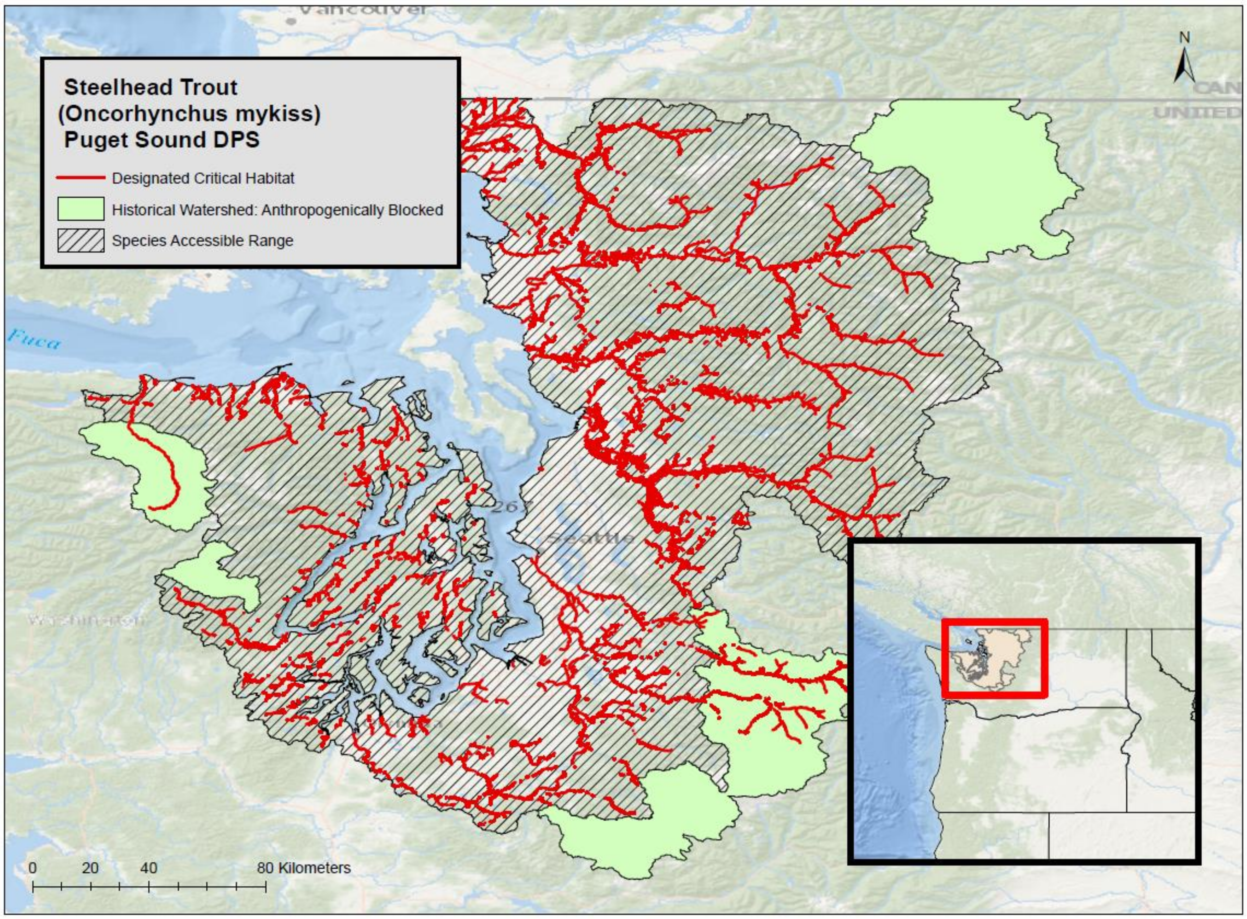


Figure 35**. Steelhead, Puget Sound DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On June 11, 2007 NMFS listed the Puget Sound (PS) DPS of steelhead as threatened (72 FR 26722). This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia. Also, steelhead from 6 artificial propagation programs are included in this DPS.

Status. Puget Sound steelhead DPS status has not substantively changed since the listing in 2007, or since the 2011 status review. Furthermore, the PS steelhead Technical Recovery Team (TRT) recently concluded that the DPS was at very low viability, as were all 3 of its constituent MPGs, and many of its 32 DIPs (Hard et al. 2015). Recent analyses indicate that the viability of the Puget Sound steelhead DPS has improved somewhat since the PSTRT concluded that the DPS was at very low viability, as were all 3 of its constituent MPGs, and many of its 32 DIPs (Hard et al. 2015). Increases in spawner abundance were observed in a number of populations over the last 5 years. These improvements were disproportionately found within the Central & South Puget Sound and the Hood Canal & Strait of Juan de Fuca MPGs, primarily among smaller populations. The apparent reversal of strongly negative trends among winter-run populations in the White, Nisqually, and Skokomish Rivers abated somewhat the demographic risks facing those populations. Certainly, improvement in the status of the Elwha River steelhead (both winter- and summer-run) following the removal of the Elwha dams reduced the demographic and diversity risk for the DIP and the MPG. Improvements in abundance were not as widely observed in the Northern Cascades MPG. Foremost among the declines were summer- and winter-run populations in the Snohomish River basin. These populations figure prominently as sources of abundance for the MPG and DPS. Additionally, the decline in the Tolt River summer-run steelhead population was especially of concern given that it is the only population for which we have abundance estimates. The demographic and diversity risks to the Tolt River summer-run DIP are very high. In fact, all summer-run steelhead populations in the Northern Cascades MPG are likely at a very high demographic risk. In spite of improvements in some areas, most populations are still at relatively low abundance levels, with about a third of the DIPs unmonitored and presumably at very low levels.

Life History. The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS 2005c). Spawning occurs from January to mid-June, with peak spawning occurring from mid-April through May. Prior to spawning, maturing adults hold in pools or in side channels to avoid high winter flows. Less information exists for summer-run steelhead as their smaller run size and higher altitude headwater holding areas have not been conducive for monitoring. Based on information from 4 streams, adult run time occur from mid-April to October with a higher concentration from July through September (NMFS 2005c).
The majority of juveniles reside in the river system for 2 years with a minority migrating to the ocean as 1 or 3-year olds. Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from 1 to 3 years in the ocean (Busby et al. 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

Table 63. Temporal distribution of Steelhead, Puget Sound DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="12">Present</td></tr>
<tr><td>Spawning</td><td></td><td colspan="6">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td colspan="7">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td colspan="6">Present</td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. In the 1996 and 2005 status reviews, the Skagit and Snohomish Rivers (North Puget Sound) winter-run steelhead were found to produce the largest escapements (Busby et al. 1996; NMFS 2005c). The 2 rivers still produce the largest wild escapement with the 2005 to 2008 4-year geometric mean of 5,468 for the Skagit River and an average 2,944 steelhead in Snohomish River for the 2 years 2005 and 2006 (Washington Department of Fish and Wildlife (WDFW) 2009).

The Hood Canal and Strait of Juan de Fuca MPG populations experienced an increase in abundance during the 2015–19 period. The 5-year geomean for the Elwha River DIP increased to 1,241 winter-run steelhead, an 82% increase over the 2010–14 period. Productivity estimates for recent broodyears have also been strongly positive. In addition, summer-run steelhead have been observed in the upper Elwha River, with recent counts in the low hundreds of returning adults (Pess et al. 2020). Rather than a recolonization, these fish appear to be reanadromized *O. mykiss* from summer-run steelhead originally isolated behind the Elwha and Glines Canyon Dams.

Within the Northern Cascades MPG over the last 5 years, there has been considerable variability in the performance of individual basins. The winter-run populations in the Samish River and Bellingham Bay tributaries exhibited a 74% increase in the 5-year geometric mean abundance, with an average 1,305 natural-origin spawners for the present review period. Additionally, this estimate is an underestimate as it does not include tributaries to Bellingham Bay. Winter-run DIPs in the Nooksack and Skagit River basins exhibited slight increases in their average 5-year abundances, although within the 2015–19 period a negative trend in abundances is evident in the Skagit River populations with a geomean of 7,181. The Stillaguamish River winter-run DIP exhibited a moderate increase in its 5-year geomean abundance of 26%, although the longer-term trend for this population from abundance levels in the 1980s is strongly downward. DIPs in the Snohomish River basin were stable or negative. This MPG represents the majority of the abundance for the entire Puget Sound steelhead DPS, with a total abundance (based on 5-year geomeans) of over 10,000 natural spawners. Several populations have abundances over 1,000, and others over 250; however, over a third of the populations are not sufficiently monitored to develop population abundance estimates and likely have very low numbers of spawners. Except for the Samish River/Bellingham Bay Tributaries DIP and perhaps the Nooksack and Skagit Rivers, productivity for most populations was negative (in contrast to the 5-year geomean trends), suggesting a downward trend into the near future.

Steelhead populations in the Central and South Puget Sound MPG exhibited strongly positive increases in their 5-year abundances. Four populations represent the major basins in this MPG: Green River, Puyallup River, White River, and Nisqually River winter-run DIPs exhibited 94–187% increases in 5-year abundances. Long-term (15-year) trends for 3 of these populations—White, Puyallup, and Nisqually—were positive, with annual growth rates of 6–8%, while the Green River DIP long-term trend remained stable at 0%. Abundances for the White and Puyallup River winter-run DIPs remain in the low hundreds and continue to be at some demographic risk, although estimates include counts from only portions of the DIPs. Further, abundances for the Puyallup/Carbon River DIP include data series for the Puyallup and Carbon Rivers that could not be combined due to differences in survey protocols. Recent productivity for these 4 populations has been predominately positive. Two DIPs in the Lake Washington watershed, North Lake Washington Tributaries and Cedar River, had adult abundances near zero.
Productivity / Population Growth Rate. Long-term trends (1980 to 2004) for the Puget Sound steelhead natural escapement have declined significantly for most populations, especially in southern Puget Sound, and in some populations in northern Puget Sound (Stillaguamish winter-run), Canal (Skokomish winter-run), and along the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005c). Positive trends were observed in the Samish winter-run (northern Puget Sound) and the Hamma Hamma winter-run (Hood Canal) populations. The increasing trend on the Hamma Hamma River may be due to a captive rearing program rather than to natural escapement (NMFS 2005c).

The negative trends in escapement of naturally produced fish resulted from peaks in natural escapement in the early 1980s. Still, the period 1995 through 2004 (short-term) showed strong negative trends for several populations. This is especially evident in southern Puget Sound (Green, Lake Washington, Nisqually, and Puyallup winter-run), Hood Canal (Skokomish winter-run), and the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005c). As with the long-term trends, positive trends were evident in short-term natural escapement for the Samish and Hamma Hamma winter-run populations, and also in the Snohomish winter-run populations. Median population growth rates (λ) using 4-year running sums is less than 1, indicating declining population growth, for nearly all populations in the DPS (NMFS 2005c). However, some of the populations with declining recent population growth show only slight declines, (*e.g.*, Samish and Skagit winter-run in northern Puget Sound, and Quilcene and Tahuya winter-run in Hood Canal).

The long-term abundance of adult steelhead returning to many Puget Sound rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s; however, in the nearer term, there has been a relative improvement in abundance and productivity. Of the 20 datasets analyzed, abundance trends were available for 7 of the 8 winter-run DIPs in the Hood Canal & Strait of Juan de Fuca MPG; for 5 of the 8 winter-run DIPs in the Central & South Puget Sound MPG; and for 7 of the 11 winter-run DIPs, but only 1 of the 5 summer-run DIPs, in the Northern Cascades MPG (Table 54). One-third of the populations lack monitoring and abundance data; in most cases it is likely that abundances are very low. The data submitted only included natural-origin spawners, therefore statistical analyses for natural spawners and total spawners were identical.

Genetic Diversity. Only 2 hatchery stocks genetically represent native local populations (Hamma Hamma and Green River natural winter-run). The remaining programs, which account for the vast preponderance of production, are either out-of-DPS derived stocks or were within-DPS stocks that have diverged substantially from local populations. The WDFW estimated that 31 of the 53 stocks were of native origin and predominantly natural production (Washington Department of Fish and Wildlife (WDFW) 1993).

Distribution. NMFS listed Puget Sound steelhead as threatened on May 11, 2007 (72 FR 26722). Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run steelhead in the rest of the DPS. The Elwha River run, however, is descended from the introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s.

Designated Critical Habitat. NMFS designated critical habitat for Puget Sound steelhead on February 2, 2016 (81 FR 9251). PBFs considered essential for the conservation of Steelhead, Puget Sound DPS are described in Appendix B.

Recovery Goals. See the 2019 Recovery Plan (NMFS 2019) for a complete description of recovery goals and criteria. The overarching viability criteria for this DPS is that "all 3 of the species MPGs (the Central and South Puget Sound MPG, Hood Canal and Strait of Juan de Fuca MPG, and North Cascades MPG) need to be viable for the DPS to be removed from the ESA's threatened and endangered species list." Currently, all 3 MPGs remain at low viability.

8.26 Steelhead, Snake River Basin

Table 64. Steelhead, Snake River Basin DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Snake River Basin	Threatened	2022	2006 71 FR 834	2017	2005 70 FR 52630

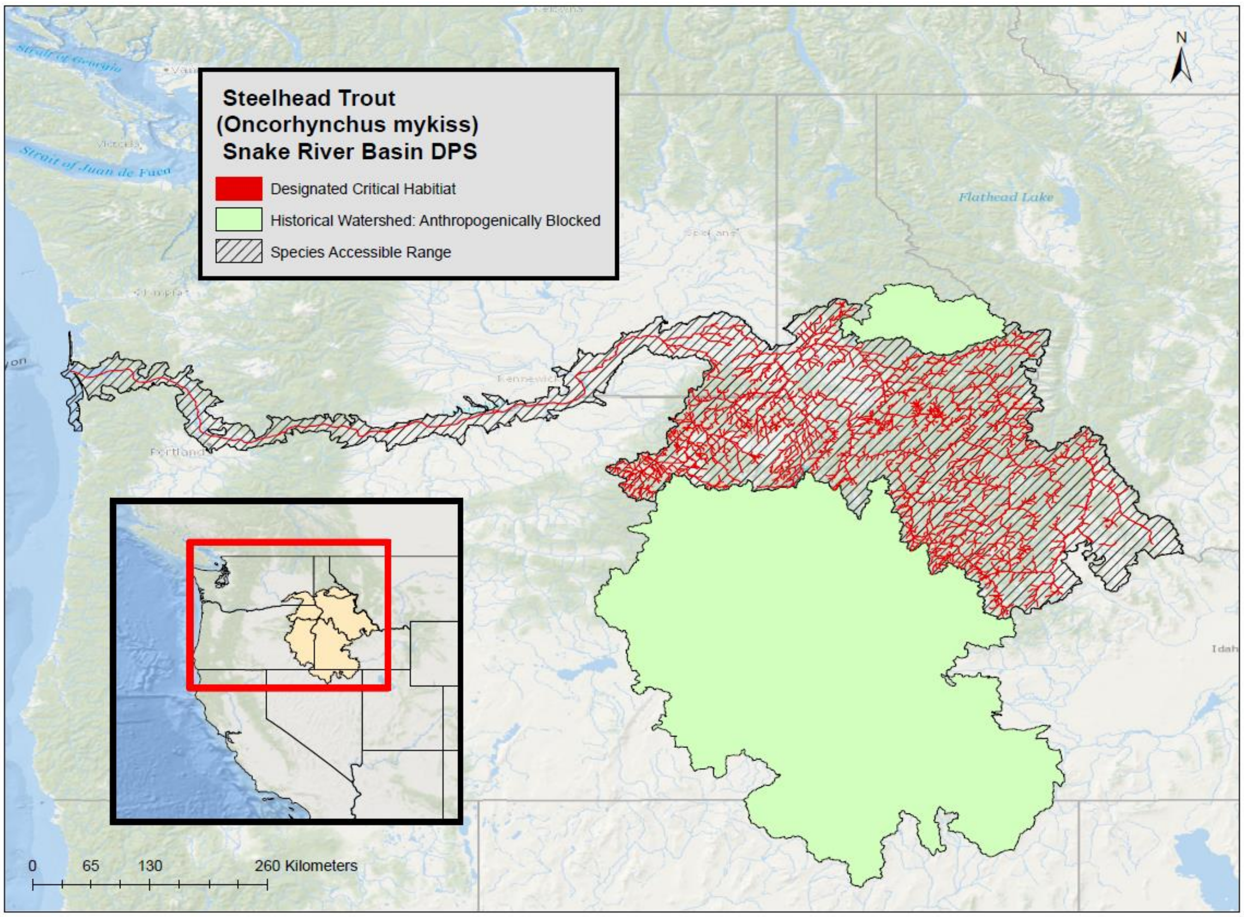


Figure 36**. Steelhead, Snake River Basin DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On August 18, 1997 NMFS listed the Snake River Basin (SRB) DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). The 2022 Status Update found no new information that would justify a change in the delineation of the SRB steelhead DPS (Ford 2022).This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from the Snake River basin, and also steelhead from 6 artificial propagation programs. The 2022 status review updated the SRB DPS listing to reflect the following 6 changes made by NMFS to hatchery programs (85 FR 81822): (1) added the Salmon River B-run Program because the existing release is now classified as a separate and distinct program; (2) added the South Fork Clearwater (Clearwater Hatchery) B-run program because the existing release is now classified as a separate and distinct program; (3) changed the name of the East Fork Salmon River Program to the East Fork Salmon River Natural Program; (4) removed the Lolo Creek Program because it is now considered part of the listed Dworshak National Fish Hatchery Program; (5) removed the North Fork Clearwater Program because it is now considered part of the listed Dworshak National Fish Hatchery Program, and; (6) changed the name of the Little Sheep Creek/Imnaha River Hatchery Program to the Little Sheep Creek/Imnaha Program.

Status. Based on the updated viability information available for this review, all 5 MPGs are not meeting the specific objectives in the draft recovery plan, and the viability of many individual populations remains uncertain (NWFSC 2015b). The Grande Ronde MPG is tentatively rated as viable; more specific data on spawning abundance and the relative contribution of hatchery spawners for the Lower Grande Ronde and Wallowa populations would improve future assessments. A great deal of uncertainty still remains regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites within individual populations. Overall, the information analyzed for this viability review indicates that the Snake River Basin steelhead DPS remains at "moderate" risk of extinction, with viability largely unchanged from the prior review. Of particular note, the updated, population-level abundance estimates have made very clear the recent (last 5 years) sharp declines that are extremely worrisome, were they to continue.

Life History. SR basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, SR basin steelhead spawn the following spring (March to May). Managers recognize 2 life history patterns within this DPS primarily based on ocean age and adult size upon return: A-run and B-run. A-run steelhead are typically smaller, have a shorter freshwater and ocean residence (generally 1 year in the ocean), and begin their up-river migration earlier in the year. B-run steelhead are larger, spend more time in fresh water and the ocean (generally 2 years in ocean), and appear to start their upstream migration later in the year. SR basin steelhead usually smolt after 2 or 3 years.

Table 65. Temporal distribution of Steelhead, Snake River Basin DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td></td><td></td><td></td><td></td><td colspan="6">Present</td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance / Productivity.
There is uncertainty for wild populations given limited data for adult spawners in individual populations. Regarding population growth rate, there are mixed long- and short-term trends in abundance and productivity. Overall, the abundances remain well below interim recovery criteria. The 5-year geometric mean abundance estimates for the populations in this DPS all show significant declines in the recent past. Each of the populations decreased by roughly 50% in the past 5-year period, resulting in a near-zero population change in the past 15 years for the 3 populations with sufficiently long data time series. Hatchery-origin spawner estimates for these populations continued to be low.

Populations in the Snake River Basin steelhead DPS exhibited similar temporal patterns in broodyear returns per spawner, oscillating with a rough period of 10 years. Return rates for broodyears 1995−99 generally exceeded replacement (1:1). Spawner-to-spawner ratios for broodyears 2001−03 were generally well below replacement for many populations, cycling above replacement during 2005–10, and strongly below replacement since 2010. Broodyear return rates reflect the combined impacts of year-to-year patterns in marine life-history stages, upstream and downstream passage survivals, as well as density-dependent effects resulting from capacity or survival limitations on tributary spawning or juvenile rearing habitats.

Genetic Diversity. The ICTRT identified 24 extant populations within this DPS, organized into 5 major population groups (ICTRT 2003). They also identified a number of potential historical populations associated with tributary habitat above the Hells Canyon Dam complex on the mainstem Snake River, a barrier to anadromous migration. The 5 MPGs with extant populations are Lower Snake River (2 populations), Clearwater River (5 extant populations, 1 extirpated), Grande Ronde River (4 populations), Imnaha River (1 population), and Salmon River (12 populations). In addition, the ICTRT concluded that small tributaries entering the mainstem Snake River below Hells Canyon Dam may have historically been part of a larger population with a core area currently cut off from anadromous access. That population would have been part of 1 of the historical upstream MPGs.

Distribution. The ICTRT (ICTRT 2003) identified 23 populations. SR basin steelhead remain spatially well distributed in each of the 6 major geographic areas in the Snake River basin (Good et al. 2005b). The SR basin steelhead B- run populations remain particularly depressed. Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). PBFs considered essential for the conservation of Steelhead, Snake River Basin DPS are described in Appendix B.

The current condition of critical habitat designated for SR basin steelhead is moderately degraded. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds. Contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. These factors have substantially reduced the rearing PBFs contribution to the conservation value necessary for species recovery. Several dams affect adult migration PBF by obstructing the migration corridor.

Recovery Goals. See the 2017 Recovery Plan (NMFS 2017d)for a complete list of recovery goals and criteria. The recovery plan outlines a variety of different scenarios that may lead to a viable DPS, however, the overall criteria for the DPS is that all extant MPGs and any extirpated MPGs critical for proper functioning of the DPS should be at low risk.

8.27 Steelhead, South-Central California Coast DPS

Table 66. Steelhead, South-Central California Coast DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	South-Central California Coast	Threatened	2023	2006 71 FR 834	2013	2005 70 FR 52488

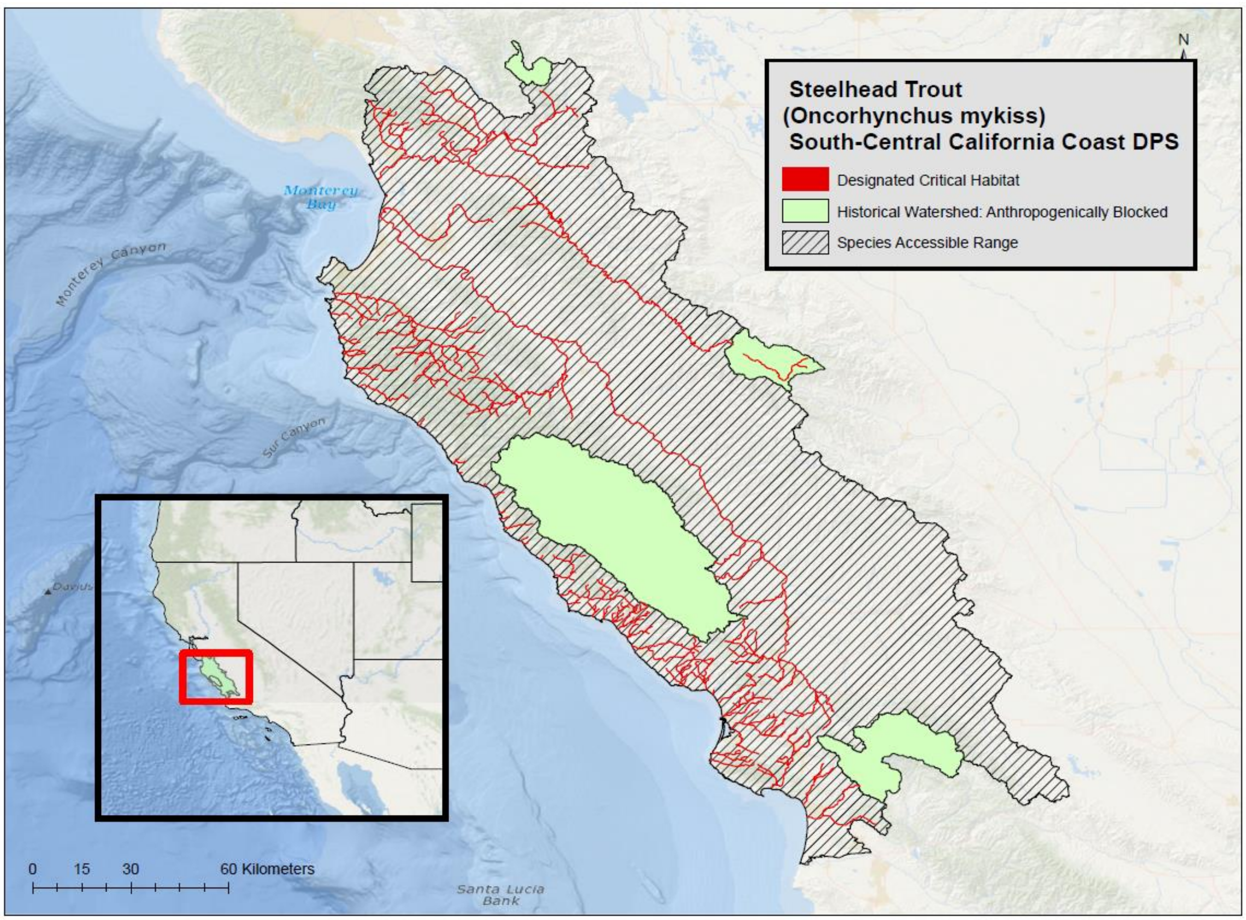


Figure 37**. Steelhead, South-Central California Coast DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On August 18, 1997 NMFS listed the South-Central California Coast (SCCC) DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 5248). This DPS includes naturally spawned anadromous O. mykiss (steelhead) originating below natural and manmade impassable barriers from the Pajaro River to (but not including) the Santa Maria River.

Status. Following the dramatic rise in South-Central California's human population after World War II and the associated land and water development within coastal drainages (particularly major dams and water diversions), steelhead abundance rapidly declined, leading to the extirpation of populations in many watersheds and leaving only sporadic and remnant populations in the remaining, more highly modified watersheds such as the Salinas River and Arroyo Grande Creek watersheds (Boughton et al. 2007; Good et al. 2005b). As conditions in South-Central California coastal rivers and streams continued to deteriorate, put-and-take trout stocking became more focused on suitable man made reservoirs. Since the listing of the SCCC DPS as threatened in 1997, the California Department of Fish and Wildlife has ceased stocking hatchery reared fish in the anadromous waters of South-Central California (California Department of Fish and Wildlife and U.S. Fish and Wildlife Service 2010). A substantial portion of the upper watersheds, which contain the majority of historical spawning and rearing habitats for anadromous *O. mykiss*, remain intact (though inaccessible to anadromous fish) and protected from intensive development as a result of their inclusion in the Los Padres National Forest (Blakley and Barnette 1985).

Life History. Only Winter run steelhead are found in this DPS. Migration and spawn timing are similar to adjacent steelhead populations. There is limited life history information for steelhead in this DPS.

Table 67. Temporal distribution of Steelhead, South-Central California Coast DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td colspan="2">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance / Productivity. The data summarized in this status review indicate small (generally <10 adult spawners) but surprisingly persistent annual runs of anadromous O. mykiss are currently being monitored across a limited but diverse set of basins within the range of this DPS, but interrupted in years when the mouth of the coastal estuaries fail to open to the ocean due to low flows (Williams et al. 2011; Williams et al. 2016).

Genetic Diversity / Distribution. South-Central California Coast (SCCC) steelhead include all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The 2 largest basins overlapping within the range of this DPS include the inland basins of the Pajaro River and the Salinas River.

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). PBFs considered essential for the conservation of Steelhead, South-Central California Coast DPS are described in Appendix B.

Migration and rearing PBFs are degraded throughout critical habitat by elevated stream temperatures and contaminants from urban and agricultural areas. Estuarine PBF is impacted by most estuaries being breached, removal of structures, and contaminants.

Recovery Goals. See the 2013 recovery plan (NMFS 2013c) for the South-Central California Coast steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species.

8.28 Steelhead, Southern California DPS

Table 68. Steelhead, Southern California DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Southern California Coast	Endangered	2016	2006 71 FR 834	2012	2005 70 FR 52488

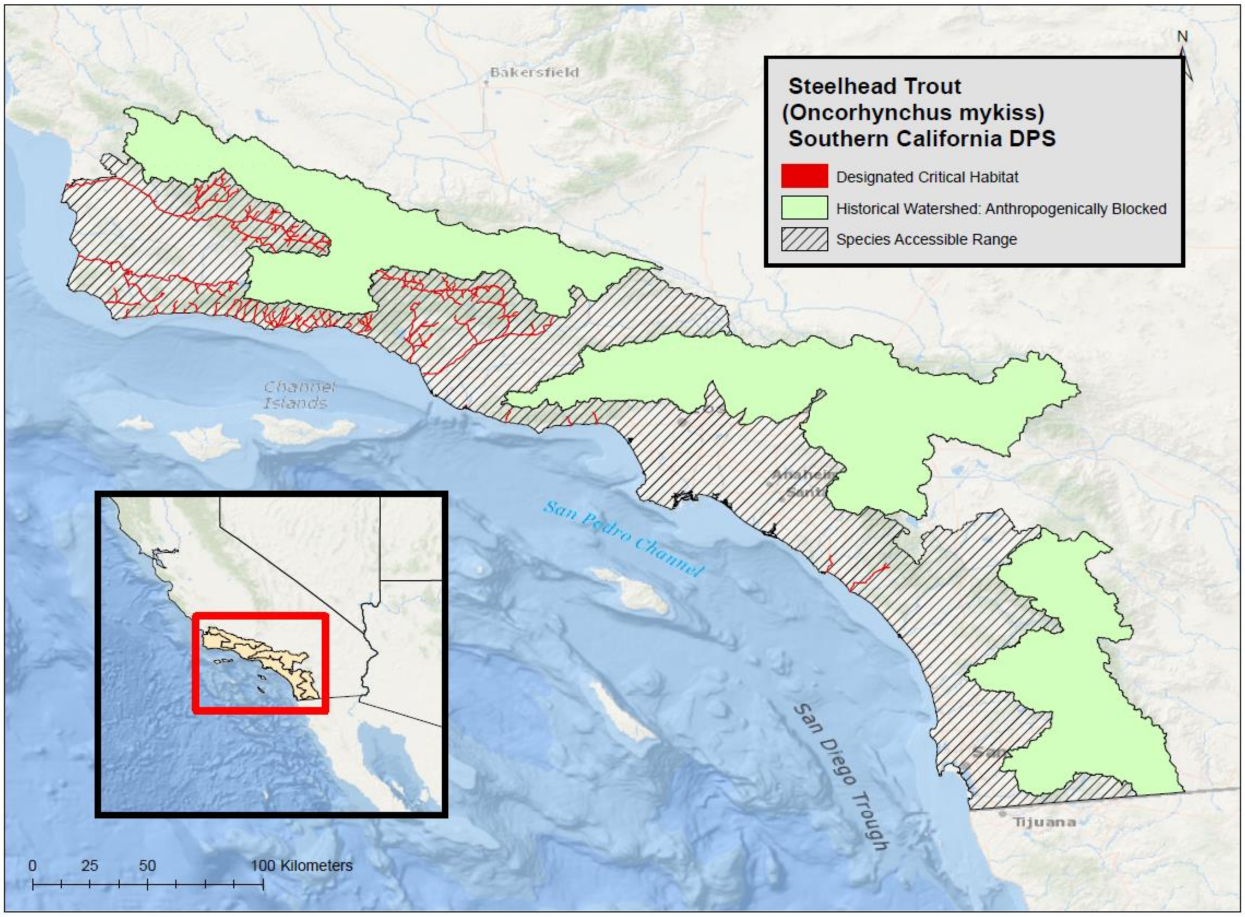


Figure 38**. Steelhead, Southern California DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On August 18, 1997 NMFS listed the Southern California (SC) DPS of steelhead as endangered (62 FR 43937) and reaffirmed the DPS's status as endangered on January 5, 2006 (71 FR 5248). This DPS includes naturally spawned anadromous *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from the Santa Maria River to the U.S.-Mexico Border.

Status. There is little new evidence to indicate that the status of the Southern California Coast Steelhead DPS has changed appreciably in either direction since the last status review (Williams et al. 2011). The extended drought and the recent genetic data documenting the high level of introgression and extirpation of native *O. mykiss* stocks in the southern portion of the DPS has elevated the threats level to the already endangered populations; the drought, and the lack of comprehensive monitoring, has also limited the ability to fully assess the status of individual populations and the DPS as a whole. The systemic anthropogenic threats identified at the time of the initial listing have remained essentially unchanged over the past 5 years, though there has been significant progress in removing fish passage barriers in a number of the smaller and mid-sized watersheds. Threats to the Southern California Steelhead DPS posed by environmental variability resulting from projected climate change are likely to exacerbate the factors affecting the continued existence of the DPS.

Life History. There is limited life history information for SC steelhead. In general, migration and life history patterns of SC steelhead populations are dependent on rainfall and streamflow (Moore 1980). Steelhead within this DPS can withstand higher temperatures compared to populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead compared to the more northerly populations (Moore 1980).

Table 69. Temporal distribution of Steelhead, Southern California DPS in freshwater habitats

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)		Present	Present	Present	Present							
Spawning				Present	Present							
Incubation (eggs)				Present	Present	Present						
Emergence (alevin to fry phases)						Present	Present					
Rearing and migration (juveniles)	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present	Present

Population Dynamics

Abundance / Productivity. Limited information exists on SC steelhead runs. Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS historically. In contrast, less than 500 adults are estimated to occupy the same 4 waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults (Busby et al. 1996).

Genetic Diversity / Distribution. Limited information is available regarding the structural and genetic diversity of the Southern California steelhead.

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). PBFs considered essential for the conservation of Steelhead, Southern California DPS are described in Appendix B.

All PBFs have been affected by degraded water quality by pollutants from densely populated areas and agriculture within the DPS. Elevated water temperatures impact rearing and juvenile migration PBFs in all river basins and estuaries. Rearing and spawning PBFs have also been affected throughout the DPS by management or reduction in water quantity. The spawning PBF has also been affected by the combination of erosive geology and land management activities that have resulted in an excessive amount of fines in the spawning gravel of most rivers.

Recovery Goals. See the 2012 recovery plan (NMFS 2012c) for the California Central Valley steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species.

8.29 Steelhead, Upper Columbia River DPS

Table 70. Steelhead, Upper Columbia River DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Upper Columbia River	Threatened	2022	2009 74 FR 42605	2007	2005 70 FR 52630

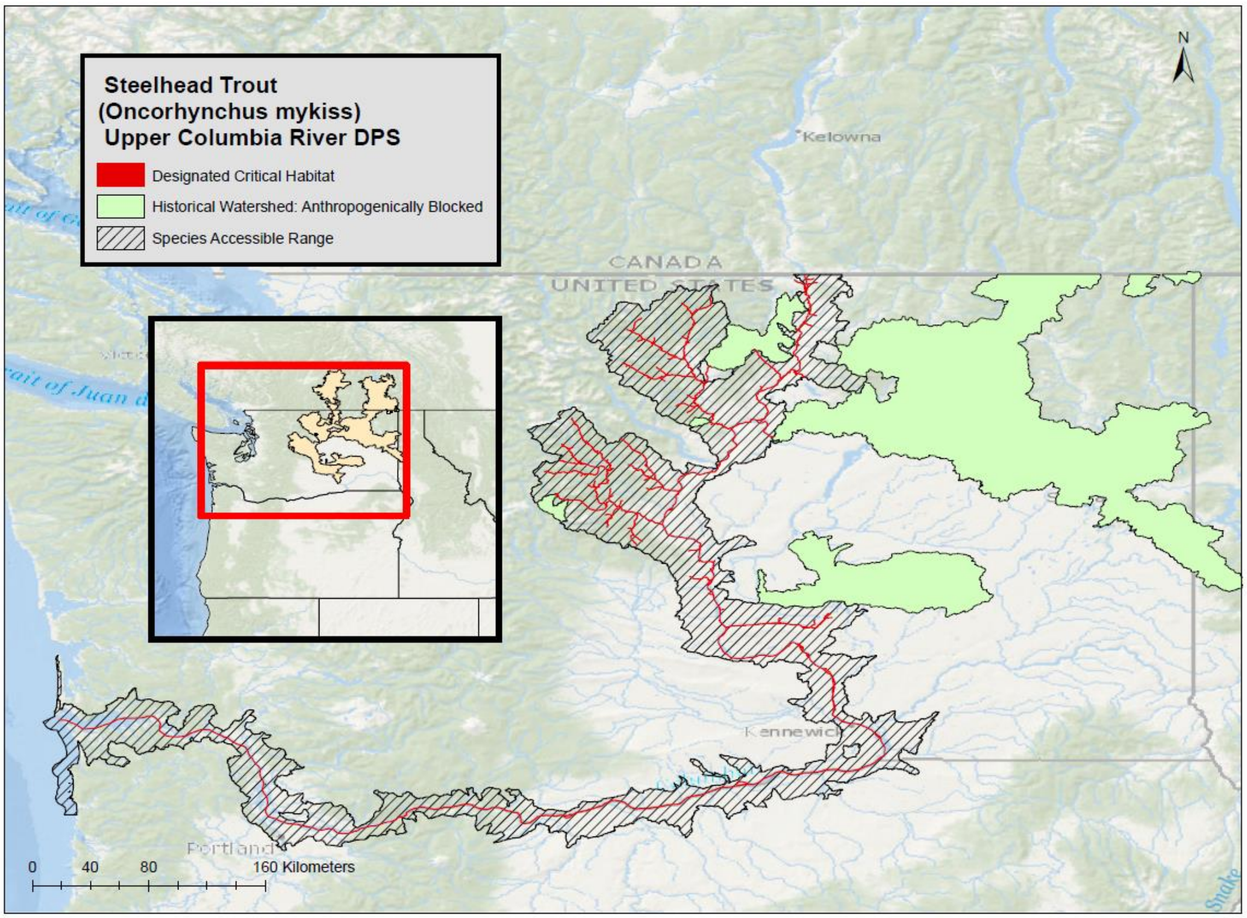


Figure 39**. Steelhead, Upper Columbia River DPS range and designated critical habitat**

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On August 18, 1997 NMFS listed the Upper Columbia River (UCR) DPS of steelhead as endangered (62 FR 43937) and reclassified the DPS's status as threatened on January 5, 2006 (71 FR 833) and this was reaffirmed on August 24, 2009 (74 FR 42605); updated April 14, 2014 (79 FR 20802). The 2022 Status Update found no new information that would justify a change in the delineation of the

UCR steelhead DPS (Ford 2022). This DPS includes naturally spawned anadromous *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from the Columbia River and its tributaries upstream of the Yakima River to the U.S.-Canada border. Also, steelhead from 6 artificial propagation programs.

Status. The most recent estimates (5-year geometric mean) of total and natural-origin spawner abundance have declined dramatically, erasing gains observed over the past 2 decades for all 4 populations (Wenatchee, Entiat, Methow and Okanogan Rivers). Recent declines are persistent and large enough to result in small, but negative, 15-year trends in abundance for all 4 populations. The abundance and productivity viability rating for the Wenatchee River exceeds the minimum threshold for 5% extinction risk. The overall Upper Columbia River steelhead DPS viability remains largely unchanged from the prior review, and the DPS is at high risk driven by low abundance and productivity relative to viability objectives and diversity concerns. The Northwest Fisheries Science Center's review of updated information (Ford 2022) does not indicate a change in the biological risk category for this species since the time of the last 5-year review (NWFSC 2015). Analysis of the ESA section 4(a)(1) factors indicates that the collective risk to the UCR Steelhead's persistence has not changed significantly since our previous 5-year review for the UCR Steelhead DPS.

Life History. All UCR steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinters in mainstem reservoirs, passing over upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend 1 to 7 years rearing in fresh water before migrating to sea. Smolt outmigrations are predominantly year class 2 and 3 (juveniles), although some of the oldest smolts are reported from this DPS at 7 years. Most adult steelhead return to fresh water after 1 or 2 years at sea.

Table 71. Temporal distribution of Steelhead, Upper Columbia River DPS in freshwater habitats

<table>
<tr><th>Life History phase</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="12">Present</td></tr>
<tr><td>Spawning</td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td colspan="4">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases</td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. The average 1997 to 2001 return counted through the Priest Rapids fish ladder was approximately 12,900 returning adults. The average for the previous 5 years (1992 to 1996) was 7,800 returning adult fish. Abundance estimates of returning naturally produced UCR steelhead were based on extrapolations from mainstem dam counts and associated sampling information (Good et al. 2005b). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICTRT 2003). Adult abundances for the Wenatchee and Entiat aggregate population and the Methow population were low with a 5-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat Rivers (combined).

The most recent estimates (5-year geometric mean) of total and natural-origin spawner abundance have declined dramatically, erasing gains observed over the past 2 decades for all 4 populations. Naturally produced spawner 5-year geomean were estimated at 554, 92, 595, and 223 for the Wenatchee, Entiat, Methow, and Okanogan Rivers, respectively. Recent declines are persistent and large enough to result in small, but negative, 15-year trends in abundance for all 4 populations. Updated spawner estimation methods show a strong concordance with existing methods, which is extremely encouraging as the estimation process based on detecting tags from a run-at-large tagging program is a very robust approach to monitoring across the DPS.

Hatchery-origin returns continue to constitute a high fraction of total spawners in natural spawning areas for this DPS. The estimated proportion of natural-origin spawners has increased consistently since the late 1990s for all 4 populations. Natural-origin proportions were highest in the Wenatchee River (58%). Although increasing, natural-origin proportions in the Methow and Okanogan Rivers remained at low levels.

Productivity / Population Growth Rate. Annual brood-year R/S estimates have been well below replacement in recent years for all 4 populations. The R/S estimates summarized in Figure 14 are ratios of the estimated natural-origin returns produced from spawners in each brood year, under the assumption that both hatchery- and natural-origin fish contribute to production as parent spawners. All populations are consistently exhibiting natural production rates well below replacement, and natural production has also declined consistently, resulting in an increasing fraction of hatchery fish on the spawning grounds each year.

Genetic Diversity. It was initially determined that all UCR steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman et al. 1994). Genetics samples taken in the 1980s indicate little differentiation within populations in the Upper Columbia River steelhead DPS. More recent studies within the Wenatchee River basin have found differences between samples from the Peshastin River, believed to be relatively isolated from hatchery spawning, and those from other reaches in the basin. This suggests that there may have been a higher level of within- and among-population diversity prior to the advent of major hatchery releases (Seamons et al. 2012).

Distribution. The UCR steelhead consisted of 4 historical independent populations: the Wenatchee, Entiat, Methow, and Okanogan. All populations are extant. The UCR steelhead must navigate over several dams to access spawning areas. The construction of Grand Coulee Dam in 1939 blocked access to over 50% of the river miles formerly available to UCR steelhead (ICTRT 2003). With the exception of the Okanogan population, the upper Columbia River steelhead populations were rated as low-risk for spatial structure. The high-risk ratings for diversity are largely driven by high levels of hatchery spawners within natural spawning areas, and lack of genetic diversity among the populations. The basic major life-history patterns (summer A-run type, tributary and mainstem spawning/rearing patterns, and the presence of resident populations and subpopulations) appear to be present. All of the populations were rated at high risk for current genetic characteristics by the ICTRT.

Designated Critical Habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). PBFs considered essential for the conservation of Steelhead, Upper Columbia River DPS are described in Appendix B.

The current condition of critical habitat designated for the UCR steelhead is moderately degraded. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PBF by obstructing the migration corridor.

Recovery Goals. See the 2007 recovery plan (Upper Columbia Salmon Recovery Board 2007) for the Upper Columbia River steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species.

8.30 Steelhead, Upper Willamette River DPS

Table 72. Steelhead, Upper Willamette River DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	California Central Valley	Threatened	2022	2006 71 FR 834	2011	70 FR 52630

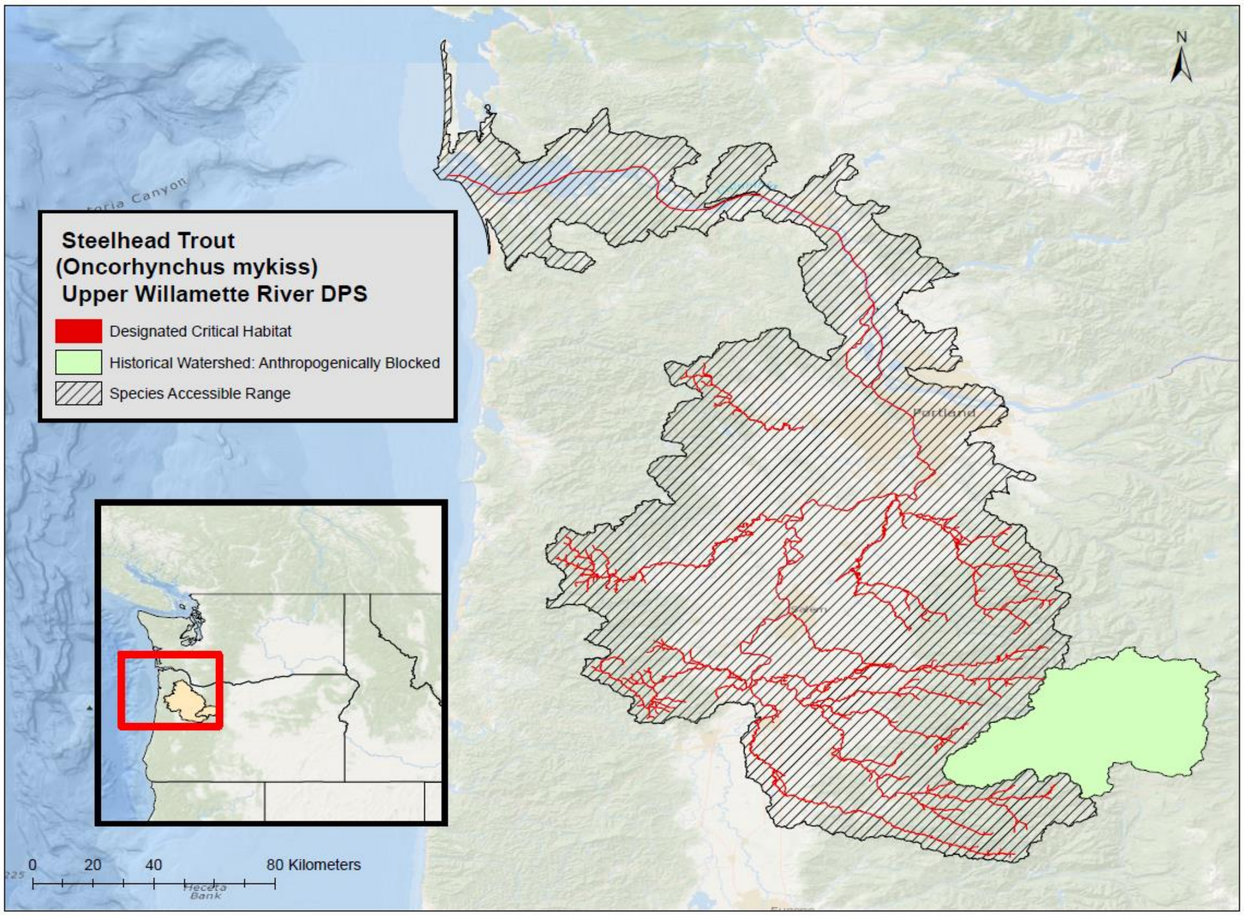


Figure 40. Steelhead, Upper Willamette River DPS range and designated critical habitat

Species Description. Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though the average size is much smaller. On March 25, 1999 NMFS listed the Upper Willamette River (UWR) DPS of steelhead as threatened (64 FR 14517) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834). This DPS includes naturally spawned anadromous winter-run *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from the Willamette River and its tributaries upstream of Willamette Falls to and including the Calapooia River.

Status. Four basins on the east side of the Willamette River historically supported independent populations for the UWR steelhead, all of which remain extant. Data reported in McElhaney et al. (2007) indicate that currently the 2 largest populations within the DPS are the Santiam River populations. Mean spawner abundance in both the North and South Santiam River is about 2,100 native winter-run steelhead. However, about 30% of all habitat has been lost due to human activities (McElhany et al. 2007a). The North Santiam population has been substantially affected by the loss of access to the upper North Santiam basin. The South Santiam subbasin has lost habitat behind non-passable dams in the Quartzville Creek watershed. Notwithstanding the lost spawning habitat, the DPS continues to be spatially well distributed, occupying each of the 4 major subbasins.

Overall, the Upper Willamette River steelhead DPS continued to decline in abundance. Although the most recent counts at Willamette Falls and the Bennett Dams in 2019 and 2020 suggest a rebound from the record 2017 lows, it should be noted that current "highs" are equivalent to past lows. Uncertainty in adult counts at Willamette Falls are a concern, given that the counts represent an upper bound on DPS abundance. Radio-tagging studies suggest that a considerable proportion of "winter" steelhead ascending Willamette Falls do not enter the tributaries that are considered part of this DPS; these fish may be non-native early-winter steelhead that appear to have colonized the western tributaries, misidentified summer steelhead, late-winter steelhead that have colonized tributaries not historically part of the DPS, or hybrids between native and non-native steelhead. More definitive genetic monitoring of steelhead ascending Willamette Falls, in tandem with radio tagging work, needs to be undertaken to estimate the total abundance of the DPS.

Life History. Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al. 1985). UWR steelhead do not ascend to their spawning areas until late March or April, which is late compared to other West Coast winter steelhead. Spawning occurs from April to June 1. The unusual run timing may be an adaptation for ascending the Willamette Falls, which may have facilitated reproductive isolation of the stock. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell et al. 1985). Smolts generally migrate through the Columbia via the Multnomah Channel rather than the mouth of the Willamette River. As with other coastal steelhead, the majority of juveniles smolt and outmigrate after 2 years; adults return to their natal rivers to spawn after spending 2 years in the ocean. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby et al. 1996).

Table 73. Temporal distribution of Steelhead, Upper Willamette River DPS in freshwater habitats

<table>
<tr><td>Life History phase</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sep</td><td>Oct</td><td>Nov</td><td>Dec</td></tr>
<tr><td>Entering Fresh Water (adults/jacks)</td><td colspan="6">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Spawning</td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Incubation (eggs)</td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Emergence (alevin to fry phases)</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="3">Present</td><td></td><td></td><td></td></tr>
<tr><td>Rearing and migration (juveniles)</td><td colspan="12">Present</td></tr>
</table>

Population Dynamics

Abundance. UWR steelhead are moderately depressed from historical levels (McElhany et al. 2007a). Average number of late-fall steelhead passing Willamette Falls decreased during the 1990s to less than 5,000 spawners. The number again increased to over 10,000 spawners in 2001 and 2002. The geometric and arithmetic mean number of late-run steelhead passing Willamette Falls for the period 1998 to 2001 were 5,819 and 6,795, respectively.

Winter steelhead counts at Willamette Falls provide a complete count of fish returning to the DPS. In the last 5 years, counts of steelhead at Willamette Falls experienced a marked decrease, with a record low count in 2017 of 822. During the 2016–17 return year, pinniped predation at Willamette Falls became a concern. Increases in the pinniped population at the falls, in conjunction with low steelhead return, resulted in an estimated 25% predation rate on winter steelhead (Steingass et al. 2019). With the initiation of pinniped control measures in 2019 and improvements in the steelhead run size, predation levels fell to an estimated 8% in 2019 (Steingass et al. 2019). Overall, there was a 59% decrease in the geometric average for 2015–19 relative to 2010–14. Abundances at Willamette Falls appear to have recovered since the 2017 low, with a recent (unofficial) count of 5,510 winter-run steelhead.

Productivity / Population Growth Rate. Population information for individual basins exist as redds per (river) mile. These redd counts show a declining long-term trend for all populations (Good et al. 2005b). One population, the Calapooia, had a positive short-term trend during the years from 1990 to 2001. McElhany *et al.* (2007a). While the viability of the ESU appears to be declining, the recent uptick in abundance may provide a short-term demographic buffer. Furthermore, increased monitoring is necessary to provide quantitative verification of sustainability for most of the populations. In the absence of substantial changes in accessibility to high-quality habitat, the DPS will remain at "moderate-to-high" risk. Overall, the Upper Willamette River steelhead DPS is therefore at "moderate-to-high" risk, with a declining viability trend.

Genetic Diversity. Introgression by non-native summer-run steelhead continues to be a concern. Genetic analysis suggests that there is introgression among native late-winter steelhead and summer-run steelhead (Van Doornik et al. 2015, Johnson et al. 2018, 2021).
Distribution. The UWR steelhead DPS includes all naturally spawned winter-run steelhead populations in the Willamette River and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive). The North Santiam and South Santiam rivers are thought to have

been major production areas (McElhany et al. 2003) and these populations were designated as “core” and “genetic legacy”. The 4 “east-side” subbasin populations are part of 1 stratum, the Cascade Tributaries Stratum, for UWR winter steelhead. Accessibility to historical spawning habitat is still limited, especially in the North Santiam River. Efforts to provide juvenile downstream passage at Detroit Dam are well behind the proscribed timetable (NMFS 2008), and passage at Green Peter Dam has not yet entered the planning stage. Much of the accessible habitat in the Molalla, Calapooia, and the lower reaches of the North and South Santiam Rivers is degraded and under continued development pressure. Although habitat restoration efforts are underway, the time scale for restoring functional habitat is considerable. There are no hatchery programs supporting this DPS (Myers et al. 2006). The hatchery summer-run steelhead that are produced and released in the subbasins are from an out-of-basin stock and not considered part of the DPS. Accessibility to historical spawning habitat is still limited, especially in the North Santiam River. Much of the accessible habitat in the Molalla, Calapooia, and lower reaches of North and South Santiam rivers is degraded and under continued development pressure. Although habitat restoration efforts are underway, the time scale for restoring functional habitat is considerable (NWFSC 2015b).

Designated Critical Habitat. NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488). PBFs considered essential for the conservation of Steelhead, Upper Willamette River DPS are described in Appendix B.

The current condition of critical habitat designated for the UWR steelhead is degraded, and provides a reduced conservation value necessary for species recovery. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PBF by obstructing the migration corridor.

Recovery Goals. See the 2011 recovery plan (NMFS 2011b) for the Upper Willamette River steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species.

8.31 Eulachon, Southern DPS

Table 74. Eulachon. Southern DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Thaleichthys pacificus	Eulachon	Southern	Threatened	2022	2010 75 FR 13012	2017	2011 76 FR 65323

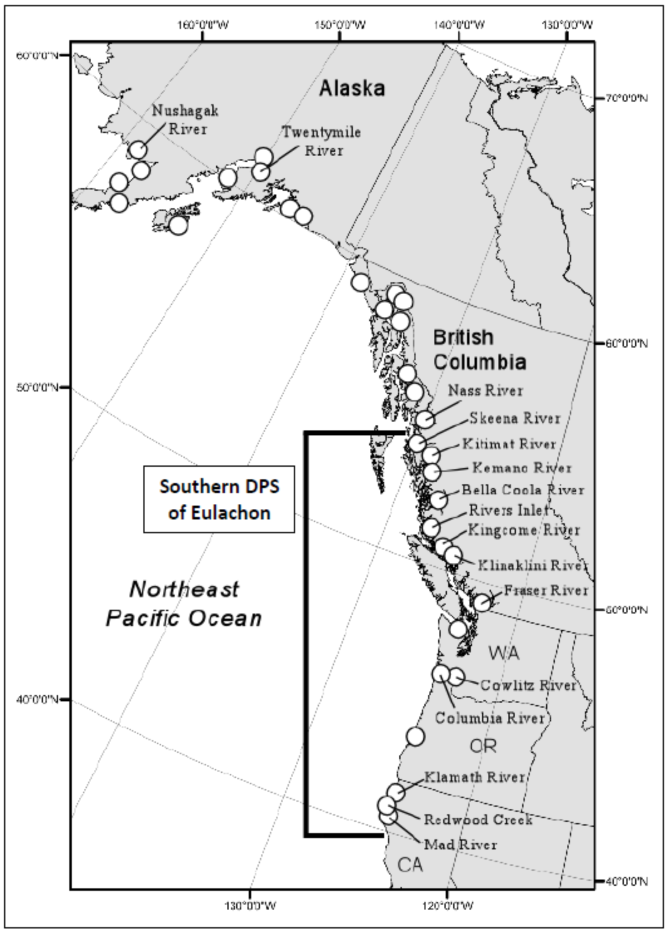


Figure 41**. Eulachon, Southern DPS range. From the NMFS 2016 5-Year Review (NMFS 2016).**

Species Description. The eulachon is a small, cold-water species of anadromous fish, occupying the eastern Pacific Ocean in nearshore waters to depths of about 1,000 feet (300 meters) from California to the Bering Sea. Eulachon will return to their natal river spawn. Southern DPS eulachon are those that spawn in rivers south of the Nass River in British Columbia to the Mad River in California (NMFS 2016b).

Status. Eulachon formerly experienced widespread, abundant runs and have been a staple of Native American diets for centuries along the northwest coast. However, such runs that were formerly present in several California rivers as late as the 1960s and 1970s (i.e., Klamath River, Mad River and Redwood Creek) no longer occur (Larson and Belchik 2000). This decline likely began in the 1970s and continued until, in 1988 and 1989, the last reported sizeable run occurred in the Klamath River and no fish were found in 1996, although a moderate run was noted in 1999 (Larson and Belchik 2000; Moyle 2002b). Eulachon have not been identified in the Mad River and Redwood Creek since the mid-1990s (Moyle 2002b). The species is considered to be at moderate risk of extinction throughout its range because of a variety of factors, including ocean conditions, predation, commercial and recreational fishing pressure (directed and bycatch), water quality, and loss of habitat. Warmer water temperatures associated with climate change could alter the timing of spawning, and the availability of prey for larval and juvenile eulachon (NMFS 2016b). Further population decline is anticipated to continue as a result of climate change and bycatch in commercial fisheries. However, because of their fecundity, eulachon are assumed to have the ability to recover quickly if given the opportunity associated with favorable ocean conditions (Bailey and Houde 1989).

Ocean Conditions. From late 2013 to mid-2017 the California Current Ecosystem (CCE) experienced both a severe marine heat wave (MHW) in the form of the “Blob” (2013–16) and a strong El Niño event (2015– 16). The impact of the “Blob” on eulachon abundance is likely reflected in the 2018 Columbia River SSB estimate of slightly more than 4 million fish, the lowest since 2010. Eulachon returning to the Columbia River in 2018 were mostly from the broodyears 2015 or 2016, which would have entered the CCE in spring to summer of those years, when both the “Blob” and the strong El Niño of 2015–16 were active. In 2015 and 2016, the biological spring transition never occurred, as northern copepods were absent from surveys along the Newport Hydrographic Line during both years. Euphausiids (commonly called krill), the primary prey of juvenile/adult eulachon, experienced very low densities during 2015–2016, which likely had negative impacts on eulachon growth and survival. Additional MHWs developed in May 2019, and again in 2020 and 2021, although the latter 2 MHWs mostly stayed offshore and had low impact on the CCE.

The near-term outlook for eulachon productivity in the CCE is positive, based on the presence of good ocean conditions. The current abundance of northern copepods and depressed numbers of southern copepods in the CCE would be expected to result in increased eulachon survival. The return of good ocean conditions and the likelihood that these conditions will persist into the near future suggests that population stabilization or increases may be widespread in the upcoming return years. The productivity potential as indicated by life history characteristics such as low age-at maturity, small body size, planktonic larvae, and perhaps their high fecundity, confers eulachon with some resilience to environmental perturbations, as they retain the ability to quickly respond to favorable ocean conditions.

Montgomery (2020) used a multivariate analyses to look at ocean ecosystem indicators in years when ocean residency is correlated with eulachon abundance in the Columbia River. Large-scale and bottom-up indicators such as the status of the Pacific Decadal Oscillation and prey abundance describe much of the variation in eulachon abundance. The time series analysis also indicates eulachon abundance correlates strongly with ocean conditions in the 2 and 3 years prior to their return, suggesting dominant life histories of 2- and 3-year ocean types.

Total fleetwide bycatch in U.S. West Coast groundfish and ocean shrimp trawl fisheries continue to affect productivity of Eulachon. However in recent years the numbers caught in these fisheries have been in decline.

2022 Updated Risk Summary

California
Although the Yurok Tribal Fisheries Program has not conducted any official eulachon surveys since 2014, eulachon have been observed in small numbers (no more than 2 per night) at the mouth of the Klamath River every year since then (Gustafson et al. 2022).

Oregon/Washington
Since the 2016 5-year status review, annual monitoring of eulachon spawning stock biomass (SSB) has continued in the Columbia (2011–2021) River; and expanded to the Grays (2011–2013, 2015– 2016), Cowlitz (2015–2018), Naselle (2015–2017), and Chehalis (2015–2018) Rivers (Gustafson et al. 2022). In 2018, eulachon were 2.4 times more abundant in the Fraser River than in the Columbia River. Although the cause of the decline in Columbia River SSB in 2017–2018 is unknown, it is possibly related to the 2013–2016 MHW, known as the “Blob,” that reduced productivity of northern copepods and euphausiids, critical prey for eulachon in the CCE. The Lower Elwha S’klallam Tribe has sampled eulachon adults and/or larvae in the Elwha, Dungeness, and Lyre Rivers on the Olympic Peninsula of Washington.

Columbia River Basin
In the Columbia River, eulachon abundance decreased markedly since the 2016 status review. The decrease in abundance reflects both changes in biological status and changes in ocean conditions. For the years 2011 through 2015, the 5-year SSB mean was 97.9 million spawners. For the years 2016 through 2021, the 6-year SSB mean was 40.2 million spawners. However, the 2021 estimate (96.4 million spawners) was nearly equivalent to the 2011-2015 mean.

Canada
No new information on the status of Klinaklini River eulachon has been located since the 2016 status review (Gustafson et al. 2022); however, there were anecdotal reports that large numbers of eulachon were observed in Kingcome River during 2015–17. Wuikinuxv Nation conducts an annual eulachon monitoring survey on the Wannock, Kilbella and Chckwalla Rivers in Rivers Inlet; however, results have not been released. Anecdotal information indicates that eulachon began to return to the Bella Coola River in 2012 and the run has been slowly building in numbers, such that multiple schools of eulachon were observed in 2018; however, the run was not large enough to support a fishery.

Life History. Eulachon spend 95–98% of their lives at sea (Hay and McCarter 2000) and return to freshwater to spawn. In the portion of the species’ range that lies south of the U.S.–Canada border, most eulachon production originates in the Columbia River Basin, including the Columbia River, the Cowlitz River the Grays River, the Kalama River, the Lewis River, and the Sandy River (Gustafson et al. 2010). Spawning usually occurs between ages 2 and 5. Spawning is strongly influenced by water temperatures and tides, and the timing of migration typically occurs between December and June during high tides, when water temperatures are between 0°C and 10°C (Gustafson 2016). In the Columbia River and further south, spawning occurs from late January to March (Hay and McCarter 2000). Further north, the peak of eulachon runs in Washington State is from February through March (Hay and McCarter 2000). Females lay between 7,000 and 60,000 eggs over sand, course gravel or detritial substrate. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters. Larvae and young juveniles become widely distributed in coastal waters, mostly at depths up to 15 meters (Hay and McCarter 2000) but sometimes as deep as 182 meters (Barraclough 1964, as cited in Willson et al. 2006). Adult eulachon are found in coastal and offshore marine habitats. With the exception of some individuals in Alaska, eulachon generally die after spawning (Gustafson 2016). The maximum known lifespan is 9 years of age, but 20 to 30% of individuals live to 4 years and most individuals survive to 3 years of age, although spawning has been noted as early as 2 years of age. Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). The primary prey of adult eulachon are copepods and euphausiids, malacostracans and cumaceans.

Population Dynamics

Abundance. There are no reliable fishery-independent, historical abundance estimates for eulachon. Spawning stock biomass estimations of eulachon in the Columbia River for the years 2000 through 2017 have ranged from a low of 783,400 fish in 2005 to a high of 185,965,200 fish in 2013, with an estimated 18,307,100 fish in 2017. Spawning stock biomass estimations of eulachon in the Fraser River for the years 1995 through 2017 have ranged from a low of 109,129 to 146,606 fish in 2010 to a high of 41,709,035 to 56,033,332 fish in 1996, with an estimated 763,330 to 1,026,251 fish in 2017.

Productivity / Population Growth Rate. There is no population growth rate available for Southern DPS eulachon, although some indices show an increasing temporal trend. (Gustafson 2016).

Genetic Diversity. Southern DPS eulachon are genetically distinct from eulachon in the northern parts of its range (i.e., Alaska). Recent genetic analysis indicates that the Southern DPS exhibits a regional population structure, with a 3-population southern Columbia-Fraser group, coming from the Cowlitz, Columbia, and Fraser rivers (Candy et al. 2015; Gustafson 2016). Sutherland et al. (2021) developed an improved genetic baseline of 521 variant single nucleotide polymorphism (SNP) loci, genotyped in 1,989 individuals from 14 populations ranging from southcentral Alaska (Twentymile River) to northern California (Klamath River). Three main groupings were evident: southern rivers (Klamath, Columbia, Cowlitz, Sandy, and Fraser), northern rivers (Kingcome, Klinaklini, Wannock, Bella Coola, Kemano, Skeena, Nass, and Unuk), and the Gulf of Alaska (Twentymile River). These results were similar to those of Candy et al. (2015), and "the general trend of the data was similar between the SNP and microsatellite results, with a large divide between the populations to the south of the Fraser River, inclusive, and the populations to the north of the Fraser River, with Twentymile River as an outgroup" (Sutherland et al. 2021, p. 84–85). Separation of the southern rivers group from northern rivers "had high bootstrap support (>99.99%)" in dendrograms (Sutherland et al. 2021, p. 82). Within the southern grouping, there was some clustering of Columbia River populations together, but the Cowlitz River population grouped into a cluster with Klamath River, and more broadly with the Fraser River rather than with the other Columbia River populations (Columbia River, Sandy River). Cowlitz River and Klamath River are grouped closely together, and in 87% of trees Cowlitz and Klamath rivers group together without the Fraser River. In general these populations were very similar (e.g., Fraser River versus Columbia River: Fst = 0.0079 Fraser River versus Klamath River: FST = 0.0021, and Klamath River versus Columbia River: Fst = 0.0091 [Sutherland et al. 2021, p. 82].

Within the northern grouping, the Kingcome, Klinaklini, and Bella Coola Rivers "had high genetic similarity with each other (mean Fst = 0.0021)," as did the Kemano and Wannock Rivers (Fst = 0.0043) (Sutherland et al. 2021, p. 82). The Skeena and Nass Rivers "were nearly indistinguishable Fst = 0.0009" and the "Unuk River clustered outside of the north coast and central coast groupings, but still within the larger northern grouping" (Sutherland et al. 2021, p. 82). In the future, this improved genetic baseline will be applied to mixed stock analysis of at-sea sampled eulachon to improve estimates of where eulachon from specific rivers are distributed and which rivers are most impacted from at-sea bycatch risk (Sutherland et al. 2021).

Distribution. Adult and juvenile Southern DPS eulachon can be found in the Pacific Ocean, along the continental shelf, in waters from 50 to 200 meters deep (Gustafson 2016). In the portion of the species' range that lies south of the U.S.–Canada border, most eulachon production originates in the Columbia River Basin, including the Columbia River, the Cowlitz River the Grays River, the Kalama River, the Lewis River, and the Sandy River (Gustafson et al. 2010) and sometimes in the Klamath River, California.

Designated Critical Habitat. On October 20, 2011, NMFS designated critical habitat for Southern DPS eulachon (76 FR 65324). Sixteen areas were designated in the states of Washington, Oregon, and California. These areas include: the Mad River, CA, Redwood Creek, CA, Klamath River, CA, Umpqua River/Winchester Bay, OR, Tenmile Creek, OR, Sandy River, OR, Lower Columbia River, OR and WA, Grays River, WA, Skamokawa Creek, WA, Elochoman River, WA, Cowlitz River, WA, Toutle River, WA, Kalama River, WA, Lewis River, WA, Quinault River, WA, and the Elwha River, WA. The designated areas are a combination of freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km (335 mi) of habitat. The physical or biological features essential to the conservation of the DPS include:

- Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles.
- Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted.
- Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

Recovery Goals. See the 2017 Recovery Plan (NMFS 2017e)for the Southern DPS eulachon, for complete down listing/delisting criteria for each of their respective recovery goals. The Eulachon

Recovery Team identified 4 recovery objectives: 1) ensure subpopulation viability, 2) conserve spatial structure and temporal distribution patterns, 3) conserve existing genetic and life history diversity and provide opportunities for interchange of genetic material between and within subpopulations and, 4) eliminate or sufficiently reduce the severity of threats.

8.32 Green Sturgeon, Southern DPS

Table 75. Green Sturgeon, Southern DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser medirostris	Green Sturgeon	Southern	Threatened	2021	2006 71 FR 17757	2018	2009 74 FR 52299

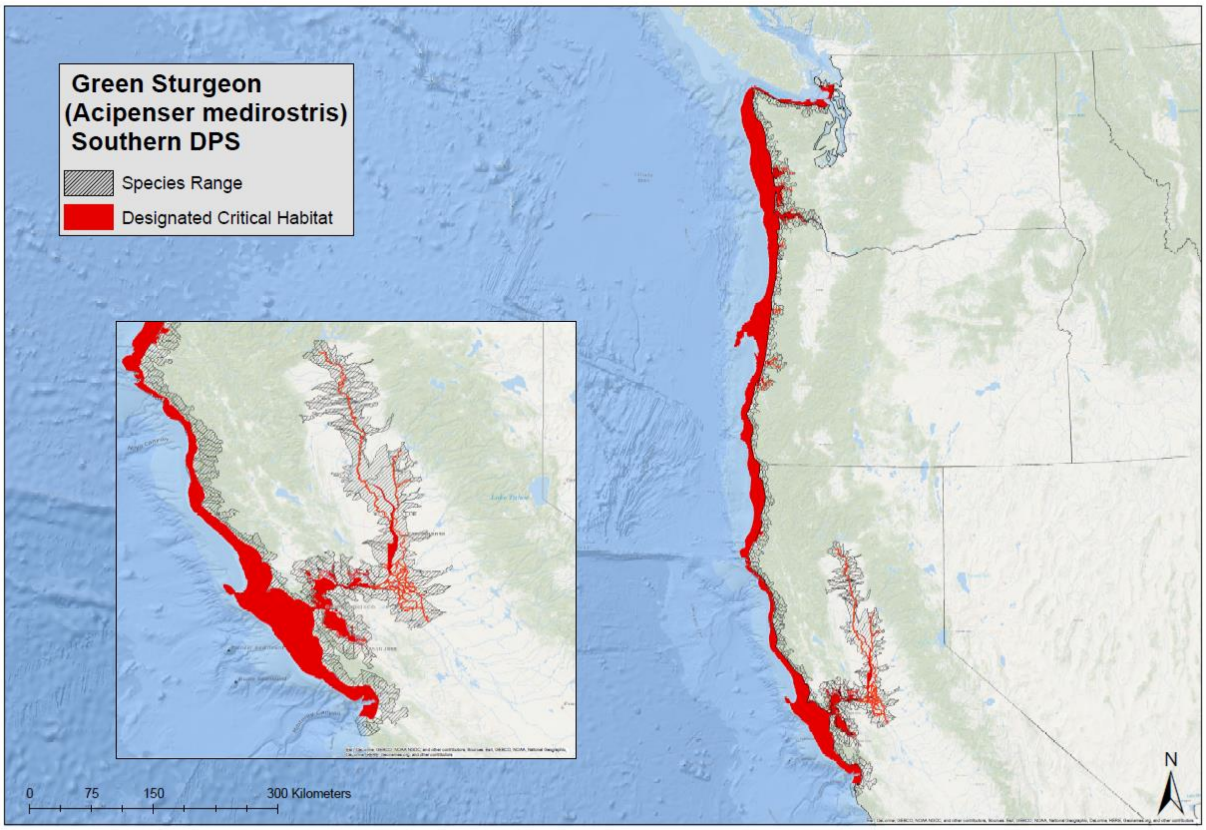


Figure 42. Green Sturgeon, Southern DPS range (within the contiguous US) and designated critical habitat

Species Description. The North American green sturgeon, *Acipenser medirostris*, is an anadromous fish that occurs in the nearshore Eastern Pacific Ocean from Alaska to Mexico (Moyle 2002a). Green sturgeon are long-lived, late-maturing, iteroparous, anadromous species that spawn infrequently in natal streams, and spend substantial portions of their lives in marine waters. Although they are members of the class of bony fishes, the skeleton of sturgeons is composed mostly of cartilage. Sturgeon have 5 rows of characteristic bony plates on their body (called scutes). Green sturgeon have an olive green to dark green back, a yellowish green-white belly, and a white stripe beneath the lateral scutes (Adams et al. 2002). NMFS has identified 2 DPS of green sturgeon; northern and southern (Israel et al. 2009). In 2006, NMFS determined that the southern DPS green sturgeon warranted listing as a threatened species under the ESA (71 FR 17757). Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the US, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco bay and Monterey bay.

Status. Attempts to evaluate the status of southern DPS green sturgeon have been met with limited success due to the lack of reliable long term data, however based on available scientific data (Adams et al. 2007) and ongoing conservation efforts, NMFS concluded in the final rule designating this species that southern DPS green sturgeon were likely to become endangered in the foreseeable future throughout all of its range. The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River (71 FR 17757). In general, the primary threats to southern DPS green sturgeon are the reduction of potential spawning habitat (most notably by impoundments), water temperature and flow, and commercial and recreational bycatch. Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015e).

Life History. Green sturgeon reach sexual maturity at approximately 15 years of age (Van Eenennaam et al. 2006), and may spawn every 3-5 years throughout their long lives (Tracy 1990). Southern DPS green sturgeon spawn in cool (14-17ºC), deep, turbulent areas with clean, hard substrates. Spawning occurs primarily in the Sacramento River (Brown 2007, Poytress et al. 2015, Mora et al. 2018). Since 2015, spawning has also been documented in the Feather and Yuba rivers, which are tributaries to the Sacramento River (Seesholtz et al. 2015, Beccio 2018, 2019). Adult Southern DPS green sturgeon enter San Francisco Bay in late winter through early spring, migrate upstream, and spawn from April through early July, with peaks of activity influenced by factors including water flow and temperature (Heublein et al. 2009, Poytress et al. 2015, Miller et al. 2020). Post-spawn fish typically congregate and hold for several months in a few deep pools in the upper mainstem Sacramento River near spawning sites and migrate back downstream when river flows increase in fall. They re-enter the ocean during the winter months (November through January) and begin their marine migration north along the coast (California Fish Tracking Consortium database).

Green sturgeon eggs primarily adhere to gravel or cobble substrates, or settle into crevices (Van Eenennaam et al. 2001, Poytress et al. 2011). Larvae disperse at approximately 12 days post hatch (dph) in the laboratory (Kynard et al. 2005), and are suspected to remain near spawning habitats. It is unknown how long juveniles remain in upriver rearing habitats after metamorphosis. Based on length distribution data from salvage and recent upstream surveys, juveniles typically enter the Delta as sub-yearlings or yearlings to rear prior to ocean entry. The San Francisco Bay Delta Estuary provides year-round rearing habitat for juveniles, as well as foraging habitat for non-spawning adults and subadults in the summer months (NMFS 2009c). Once at sea, subadults and adults occupy coastal waters to a depth of 110 meters from Baja California, Mexico to the Bering Sea, Alaska (Hightower 2007). Seasonal migrations are known to occur. Fish congregate in coastal bays and estuaries of Washington, Oregon, and California during summer and fall. In winter and spring, similar aggregations can be found from Vancouver Island to Hecate Strait, British Columbia, Canada (Lindley et al. 2008)

Green sturgeon are opportunistic feeders that consume a variety of prey items such as insect larvae, oligochaetes, and decapods (NMFS 2009a). In the San Francisco Bay Delta Estuary, juvenile green sturgeon feed on shrimp, amphipods, isopods, clams, annelid worms, and an assortment of crabs and fish (Ganssle 1966; Radtke 1966). Post-spawn adult green sturgeon in freshwater likely feed on benthic prey species (e.g., lamprey ammocoetes, crayfish). In coastal bays and estuaries, adult and subadult green sturgeon feed on shrimp, clams, crabs, and benthic fish (Moyle et al. 1995; Dumbauld et al. 2008). Nearshore marine prey resources likely include species similar to those of coastal bays and estuaries.

Population Dynamics

Abundance. Mora et al. (2018) estimated the Southern DPS total population size to be 17,548 individuals (95% confidence interval [CI] = 12,614-22,482). The SWFSC recently updated the total population estimate to 17,723 (Dudley 2021).These surveys estimate the abundance of Southern DPS adults at 2,106 individuals (95% CI = 1,246-2,966) (Mora 2016, Mora et al. 2018). A conceptual demographic structure applied to the adult population estimate resulted in a Southern DPS subadult population estimate of 11,055 (95% CI = 6,540-15,571) and juvenile population estimate of 4,387 (95% CI = 2,595-6,179) (Mora et al. 2018).

Productivity / Population Growth Rate. Attempts to evaluate the status of southern DPS green sturgeon have been met with limited success due to the lack of reliable long term data. No estimate of λ is available for southern DPS green sturgeon.

Genetic Diversity. The available genetic data do not change the status of the species or the imminence or magnitude of any threat; data only confirm the DPS structure and add detail to the DPS composition in different estuaries during the sampling periods (NMFS 2015e). Green sturgeon stocks from the DPSs have been found to be genetically differentiated (Israel et al. 2009; Israel et al. 2004).

Distribution. In general, subadult (from the age of ocean entry to age of first spawning) and adult North American green sturgeon occur from Graves Harbor, Alaska to Monterey Bay, California (Moser and Lindley 2007; Lindley et al. 2008, 2011; Schreier et al. 2016). Within this range, green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the US, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco bay and Monterey bay (Huff et al. 2012; Lindley et al. 2011; Lindley et al. 2008; Moser and Lindley 2007).

Designated Critical Habitat. Critical habitat was designated for Southern DPS green sturgeon on October 9, 2009, and includes marine, coastal bay, estuarine, and freshwater areas (74 FR 52300). PBFs considered essential for the conservation of Green Sturgeon, Southern DPS are:

Freshwater areas

- *Food resources.* Abundant prey items for larval, juvenile, subadult, and adult life stages.
- Substrate type or size (i.e., structural features of substrates)

- Water flow. A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages.
- Water quality. Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.
- Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within riverine habitats and between riverine and estuarine habitats (e.g., an unobstructed river or dammed river that still allows for safe and timely passage).
- Water depth. Deep (≥5 meters) holding pools for both upstream and downstream holding of adult or subadult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or subadult fish.
- Sediment quality. Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

Estuarine areas

- Food resources. Abundant prey items within estuarine habitats and substrates for juvenile, subadult, and adult life stages.
- Water flow. Within bays and estuaries adjacent to the Sacramento River (*i.e.*, the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds.
- *Water quality*. Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.
- Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats.
- Water depth. A diversity of depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages.
- Sediment quality. Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of elevated levels of contaminants

Coastal Marine Areas

- Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within marine and between estuarine and marine habitats.
- Water quality. Coastal marine waters with adequate dissolved oxygen levels and acceptably low levels of contaminants (e.g., pesticides, PAHs, heavy metals that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon).
- Food resources. Abundant prey items for subadults and adults, which may include benthic invertebrates and fish.

Recovery Goals See the 2018 Recovery Plan (NMFS 2018a) for a complete description of recovery goals, objectives and criteria. The 5 demographic recovery criteria are as follows:

- The adults DPS green sturgeon census population remains at or above 3,000 for 3 generations. In addition, the effective population size must be at least 500 individuals in any given year and each annual spawning run must be comprised of a combined total, from all spawning locations, of at least 500 adult fish in any given year.
- Southern DPS green sturgeon spawn successfully in at least 2 rivers within their historical range. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
- A net positive trend in juvenile and subadult abundance is observed over the course of at least 20 years.
- The population is characterized by a broad distribution of size classes representing multiple cohorts that are stable over the long term (20 years or more).
- There is no net loss of DPS green sturgeon diversity from current levels.

8.33 Shortnose Sturgeon

Table 76. Shortnose Sturgeon; overview table

Species	Common Name	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser brevirostrum	Sturgeon, Shortnose	Endangered	2010	1967 32 FR4001	1998	None Designated

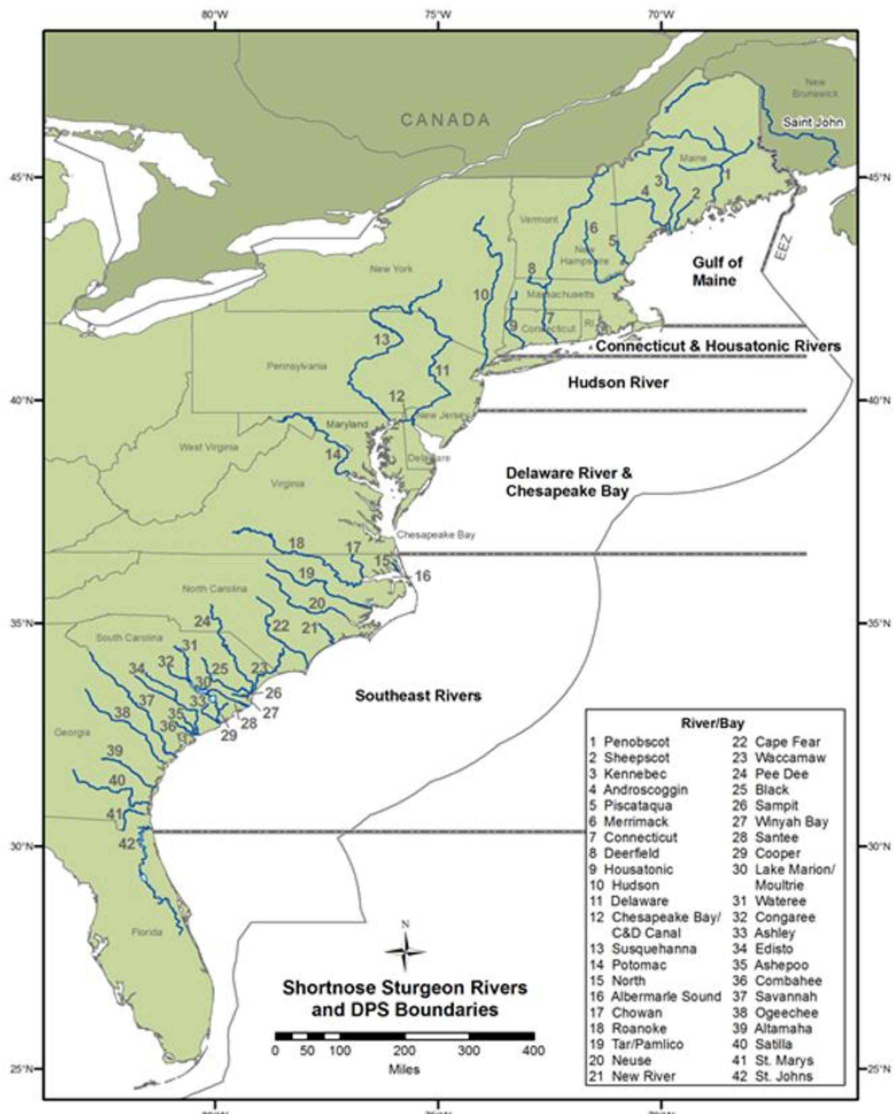


Figure 43**. Shortnose Sturgeon range (https://www.fisheries.noaa.gov/species/shortnose-sturgeon#populations)**

Species Description. The shortnose sturgeon (*Acipenser brevirostrum*) is the smallest of the 3 sturgeon species that occur in eastern North America. It has a benthic fusiform body and its head and snout are smaller while its mouth is larger relative to Atlantic sturgeon (Dadswell 1984). Shortnose sturgeon vary in color but are generally dark brown to olive/black on the dorsal surface, lighter along the row of lateral scutes and nearly white on the ventral surface (Gilbert 1989). The shortnose sturgeon was listed as endangered on March 11, 1967 (32 FR 4001) under the Endangered Species Preservation Act of 1966. Shortnose sturgeon remained on the endangered species list with the enactment of the ESA in 1973. Shortnose sturgeon occur in estuaries and rivers along the east coast of North America (Vladykov and Greeley 1963). Their northerly distribution extends to the Saint John River, New Brunswick, Canada, and their southerly distribution historically extended to the Indian River, Florida (Evermann and Bean 1898; Scott and Scott 1988).

Status. The decline in abundance and slow recovery of shortnose sturgeon has been attributed to pollution, overfishing, bycatch in commercial fisheries, and an increase in industrial uses of the nation's large coastal rivers during the 20th century (e.g., hydropower, nuclear power, treated sewage disposal, dredging, construction) (NMFS 2010a). In addition, the effects of climate change may adversely impact shortnose sturgeon by reducing the amount of available habitat, exacerbating existing water quality problems, and interfering with migration and spawning cues (NMFS 2010a). Without substantial mitigation and management to improve access to historical habitats and water quality of these systems, shortnose sturgeon populations will likely continue to be depressed. This is particularly evident in some southern rivers that are suspected to no longer support reproducing populations of shortnose sturgeon (NMFS 2010a). The number of river systems in which spawning has been confirmed has been reduced to around 12 locations (NMFS 2010a).

Life History. Shortnose sturgeon are relatively slow growing, late maturing and long-lived. Growth rate, maximum age and maximum size vary with latitude; populations in southern areas grow more rapidly and mature at younger ages but attain smaller maximum sizes than those in the north (Dadswell et al. 1984). In general, females reach sexual maturity in the south as early as age 4 and in the north as late as age 18, and males display similar difference in latitudinal development, maturing between ages 2 and 11 (NMFS 2010a). Shortnose sturgeon overwinter in the lower portions of rivers and migrate upriver to spawn in the spring. Spawning periodicity is poorly understood, but males seem to spawn more frequently than females. Dadswell (1984) estimated that Saint John River males spawned at 2-year intervals; females at 3-5 year intervals. Spawning females deposit their eggs over gravel, rubble, and/or cobble often in the farthest accessible upstream reach of the river (Kynard 1997b). After spawning, adult shortnose sturgeon move rapidly to downstream feeding areas where they forage on benthic insects, crustaceans, mollusks, and polychaetes (Buckley and Kynard 1985; Dadswell 1984; Kieffer and Kynard 1993; O'herron et al. 1993).

Upon hatching, shortnose sturgeon shelter in dark substrate or are found in schools swimming against the current. Around 4-12 days after hatching individuals begin to feed exogenously and are dispersed downstream. These larvae are often found in the deepest water, usually within the channel (Kieffer and Kynard 1993; O'Connor et al. 1981; Parker and Kynard 2014; Taubert and Dadswell 1980). Young-of-the-year remain in freshwater habitats upstream of the salt wedge for about 1 year (Dadswell et al. 1984; Kynard 1997b). The age at which juveniles begin to utilize habitat associated with the salt/fresh water interface varies with river system from age 1 to 8 (Collins et al. 2002; Dadswell 1979; Flournoy et al. 1992). Overwintering habitat and behavior of shortnose sturgeon varies with latitude: fish in northern rivers form tight aggregations with

little movement and will inhabit either freshwater or saline reaches of the river, while fish in the south are more active and are found predominantly near the fresh/saltwater interface (Collins and Smith 1993; Kynard et al. 2012; Weber et al. 1998).
The general pattern of coastal migration of shortnose sturgeon indicates movement between groups of rivers proximal to each other across the geographic range (Altenritter et al. 2015; Dionne et al. 2013; Quattro et al. 2002; Wirgin et al. 2005). NMFS's 2010 biological assessment of shortnose sturgeon grouped the species into 5 regional population clusters: Gulf of Maine, Connecticut/Housatonic rivers, Hudson River, Delaware River/Chesapeake Bay, and Southeast. King et al. (King et al.) identified 3 metapopulations: 1) Maine rivers, 2) Delaware River and Chesapeake Bay proper, and 3) the Southeast assemblage. The shortnose sturgeon status review team recommends that recovery and management actions consider each riverine population as a management/recovery unit (NMFS 2010a).

Population Dynamics

Abundance. The 2010 biological assessment of shortnose sturgeon identified 5 regional population clusters of shortnose sturgeon. See table below for abundance estimates for populations within each of these population clusters.

Table 77. Shortnose sturgeon populations and estimated abundances

Regional Population Cluster	Location[a]	Abundance Estimate (Upper/Lower 95% CI)[b]	(Source) Year of Collection Data
Gulf of Maine	Penobscot River	1,049 (673 / 6,939)	(NMFS 2012a) 2006 – 2007
	Kennebec Complex	9,488 (6,942 / 13,358)	(Squiers 2004) 1998 – 2000
	Merrimack River	2000 (NA)	(NMFS 2010a) 2009
Connecticut and Housatonic Rivers	Connecticut River – upper*	143 (14 / 360)	(Kynard et al. 2012) 1994 – 2001
	Connecticut River – lower*	1,297 (NA)	(Savoy 2004) 1996 – 2002
Hudson River	Hudson River	30,311 (NA)	(NMFS 2010a) 1980
Delaware River/Chesapeake Bay	Delaware River	12,047 (10,757 / 13,580)	(Brundage III 2006) 1999 – 2003
Southeast Rivers	Cape Fear River	50 (NA)	(NMFS 2010a) NA
	Cooper River	301 (150 / 659)	(Cooke et al. 2004) 1996 – 1998
	Lake Marion	Unknown (NA)	(NMFS 2010a) NA

Regional Population Cluster	Location[a]	Abundance Estimate (Upper/Lower 95% CI)[b]	(Source) Year of Collection Data
	Savannah River	940 adults (535 / 1753)	(Bahr and Peterson 2017) 2015
	Ogeechee River	147 (104 / 249)	(Fleming et al. 2003) 1999 – 2000
	Altamaha River	1,209 (556 / 2759)	(Bednarski 2012) 2004 – 2010

[a]Locations listed here are those for which population estimates are available. Additional waterbodies with confirmed shortnose sturgeon include Piscataqua River, Housatonic River, Chesapeake Bay, Susquehanna River, Potomac River, Roanoke River, Chowan River, Tar/Pamlico River, Neuse River, New River, North River, Santee River, ACE Basin – Edisto (Smith et al. 2002), Satilla River, St. Mary's River, St. Johns River (NMFS 2010a).

[b]Abundance estimates are established using different techniques and should be viewed with caution. Estimates listed here are those identified by NMFS in the 2010 Biological Assessment of Shortnose Sturgeon (NMFS 2010a).

**The Connecticut River population of shortnose sturgeon is separated into an upstream and downstream segment bisected by the Holyoke Dam.*

Productivity / Population Growth Rate. Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data.

Table 78. Shortnose sturgeon populations and productivity estimates

Regional Population Cluster	Location[a]	Evidence of Spawning	Abundance Trend Estimate (Population Health Score)[b]
Gulf of Maine	Penobscot River	No spawning locations found; no juveniles or larvae observed.	No estimates (4.35)
	Kennebec Complex	Spawning confirmed on Kennebec and Androscoggin rivers.	Increasing (10.42)
	Merrimack River	Spawning confirmed	Potentially stable (5.65)
Connecticut and Housatonic Rivers	Connecticut River – upper	Spawning confirmed	Potentially stable (8.35)
	Connecticut River - lower	Minimal spawning	Potentially stable (8.35)
Hudson River	Hudson River	Spawning confirmed	Potentially stable (10.00)

Regional Population Cluster	Location[a]	Evidence of Spawning	Abundance Trend Estimate (Population Health Score)[b]
Delaware River/Chesapeake Bay	Delaware River	Spawning confirmed	Potentially stable (9.56)
Southeast Rivers	Cape Fear River	Gravid females documented	Declining (3.12)
	Winyah Bay System	Spawning confirmed	Potentially stable (6.23)
	Cooper River	Spawning confirmed	Potentially stable (6.23)
	Lake Marion	Spawning confirmed	No estimates (4.12)
	Savannah River	Spawning confirmed	Potentially stable (8.35)
	Ogeechee River	No spawning locations found; gravid females and juveniles confirmed	Potentially stable (7.23)
	Altamaha River	Spawning confirmed	Potentially stable (9.22)

[a] Locations listed here are those for which population estimates are available, and/or those in which spawning has been confirmed. Additional waterbodies with confirmed shortnose sturgeon include Piscataqua River, Housatonic River, Chesapeake Bay, Susquehanna River, Potomac River, Roanoke River, Chowan River, Tar/Pamlico River, Neuse River, New River, North River, Santee River, ACE Basin, Satilla River, St. Mary's River, St. Johns River (NMFS 2010a).

[b] Population Health Scores taken from NMFS 2010 Biological Assessment of shortnose sturgeon. Scale from 0 – 12, with larger values representing healthier populations (NMFS 2010a).

Genetic Diversity. Genetic diversity estimates for shortnose sturgeon have been shown to be moderately high in both mitochondrial (Quattro et al. 2002; Wirgin et al. 2005; Wirgin et al. 2010) and nuclear genomes (King et al. 2014). The mtDNA and nDNA studies performed to date suggest that dispersal is a very important factor in maintaining these high levels of genetic diversity.

Distribution. Shortnose sturgeon occur along the East Coast of North America in rivers, estuaries and the sea. They were once present in most major rivers systems along the Atlantic coast (Evermann and Bean 1898; Scott and Scott 1988). Their current distribution extends north to the Saint John River, New Brunswick, Canada, and south to the St. Johns River, FL (NMFS 1998). Currently, the distribution of shortnose sturgeon across their range is disjunct, with northern populations separated from southern populations by a distance of about 400 km near their geographic center in North Carolina and Virginia. Some river systems host populations which rarely leave freshwater while in other areas coastal migrations between river systems are common. Spawning locations have been identified within a number of river systems (NMFS 2010a).

Designated Critical Habitat. Critical habitat has not been proposed for shortnose sturgeon.

Recovery Goals. The long-term recovery objective for the shortnose sturgeon is to recover all discrete population segments (as defined in the 1998 shortnose sturgeon recovery plan) to levels of abundance at which they no longer require protection under the ESA. Each population segment may become a candidate for downlisting when it reaches a minimum population size that: 1) is large enough to prevent extinction, and 2) will make the loss of genetic diversity unlikely. The minimum population size for each population segment has not yet been determined (NMFS 1998; NMFS 2010a).

8.34 Atlantic Sturgeon, Gulf of Maine DPS

Table 79. Atlantic Sturgeon, Gulf of Maine DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Sturgeon, Atlantic	Gulf of Maine	Threatened	2022	2012 77 FR 5880	2018 (Recovery Outline)	2017 82 FR 39160

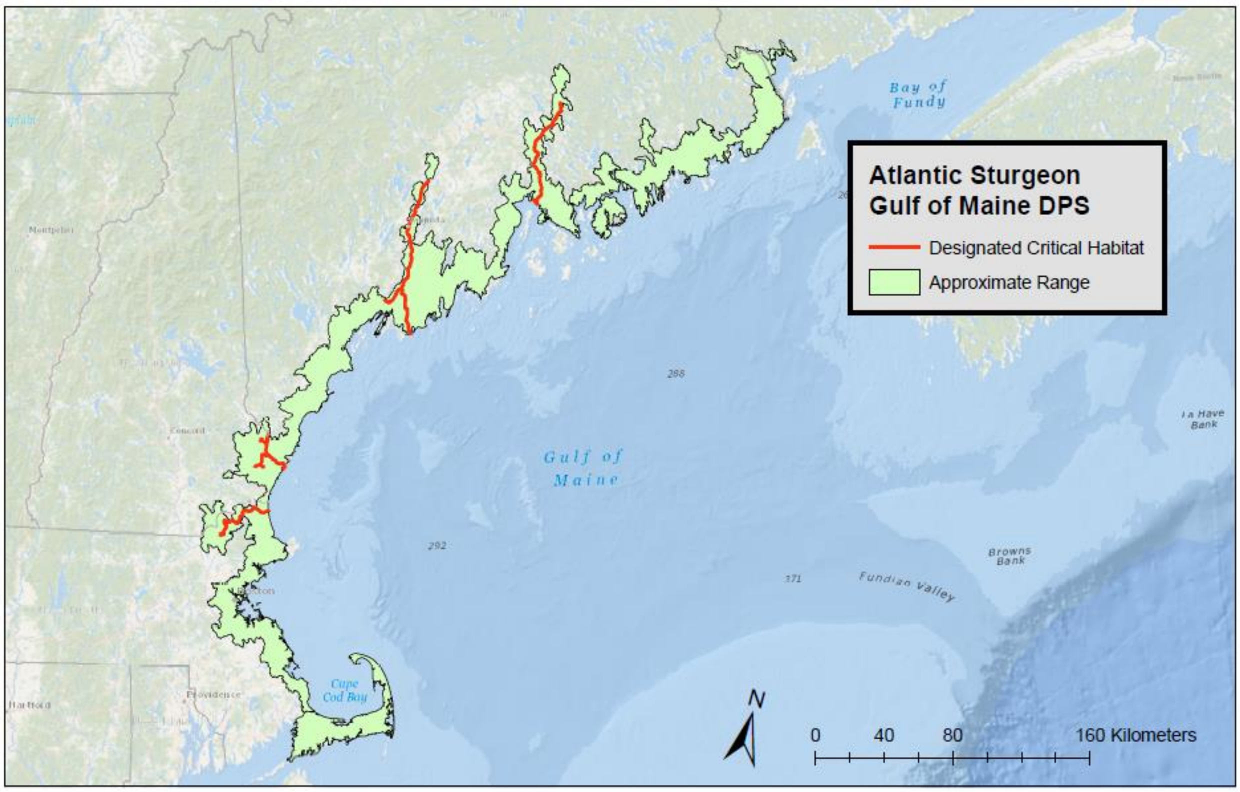


Figure 44. Atlantic Sturgeon, Gulf of Maine DPS range and designated critical habitat

Species Description. The Atlantic sturgeon is a long lived, late maturing, anadromous species. Atlantic sturgeon attain lengths of up to approximately 14 feet, and weights of more than 800 pounds. They are bluish black or olive brown dorsally with paler sides and a white ventral surface and have 5 major rows of dermal scutes (Colette and Klein-MacPhee 2002). On February 6, 2012, 4 DPSs of Atlantic sturgeon: New York Bight, Chesapeake Bay, Carolina, and South Atlantic, were listed as endangered and the Gulf of Maine DPS was listed as threatened (77 FR 5880; 77 FR 5914). Atlantic sturgeon from the Gulf of Maine DPS spawn in the rivers of Maine, as well as rivers that drain into the Gulf of Maine from as far south as Chatham, Massachusetts.

Status. The Kennebec River remains the only known spawning population for the Gulf of Maine DPS despite the availability of suitable spawning and rearing habitat in other Gulf of Maine rivers such as the Anroscoggin River. The estimated effective population size is less than 70 adults which suggests a relatively small spawning population. It is currently the only DPS with only 1 known spawning population. The new information from the 2022 Status Update further supports NMFS determination in the listing rule that the Gulf of Maine DPS has low abundance, and that the current numbers of spawning adults are 1–2 orders of magnitude smaller than historical levels. In 2017, the Atlantic States Marine Fisheries Commission (ASMFC) conducted a benchmark stock assessment of Atlantic sturgeon (ASMFC 2017). The assessment contains the latest and best available information on the status of U.S. Atlantic sturgeon populations. The stock assessment concluded that the abundance of the Gulf of Maine DPS is "depleted" relative to historical levels. The assessment also concluded that there was a 51% probability that the abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium, but there was a 74% probability that mortality for the Gulf of Maine DPS exceeds the mortality threshold used for the assessment (ASMFC 2017). General threats include: habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries.

Life History. Atlantic Sturgeon size at sexual maturity varies with latitude (Scott and Crossman 1973). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Spawning adults generally migrate upriver in May-July in Canadian systems (Bain 1997; Caron et al. 2002; Murawski and Pacheco 1977; Smith 1985; Smith and Clugston 1997). Atlantic sturgeon spawning is believed to occur in flowing water between the salt front and fall line of large rivers at depths of 3-27 meters (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year; spawning intervals range from 1-5 years for males (Caron et al. 2002; Collins et al. 2000; Smith 1985) and 2-5 for females (Stevenson and Secor 2000; Van Eenennaam et al. 1996; Vladykov and Greeley 1963).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (Gilbert 1989; Smith and Clugston 1997) between the salt front and fall line of large rivers (Bain et al. 2000; Borodin 1925; Crance 1987; Scott and Crossman 1973). Following spawning in northern rivers, males may remain in the river or lower estuary until the fall; females typically exit the rivers within 4 to 6 weeks (Savoy and Pacileo 2003). Hatching occurs approximately 94-140 hours after egg deposition at temperatures of 20° and 18° Celsius, respectively (Theodore et al. 1980). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from 0 to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend 2 to 5 years in freshwater before eventually becoming coastal residents as sub-adults (Boreman 1997; Schueller and Peterson 2010; Smith 1985).

Upon reaching the subadult phase individuals may move to coastal and estuarine habitats (Dovel and Berggren 1983; Murawski and Pacheco 1977; Smith 1985; Stevenson 1997). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Grunwald et al. 2008; King et al. 2001; Waldman et al. 2002). Because of high natal river fidelity, it appears that most rivers support independent populations

(Grunwald et al. 2008; King et al. 2001; Waldman and Wirgin 1998; Wirgin et al. 2002; Wirgin et al. 2000). Atlantic sturgeon feed primarily on polychaetes, isopods, American sand lances and amphipods in the marine environment, while in fresh water they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Guilbard et al. 2007; Johnson et al. 1997a; Moser and Ross 1995; Novak et al. 2017; Savoy 2007).

Population Dynamics

Abundance. Historically, the Gulf of Maine DPS likely supported more than 10,000 spawning adults (ASSRT 2007; KRRMP 1993; Secor 2002; NMFS 2007). The current abundance is estimated to be 1-2 orders of magnitude smaller than historical levels (ASSRT 2007). New information provided in the 2022 update suggests that the observed seasonal abundance of Atlantic sturgeon in the Saco River is a large feeding aggregation and may be, but is not necessarily, indicative of an increased abundance for the DPS, overall. New information also supports that the Gulf of Maine is 1 of the fastest warming areas of the world as a result of global climate change.

Productivity / Population Growth Rate. There are some positive signs for the Gulf of Maine DPS, which include observations of Atlantic sturgeon in rivers from which sturgeon observations have not been reported for many years (Saco, Presumpscot, and Charles rivers) and potentially higher catch-per-unit-effort levels than in the past (Kennebec) (NMFS 2007c). Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data.

Genetic Diversity. The genetic diversity of Atlantic sturgeon throughout its range has been well documented (Bowen and Avise 1990; Ong et al. 1996; Waldman et al. 1996; Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (King et al. 2001; Waldman et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters.

Distribution. The geomorphology of most small coastal rivers in Maine is not sufficient to support Atlantic sturgeon spawning populations, except for the Penobscot and the estuarial complex of the Kennebec, Androscoggin, and Sheepscot rivers. Spawning still occurs in the Kennebec and Androscoggin Rivers, and may occur in the Penobscot River. Atlantic sturgeon have more recently been observed in the Saco, Presumpscot, and Charles rivers. New information demonstrates that the Saco River supports a large aggregation of Atlantic sturgeon that forage on sand lance in Saco Bay and within the first few kilometers of the Saco River, primarily from May through October. Detections of acoustically-tagged sturgeon indicate that both adult and subadult Atlantic sturgeon use the area for foraging and come back to the area year after year (Little 2013; Novak et al. 2017). Some sturgeon also overwinter in Saco Bay (Little et al. 2013; Hylton et al. 2018) which suggests that the river provides important wintering habitat as well, particularly for subadults. However, none of the new information indicates recolonization of the Saco River for spawning. It remains questionable whether sturgeon larvae could survive in the Saco River even if spawning were to occur because of the presence of the Cataract Dam at rkm 10 of the river (Little 2013) which limits access to the freshwater reach.

Some sturgeon that spawn in the Kennebec have subsequently been detected foraging in the Saco River and Bay (Novak et al. 2017; Wippelhauser et al. 2017).

Designated Critical Habitat. Designated Critical Habitat was effective September 18, 2017. Based on the best scientific information available for the life history needs of the Gulf of Maine, DPS, the physical features essential to the conservation of the species and that may require special management considerations or protection are:

- Hard bottom substrate (*e.g.*, rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (*i.e.*, 0.0 to 0.5 parts per thousand (ppt) range) for settlement of fertilized eggs, refuge, growth, and development of early life stages;
- Aquatic habitat with a gradual downstream salinity gradient of 0.5 up to as high as 30 ppt and soft substrate (*e.g.*, sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development;
- Water of appropriate depth and absent physical barriers to passage (*e.g.*, locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support:
- Unimpeded movement of adults to and from spawning sites;
- Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- Staging, resting, or holding of subadults or spawning condition adults.
- Water depths in main river channels must also be deep enough (*e.g.*, at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.
- Water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support:
- Spawning;
- Annual and interannual adult, subadult, larval, and juvenile survival; and
- Larval, juvenile, and subadult growth, development, and recruitment (*e.g.*, 13 °C to 26 °C for spawning habitat and no more than 30 °C for juvenile rearing habitat, and 6 milligrams per liter (mg/L) dissolved oxygen (DO) or greater for juvenile rearing habitat).

Recovery Goals. The Recovery Plan has not yet been finalized. A 2018 recovery outline (NMFS 2018b) provides the following recovery vision statement:

Subpopulations of all 5 Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future.

8.35 Atlantic Sturgeon, New York Bight DPS

Table 80. Atlantic Sturgeon, New York Bight DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Sturgeon, Atlantic	New York Bight	Endangered	2022	2012 77 FR 5880	2018 (Recovery Outline)	2017 82 FR 39160

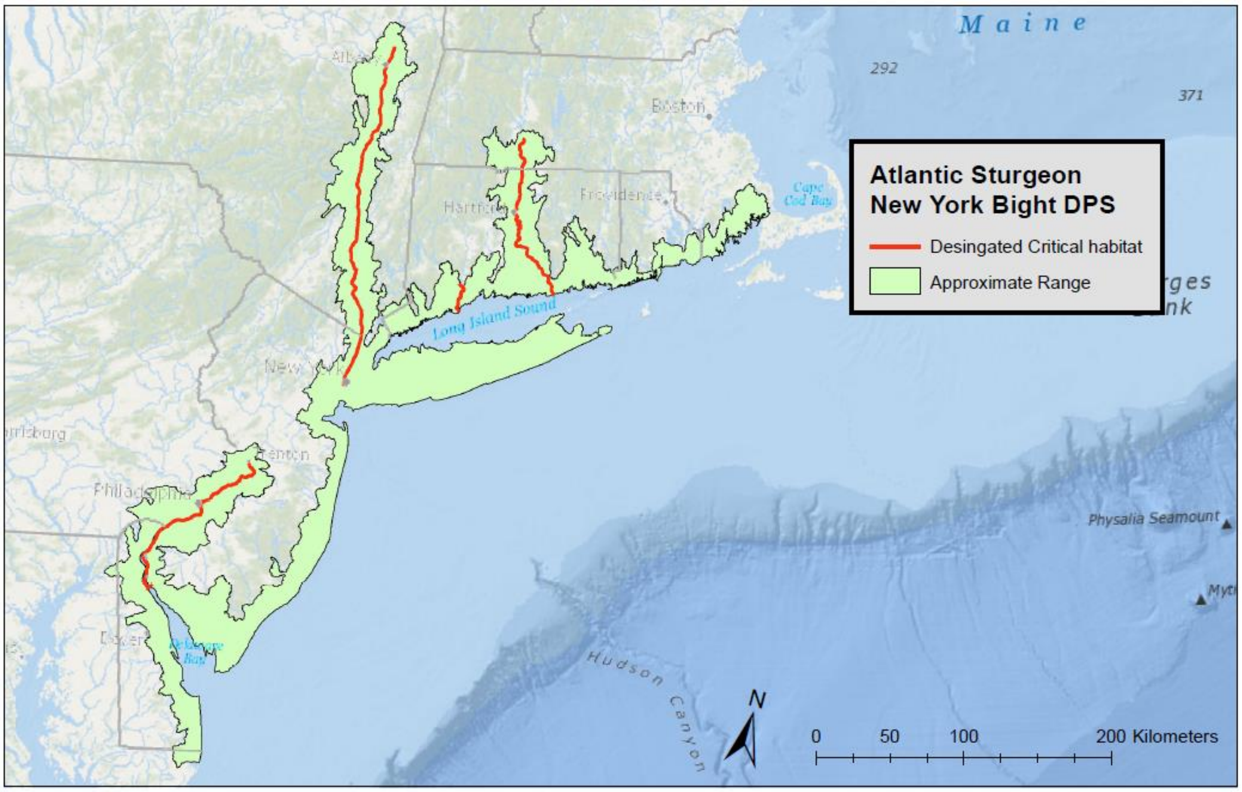


Figure 45**. Atlantic Sturgeon, New York Bight DPS range and designated critical habitat**

Species Description. The Atlantic sturgeon is a long lived, late maturing, anadromous species. Atlantic sturgeon attain lengths of up to approximately 14 feet, and weights of more than 800 pounds. They are bluish black or olive brown dorsally with paler sides and a white ventral surface and have 5 major rows of dermal scutes (Colette and Klein-MacPhee 2002). On February 6, 2012, 4 DPSs of Atlantic sturgeon: New York Bight, Chesapeake Bay, Carolina, and South Atlantic, were listed as endangered and the Gulf of Maine DPS was listed as threatened (77 FR 5880; 77 FR 5914). The New York Bight DPS of Atlantic sturgeon originates from rivers that drain into the coastal waters from Chatham, Massachusetts, to the Delaware-Maryland border at Fenwick Island.

Status. There were 2 known spawning subpopulations when the New York Bight DPS was listed as endangered under the ESA: the Hudson River and Delaware River spawning subpopulations. Since then, new information provided from the capture of juvenile Atlantic sturgeon suggests the Connecticut River likely also supports a spawning subpopulation of Atlantic sturgeon for the New York Bight DPS. The 2017 ASMFC stock assessment determined that abundance of the New York Bight DPS is "depleted" relative to historical levels (ASMFC 2017). However, the assessment also determined there is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 31% probability that mortality for the New York Bight DPS exceeds the mortality threshold used for the assessment (ASMFC 2017). General threats include: habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; dredging activities; and bycatch in commercial fisheries. New information in the 2022 Status Update supports NMFS's determination in the listing rule that the New York Bight DPS has low abundance, and that the current numbers of spawning adults are 1 to 2 orders of magnitude smaller than historical levels.

Life History. Atlantic Sturgeon size at sexual maturity varies with latitude with individuals reaching maturity in the Hudson River at 11 – 21 years (Young et al. 1988). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Spawning adults generally migrate upriver in April-May in mid-Atlantic systems, and May-July in Canadian systems (Bain 1997; Caron et al. 2002; Murawski and Pacheco 1977; Smith 1985; Smith and Clugston 1997). Atlantic sturgeon spawning is believed to occur in flowing water between the salt front and fall line of large rivers at depths of 3-27 meters (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year; spawning intervals range from 1-5 years for males (Caron et al. 2002; Collins et al. 2000; Smith 1985) and 2-5 for females (Stevenson and Secor 2000; Van Eenennaam et al. 1996; Vladykov and Greeley 1963).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (Gilbert 1989; Smith and Clugston 1997) between the salt front and fall line of large rivers (Bain et al. 2000; Borodin 1925; Crance 1987; Scott and Crossman 1973). Following spawning in northern rivers, males may remain in the river or lower estuary until the fall; females typically exit the rivers within 4 to 6 weeks (Savoy and Pacileo 2003). Hatching occurs approximately 94-140 hours after egg deposition at temperatures of 20° and 18° Celsius, respectively (Theodore et al. 1980). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from 0 to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend 2-5 years in freshwater before eventually becoming coastal residents as sub-adults (Boreman 1997; Schueller and Peterson 2010; Smith 1985).

Upon reaching the subadult phase individuals may move to coastal and estuarine habitats (Dovel and Berggren 1983; Murawski and Pacheco 1977; Smith 1985; Stevenson 1997). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Grunwald et al. 2008; King et al. 2001; Waldman et al. 2002).

Because of high natal river fidelity, it appears that most rivers support independent populations (Grunwald et al. 2008; King et al. 2001; Waldman and Wirgin 1998; Wirgin et al. 2002; Wirgin et al. 2000). Atlantic sturgeon feed primarily on polychaetes, isopods, American sand lances and amphipods in the marine environment, while in fresh water they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Guilbard et al. 2007; Johnson et al. 1997a; Moser and Ross 1995; Novak et al. 2017; Savoy 2007).

Population Dynamics

Abundance. There are no abundance estimates at this time for the Connecticut River. The Hudson River spawning subpopulation is believed to be the most robust because animals from the Hudson River show up most frequently in genetic samples collected from Atlantic sturgeon in coastal aggregations, with the exception of the summer aggregation in the Bay of Fundy, Canada. Conversely, Atlantic sturgeon from the Delaware River subpopulation show up less frequently even when the sampling area is in proximity to the Delaware River. Researchers have had recent success capturing juvenile Atlantic sturgeon in the Delaware River and estimate there were 3,656 (95% CI = 1,935–33,041) age 0-1 juvenile Atlantic sturgeon in the Delaware River subpopulation in 2014 (Hale et al. 2016). The 2017 ASMFC stock assessment determined that abundance of the New York Bight DPS is "depleted" relative to historical levels (ASMFC 2017).

Productivity / Population Growth Rate. Historically the Delaware River is believed to have supported around 180,000 individuals (Secor 2002). In 2007, NMFS status review estimated that the population had declined to fewer than 300 individuals. In 2014 Hale et al. (2016) estimated that 3,656 (95% CI = 1,935-33,041) early juveniles (age 0-1) utilized the Delaware River estuary as a nursery. Based on commercial fishery landings from the mid-1980s to the mid-1990s. The total abundance of adult Hudson River Atlantic sturgeon was estimated to be 870 individuals (Kahnle et al. 2007). Based on the juvenile assessments from Peterson et al. (2000), the Hudson River suffered a series of recruitment failures, which triggered the ASMFC fishing moratorium to allow the populations to recover. Long-term juvenile surveys indicate that the Hudson River population supports successful annual year classes since 2000 and the annual production has been stable and/or slightly increasing in abundance (NMFS 2007c). Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data.

The 2017 ASMFC stock assessment determined there is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 31% probability that mortality for the New York Bight DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

Genetic Diversity. The genetic diversity of Atlantic sturgeon throughout its range has been well documented (Bowen and Avise 1990; Ong et al. 1996; Waldman et al. 1996; Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (King et al. 2001; Waldman et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters.

Distribution. The Connecticut River has long been known as a seasonal aggregation area for subadult Atlantic sturgeon, and both historical and contemporary records document presence of Atlantic sturgeon in the river as far upstream as Hadley, MA (Savoy and Shake, 1993; Savoy and Pacileo, 2003; NMFS and USFWS, 2007). The upstream limit for Atlantic sturgeon on the Hudson River is the Federal Dam at the fall line, approximately river kilometer 246 (Dovel and Berggren, 1983; Bain, 1998; Kahnle et al., 1998; Everly and Boreman, 1999). In the Delaware River, there is evidence of Atlantic sturgeon presence from the mouth of the Delaware Bay to the head-of-tide at the fall line near Trenton on the New Jersey side and Morrisville on the Pennsylvania side of the River, a distance of 220 river kilometers (Breece et al., 2013).

Designated Critical Habitat. Designated Critical Habitat was effective September 18, 2017. Based on the best scientific information available for the life history needs of the New York Bight DPS, the physical features essential to the conservation of the species and that may require special management considerations or protection are:

- Hard bottom substrate (*e.g.*, rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (*i.e.*, 0.0 to 0.5 parts per thousand (ppt) range) for settlement of fertilized eggs, refuge, growth, and development of early life stages;
- Aquatic habitat with a gradual downstream salinity gradient of 0.5 up to as high as 30 ppt and soft substrate (*e.g.*, sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development;
- Water of appropriate depth and absent physical barriers to passage (*e.g.*, locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support:
- Unimpeded movement of adults to and from spawning sites;
- Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- Staging, resting, or holding of subadults or spawning condition adults.
- Water depths in main river channels must also be deep enough (*e.g.*, at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.
- Water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support:
- Spawning;
- Annual and interannual adult, subadult, larval, and juvenile survival; and
- Larval, juvenile, and subadult growth, development, and recruitment (*e.g.*, 13 °C to 26 °C for spawning habitat and no more than 30 °C for juvenile rearing habitat, and 6 milligrams per liter (mg/L) dissolved oxygen (DO) or greater for juvenile rearing habitat).

Habitat, including critical habitat, for the New York Bight DPS continues to be lost or altered because of anthropogenic activities. In the Delaware River, water quality is still a concern and is likely a threat to the survival of an entire year class in some years when dissolved oxygen levels are low. New information indicates that all Atlantic sturgeons are highly vulnerable to climate change, and that the Atlantic sturgeon's low likelihood to change distribution in response to current global climate change will also expose them to effects of climate change on estuarine habitat such as changes in the occurrence and abundance of prey species in currently identified key foraging areas.

Recovery Goals. The Recovery Plan has not yet been finalized. A 2018 recovery outline (NMFS 2018b)provides the following Recovery vision statement:

Subpopulations of all 5 Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future.

8.36 Atlantic Sturgeon, Chesapeake Bay DPS

Table 81. Atlantic Sturgeon, Chesapeake Bay DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Sturgeon, Atlantic	Chesapeake Bay	Endangered	2022	2012 77 FR 5880	2018 (Recovery Outline)	2017 82 FR 39160

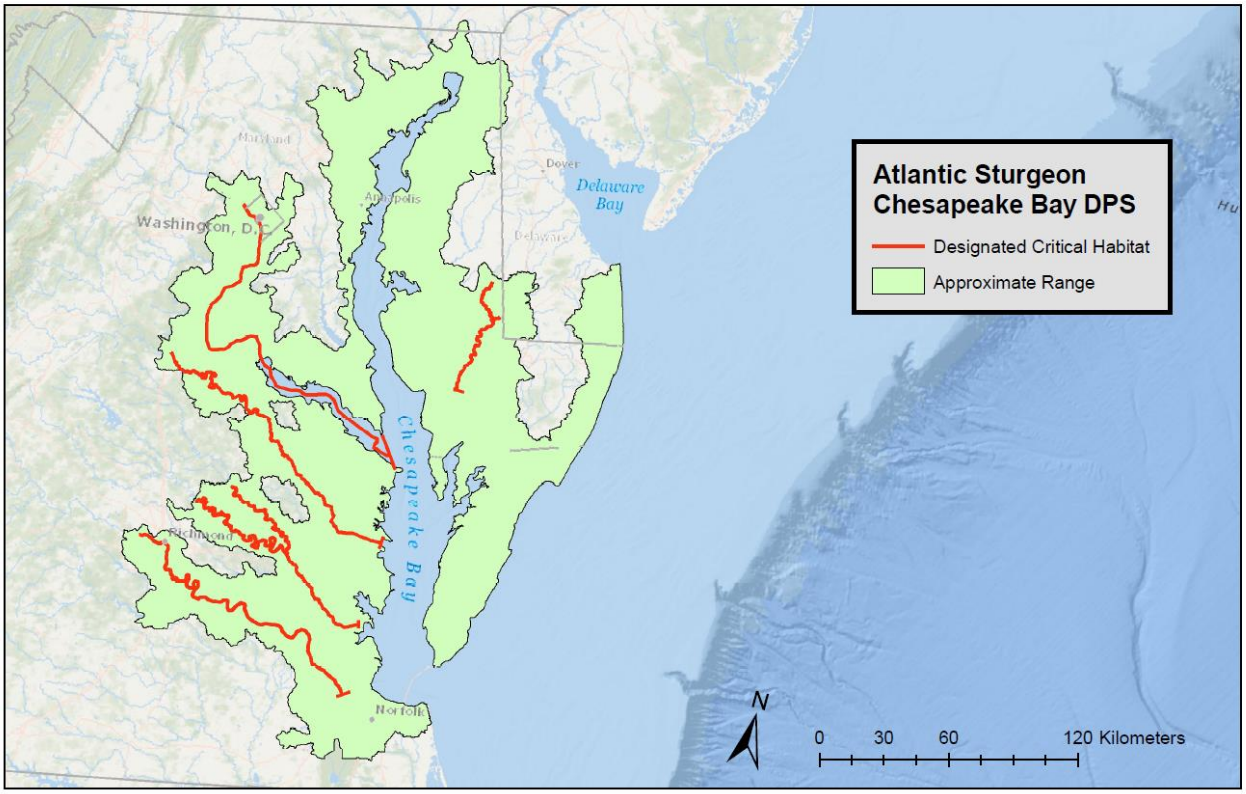


Figure 46**. Atlantic Sturgeon, Chesapeake Bay DPS range and designated critical habitat**

Species Description. The Atlantic sturgeon is a long lived, late maturing, anadromous species. Atlantic sturgeon attain lengths of up to approximately 14 feet, and weights of more than 800 pounds. They are bluish black or olive brown dorsally with paler sides and a white ventral surface and have 5 major rows of dermal scutes (Colette and Klein-MacPhee 2002). On February 6, 2012, 4 DPSs of Atlantic sturgeon: New York Bight, Chesapeake Bay, Carolina, and South Atlantic, were listed as endangered and the Gulf of Maine DPS was listed as threatened (77 FR 5880; 77 FR 5914). The Chesapeake Bay DPS is comprised of Atlantic sturgeon that originate from rivers that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, Virginia.

Status. There are 3 known spawning subpopulations: the James River, the Pamunkey River of the York River system, and Marshyhope Creek of the Nanticoke River system (NMFS 2017). Comprehensive information on current abundance and population trends for any of the Chesapeake Bay spawning subpopulations is lacking (ASSRT 2007). The 2017 ASMFC stock assessment determined that abundance of the Chesapeake Bay DPS is "depleted" relative to historical levels (ASMFC 2017). The assessment also determined there is a relatively low probability (37%) that abundance of the Chesapeake Bay DPS has increased since the implementation of the 1998 fishing moratorium, and a 30% probability that mortality for the Chesapeake Bay DPS exceeds the mortality threshold used for the assessment (ASMFC 2017). The number of spawning adults in the Pamunkey River spawning population is only hundreds per year. There are no spawning run estimates for the James River spawning populations or for the spawning population in the Nanticoke River system. Despite research effort, natal juveniles are rarely captured which suggests that the Chesapeake Bay DPS has low reproductive success.

New information supports NMFS determination in the listing rule that the Chesapeake Bay DPS has low abundance, and that the current numbers of spawning adults are 1–2 orders of magnitude smaller than historical levels. Also, the new information provided in the 2022 update supports NMFS determination in the listing rule that the Chesapeake Bay DPS continues to be significantly affected by threats from bycatch and vessel strikes as well as threats to habitat from continued degraded water quality, dredging, and global climate change, and that these threats are considered to be unsustainable at present. Further, the new information supports our determinations in the listing rule that there is a lack of existing regulatory mechanisms to adequately address these threats, particularly to address the threat of vessel strikes.

Life History. Atlantic Sturgeon size at sexual maturity varies with latitude with individuals reaching maturity in the Hudson River at 11 – 21 years (Young et al. 1988). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Spawning adults generally migrate upriver in April-May in mid-Atlantic systems (Bain 1997; Caron et al. 2002; Murawski and Pacheco 1977; Smith 1985; Smith and Clugston 1997). There is a growing body of evidence that some Atlantic sturgeon river populations have 2 spawning seasons comprised of different spawning adults (Balazik and Musick 2015). Evidence of fall as well as spring spawning has been obtained for the Chesapeake Bay, Carolina, and South Atlantic DPSs (77 FR 5914; Balazik et al. 2012; Collins et al. 2000; Hager et al. 2014; Kahn et al. 2014a; NMFS 1998; Smith 1985). Atlantic sturgeon spawning is believed to occur in flowing water between the salt front and fall line of large rivers at depths of 3-27 meters (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year; spawning intervals range from 1-5 years for males (Caron et al. 2002; Collins et al. 2000; Smith 1985) and 2-5 for females (Stevenson and Secor 2000; Van Eenennaam et al. 1996; Vladykov and Greeley 1963).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (Gilbert 1989; Smith and Clugston 1997) between the salt front and fall line of large rivers (Bain et al. 2000; Borodin 1925; Crance 1987; Scott and Crossman 1973). Following spawning in northern rivers, males may remain in the river or lower estuary until the fall; females typically exit the rivers within 4 to 6 weeks (Savoy and Pacileo 2003). Hatching occurs approximately 94-140 hours after egg deposition at temperatures of 20° and 18° Celsius, respectively (Theodore et al. 1980). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from 0 to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend 2-5 years in freshwater before eventually becoming coastal residents as sub-adults (Boreman 1997; Schueller and Peterson 2010; Smith 1985).

Upon reaching the subadult phase individuals may move to coastal and estuarine habitats (Dovel and Berggren 1983; Murawski and Pacheco 1977; Smith 1985; Stevenson 1997). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Grunwald et al. 2008; King et al. 2001; Waldman et al. 2002). Because of high natal river fidelity, it appears that most rivers support independent populations (Grunwald et al. 2008; King et al. 2001; Waldman and Wirgin 1998; Wirgin et al. 2002; Wirgin et al. 2000). Atlantic sturgeon feed primarily on polychaetes, isopods, American sand lances and amphipods in the marine environment, while in fresh water they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Guilbard et al. 2007; Johnson et al. 1997a; Moser and Ross 1995; Novak et al. 2017; Savoy 2007).

Population Dynamics

Abundance. Historically, Atlantic sturgeon were common throughout the Chesapeake Bay and its tributaries (Kahnle et al. 1998, Wharton 1957, Bushnoe et al. 2005). At the time of listing, the James River was the only known spawning river for the Chesapeake Bay DPS (NMFS and USFWS, 2007; Hager, 2011; Balazik et al., 2012). Comprehensive information on current abundance and population trends for any of the Chesapeake Bay spawning subpopulations is lacking (ASSRT 2007). Based on research captures of tagged adults, an estimated 75 Chesapeake Bay DPS Atlantic sturgeon spawned in the Pamunkey River in 2013 (Kahn et al. 2014). In the James River, the total number of adult-sized Atlantic sturgeon captured in the spring and fall for 2012 through spring 2014 is 239 sturgeon. This is a minimum count of the number of adult Atlantic sturgeon in the James River during the time period because capture efforts did not occur in all areas and at all times when Atlantic sturgeon were present in the river.

The York River has a much smaller population, with annual spawning abundance estimates for 2013 of 75 (Kahn et al. 2014b). The effective population size of the York River population ranges from 6 to 12 individuals, the smallest effective population size for any Atlantic sturgeon subpopulation along the Atlantic Coast. The total York River adult Atlantic sturgeon abundance is estimated at 289 individuals.

Productivity / Population Growth Rate. The Chesapeake Bay once supported at least 6 historical Atlantic sturgeon spawning populations; however, today the bay is believed to support at the most, 4-5 spawning populations. Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data. The status review team (NMFS and USFWS, 2007) concluded that the populations in the James and York Rivers are at a moderate and moderately high risk of extinction.

Genetic Diversity. The genetic diversity of Atlantic sturgeon throughout its range has been well documented (Bowen and Avise 1990; Ong et al. 1996; Waldman et al. 1996; Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (King et al. 2001; Waldman et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters. Recent genetic evidence suggests that the James River spring and fall spawning Atlantic sturgeon are separate subpopulations (Balazik and Musick 2015).

Distribution. At the time of listing, the James River was the only known spawning river for the Chesapeake Bay DPS (NMFS and USFWS, 2007; Hager, 2011; Balazik et al., 2012). There are currently 3 known spawning subpopulations: the James River, the Pamunkey River of the York River system, and Marshyhope Creek of the Nanticoke River system (NMFS 2017). Adult Atlantic sturgeon enter the James River in the spring, with at least some eventually moving as far upstream as Richmond (river kilometer 155). Adults disperse through downriver sites and begin to move out of the river in late September to early October, occupy only lower river sites by November, and leave the river for the winter (Hager, 2011; Balazik et al., 2012). The condition of Atlantic sturgeon captured in the late summer-fall in the James and Pamunkey Rivers (e.g., adults expressing milt or eggs), the rapid upstream movement of adults in the fall, and the aggregation of adults relative to the salt wedge provide evidence that Chesapeake DPS Atlantic sturgeon also spawn in the fall. Genetic analyses suggest that Chesapeake Bay DPS Atlantic sturgeon travel great distances, including into Canadian waters, but occur most predominantly in marine waters of the New York and Mid-Atlantic Bight (Waldman et al., 2013; O'Leary et al., 2014; Wirgin et al., 2015a).

Designated Critical Habitat. Designated Critical Habitat was effective September 18, 2017. Based on the best scientific information available for the life history needs of the Chesapeake Bay DPS, the physical features essential to the conservation of the species and that may require special management considerations or protection are:

- Hard bottom substrate (*e.g.,* rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (*i.e.,* 0.0 to 0.5 parts per thousand (ppt) range) for settlement of fertilized eggs, refuge, growth, and development of early life stages;
- Aquatic habitat with a gradual downstream salinity gradient of 0.5 up to as high as 30 ppt and soft substrate (*e.g.,* sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development;
- Water of appropriate depth and absent physical barriers to passage (*e.g.,* locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support:
- Unimpeded movement of adults to and from spawning sites;
- Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- Staging, resting, or holding of subadults or spawning condition adults.
- Water depths in main river channels must also be deep enough (*e.g.,* at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.

- Water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support:
- Spawning;
- Annual and interannual adult, subadult, larval, and juvenile survival; and
- Larval, juvenile, and subadult growth, development, and recruitment (*e.g.*, 13 °C to 26 °C for spawning habitat and no more than 30 °C for juvenile rearing habitat, and 6 milligrams per liter (mg/L) dissolved oxygen (DO) or greater for juvenile rearing habitat).

Recovery Goals. The Recovery Plan has not yet been finalized. A 2018 recovery outline (NMFS 2018b)provides the following Recovery vision statement:

Subpopulations of all 5 Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future.

8.37 Atlantic Sturgeon, Carolina DPS

Table 82. Atlantic Sturgeon, Carolina DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Sturgeon, Atlantic	Carolina	Endangered	2023	2012 77 FR 5914	2018 (Recovery Outline)	2017 82 FR 39160

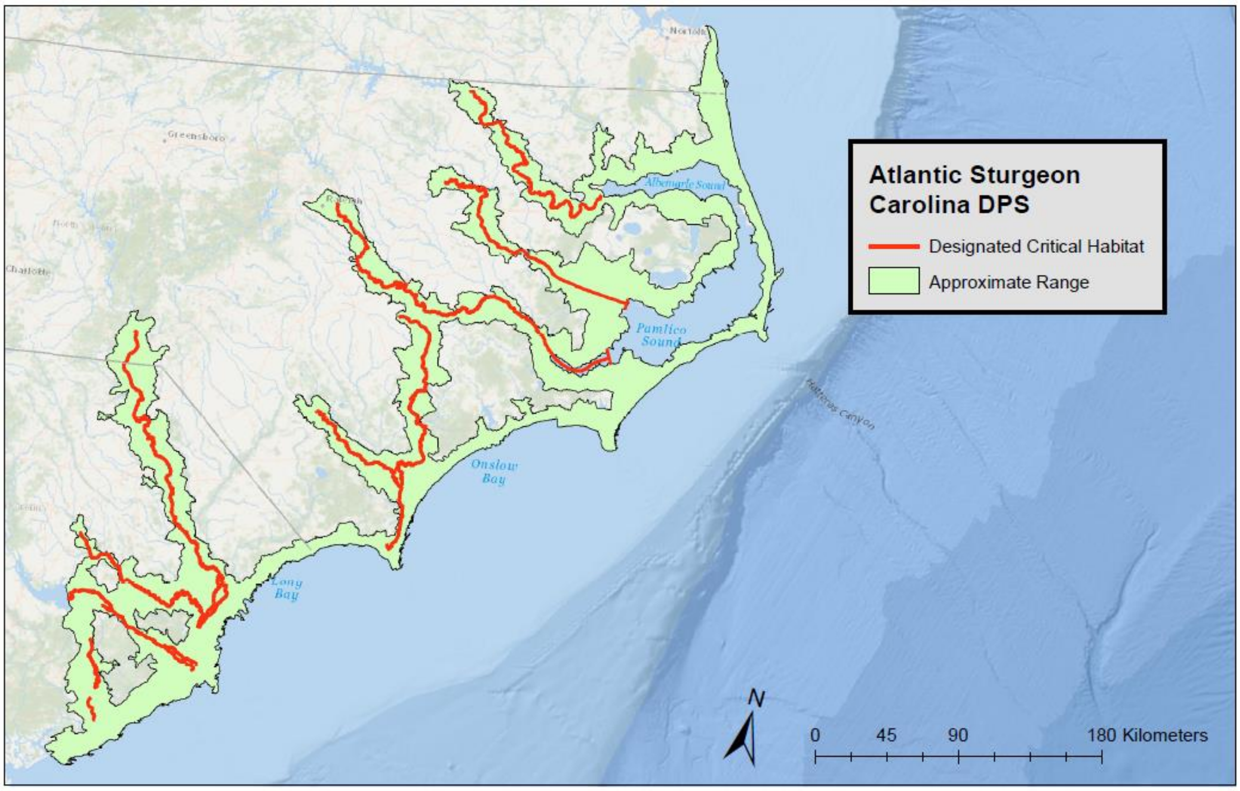


Figure 47**. Atlantic Sturgeon, Carolina DPS range and designated critical habitat**

Species Description. The Atlantic sturgeon is a long lived, late maturing, anadromous species. Atlantic sturgeon attain lengths of up to approximately 14 feet, and weights of more than 800 pounds. They are bluish black or olive brown dorsally with paler sides and a white ventral surface and have 5 major rows of dermal scutes (Colette and Klein-MacPhee 2002). On February 6, 2012, 4 DPSs of Atlantic sturgeon: New York Bight, Chesapeake Bay, Carolina, and South Atlantic, were listed as endangered and the Gulf of Maine DPS was listed as threatened (77 FR 5880; 77 FR 5914). Atlantic sturgeon from the Carolina DPS spawn in the rivers of North Carolina south to the Cooper River, South Carolina.

Status. There are currently 7 spawning subpopulations within the Carolina DPS: Roanoke River, Tar-Pamlico River, Neuse River, Northeast Cape Fear and Cape Fear Rivers, Waccamaw and

Great Pee Dee Rivers, Black River, Santee and Cooper Rivers; 1 is likely extinct (Sampit River). The existing subpopulations are likely at less than 3% of their historical abundance (ASSRT 2007). The 2017 ASMFC stock assessment determined the Carolina DPS abundance is "depleted" relative to historical levels (ASMFC 2017). The assessment also determined there is a relatively high probability (67%) that the Carolina DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a relatively high probability (75%) that mortality for the Carolina DPS exceeds the mortality threshold used for the assessment (ASMFC 2017). General threats include: habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries.

Life History. Atlantic Sturgeon size at sexual maturity varies with latitude with individuals reaching maturity in South Carolina at 5 – 19 years (Smith et al. 1982). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Spawning adults generally migrate upriver in the spring/early summer; February- March in southern systems. There is a growing body of evidence that some Atlantic sturgeon river populations have 2 spawning seasons comprised of different spawning adults (Balazik and Musick 2015). Evidence of fall as well as spring spawning has been obtained for the Chesapeake Bay, Carolina, and South Atlantic DPSs (77 FR 5914; Balazik et al. 2012; Collins et al. 2000; Hager et al. 2014; Kahn et al. 2014a; NMFS 1998; Smith 1985). Atlantic sturgeon spawning is believed to occur in flowing water between the salt front and fall line of large rivers at depths of 3-27 meters (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year; spawning intervals range from 1-5 years for males and 2-5 for females (Caron et al. 2002; Collins et al. 2000; Smith 1985).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (Gilbert 1989; Smith and Clugston 1997) between the salt front and fall line of large rivers (Bain et al. 2000; Borodin 1925; Crance 1987; Scott and Crossman 1973). Following spawning in northern rivers, males may remain in the river or lower estuary until the fall; females typically exit the rivers within 4 to 6 weeks (Savoy and Pacileo 2003). Hatching occurs approximately 94-140 hours after egg deposition at temperatures of 20° and 18° Celsius, respectively (Theodore et al. 1980). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from 0 to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend 2-5 years in freshwater before eventually becoming coastal residents as sub-adults (Boreman 1997; Schueller and Peterson 2010; Smith 1985).

Upon reaching the subadult phase individuals may move to coastal and estuarine habitats (Dovel and Berggren 1983; Murawski and Pacheco 1977; Smith 1985; Stevenson 1997). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Grunwald et al. 2008; King et al. 2001; Waldman et al. 2002). Because of high natal river fidelity, it appears that most rivers support independent populations (Grunwald et al. 2008; King et al. 2001; Waldman and Wirgin 1998; Wirgin et al. 2002; Wirgin et al. 2000). Atlantic sturgeon feed primarily on polychaetes, isopods, American sand lances and amphipods in the marine environment, while in fresh water they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Guilbard et al. 2007; Johnson et al. 1997a; Moser and Ross 1995; Novak et al. 2017; Savoy 2007).

Population Dynamics

Abundance. The Carolina DPS spawning populations are estimated to be at less than 3% of their historic levels. Prior to 1890, there were estimated to be 7,000 – 10,500 adult female Atlantic sturgeon in North Carolina and approximately 8,000 adult females in South Carolina. Currently, the existing spawning populations in each of the rivers in the Carolina DPS are thought to have less than 300 adults spawning each year.

Productivity / Population Growth Rate. Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data. The status review team (ASSRT 2007) concluded that the populations in the Roanoke, Tar/Pamlico, Neuse, Waccamaw, and Pee Dee river systems are at a moderate extinction risk and the populations in the Cape Fear and Santee-Cooper river systems are at a moderately high risk of extinction.

Genetic Diversity. The genetic diversity of Atlantic sturgeon throughout its range has been well documented (Bowen and Avise 1990; Ong et al. 1996; Waldman et al. 1996; Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (King et al. 2001; Waldman et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters.

Distribution. There are currently 7 spawning subpopulations within the Carolina DPS: Roanoke River, Tar-Pamlico River, Neuse River, Northeast Cape Fear and Cape Fear Rivers, Waccamaw and Great Pee Dee Rivers, Black River, Santee and Cooper Rivers; 1 is likely extinct (Sampit River). In the Roanoke River, Atlantic sturgeon are restricted to the lower 17 RKM of fall zone habitat, which extends from the Roanoke Rapids Dam to Weldon, North Carolina at RKM 204 (Armstrong and Hightower, 2002; Smith et al., 2014). The Tar-Pamlico riverine habitat is fully accessible to Atlantic sturgeon because the lowermost dam, the Rocky Mount Mill Pond Dam (RKM199), is located at the fall line. Spatial distribution of Atlantic sturgeon within the Neuse River is unknown. The Cape Fear River is tidally influenced by diurnal tides up to at least RKM 96. While telemetry data have not indicated Atlantic sturgeon presence above Lock and Dam #1 (RKM 95), other evidence indicates fish passage at the dam is successful or that fish pass through the lock. Pee Dee River system appears to be utilized by Atlantic sturgeon for summer/winter seasonal habitat as well as for spawning. Exact spatial distribution within the Pee Dee river system in unknown (Post et al. 2014). During a telemetry study from 2011 to 2014, Post et al. (2014) detected 10 juveniles and 10 adults utilizing the Black River. An adult male was detected at the last receiver station in the river 1 year (RKM 70.4) and the next to last receiver station in a subsequent year. Access to suitable spawning habitat is limited in the Santee-Cooper River system due to the locations of the Wilson Dam and St. Stephen Powerhouse on the Santee River and the Pinopolis Dam on the Cooper River. Nonetheless, the Santee-Cooper River system appears to be important foraging and refuge habitat and could serve as important spawning habitat once access to historical spawning grounds is restored through a fishway prescription under the Federal Power Act (NMFS 2007).

Designated Critical Habitat. Designated Critical Habitat was effective September 18, 2017. NMFS determined that the key conservation objectives for the Carolina DPS of Atlantic sturgeon are to increase the abundance of each DPS by facilitating increased survival of all life stages and facilitating adult reproduction and juvenile and subadult recruitment into the adult population. NMFS determined the physical features essential to the conservation of the species and that may require special management considerations or protection, which support the identified conservation objectives, are:

- Hard bottom substrate (*e.g.,* rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (*i.e.,* 0.0-0.5 ppt range) for settlement of fertilized eggs and refuge, growth, and development of early life stages;
- Transitional salinity zones inclusive of waters with a gradual downstream gradient of 0.5- up to 30 ppt and soft substrate (*e.g.,* sand, mud) between the river mouths and spawning sites for juvenile foraging and physiological development;
- Water of appropriate depth and absent physical barriers to passage (*e.g.,* locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouths and spawning sites necessary to support:
- Unimpeded movement of adults to and from spawning sites;
- Seasonal and physiologically-dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- Staging, resting, or holding of subadults or spawning condition adults.
- Water depths in main river channels must also be deep enough (at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.
- Water quality conditions, especially in the bottom meter of the water column, between the river mouths and spawning sites with temperature and oxygen values that support:
- Spawning;
- Annual and inter-annual adult, subadult, larval, and juvenile survival; and
- Larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L DO or greater likely supports juvenile rearing habitat, whereas DO less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25 °C. In temperatures greater than 26 °C, DO greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26 °C likely to support spawning habitat.

Recovery Goals. The Recovery Plan has not yet been finalized. A 2018 recovery outline (NMFS 2018b)provides the following Recovery vision statement:

Subpopulations of all 5 Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future.

8.38 Atlantic Sturgeon, South Atlantic DPS

Table 83. Atlantic Sturgeon, South Atlantic DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Sturgeon, Atlantic	South Atlantic	Endangered	2023	2012 77 FR 5914	2018 (Recovery Outline)	2017 82 FR 39160

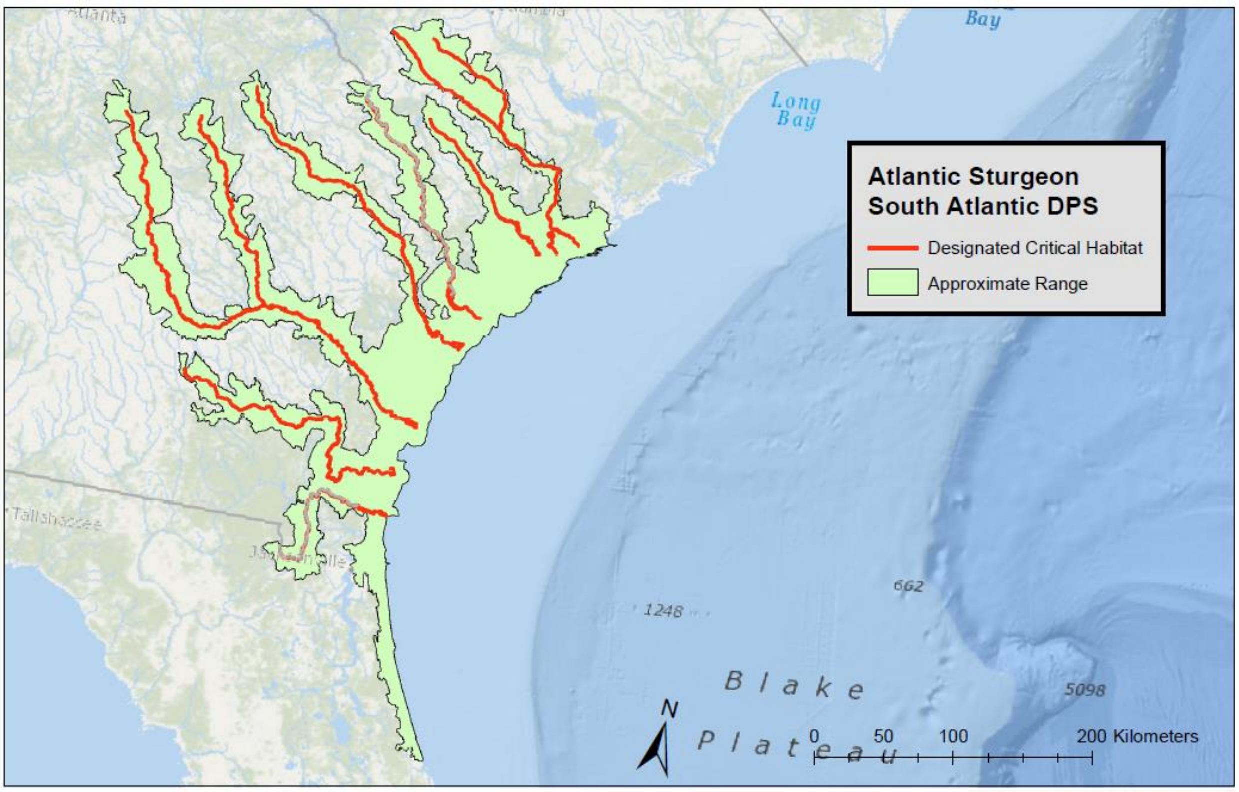


Figure 48. Atlantic Sturgeon, South Atlantic DPS range and designated critical habitat

Species Description. The Atlantic sturgeon is a long lived, late maturing, anadromous species. Atlantic sturgeon attain lengths of up to approximately 14 feet, and weights of more than 800 pounds. They are bluish black or olive brown dorsally with paler sides and a white ventral surface and have 5 major rows of dermal scutes (Colette and Klein-MacPhee 2002). On February 6, 2012, 4 DPSs of Atlantic sturgeon (New York Bight, Chesapeake Bay, Carolina, and South Atlantic), were listed as endangered, and 1 (Gulf of Maine) was listed as threatened (77 FR 5880;

77 FR 5914). Atlantic sturgeon from the South Atlantic DPS spawn from the Edisto River, South Carolina, to the St. Marys River at the Florida/Georgia border.

Status. The South Atlantic DPS historically supported 8 spawning subpopulations. At the time of listing only 6 spawning subpopulations were believed to have existed: the Combahee River, Edisto River, Savannah River, Ogeechee River, Altamaha River, and Satilla River. The 2 remaining spawning subpopulations in the Broad-Coosawatchie River and St. Marys River were believed to be extinct. However, new information provided from the capture of juvenile Atlantic sturgeon suggests the spawning subpopulation in the St. Marys River is not extinct and continues to exist, albeit at very low levels. Two of the spawning subpopulations in the South Atlantic DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. These 2 spawning subpopulations are likely less than 6% of their historic abundance. The 2017 ASMFC stock assessment determined the South Atlantic DPS abundance is "depleted" relative to historical levels (ASMFC 2017). The assessment concluded there was not enough information available to assess the abundance of the DPS relative to the 1998 fishing moratorium, but did conclude there was a (40% probability that mortality for the South Atlantic DPS exceeds the mortality threshold used for the assessment (ASMFC 2017). General threats include: habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries.

Life History. Atlantic Sturgeon age at sexual maturity varies with latitude with individuals reaching maturity in South Carolina at 5 – 19 years (Smith et al. 1982). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Spawning adults generally migrate upriver in the late summer/early fall; August-November in southern systems (77 FR 5914; Balazik et al. 2012; Collins et al. 2000; Hager et al. 2014; Kahn et al. 2014a; NMFS 1998; Smith 1985). Atlantic sturgeon spawning is believed to occur in flowing water between the salt front and fall line of large rivers at depths of 3-27 meters (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year; spawning intervals range from 1-5 years for males (Caron et al. 2002; Collins et al. 2000; Smith 1985) and 2-5 for females (Stevenson and Secor 2000; Van Eenennaam et al. 1996; Vladykov and Greeley 1963).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (Gilbert 1989; Smith and Clugston 1997) between the salt front and fall line of large rivers (Bain et al. 2000; Borodin 1925; Crance 1987; Scott and Crossman 1973). Following spawning in northern rivers, males may remain in the river or lower estuary until the fall; females typically exit the rivers within 4 to 6 weeks (Savoy and Pacileo 2003). Hatching occurs approximately 94-140 hours after egg deposition at temperatures of 20° and 18° Celsius, respectively (Theodore et al. 1980). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from 0 to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend 2-5 years in freshwater before eventually becoming coastal residents as sub-adults (Boreman 1997; Schueller and Peterson 2010; Smith 1985).

Upon reaching the subadult phase individuals may move to coastal and estuarine habitats (Dovel and Berggren 1983; Murawski and Pacheco 1977; Smith 1985; Stevenson 1997). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Grunwald et al. 2008; King et al. 2001; Waldman et al. 2002). Because of high natal river fidelity, it appears that most rivers support independent populations (Grunwald et al. 2008; King et al. 2001; Waldman and Wirgin 1998; Wirgin et al. 2002; Wirgin et al. 2000). Atlantic sturgeon feed primarily on polychaetes, isopods, American sand lances and amphipods in the marine environment, while in freshwater they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Guilbard et al. 2007; Johnson et al. 1997a; Moser and Ross 1995; Novak et al. 2017; Savoy 2007).

Population Dynamics

Abundance. Two of the spawning subpopulations in the South Atlantic DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. These 2 spawning subpopulations are likely less than 6% of their historic abundance. There are an estimated 343 adults that spawn annually in the Altamaha River and less than 300 adults spawning annually (total of both sexes) in the river systems where spawning still occurs (75 FR 61904; October 6, 2010). The abundance of the remaining 3 spawning subpopulations in the South Atlantic DPS is likely less than 1% of their historical abundance (ASSRT 2007).

Productivity / Population Growth Rate. Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data. During the last 2 decades, Atlantic sturgeon have been observed in most South Carolina coastal rivers, although it is not known if all rivers support a spawning population (Collins and Smith 1997). The status review team (ASSRT 2007) found that, overall, the South Atlantic DPS had a moderate risk (<50% chance) of becoming endangered over the next 20 years.

Genetic Diversity. The genetic diversity of Atlantic sturgeon throughout its range has been well documented (Bowen and Avise 1990; Ong et al. 1996; Waldman et al. 1996; Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (King et al. 2001; Waldman et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters.

Distribution. The South Atlantic DPS historically supported 8 spawning subpopulations. At the time of listing only 6 spawning subpopulations were believed to have existed: the Combahee River, Edisto River, Savannah River, Ogeechee River, Altamaha River, and Satilla River. The 2 remaining spawning subpopulations in the Broad-Coosawatchie River and St. Marys River were believed to be extinct. However, new information provided from the capture of juvenile Atlantic sturgeon suggests the spawning subpopulation in the St. Marys River is not extinct and continues to exist, albeit at very low levels. Seventy-six Atlantic sturgeon were tagged in the Edisto River during a 2011 to 2014 telemetry study (Post et al., 2014). Fish entered the river between April and June and were detected in the saltwater tidal zone until water temperature decreased below

25° C. They then moved into the freshwater tidal area, and some fish made presumed spawning migrations in the fall around September–October. Spawning migrations were thought to be occurring based on fish movements upstream to the presumed spawning zone between RKM 78 and 210. Fish stayed in these presumed spawning zones for an average of 22 days. The tagged Atlantic sturgeon left the river system by November. In the winter and spring, Atlantic sturgeon were generally absent from the system except for a few fish that remained in the saltwater tidal zone (Post et al., 2014). The Combahee—Salkehatchie River was identified as a spawning river for Atlantic sturgeon based on capture location and tracking locations of adults and the spawning condition of an adult (Collins and Smith, 1997; ASSRT, 2007). The farthest upstream detection of any tagged Atlantic sturgeon was RKM 56 (Post et al., 2014). Atlantic sturgeon in the Savannah River were documented displaying similar behavior 3 years in a row—migrating upstream during the fall and then being absent from the system during spring and summer. Forty-three Atlantic sturgeon larvae were collected in upstream locations (RKM 113–283) near presumed spawning locations (Collins and Smith, 1997). The Altamaha River supports 1 of the healthiest Atlantic sturgeon subpopulations in the Southeast In a telemetry study by Peterson et al. (2006), most tagged adult Atlantic sturgeon were found between RKM 215 and 420 in October and November when water temperatures were appropriate for spawning. Two general migration patterns were observed for fish in this system. Early upriver migrations that began in April— May typically occurred in 2 steps, with fish remaining at mid-river locations during the summer months before continuing upstream in the fall. The late-year migrations, however, were typically initiated in August or September and were generally non-stop. Regardless of which migration pattern was used during upstream migration, all fish exhibited a 1-step pattern of migrating downstream in December and early January (Ingram and Peterson in Post et al., 2014). The spatial distribution of Atlantic sturgeon in the Satilla, St. Marys, and St. Johns rivers is unknown.

Designated Critical Habitat. Designated Critical Habitat was effective September 18, 2017. NMFS determined that the key conservation objectives for the South Atlantic DPS of Atlantic sturgeon are to increase the abundance of each DPS by facilitating increased survival of all life stages and facilitating adult reproduction and juvenile and subadult recruitment into the adult population. NMFS determined the physical features essential to the conservation of the species and that may require special management considerations or protection, which support the identified conservation objectives, are:

- Hard bottom substrate (*e.g.*, rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (*i.e.*, 0.0-0.5 ppt range) for settlement of fertilized eggs and refuge, growth, and development of early life stages;
- Transitional salinity zones inclusive of waters with a gradual downstream gradient of 0.5-up to 30 ppt and soft substrate (*e.g.*, sand, mud) between the river mouths and spawning sites for juvenile foraging and physiological development;
- Water of appropriate depth and absent physical barriers to passage (*e.g.*, locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouths and spawning sites necessary to support:
- Unimpeded movement of adults to and from spawning sites;
- Seasonal and physiologically-dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- Staging, resting, or holding of subadults or spawning condition adults.

- Water depths in main river channels must also be deep enough (at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.
- Water quality conditions, especially in the bottom meter of the water column, between the river mouths and spawning sites with temperature and oxygen values that support:
- Spawning;
- Annual and inter-annual adult, subadult, larval, and juvenile survival; and
- Larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L DO or greater likely supports juvenile rearing habitat, whereas DO less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25 °C. In temperatures greater than 26 °C, DO greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26 °C likely to support spawning habitat.

Recovery Goals. The Recovery Plan has not yet been finalized. A 2018 recovery outline (NMFS 2018b)provides the following Recovery vision statement:

Subpopulations of all 5 Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future.

8.39 Gulf Sturgeon

Table 84. Gulf Sturgeon; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus desotoi	Sturgeon, Gulf	Entire	Threatened	2022	1991 56 FR 49653	1995	2003 68 FR 13369

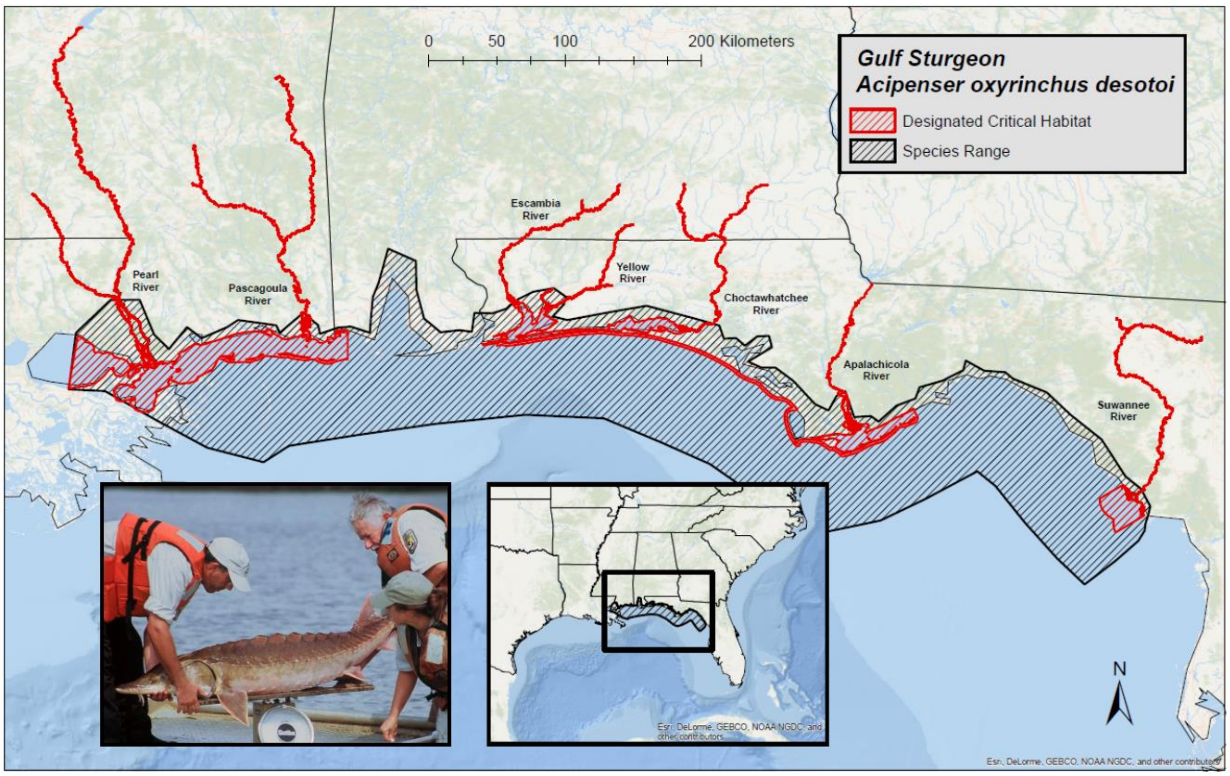


Figure 49. Gulf Sturgeon range and designated critical habitat

Species Description. Gulf sturgeon are benthic fusiform fish with an extended snout, vertical mouth, 5 rows of scutes (bony plates surrounding the body), 4 barbels (slender, whisker-like feelers anterior to the mouth used for touch and taste), and a heterocercal (upper lobe is longer than lower) caudal fin. Adults range from 6-8 feet in length and weigh up to 200 pounds; females grow larger than males (USFWS 2009b). The Gulf sturgeon was listed as Threatened on September 30, 1991.

Status. Past declines in the abundance of Gulf sturgeon has been attributed to targeted fisheries in the late 19th and early 20th centuries, habitat loss associated with dams and sills, habitat degradation associated with dredging, de-snagging, and contamination by pesticides, heavy metals, and other industrial contaminants, and certain life history characteristics (e.g., slow growth and late maturation) (56 FR 49653). Recent abundance data described in the 2022 Status Update indicate a roughly stable or slightly increasing population trend over the last decade in the eastern river systems (Florida), with a much stronger increasing trend in the Suwannee River. Populations in the western portion of the range (Mississippi and Louisiana) are believed to exhibit lower abundance than those in the eastern portion of the range. Effects of climate change (warmer water, sea level rise and higher salinity levels) could lead to accelerated changes in habitats utilized by Gulf sturgeon. The rate that climate change and corollary impacts are occurring may outpace the ability of the Gulf sturgeon to adapt given its limited geographic distribution and low dispersal rate. In general, Gulf sturgeon populations in the eastern portion of the range appear to be stable or slightly increasing, while populations in the western portion are associated with lower abundances and higher uncertainty (USFWS 2009b).

Life History. Gulf sturgeon are long-lived, with some individuals reaching at least 42 years in age. Surveys in the Suwannee River suggest that a more common maximum age may be around 25 years (Sulak and Clugston 1999). Age at sexual maturity for females ranges from 8 to 17 years, and for males from 7 to 21 years (Huff 1975). In general, Gulf sturgeon spawn up-river in spring, spend winter months in near-shore marine environments, and utilize pre- and post-spawn staging and nursery areas in the lower rivers and estuaries (Heise et al. 2005; Heise et al. 2004). There is some evidence of autumn spawning in the Suwannee River, however there is uncertainty as to whether this spawning is due to environmental conditions or represents a genetically distinct population (Randall and Sulak 2012). Gulf sturgeon spawn at intervals ranging from 3-5 years for females and 1-5 years for males (Fox et al. 2000; Smith 1985). The spring migration to up-river spawning sites begins in mid-February and continues through May. Fertilization is external; females deposit their eggs in the upper reaches of and show preference for hard, clean substrate (e.g., bedrock covered in gravel and small cobble).

Upon hatching from their eggs, Gulf sturgeon larvae spend the first few days of life sheltered in interstitial spaces at the spawning site (Kynard and Parker 2004). At the onset of feeding, age-0 Gulf sturgeon disperse and are often found on shallow sandbars and rippled sand shoals (<4 meters depth) (Sulak and Clugston 1998). Young-of-the-year spend 6-10 months slowing working their way downstream feeding on aquatic insects (e.g., mayflies and caddisflies), worms (oligochaetes), and bivalve molluscs, and arrive in estuaries and river mouths by mid-winter (Sulak and Clugston 1999) where they will spend their next 6 years developing. After spawning, adult Gulf sturgeon migrate downstream to summer resting and holding areas in the mid to lower reaches of the rivers where they may hold until November (Wooley and Crateau 1985). While in freshwater adults lose a substantial amount of their weight, but regain it upon entering the estuaries. Sub adult and non-spawning adults also spend late spring through fall in these holding areas (Foster and Clugston 1997). By early December all adult and sub-adult Gulf sturgeon return to the marine environment to forage on benthic (bottom dwelling) invertebrates along the shallow nearshore (2-4 meter depth), barrier island passes, and in unknown off-shore locations in the gulf (Carr et al. 1996; Fox et al. 2002; Huff 1975; Ross et al. 2009). Juvenile Gulf sturgeon overwinter in estuaries, river mouths, and bays; juveniles do not enter the nearshore/offshore marine environments until around age 6 (Sulak and Clugston 1999). Gulf sturgeon show a high degree of river-specific fidelity (Rudd et al. 2014). Adult and sub-adult Gulf sturgeon fast while in freshwater environments and are almost entirely dependent on the estuarine/marine environment for food (Gu et al. 2001; Wooley and Crateau 1985). Some juveniles (ages 1-6) will also fast in the freshwater summer holding areas, but the majority feed year round in the estuaries, river mouths, and bays (Sulak et al. 2009).

Population Dynamics

Abundance. Currently, 7 rivers are known to support reproducing populations of Gulf sturgeon. The most recent abundance estimates reported in the 5-Year Review are shown in Table 85 (USFWS 2009b).

Table 85. Gulf sturgeon abundance estimates by river and year.

River	Year of Data Collection	Abundance Estimate[a]	Lower/Upper 95% CI[b]	Source
Pearl	2001	430	323/605	(Rogillio et al. 2001)
Pascagoula	2000	216	124/429	(Ross et al. 2001)
Escambia	2015	372	241/576	(USFWS 2007)
Yellow	2012	398	111/1,859	(Berg et al. 2007)
Choctawhatchee	2008	3,314	not reported	(USFWS 2009a)
Apalachicola	2014	1,288	not reported	(Sulak et al. 2016)
Suwannee	2012-2013	9,743	not reported	(Sulak al. 2016)

a Estimates refer to numbers of individuals greater than a certain size, which varies between studies depending on sampling gear, and in some cases, numbers of individuals that use a particular portion of the river. Refer to original publication for details.

b Large confidence intervals (CI) around the mean estimates reflect the low capture probability in mark-recapture survey.

Confidence intervals (CI) for the 7 major rivers with reproducing populations. Table modified from USFWS (2009b)

Productivity / Population Growth Rate. Gulf sturgeon abundance trends are typically assessed on a riverine basis. In general, Gulf sturgeon populations in the eastern portion of the range appear to be stable or slightly increasing, while populations in the western portion are associated with lower abundances and higher uncertainty (USFWS 2009b). Pine and Martell (2009) reported that, due to low recapture rates and sparse data, the population viability of Gulf sturgeon is currently uncertain.

Genetic Diversity. When grouped by genetic relatedness, 5 regional or river-specific stocks emerge: (1) Lake Pontchartrain and Pearl River; (2) Pascagoula River; (3) Escambia, Blackwater and Yellow Rivers; (4) Choctawhatchee River; and (5) Apalachicola, Ochlocknee and Suwanee Rivers (Rudd et al. 2014; Stabile et al. 1996). Gene flow is low in Gulf sturgeon stocks, with each stock exchanging less than 1 mature female per generation (Waldman and Wirgin 1998).

Distribution. The Gulf sturgeon is 1 of 2 subspecies of the Atlantic Sturgeon (USFWS 1995). The Gulf sturgeon is anadromous, and historically occurred in most river systems from the Mississippi river east to Tampa Bay, and in marine coastal/estuarine areas from the Central and Eastern Gulf of Mexico south to Florida Bay (Wooley and Crateau 1985). The current range of the sub-species extends from Lake Pontchartrain in Louisiana east to the Suwannee river system in Florida. Within that range, 7 major rivers are known to support reproducing populations: Pearl, Pascagoula, Escambia, Yellow, Choctawhatchee, Apalachicola, and Suwannee (USFWS 2009b).

Designated Critical Habitat. Critical Habitat for Gulf sturgeon was established in 2003 68 FR 13370) and consists of 14 geographic units encompassing 2,783 river kilometers as well as 6,042 square kilometers of estuarine and marine habitat. PBFs considered essential for the conservation of Gulf sturgeon are:

- Abundant food items, such as detritus, aquatic insects, worms, and/or molluscs, within riverine habitats for larval and juvenile life stages; and abundant prey items, such as amphipods, lancelets, polychaetes, gastropods, ghost shrimp, isopods, molluscs and/or crustaceans, within estuarine and marine habitats and substrates for subadult and adult life stages.
- Riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone, or hard clay;
- Riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during fresh water residency and possibly for osmoregulatory functions;
- A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging, and for maintaining spawning sites in suitable condition for egg attachment, egg sheltering, resting, and larval staging;
- Water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages;
- Sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
- Safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g., an unobstructed river or a dammed river that still allows for passage).

Recovery Goals. The 1995 Recovery Plan outlined 3 recovery objectives: (1) to prevent further reduction of existing wild populations of Gulf sturgeon within the range of the subspecies; (2) to establish population levels that would allow delisting of the Gulf sturgeon by management units (management units could be delisted by 2023 if required criteria are met); (3) to establish, following delisting, a self-sustaining population that could withstand directed fishing pressure within management units (USFWS 1995). Although the tasks outlined in the 1995 Recovery Plan address threats relative to listing factors (e.g., habitat modification, overutilization, water quality, etc.), the plan lacks criteria that would measure progress towards reducing these threats. The most recent Gulf sturgeon 5-year review recommended that criteria be developed in a revised recovery plan (USFWS 2009b).

8.40 Yelloweye Rockfish, Puget Sound/Georgia Basin DPS

Table 86. Yelloweye Rockfish, Puget Sound/Georgia Basin DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Sebastes ruberrimus	Rockfish, Yelloweye	Puget Sound/ Georgia Basin	Threatened	2016	2010 75 FR 22276	2017	2014 79 FR 68041

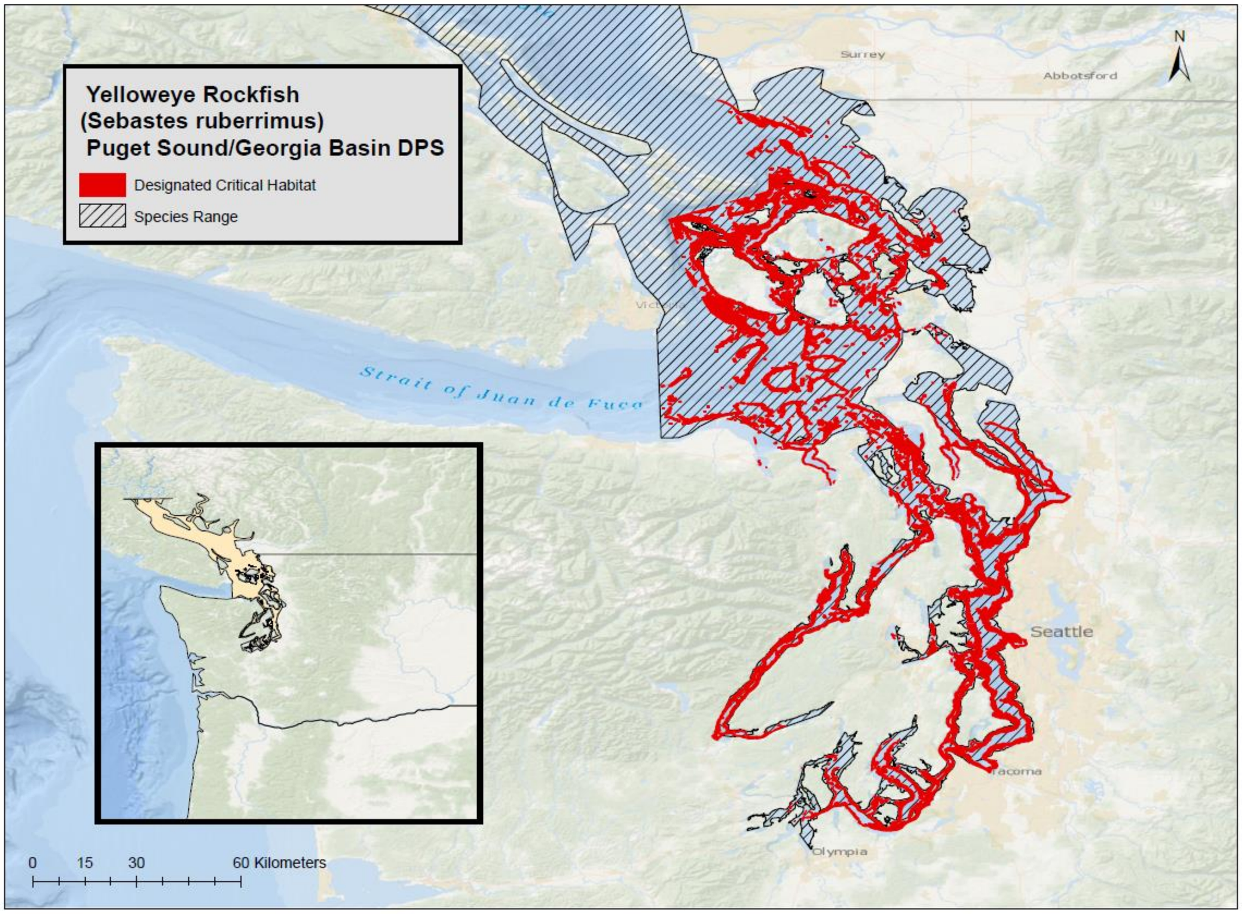


Figure 50**. Yelloweye Rockfish, Puget Sound/Georgia Basin DPS range and designated critical habitat**

Species Description. Yelloweye rockfish occur throughout most of the eastern Pacific Ocean ranging from northern Baja California to the Aleutian Islands, Alaska. The Puget Sound/Georgia Basin DPS is located along the coastal/inlet waters off the state of Washington and province of British Columbia and is the only population listed on the Endangered Species Act. Yelloweye rockfish is 1 of the largest species belonging to the genus *Sebastes*. They are orange-red to orange-yellow in color and may have black fin tips with bright yellow eyes. Adults usually have a light to white stripe on the lateral line; juveniles have 2 light stripes, 1 on the lateral line and a shorter 1 below the lateral line (Yamanaka et al. 2006).

Status. Puget Sound/Georgia Basin yelloweye rockfish were listed on the ESA as threatened on April 28, 2010 (75 FR 22276). Puget Sound/Georgia Basin yelloweye rockfish abundance is much less than it was historically. The fish face several threats including bycatch in commercial and recreational harvest, non-native species introductions, and habitat degradation. Results from a recent genetic study comparing yelloweye rockfish individuals from within the Puget Sound/Georgia Basin DPS to those outside the DPS concluded that a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait (Tonnes et al. 2016). Further, individuals within Hood Canal are genetically differentiated from the rest of the Puget Sound/Georgia Basin DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (Tonnes et al. 2016). NMFS has determined that this DPS is likely to be in danger of extinction in the foreseeable future throughout all of its range; and in its 2016 status review (Tonnes et al. 2016), NMFS has recommended no change in the Puget Sound/Georgia Basin yelloweye rockfish’s threatened classification.

Life History. Female yelloweye rockfish and bocaccio produce from 1 to 3 million larvae annually, depending upon age and body size. Rockfish are viviparous, meaning the eggs are fertilized internally, the embryonic fish develop within the mother, and the young are released as larvae (Love et al. 2002). Larval rockfish are often observed under free-floating algae, seagrass, and detached kelp (Shaffer et al. 1995; Love et al. 2002), and also occupy the full water column (Weis 2004). Generally, juvenile rockfish move from the pelagic environment and associate with benthic environments when they reach about 1.2 to 3.6 inches (3 to 9 cm) in length and approximately the age of 3 to 6 months (Love et al. 2002). As they grow, juveniles of each species gradually move to areas of high rugosity (roughness) and rocky habitat in deeper waters (Love et al. 1991; Johnson et al. 2003; Love et al. 2002). Adult yelloweye rockfish remain near the substrate and have relatively small home ranges, while some bocaccio have larger home ranges, move long distances, and spend time suspended in the water column (Demott 1983; Love et al. 2002; Friedwald 2009). Depth is generally the most important determinant in the distribution of many rockfish species of the Pacific Coast (Chen 1971; Williams and Ralston 2002; Anderson and Yoklavich 2007; Young et al. 2010). Adult yelloweye rockfish and bocaccio generally occupy habitats from approximately 90 to 1,394 feet (30 to 425 meters) (Orr et al. 2000; Love et al. 2002). Larval and juvenile rockfish feed on very small organisms such as zooplankton. Larger juveniles also feed upon small fish (Love et al. 1991). Adult yelloweye rockfish and bocaccio have diverse diets that include many species of fish and invertebrates.

Population Dynamics

Abundance. The apparent steep reduction of ESA-listed rockfish in Puget Sound proper (and their consequent fragmentation) has led to concerns about the viability of these populations (Drake et al. 2010). Recreationally caught yelloweye rockfish in the 1970s spanned a broad size range. By the 2000s, fewer older fish in the population were observed (Drake et al. 2010). However, overall fish numbers in the database were also much lower, making it difficult to determine if clear size truncation occurred. With age truncation, the reproductive burden may have shifted to younger and smaller fish. This could alter larval release timing and condition, which may create a mismatch with habitat conditions and potentially reduce offspring viability (Drake et al. 2010).

The Washington Department of Fish and Wildlife has generated several population estimates of the Puget Sound/Georgia Basin yelloweye rockfish DPS in recent years. ROV surveys in the San Juan Island region in 2008 (focused on rocky substrate) and 2010 (across all habitat types) estimated a population of 47,407±11,761 and 114,494±31,036 individuals, respectively. A 2015 ROV survey of that portion of the DPSs south of the entrance to Admiralty Inlet encountered 35 yelloweye rockfish, producing a preliminary population estimate of 66,998 ± 7,370 individuals (WDFW 2017).

Productivity / Population Growth Rate. Productivity measures a population's growth rate through all or a portion of its life-cycle. Yelloweye rockfish life-history traits suggest generally low inherent productivity levels because they are long-lived, mature slowly, and have sporadic episodes of successful reproduction (Tolimieri and Levin 2005, Drake et al. 2010). Adult yelloweye rockfish typically occupy relatively small ranges (Love et al. 2002) and may not move to find suitable mates. So as the density of mature fish has decreased, productivity may have also been impacted by Allee effects. Further, past commercial and recreational fishing may have depressed the DPS to a threshold beyond which optimal productivity is unattainable (Drake et al. 2010). Also, historic over-fishing may have had dramatic impacts on population size or age structure.

Genetic Diversity. Rockfish diversity characteristics include fecundity, larvae release timing, larvae condition, morphology, age at reproductive maturity, physiology, and molecular genetic characteristics. The leading factors affecting diversity are the relatively small home ranges of juveniles and subadults (Love et al. 2002) and low population size of all life stages. Yelloweye rockfish spatial structure and connectivity are likely threatened by the apparently severe reduction of fish numbers throughout Hood Canal and South Puget Sound. At 2,330 square km, Puget Sound is a small geographic area compared with the entire yelloweye rockfish range in the northeastern Pacific.

Results from a recent genetic study comparing yelloweye rockfish individuals from within the PS/GB DPS (n=52) to those outside the DPS (n=52) provided multiple results (Tonnes et al. 2016). First, yelloweye rockfish in inland Canadian waters as far north as Johnstone Strait were genetically similar to those within the PS/GB DPS (the DPS was subsequently revised to include Johnstone strait individuals). Second, a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait. Lastly, individuals within Hood Canal are genetically differentiated from the rest of the DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (Tonnes et al. 2016).

Distribution. Spatial distribution provides a protective measure from larger scale anthropogenic changes that damage habitat suitability, such as oil spills or hypoxia, which can occur within 1 basin but not necessarily the other basins. When localized depletion of rockfish occurs, it can reduce stock resiliency, especially when exacerbated by the natural hydrologic constrictions within Puget Sound (Levin 1998, Hilborn et al. 2003, Hamilton 2008). Combining this with limited adult movement, yelloweye rockfish population viability may be highly influenced by the probable localized loss of populations within the DPS, thus decreasing spatial structure and connectivity.

Designated Critical Habitat. Critical habitat was designated for Puget Sound/Georgia Basin yelloweye rockfish on November 13, 2014, when NMFS published a final rule in the *Federal Register* (79 FR 68042). The critical habitat in the U.S. is spread amongst 5 interconnected, biogeographic basins (San Juan/Strait of Juan de Fuca basin, Main basin, Whidbey basin, South Puget Sound, and Hood Canal) based upon presence and distribution of adult and juvenile yelloweye rockfish, geographic conditions, and habitat features. PBFs considered essential for the conservation of yelloweye rockfish, Puget Sound/Georgia Basin DPS are:

- **Adults**
    - Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities,
    - Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities, and
    - The type and amount of structure and rugosity that supports feeding opportunities and predator avoidance.
- **Juvenile**
    - Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities; and
    - Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities.

Recovery Goals. See the 2017 Recovery Plan (NMFS 2017f) for a complete description of recovery goals and criteria. Below are the delisting criteria:

Table 87. Recovery targets for yelloweye rockfish

	Overall Minimum Productivity (SPR)	**Minimum Time at Target**
Hood Canal population		
Scenario A	20% to 24%	15 years (no less than 4 systematic sampling events with 80% probability)
Scenario B	25% (and above)	10 years (no less than 3 systematic sampling events with 80% probability)
non-Hood Canal population		
Scenario A	15% (and increasing after first sampling event finds 15%)	25 years (no less than 5 systematic sampling events with 80% probability/confidence interval)
Scenario B	20% to 24%	15 years (no less than 4 systematic sampling events with 80% probability)
Scenario C	25% (and above)	10 years (no less than 3 systematic sampling events with 80% probability)

8.41 Boccacio, Puget Sound/Georgia Basin DPS

Table 88. Bocaccio, Puget Sound/Georgia Basin DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Sebastes paucispinis	Bocaccio	Puget Sound/ Georgia Basin	Endangered	2016	2010 75 FR 22276	2017	2014 79 FR 68042

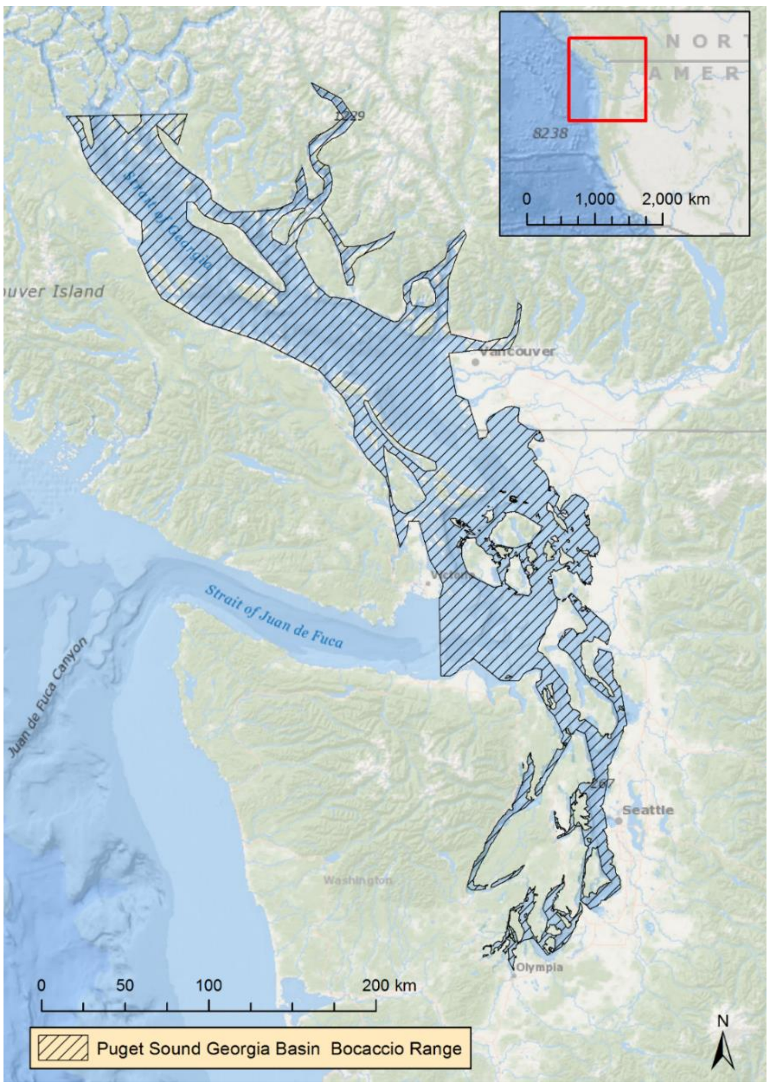


Figure 51. Bocaccio, Puget Sound/Georgia Basin DPS range

Species Description. The bocaccio is a long-lived species of rockfish, occupying the eastern Pacific Ocean in waters from California to Alaska. Bocaccio are a large (3 feet, 1 meter) Pacific rockfish, olive to burnt orange-brown, with a distinctively long jaw. The Puget Sound/Georgia Basin DPS was first listed as endangered by NMFS on April 28, 2010 (75 FR 22276). The listing was updated on January 23, 2017 (82 FR 7711), when NMFS amended the listing description to include fish residing within the Puget Sound/Georgia Basin rather than only fish originating from the Puget Sound/Georgia Basin.

Status. Bocaccio resistance to depletion and recovery is hindered by demographic features (Love et al. 1998a). Bocaccio are long-lived fishes, taking several years to reach sexual maturity and becoming more fecund with age (Dorn 2002). As harvesting targeted the largest individuals available, bocaccio have become less capable of recovering population numbers (Love et al. 1998b). Bocaccio reproduction appears to be characterized by frequent recruitment failures, punctuated by occasional high success years (Love et al. 1998b; MacCall and He 2002). Over the past 45 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes. Recruitment success appears to be linked to oceanographic/climactic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998b; MacCall 1996; Moser et al. 2000; Sakuma and Ralston 1995). Harvey et al. (2006) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery.

Life History. Female yelloweye rockfish and bocaccio produce from 1 to 3 million larvae annually, depending upon age and body size. Rockfish are viviparous, meaning the eggs are fertilized internally, the embryonic fish develop within the mother, and the young are released as larvae (Love et al. 2002). Larval rockfish are often observed under free-floating algae, seagrass, and detached kelp (Shaffer et al. 1995; Love et al. 2002), and also occupy the full water column (Weis 2004). Generally, juvenile rockfish move from the pelagic environment and associate with benthic environments when they reach about 1.2 to 3.6 inches (3 to 9 cm) in length and approximately the age of 3 to 6 months (Love et al. 2002). Young-of-year juvenile bocaccio occur on shallow rocky reefs and nearshore areas. As they grow, juveniles gradually move to areas of high rugosity (roughness) and rocky habitat in deeper waters (Love et al. 1991; Johnson et al. 2003; Love et al. 2002). Adult yelloweye rockfish remain near the substrate and have relatively small home ranges, while some bocaccio have larger home ranges, move long distances, and spend time suspended in the water column (Demott 1983; Love et al. 2002; Friedwald 2009). Depth is generally the most important determinant in the distribution of many rockfish species of the Pacific Coast (Chen 1971; Williams and Ralston 2002; Anderson and Yoklavich 2007; Young et al. 2010). Adult yelloweye rockfish and bocaccio generally occupy habitats from approximately 90 to 1,394 feet (30 to 425 meters) (Orr et al. 2000; Love et al. 2002). Larval and juvenile rockfish feed on very small organisms such as zooplankton. Larger juveniles also feed upon small fish (Love et al. 1991). Adult yelloweye rockfish and bocaccio have diverse diets that include many species of fish and invertebrates.

Population Dynamics

Abundance. There is no current population abundance estimate for the Puget Sound/Georgia Basin DPS bocaccio. There is a lack of long-term information on the Puget Sound/Georgia Basin

DPS bocaccio abundance, although among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline. This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climactic/oceanographic conditions (MacCall and He 2002).

Productivity / Population Growth Rate. The rate of decline for rockfish in Puget Sound has been estimated at 3.1% to 3.8% annually for the period 1977 to 2014 (NMFS 2016h).

Genetic Diversity. Puget Sound/Georgia Basin DPS bocaccio are distinct from bocaccio elsewhere in its range, likely due to its inhabitance of a geographically isolated area. There is no genetic information available for bocaccio in Puget Sound/Georgia Basin (NMFS 2016h).

Distribution. Puget Sound/Georgia Basin bocaccio occupy the inland marine waters east of the Strait of Juan de Fuca and south of the northern Strait of Georgia.

Designated Critical Habitat. Critical habitat for the Puget Sound/Georgia Basin DPS for bocaccio, canary rockfish, and yelloweye rockfish was finalized in 2014 (79 FR 68041). The critical habitat designation was updated in 2017 when canary rockfish were delisted and their critical habitat removed (82 FR 7711). The specific areas designated for bocaccio include approximately 1,184.75 mi^2 (3,068.5 km^2) of marine habitat in Puget Sound, Washington. Designated habitat was divided into 2 units—nearshore, to support juveniles, and deeper, rocky habitat for adults. Features essential for adult boccacio (greater than 30 meters deep) include sufficient prey resources, water quality, and rocks or highly rugose habitat. For juvenile boccacio, features essential for their conservation include sufficient prey resources and water quality.

Recovery Goals. See the 2017 Recovery Plan (NMFS 2017f) for a complete description of recovery goals and criteria. Below are the delisting criteria:

Table 89. Recovery targets for bocaccio

	Overall Minimum Productivity (SPR)	**Minimum Time at Target**
Scenario A	15% (and increasing after first sampling event finds 15%)	15 years (no less than 4 systematic sampling events with 80% probability)
Scenario B	20% and above	10 years (no less than 3 systematic sampling events with 80% probability)
Scenario C	25% and above	5 years (no less than 2 systematic sampling events with 80% probability)

8.42 Nassau Grouper

Table 90. Nassau grouper; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Epinephelus striatus	Nassau grouper	N/A	Threatened	2013	2016 T – 81 FR 42268	2018 (Recovery Outline)	01/02/2024 89 FR 126

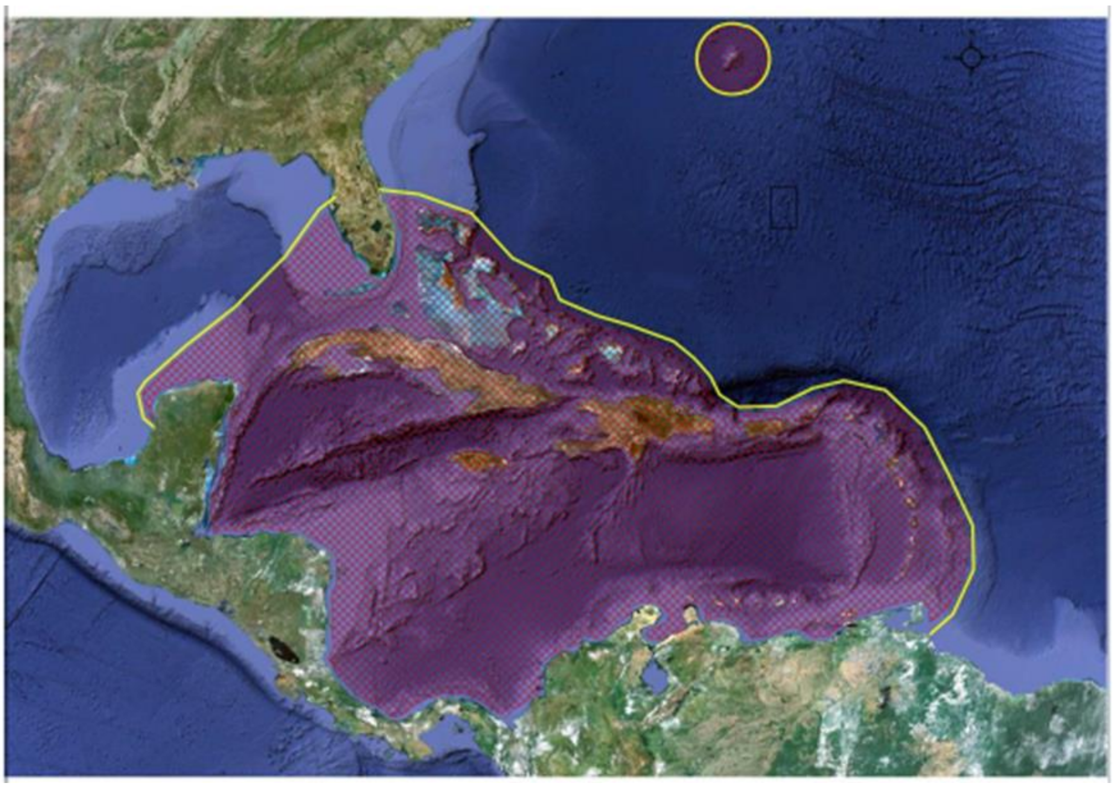


Figure 52. Nassau grouper range. From NMFS Biological Report 2013

Species Description. The Nassau grouper is a large, long-lived fish primarily occupying shallow water throughout the Caribbean, south Florida, Bermuda, and the Bahamas.
Adult Nassau grouper are large (up to 0.45 meter or 1.5 feet), have distinctive black and white stripes, and are generally found in shallow reef habitat. The Nassau grouper was listed as threatened on June 29, 2016 (81 FR 42268).

Status. Historically, tens of thousands of Nassau grouper spawned at aggregation sites throughout the Caribbean. Since grouper species were reported collectively in landings data, it is not possible to know how many Nassau grouper were harvested, or estimate historic abundance. That these large spawning aggregations occurred in predictable locations at regular times made the species susceptible to over-fishing, and was a primary cause of its decline. At some sites (e.g., Belize), spawning aggregations have decreased by over 80% in the last 25 years (Sala et al. 2001), or have disappeared entirely (e.g., Mexico) (Aguilar-Perera 2006). Nassau groupers are also targeted for fishing throughout the year during non-spawning months. In some locations, spawning aggregations are increasing. Many Caribbean countries have banned or restricted Nassau grouper harvest, and it is believed that the areas of higher abundance are correlated with effective regulations (81 FR 42268). Since Nassau groupers are dependent upon coral reefs at various points in their life history, loss of coral reef habitat due to climate change. Increasing water temperatures may change the timing and location of spawning. Habitat degradation due to water pollution also poses a threat to the species. Nassau grouper populations have been reduced from historic abundance levels, and remain vulnerable to unregulated harvest, especially the spawning aggregations.

Life History. Nassau groupers spawn once a year in large aggregations, in groups of a few dozen to thousands spawning at once. Nassau groupers move in groups towards the spawning aggregation sites parallel to the coast or along the shelf edge at depths between 20 and 33 meters. Spawning runs occur in late fall through winter (i.e., a month or 2 before spawning is likely). Sea surface temperature is thought to be a key factor in the timing of spawning, with spawning occurring at waters temperatures between 25 and 26 degrees Celsius. Spawning aggregation sites are located near significant geomorphological features, such as reef projections (as close as 50 meters to shore) and close to a drop-off into deep water over a wide depth range (6 to 60 meters). Sites are usually several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions. Nassau groupers stay on the spawning site for up to 3 months, spawning at the full moon or between the new and full moons. Spawning occurs within 20 minutes of sunset over the course of several days. There have been about fifty known spawning sites in insular areas throughout the Caribbean; many of these aggregations no longer form. Current spawning locations are found in Mexico, Bahamas, Belize, Cayman Islands, the Dominican Republic, Cuba, Puerto Rico, the U.S. Virgin Islands, and Florida.

Fertilized eggs are transported offshore by ocean currents. Thirty-five to forty days after hatching, larvae recruit from the oceanic environment to demersal habitats (at a size of about 32 millimeters total length). Juveniles inhabit macroalgae, coral clumps, and seagrass beds, and are relatively solitary. As they grow, they occupy progressively deeper areas and offshore reefs, and can be found in schools of up to forty individuals. When not spawning, adults are most commonly found in waters less than 100 meters deep. Nassau grouper diet changes with age. Juveniles eat plankton, pteropods, amphipods, and copepods. Adults are unspecialized piscivores, bottom-dwelling ambush suction predators (NMFS 2013d).

Male and female Nassau groupers reach sexual maturity at lengths between 40 and 45 centimeters standard length, about 4 to 5 years old. It is thought that sexual maturity is more determined by size, rather than age. Otolith studies indicate that the minimum age at maturity is between 4 and 8 years; most groupers have spawned by age 7 (Bush et al. 2006). Nassau groupers live to a maximum of 29 years.

Population Dynamics

Abundance. There is no range-wide abundance estimate available for Nassau grouper. The species is characterized as having patchy abundance due largely to differences in habitat availability or quality, and differences in fishing pressure in different locations (81 FR 42268). Although abundance has been reduced compared to historical levels, spawning still occurs and abundance is increasing in some locations, such as the Cayman Islands and Bermuda.

Productivity / Population Growth Rate. There is no population growth rate available for Nassau grouper. Available information from observations of spawning aggregations has shown steep declines (Aguilar-Perera 2006; Claro and Lindeman 2003; Sala et al. 2001); however, some aggregation sites are comparatively robust and show signs of increase (Vo et al. 2014; Whaylen et al. 2004).

Genetic Diversity. Recent studies on Nassau grouper genetic variation has found strong genetic differentiation across the Caribbean subpopulations, likely due to barriers created by ocean currents and larval behavior (Jackson et al. 2014a).

Distribution. Nassau grouper occur in Bermuda and Florida, throughout the Bahamas and the Caribbean Sea. Their distribution within Florida is from Cape Canaveral south through the Florida Keys and Florida Bay westward to the Dry Tortugas and Pulley Ridge. They are considered rare in the Gulf of Mexico and are fairly uncommon in Florida, although visual surveys have documented higher densities of Nassau groupers at Riley's Hump within Tortugas South Ecological Reserve as compared to the rest of Florida.

Designated Critical Habitat. NOAA fisheries designated critical habitat (89 FR 126) for the threatened Nassau grouper pursuant to section 4 of the Endangered Species Act (ESA). Specific occupied areas designated as critical habitat contain approximately 2,384.67 sq. kilometers (km) (920.73 sq. miles) of aquatic habitat located in waters off the coasts of southeastern Florida, Puerto Rico, Navassa, and the United States Virgin Islands.

Recovery Goals. NMFS has not prepared a recovery plan for the Nassau grouper. NOAA Fisheries has developed a recovery outline (PDF, 8 pages) to serve as an interim guidance document to direct recovery efforts, including recovery planning, for Nassau grouper until a full recovery plan is developed and approved. The recovery outline presents a preliminary strategy for recovery of the species and recommends high priority actions to stabilize and recover the species. The major actions recommended in the outline include:

- Building awareness and a constituency for conservation of Nassau grouper spawning aggregations through outreach and education highlighting the importance of preserving reproductive output for future fishery
- Ensuring consistent regulations across the region during the spawning period
- Trade assistance both during and after the aggregation period to ensure only legally caught fish are marketed
- Decreasing fishing pressure through increased enforcement of existing regulations

8.43 Smalltooth Sawfish, United States DPS

Table 91. Smalltooth Sawfish, United States DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Pristis pectinata	Sawfish, smalltooth	US portion of range	Endangered	2018	2003 68 FR 15674	2009	2009 74 FR 4 5353

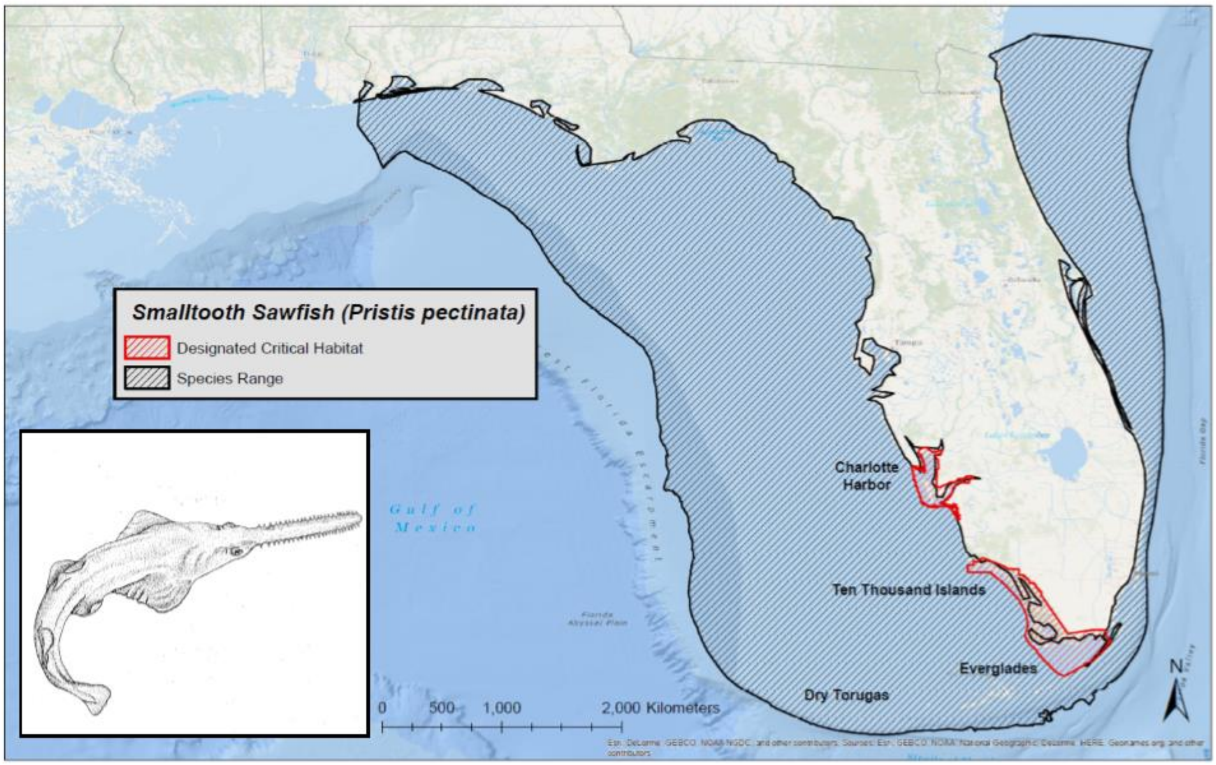


Figure 53. Smalltooth Sawfish, United States DPS range and designated critical habitat

Species Description. The smalltooth sawfish (*Pristis pectinata*) is a tropical marine and estuarine elasmobranch. Although they are rays, sawfish physically resemble sharks, with only the trunk and especially the head ventrally flattened. Smalltooth sawfish are characterized by their "saw," a long, narrow, flattened rostral blade with a series of transverse teeth along either edge (NMFS 2009c). The U.S. DPS of smalltooth sawfish was listed as endangered under the ESA effective May 1, 2003 (68 FR 15674). Although this species is reported to have a circumtropical distribution, NMFS identified smalltooth sawfish from the Southeast United States as a DPS. Within the United States, smalltooth sawfish have been captured in estuarine and coastal waters from New York southward through Texas, although peninsular Florida has historically been the region of the United States with the largest number of recorded captures (NMFS 2010b).

Status. The decline in the abundance of smalltooth sawfish has been attributed to fishing (primarily commercial and recreational bycatch), habitat modification (including changes to freshwater flow regimes as a result of climate change), and life history characteristics (i.e., slow-growing, relatively late-maturing, and long-lived species) (NMFS 2009c; Simpfendorfer et al. 2011). These factors continue to threaten the smalltooth sawfish population. Recent records indicate there is a resident reproducing population of smalltooth sawfish in south and southwest Florida from Charlotte Harbor through the Dry Tortugas, which is also the last U.S. stronghold for the species (Poulakis and Seitz 2004; Seitz and Poulakis 2002; Simpfendorfer and Wiley 2004). While the overall abundance appears to be stable, low intrinsic rates of population increase suggest that the species is particularly vulnerable to rapid population declines (NMFS 2010b).

Life History. Smalltooth sawfish size at sexual maturity has been reported as 360cm total length (TL) by Simpfendorfer (2005). Carlson and Simpfendorfer (2015) estimated that sexual maturity for females occurs between 7 and 11 years of age. As in all elasmobranchs, smalltooth sawfish are viviparous; fertilization is internal. The gestation period for smalltooth sawfish is estimated at 5 months based on data from the largetooth sawfish (Thorson 1976). Females move into shallow estuarine and nearshore nursery areas to give birth to live young between November and July, with peak parturition occurring between April and May (Poulakis et al. 2011). Litter sizes range between 10 and 20 individuals (Bigalow and Schroeder 1953; Carlson and Simpfendorfer 2015; Simpfendorfer 2005).

Neonate smalltooth sawfish are born measuring 67 – 81 cm (TL) and spend the majority of their time in the shallow nearshore edges of sand and mud banks (Poulakis et al. 2011; Simpfendorfer et al. 2010). Once individuals reach 100 – 140cm (TL) they begin to expand their foraging range. Capture data suggests smalltooth sawfish in this size class may move throughout rivers and estuaries within a salinity range of 18 and 30 (practical salinity units). Individuals in this size class also appear to have the highest affinity to mangrove habitat (Simpfendorfer et al. 2011). Juvenile sawfish spend the first 2-3 years of their lives in the shallow waters provided in the lower reaches of rivers, estuaries, and coastal bays (Simpfendorfer et al. 2008; Simpfendorfer et al. 2011). As smalltooth sawfish approach 250 cm (TL) they become less sensitive to salinity changes and begin to move out of the protected shallow-water embayments and into the shorelines of barrier islands (Poulakis et al. 2011). Adult sawfish typically occur in more open-water, marine habitats (Poulakis and Seitz 2004).

Population Dynamics

Abundance. The abundance of smalltooth sawfish in U.S. waters has decreased dramatically over the past century. Efforts are currently underway to provide better estimates of smalltooth sawfish abundance (NMFS 2014b). Current abundance estimates are based on encounter data, genetic sampling, and geographic extent. Carlson and Simpfendorfer (2015) used encounter densities to estimate the female population size to be 600. Chapman et al. (2011) analyzed genetic data from tissue samples (fin clips) to estimate the effective genetic population size as 250-350 adults (95% C.I. 142-955). Simpfendorfer (2002) estimated that the U.S. population may number less than 5% of historic levels based on the contraction of the species' range.

Productivity / Population Growth Rate. The abundance of juveniles encountered in recent studies (Poulakis et al. 2014; Seitz and Poulakis 2002; Simpfendorfer and Wiley 2004) suggests that the smalltooth sawfish population remains reproductively viable. The overall abundance appears to be stable (Wiley and Simpfendorfer 2010). Data analyzed from the Everglades portion of the smalltooth sawfish range suggests that the population growth rate for that region may be around 5% per year (Carlson and Osborne 2012; Carlson et al. 2007). Intrinsic rates of growth (λ) for smalltooth sawfish have been estimated at 1.08-1.14 per year and 1.237-1.150 per year by Simpfendorfer (2000) and Carlson and Simpfendorfer (2015) respectively. However, these intrinsic rates are uncertain due to the lack of long-term abundance data.

Genetic Diversity. Chapman et al. (2011) investigated the genetic diversity within the smalltooth sawfish population. The study reported that the remnant population exhibits high genetic diversity (allelic richness, alleles per locus, heterozygosity) and that inbreeding is rare. The study also suggested that the protected population will likely retain >90% of its current genetic diversity over the next century.

Distribution. Recent capture and encounter data suggests that the current distribution is focused primarily to south and southwest Florida from Charlotte Harbor through the Dry Tortugas (Poulakis and Seitz 2004; Seitz and Poulakis 2002). Water temperatures (no lower than 16-18°C) and the availability of appropriate coastal habitat (shallow, euryhaline waters and red mangroves) are the major environmental constraints limiting the distribution of smalltooth sawfish (Bigalow and Schroeder 1953).

Designated Critical Habitat. Critical habitat for smalltooth sawfish was designated in 2009 (74 FR 45353) and includes 2 major units: Charlotte Harbor (221,459 acres) and Ten Thousand Islands/Everglades (619,013 acres). These 2 units include essential sawfish nursery areas. The locations of nursery areas were determined by analyzing juvenile smalltooth sawfish encounter data in the context of shark nursery criteria (Heupel et al. 2007; Norton et al. 2012). Within the nursery areas, 2 features were identified as essential to the conservation of the species: red mangroves (*Rhizophora mangle*), and euryhaline habitats with water depths ≤0.9 meters (74 FR 45353). The Charlotte Harbor unit includes areas which are moderate to highly developed (Cape Coral, Fort Myers) and includes a highly altered, flow-managed system (Caloosahatchee River). In contrast, the Ten Thousand Island/Everglades unit contains relatively undeveloped, pristine smalltooth sawfish habitat (Poulakis et al. 2014; Poulakis et al. 2011).

Recovery Goals. The 2009 Smalltooth Sawfish Recovery Plan (NMFS 2009c) contains complete downlisting/delisting criteria for each of the 3 following recovery goals.
Minimize human interactions and associated injury and mortality. Specific criteria include:

- Educational programs;
- Handling and release guidelines;
- Injury and mortality regulations; and,
- Other State and/or Federal measures (not including those provided under the ESA).
- Protect and/or restore smalltooth sawfish habitats. Specific criteria include:
- protection of existing mangrove shoreline habitat;

- assurance of availability and accessibility of both mangrove and non-mangrove habitat sufficient to support subpopulations of juvenile sawfish;
- appropriate freshwater flow regimes; and,
- identification and protection of habitat areas utilized by adult smalltooth sawfish.
- Ensure smalltooth sawfish abundance increases substantially and the species reoccupies areas from which it had been previously extirpated. Specific criteria include:
- annual increases in the relative abundance of juvenile smalltooth sawfish;
- annual increases in the relative abundance of adult smalltooth sawfish;
- verified records of adult smalltooth sawfish in outer regions of the species range.

8.44 Giant Manta Ray

Table 92. Giant Manta Ray; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Manta birostris	Giant Manta Ray	None	Threatened	2017	2018 83 FR 2916	2019 (outline)	Not Designated

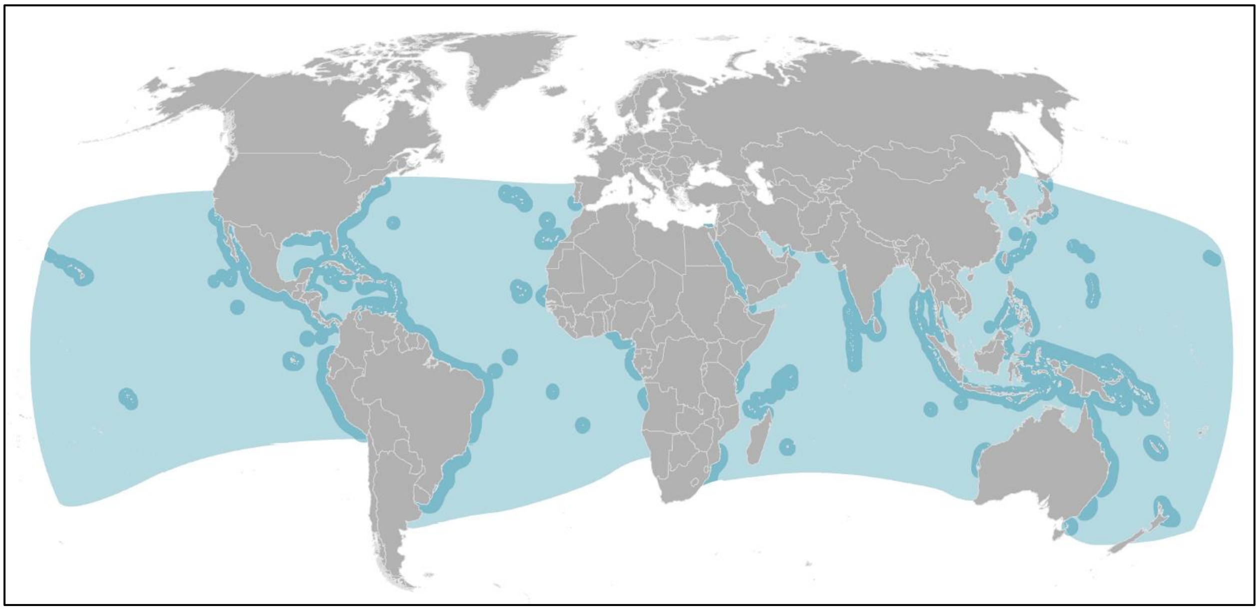


Figure 54**. Giant Manta Ray distribution. Source: Lawson et al 2016, as cited in the 2017 Status Review Report (Miller et al. 2017).**

Species Description. *Manta birostris*, the giant manta ray, is found worldwide in tropical, subtropical, and temperate bodies of water. It is commonly found offshore, in oceanic waters, and near productive coastlines. The giant manta ray is considered to be a migratory species, with estimated distances travelled of up to 1,500 km. The giant manta ray has a diamond-shaped body with wing-like pectoral fins. There are 2 distinct color types: chevron and black. Most of the chevron variants have a black dorsal surface and a white ventral surface with distinct patterns on the underside that can be used to identify individuals. The giant manta ray was listed as threatened under the ESA on January 22, 2018 (83 FR 2916).

Status. Although there is considerable uncertainty regarding historical and current abundances, the best available data indicate that the giant manta ray has experienced significant declines and continues to decline, particularly in the Indo- and eastern Pacific portions of its range. Specific country and area data are summarized in the status review report (Miller and Klimovich 2017) and suggest localized declines of 71% to 95% with possible extirpations in some areas. Yet, larger subpopulations of the species still exist, including off Mozambique, Ecuador, and Thailand. However, giant manta rays are a migratory species and continue to face fishing pressure, particularly from the industrial purse-seine fisheries and artisanal gillnet fisheries operating within the Indo-Pacific and eastern Pacific portions of its range. This significant threat, coupled with the species' low reproductive output and overall productivity may severely limit its ability for compensation and recovery.

Life History. The giant manta ray is a migratory species and seasonal visitor along productive coastlines with regular upwelling, in oceanic island groups, and near offshore pinnacles and seamounts. The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. Although the giant manta ray tends to be solitary, they aggregate at cleaning sites and to feed and mate. Manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae, and shrimp, but some studies have noted their consumption of small and moderately sized fish as well. Giant manta rays also appear to exhibit a high degree of plasticity or variation in terms of their use of depths within their habitat. During feeding, giant manta rays may be found aggregating in shallow waters at depths less than 10 meters. However, tagging studies have also shown that the species conducts dives of up to 200 to 450 meters and is capable of diving to depths exceeding 1,000 meters. Manta rays have among the lowest fecundity of all elasmobranchs (a subclass of cartilaginous fish), typically giving birth to only 1 pup every 2–3 years. Gestation is thought to last around a year. Although manta rays have been reported to live at least 40 years, not much is known about their growth and development.

Population Dynamics

Abundance. There are no current or historical estimates of the global abundance of *M. birostris*, with most estimates of subpopulations based on anecdotal diver or fisherman observations, which are subject to bias. These populations potentially range from around 100-1,500 individuals.

Productivity / Population Growth Rate. Using estimates of known life history parameters for both giant and reef manta rays, and plausible range estimates for the unknown life history parameters, Dulvy et al. (2014) calculated a maximum population growth rate of Manta spp. and found it to be one of the lowest values when compared to 106 other shark and ray species. The current net productivity of *M. birostris* is unknown due to the imprecision or lack of available abundance estimates or indices.

Genetic Diversity. The low abundance of populations may be at levels that place them at increased risk of genetic drift and potentially at more immediate risks of inbreeding depression and demographic stochasticity.

Distribution. The giant manta ray is found worldwide in tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and in productive coastal areas. The species has also been observed in estuarine waters, oceanic inlets, and within bays and intercoastal waterways. As such, giant manta rays can be found in cool water, as low as 19°C, although temperature preference appears to vary by region. For example, off the U.S. East Coast, giant manta rays are commonly found in waters from 19 to 22°C, whereas those off the Yucatan peninsula and Indonesia are commonly found in waters between 25 to 30°C.

Designated Critical Habitat. None designated.

Recovery Goals. A recovery plan has not yet been finalized for this species (see 2019 recovery plan outline). However, in advance of an approved recovery plan, the initial focus of the interim recovery program will be two-fold:

- To stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and
- To gather additional information through research and monitoring on the species' current distribution, abundance, movement, habitat, mortality rates, and other potential threats.

Because the major threat currently contributing to the species' decline is overutilization in waters outside of U.S. jurisdiction, international coordination will be critical to ensuring recovery of the species.

8.45 Black Abalone

Table 93. Black Abalone; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Haliotis cracherodii	Abalone, Black	N/A	Endangered	2009	2009 74 FR 1 937	2020	2011 76 FR 66806

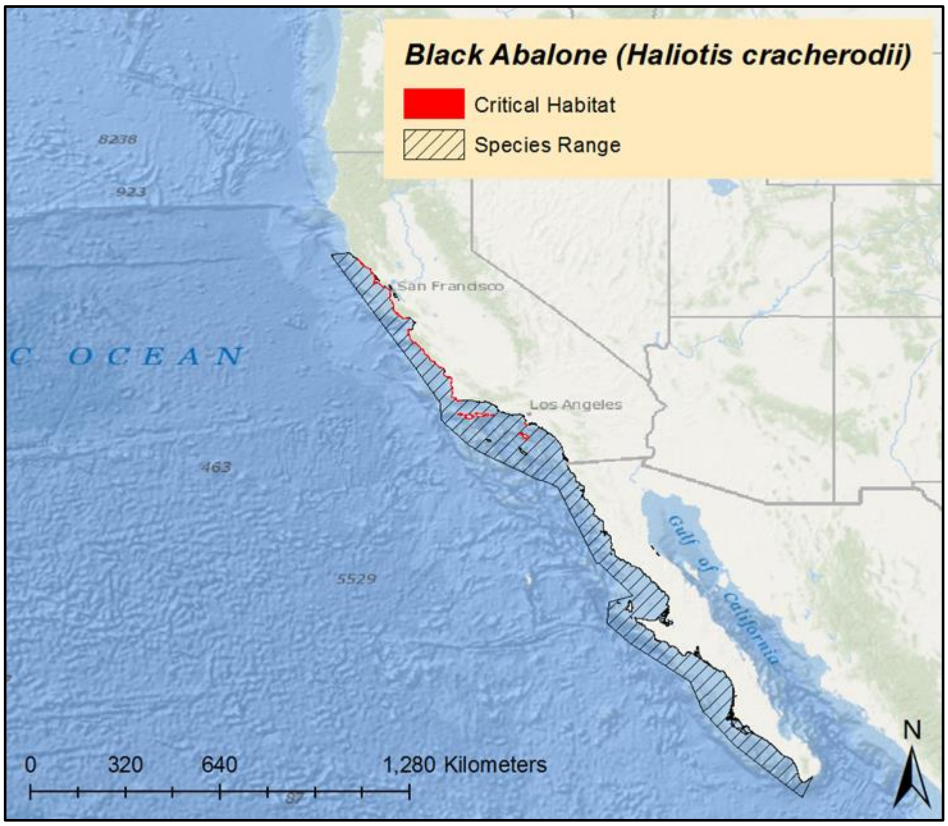


Figure 55. Black Abalone range and designated critical habitat

Species Description. Black abalone is a large marine gastropod mollusk belonging to the taxonomic family of Haliotidae, a group of sea snails with convex spiral structured shells. The majority of experts concur that the present range of black abalone extends from Point Arena (Mendocino County, California) to Northern Baja California. The black abalone is a moderately large aquatic gastropod mollusk found along rocky shorelines and coastal habitats. The majority of experts concur that the present range of black abalone extends from Point Arena (Mendocino County, California) to Northern Baja California. Black abalone are uncommon north of San Francisco (Morris et al. 1980) and south of Punta Eugenia (P.Raimondi, pers. comm. as cited in Butler et al. 2009) Black abalone, as with all abalone are benthic, occurring on hard substrata, relatively stationary, and are for the most part herbivorous, feeding on attached or floating algal material (Geiger 1999). The mollusk possesses a shell that is smooth, circular, and black to slate blue in colors (Leach 1814 as cited in Butler et al. 2009). There are 5-9 open respiratory pores, known as tremata, that are level with the shell's outer surface. Normally the shell's interior is white (Haaker et al. 1986), with ill-defined or no muscle scar (Howorth 1978). The muscular foot of the black abalone permits the animal to firmly fasten itself to rocky surfaces without being displaced by wave action. A rolling motion of the foot completes movement for the species as a column of muscle attaches the body to the shell. The epipodium, a sensory structure and extension of the foot which holds lobed tentacles of the same color (Cox 1962), circles the foot and extends beyond the shell of a healthy black abalone. The internal organs are arranged around the foot and under the shell.

Status. Black abalone has experienced substantial decline, which is reflected by the decrease in commercial catches until 1993, when commercial harvests were halted. Historic levels approached 2,200 tons in California in 1879 and declined to around 1,000 tons in the 1970's. Commercial landings then decreased to 19.1 tons in the last year of harvests, when mortality from withering syndrome devastated remaining black abalone stocks throughout southern California (Haaker 1994). Over 20 years, densities of more than 100 individuals per cubic yard disappeared from most of their former range south of Point Conception (Davis 1993). A similar mass mortality was reported at Palos Verdes Peninsula in the late 1950's, where average density decreased from more than 2.8 individuals per square yard from 1975 to 1979 down to about 0.03 individuals per square yard from 1987 to 1991 (Cox 1962). Island habitats experienced more severe trends; 99% of black abalone vanished from Anacapa, Santa Barbara, and Santa Rosa Islands in less than 5 years (Haaker et al. 1989; Richards and Davis 1993).

Black abalone have also experienced severe declines due to a temperature-related disease called withering syndrome. This bacteria-based disease prevents assimilation of nutrients in the digestive system and results in abalone that "wither" as individuals consume body tissues. The disease was first identified west of Santa Cruz and Anacapa islands in 1985 and 1986 before spreading to Santa Rosa Island and Santa Barbara Island by 1988. The disease made its appearance along the mainland in 1988 in San Luis Obispo county, where 85% of the resident black abalone died in Diablo Cove. This die-off was attributed to the presence of warm-water effluent from a nuclear power facility. From 1988 to the early 1990's, withering syndrome continued to spread throughout the Channel islands to 2000, when it was estimated that only 1% of the original population remained (Richards 2000).

However, as previously expressed, signs of possible recovery can be seen in recent Channel Island surveys that have demonstrated growth in juvenile abundance (Eckdahl 2015). Nonetheless, issues stemming from previous declines are very much ostensible on several mainland and island sites from Point Conception to San Diego, which show density estimates to be well below the minimum needed for successful recruitment of the species to occur. Densities at island sites ranged from 0.06-0.64/m^2 and mainland sites ranged from 0-0.01/m^2 whereas the estimated minimum density needed is between 0.75-1.1/m^2 (Eckdahl 2015).

Life History. Black abalone have separate sexes and are broadcast spawners. As spawning occurs, gametes are dispersed from the gonads of both parents into the sea and fertilization is entirely external. The embryos and larvae that result from this process are small and unprotected, obtain no parental care or safeguard of any kind, and are exposed to a wide range of physical and biological sources of mortality. The average life expectancy for an abalone that reaches adulthood is 30 years. Adults attain a maximum shell length of approximately 200 millimeters (indexed by linear measure of the maximum diameter of the elliptical shell). Female black abalone become sexually mature at a length of about 50 millimeters, and males at about 40 millimeters (Ault 1985). Ault (1985) projected that sexually mature female black abalone may discharge over 2 million unfertilized eggs per spawning episode and are capable of undergoing multiple episodes each spawning season. Black abalone spawning season is between April and September with peak times occurring during the late summer and early autumn (Leighton 2005 as cited in Butler et al. 2009). Black abalone are most commonly observed in the mid to low intertidal, in complex habitats with deep crevices that provide shelter for juvenile recruitment and adult survival.

Population Dynamics

Abundance. Using landings data obtained from the height of the black abalone commercial and recreational fisheries era (1972-1981), Rogers-Bennett et al. (2002) estimated baseline abundance of the species to be approximately 3.54 million animals. Due to significant declines throughout the 20th century as a result of overfishing, habitat loss, and disease (most notably withering syndrome), the abundance of black abalone currently stands as small fraction of historical numbers. Through the analysis of both fishery and fishery-independent long-term monitoring data, identification of substantial declines of black abalone throughout central and southern California have been made. Neuman et al. (2010) states that overall rates of decline exceed 95% for populations of black abalone south of Monterrey County, CA. Recent NOAA surveys off the shores of the South Farallon Islands (coastal islands located 30 miles west of San Francisco) show no current presence of black abalone (Roletto 2015). However, recent surveys on the Channel Islands have shown an increase in juvenile abundance, which may deem positive for recruitment (Eckdahl 2015). Nevertheless, in a recent 2015 survey that explored several mainland and island sites from Point Conception to San Diego, density estimates were well below the minimum density needed for successful recruitment of the species to occur.

Productivity / Population Growth Rate. As stated, population growth rates for black abalone have experienced steep declines since the late 1970s. Butler et al. (2009) states due to the large declines of the species south of Monterrey County, CA it is doubtful that black abalone populations will be able to recover naturally to their former abundance levels, at least in the near future. Furthermore, due the persistent decline of most populations and the continued northward expansion of withering syndrome as a result of warming events (Raimondi et al. 2002), it seems likely that black abalone populations will continue to decline on a large scale.

Genetic Diversity. Neuman et al. (2010) states that black abalone populations exhibit a heterogeneous genetic structure among populations and it is possible that localized genetic diversity has been lost in areas where populations have declined to extremely low abundance levels, rendering extant populations less capable of dealing with both long- and short- term environmental or anthropogenic challenges.

Distribution. As stated in the description of the species, black abalone is found off the Western Coast of the United States from Point Arena (Mendocino County, California) to Northern Baja California. Inside this broad geographic range, black abalone mostly inhabits coastal and offshore island intertidal habitats on uncovered rough shores where bedrock offers profound, protective crevice shelter (Leighton 2005 as cited in Butler et al. 2009). Compared to other native species of abalone found along California and its coastal islands, black abalone bathymetrically inhabits shallower locations situated predominantly in rocky intertidal environments (Morris et al. 1980). Bathymetry distribution for black abalone ranges from the high intertidal zone (i.e., shoreline) to 6 meters depth, with most animals found in middle and lower intertidal

Designated Critical Habitat. On October 27, 2011, the NMFS designated critical habitat for black abalone. This includes rocky areas from mean high water to 6 meters water depth in the Farallon, Channel, and Año Nuevo islands, as well as the California coastline from Del Mar Ecological Reserve south to Government Point (excluding some stretches, such as in Monterey Bay and between Cayucos and Montaña de Oros State Park) in northern and central California and between the Palos Verdes and Torrance border south to Los Angeles Harbor. PBFs considered essential for the conservation of black abalone are:

- Rocky substrate: Rocky benches, crevices, large boulders
- Food resources: Bacterial and diatom films, algae
- Juvenile settlement habitat: Rocky habitat with coralline algae and/or crevices, cryptic biogenic structures
- Suitable water quality
- Suitable nearshore circulation patterns

Recovery Goals. See the 2020 Recovery Plan (NMFS 2020b) for complete details on recovery goals and criteria. The following are the 2 general recovery objectives: 1) Increase the abundance, productivity, spatial structure, and diversity of black abalone populations to levels that support the species' long-term survival, viability, and resilience to threats; and 2) Sufficiently address the threats of concern.

8.46 White Abalone

Table 94. White Abalone; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Haliotis sorenseni	Abalone, White	N/A	Endangered	2018	2001 66 FR 29046	2008	None Designated

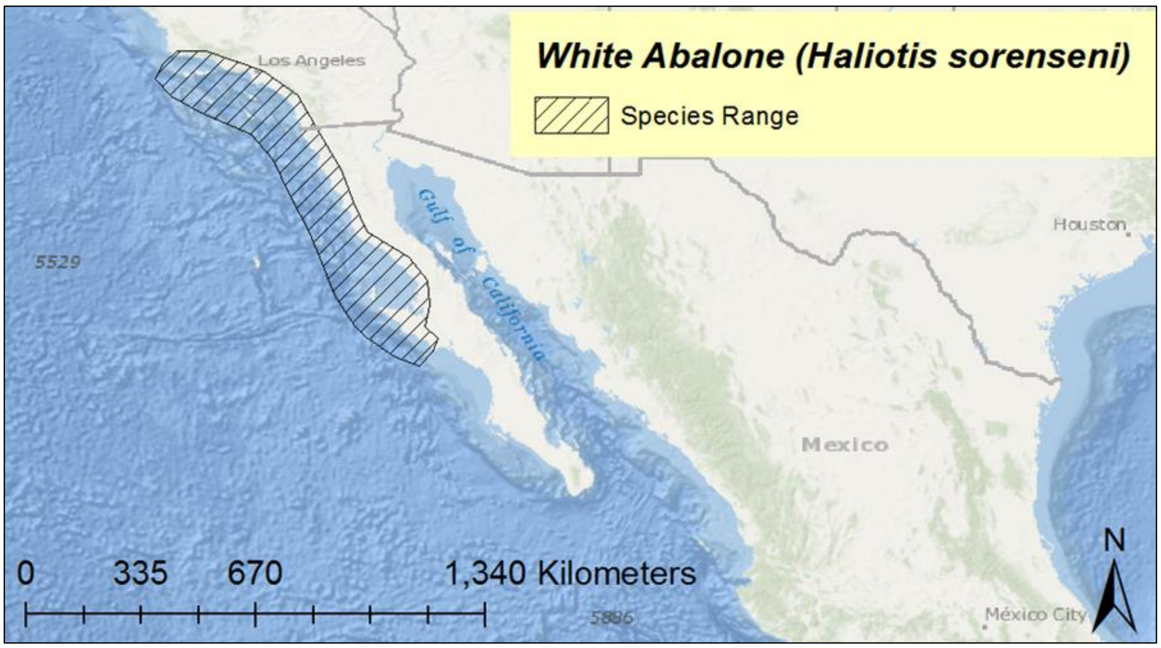


Figure 56. White Abalone range

Species Description. The white abalone is an herbivorous gastropod found in shallow rocky ocean waters. White Abalone occurs between Punta Abreojos, Baja California, Mexico and Point Conception, California, USA. White abalone occupies open low relief rock or boulder habitat surrounded by sand (Tutschulte 1976). Historically, white abalone were reported to occur at depths of 20–60 meters with the greatest abundance occurring between 25–30 meters (Cox 1960; Tutschulte 1976). However, later surveys show that they occur from 30 to 65 meters with a median depth of 48 meters (Haaker et al. 1986). Maximum shell length reached by white abalone in California is about 20-25 centimeters while in Mexico the species will only grow to 17 centimeters (Hobday and Tegner 2000). Adults have a speckled orange and tan epipodium with foliose epipodial papillae and brown cephalic tentacles. The epipodium, a sensory structure and extension of the foot which holds lobed tentacles of the same color (Cox 1962), circles the foot and extends beyond the shell of a healthy white abalone. The internal organs are organized around the foot and under the shell.

Status. On May 29, 2001, the white abalone was listed as an endangered species throughout its range under the ESA (66 FR 29046). White abalone numbers were severely reduced due to excessive harvest. This has led to below-threshold spawning densities in many areas that are blamed for recruitment failure and an inability of the species to recover. Estimates of population size have been difficult to calculate because estimates are only based upon adults, as juveniles are infrequently observed. White abalone observed during surveys were of large size which corresponds to predicted ages near the end of the anticipated life span (Davis 1996; Davis et al. 1998; Hobday and Tegner 2000; Hobday et al. 2001). Because no white abalone were observed in the smaller age/size classes during the surveys there appears to be a lack of successful recruitment since the 1960s (Hobday and Tegner 2000). Recent surveys off the southern coast of California illustrate this continued trend (Catton et al. 2016).

Life History. Recent evidence from bomb carbon research indicates that the life span for white abalone is roughly 28 to 30 years(Rogers-Bennett et al. 2016). Abalone aggregate for spawning, but low numbers and physical barriers can prevent large spawning aggregations from forming (Babcock and Keesing 1999; Leet et al. 2001). A brief annual spawning event occurs *en mass* generally between February and April (Tutschulte 1976). Although an average female is capable of producing over 20 million larvae over her lifetime, larval survival to adulthood is estimated at less than 1% (Leighton 2000). Twenty-four hours after fertilization, a free-swimming larva emerges from the fertilized egg and joins the plankton (Leighton 1989; Leighton 2000). After 2–3 weeks in the plankton, the larvae settle to the bottom. One to 3 months after settlement juveniles are fully formed and resemble adults. After 2–4 years, white abalone are mature and inhabit the tops and sides of rocky substrates. (Saunders et al. 2009a; Saunders et al. 2009b).

Population Dynamics

Abundance. The current total population of white abalone in California is estimated to be in the thousands and declining by an estimated 12% annually (Catton et al. 2016).

Productivity / Population Growth Rate. From 2002 to the present, population surveys of white abalone in southern California show declining densities (Catton et al. 2016). From 2002 to 2014, survey results at Tanner Bank, California showed population growth rates had a 12% mean decline in abundance(Catton et al. 2016). This decline was hinted at in the 2000 status review in which Hobday and Tegner (2000) cautioned that due to the prevalence of older individuals within white abalone populations at the time, the populace would vanish due to natural mortality without human intercession. This resulted in the 2001 creation of a population rebuilding strategy for white abalone, which identified hatchery production and stocking of cultured white abalone as the primary restoration action recommended. The California and federal recovery plans both promote restoration of white abalone populations through captive-rearing and stocking efforts, which have significantly increased captive-bred populations (Rogers-Bennett et al. 2016).

Genetic Diversity / Distribution. In reference to distribution, white abalone occur along the U.S. west coast among offshore islands and banks (particularly Santa Catalina and San Clemente islands) and mainland inshore waters from Point Conception, California south to Punta Abreojos, Baja California, Mexico (Cox 1960; Cox 1962; Bartsch 1940). White abalone occur primarily along the mainland coast in their northern and southern range, but are more frequently at the offshore islands (especially San Clemente and Santa Catalina islands) in the middle portion of the California range (Cox 1962; Leighton 1972). However, individuals have also been found around several Mexican islands including Isla Cedros and Isla Natividad (Guzmán Del Proó 1992). There are no recognized subspecies of white abalone although there is 1 possible subspecies of white abalone inhabiting Guadalupe Island, Mexico (Hobday and Tegner 2000). Nevertheless, recent commercial fisheries data has shown that white abalone along the Mexican coast are believed to be depleted, but their status is generally unknown (NMFS 2008b)

Designated Critical Habitat. Critical habitat has not been designated for white abalone.

Recovery Goals. The following contains requirements needed for downlisting/delisting white abalone:

- The density of emergent (detectable by human observation without substrate disturbance) animals (short term) is greater than 2,000 per hectare for 75% of the geographic localities;
- A total of 380,000 animals are maintained in the wild, distributed among all geographic localities in the USA and Mexico;
- The proportion of size of emergent animals in 75% of geographic localities includes at least 85% intermediate-size animals (90 to 130 millimeters);
- Proportion of size of emergent animals in 75% of geographic localities includes no more than 15% large animals (>130 millimeters);
- There is a stable or increasing estimate of geometric population growth (lambda ≥1) for >75% of the geographic localities over a 10 year period; and
- There is reoccupation of white abalone over a spatial scale that encompasses their historic range such that 75% of the geographic localities in the USA and Mexico are reoccupied and meet the recovery criteria.

8.47 Sunflower Sea Star

Table 95 Sunflower Sea Star; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Pycnopodia helianthoides	Sunflower Sea Star	All	Proposed Threatened	2023	2023 88 FR 16212 (Proposed)	NA	NA

Figure 57. Six potentially significant portions of the range of the Sunflower Sea Star identified in the 2022 Status Review Report (Lowry et al. 2022).

Species Description Potentially significant portions of the range of the Sunflower Sea Star are shown in Figure 57. The sunflower sea star, or *Pycnopodia helianthoides*, is an Echinoderm and is among the largest sea stars in the world, reaching over 1 m in total diameter from ray tip to ray tip across the central disk (Figure 58). *P. helianthoides* is characterized by having 15-24 rays. Its closest relative, *Lysastrosoma anthosticta*, has only 5 rays and reaches a much smaller maximum size. Very young *P. helianthoides* generally have less than a dozen rays (Figure 59), and additional rays are added by budding in symmetrical pairs as the individual grows.

Figure 58. Adult *Pycnopodia helianthoides*

Other sea stars in the northern Pacific Ocean with many rays include several sun stars of the genera *Solaster*, *Crossaster*, and *Rathbunaster*, but these species generally have 8–17 rays, as opposed to the 20 or more rays commonly found in *P. helianthoides*, and all of the sun stars are considerably smaller and less massive than *Pycnopodia*. In describing *P. helianthoides*, (Fisher 1928) said: “When under "full sail," with its thousands of tube-feet lashing back and forth, it is an impressive animal, and its numerous cushions of tenacious pedicellariae and the wide expanse of its flexible body make it a formidable engine of destruction.” Based on long-standing taxonomic analysis, and lacking any evidence to the contrary, the 2022 Status Review Report (Lowry 2022), had determined that *Pycnopodia helianthoides* represents a unique species as defined by Section 3 of the ESA.

Figure 59. Juvenile *Pycnopodia helianthoides*

Status. In 2020, the International Union for Conservation of Nature (IUCN) conducted the first ever status assessment for the sunflower sea star throughout its range (Gravem, Heady et al. 2021, Lowry 2022, Lowry, Wright et al. 2022). Estimates of population size were based on mean density in various regions and the availability of habitat. Data sources included a variety of stand-alone ecosystem monitoring efforts that regularly encounter sunflower sea stars, many of them SCUBA-based, with several geographic regions having sparse spatiotemporal coverage. The IUCN assessment concluded that the status of the sunflower sea star on a range-wide basis was critically endangered, citing a > 90% loss in overall abundance since 2013, largely as a direct consequence of the sea star wasting syndrome (SSWS) pandemic (Gravem, Heady et al. 2021). Additionally, Hamilton et al.(2021) used logistic models (general linear model with binomial errors and logit links) and presence-absence data to estimate the timing and extent of the decline in occurrence among the 12 regions used in the IUSC assessment. Range-wide occurrence declined by 52.3%, with more severe declines of 92.2% in occurrence from Oregon southward to Mexico. Where density data were available, Hamilton et al. (2021) also used zero-inflated generalized linear models (with Poisson errors and log-link) to estimate the change in density among regions and between phases (pre- and post-SSWS). Density declined by 99.2% from Baja California to the Washington coast, while declines were slightly lower (but greater than 87.8%) in regions from British Columbia through the Aleutian Islands.

Prior to 2013, the global abundance of sunflower sea stars was estimated at several billion animals, but from 2013-17 SSWS reached pandemic levels, killing an estimated 90%+ of the population. Impacts varied by region across the range of the species and generally progressed from south to north, though a notable delay occurred off Oregon for unknown reasons. By 2017, the sunflower sea star was rare south of Cape Flattery, WA, in areas where it had long been a conspicuous and ecologically important component of benthic marine ecosystems. Declines in coastal British Columbia and the Aleutian Islands were less pronounced, but still exceeded at least 60%, and more likely 80%. While the root cause of SSWS has not yet been identified, dozens of independent monitoring efforts using SCUBA, benthic trawls, and shellfish pots have documented similar declines in abundance, and sometimes in their spatial distribution, without subsequent recovery. Environmental factors such as temperature and dissolved oxygen likely contributed to the pandemic, and continue to interact with the disease agent to suppress recovery, but studies have failed to document conclusive linkages that apply on broad scales. Complex interactions among stressors, some of which have become more intense as a consequence of anthropogenic climate change, affect both the persistence of individuals and local populations.

Life-History. Little is known about several fundamental biological aspects of the life history and demography of the sunflower sea star. Parameters such as age/size at first maturity, fecundity, longevity, reproductive life span, and individual growth rate have not been validated. Furthermore, variation in these parameters over time and space, including any systematic differences among regions or habitats, have not been described. While regional asynchrony in SSWS impacts was observed during the pandemic, the degree to which this pattern aligns with population-level differences in genetics, population growth rate, disease susceptibility, and other factors is unknown. Any such relationship is also confounded by the fact that occurrence and abundance data have been collected independently using an array of methods over different timespans and seasons, resulting in a patchwork of information that is incomplete and not intended to provide information on these parameters.

Typically, sea stars with planktotrophic larval development from the temperate nearshore Northwest Pacific Ocean spawn in late winter or early spring, which serves to provide the best growing conditions for their offspring by synchronizing the presence of their obligate plankton-feeding larvae with the peak of the spring phytoplankton bloom (Menge 1975, Strathmann 1987). The spawning seasons of several other asteriid sea stars with planktotrophic larval development in the Pacific Northwest and on the U.S. West Coast, including *Pisaster ochraceus* (Brandt), *Pisaster brevispinus* (Stimpson), *Pisaster giganteus* (Stimpson), *Evasterias troschelii* (Stimpson), and *Orthasterias koehleri* (deLoriol), occurs between March and August ((Mortensen and Mortensen 1921, Farmanfarmaian, Giese et al. 1958, Mauzey 1966, Fraser, Gomez et al. 1981, Pearse and Eernisse 1982, Strathmann 1987, Pearse, McClary et al. 1988, Sanford and Menge 2007). Planktotrophic larvae of the sunflower sea star developing during winter (November to February) in the Northeast Pacific Ocean would be at a distinct disadvantage due to the scarcity of planktonic algae at that time of year. Sea stars may modify their behavior during spawning in ways that improve the chances of egg fertilization, including aggregating, modifying their positions and postures, and spawning synchronously (Strathmann 1987, Chia 1991, Dams, Blenkinsopp et al. 2018). However, it is uncertain whether the sunflower sea star does so. Kjerskog-Agersborg (1918) studied sunflower sea star in Puget Sound at Bremerton, Washington, and suggested that individuals migrated to shallower waters during the spawning season and were present in large aggregations at this time of year.(Kjerskog-Agersborg 1918) Kjerskog-Agersborg (1918, p. 246–247) stated:

> Temperature together with the impulse to breed are still other causes for migration…. That *Pycnopodia* should move up to shallow water during the spawning season seems reasonable, for the temperature in deeper waters is undoubtedly lower than at the surface even during the spring. …*Pycnopodia* was found in large numbers on a certain side of a bay, during spring, while it was totally absent from these grounds later on, during the summer. As a matter of fact, during the spring of 1915, on the … [east side of Port Washington Narrows near Bremerton] more than 100 specimens were counted, while in July of the same year not a single specimen was seen in the same area.

A number of other sea stars move into shallow water during the spawning season, supporting that movement into shallow water may be an adaptive behavior that promotes fertilization in some way (Babcock, Franke et al. 2000). Greer (1962) reported that time from fertilization to metamorphosis for larvae from the San Juan Islands, Washington, ranged from 60-70 days when reared at 10-12°C. After fertilization, the embryos quickly develop into swimming, bilateral larvae that progress through the typical echinoderm larval phases of prism, bipinnaria, and pluteus larvae (Morris, Abbott et al. 1980).

Larvae of sea stars are capable of regenerating lost body parts much like adults (Allen et al. 2018; Vickery and McClintock 2000; Vickery et al. 2002) and may also reproduce asexually through the process of larval cloning—budding off of tissue fragments that regenerate into complete larvae (Bosch, Rivkin et al. 1989, Jaeckle 1994, Knott, Balser et al. 2003). Recently, Hodin, Pearson-Lund et al. (2021) reported that larvae of the sunflower sea star also have the capability to clone in a laboratory setting, describing cloning as "commonplace" in all larval cultures. The degree to which larval sunflower sea stars clone in nature may have profound implications for life history (*e.g.*, fecundity, dispersal distance), population dynamics, and population genetic structure (Knott, Balser et al. 2003, Balser 2004, Rogers-Bennett and Rogers 2008, Allen, Reitzel et al. 2018, Allen, Richardson et al. 2019).

Sunflower sea stars are voracious predators, consuming a wide array of benthic species, and can influence ecosystem structure by virtue of their predatory habits. They prey especially on sea urchins, which consume kelp and other marine vegetation that provide habitat for many nearshore species, sometimes during crucial life stages. Kelp forests have also disappeared from parts of California where the sunflower sea star has severely declined.

Population Dynamics

To understand the population dynamics of *P. helianthoides* on a range-wide basis it is crucial to develop an understanding of larval longevity and capacity for dispersal. Time from egg fertilization to metamorphosis for *P. helianthoides* under various conditions has been described as 49-77 days (Hodin, Pearson-Lund et al. 2021), 60-70 days (Greer 1962), and 90-146 days (Strathmann 1978). As noted by Gravem, Heady et al. (2021), broadcast spawning with a long pelagic larval duration has the potential for broad larval dispersal, especially in open coastal areas with few geographic barriers. Along more heterogeneous, complex shorelines like those found inside the Salish Sea or Southeast Alaska, complex flow patterns may result in localized entrainment of larval and reduce dispersal capacity.

Abundance As part of the IUCN status assessment process, Gravem et al. (2021) contacted a broad array of government, non-government, academic, and private data holders engaged in both direct and indirect monitoring of *P. helianthoides* occurrence, abundance, density, and habitat use throughout the range of the species. After careful evaluation of the temporal span, accuracy, taxonomic resolution, and verifiability of this suite of data sources, they identified 31 data sets that met minimum criteria for use in describing abundance trends over time (Table 96).

Table 96. Summarized estimates of population growth rate, annual rate of change, percent decline since 2013 for the West Coast region from 4 models (2022 Status Review Report (Lowry et al. 2022).

Region	Model	U1	Pre-2013 annual %	U2	Post-2013 annual %	% decline since 2013
Alaska	Best-fit w/NA	-0.210	-19.0	-2.475	-91.6	100.0
	Best-fit w/min	-0.069	-6.6	-1.447	-76.5	100.0
	AK only w/NA	0.207	23.0	-0.182	-16.7	80.6
	AK only w/min	0.374	45.3	-1.017	-63.8	100.0
BC & SS	Best-fit w/NA	0.090	9.4	-0.151	-14.1	74.4
	Best-fit w/min	0.129	13.8	-0.384	-31.9	96.8
	BC only w/NA	0.005	0.5	-0.142	-13.2	72.0
	BC only w/min	-0.017	-1.7	-0.271	-23.7	91.3
West Coast	Best-fit w/NA	0.017	1.7	-0.299	-25.8	93.2
	Best-fit w/min	-0.021	-2.1	-0.246	-21.8	89.1
	WC w/NA	0.056	5.8	-0.465	-37.2	98.5
	WC w/min	0.013	1.3	-0.260	-22.9	90.3

The best-fit modes included data for the entire range with zeros replaced either by NAs or by minimum values (see Appendix A of the 2022 Status Review Report). Regional-only models contain only data for that region. Best-fit w/NA is the top model from the primary model comparison. U1 = pre-2013 growth rate, U2 = post-2013 growth rate.

Productivity / Population Growth Rate / Genetic Diversity. Little is known about the natural productivity of the sunflower sea star on both an individual and population basis. Lack of information about growth rate, longevity, age at maturity, fecundity, natural mortality, the influence of larval cloning, and other fundamental biological attributes require that broad assumptions be applied and proxy species used to inform estimates on both regional and range-wide bases. Regardless of the values of nearly all of these parameters, the loss of ~90% of the global population of sunflower sea stars from 2013-17 is likely to have had profound impacts on population-level productivity. The standing crop of individuals capable of generating new recruits has been decreased, possibly to levels where productivity will be compromised on a regional or global basis. The combined factors of spatial distribution of individuals across the seascape and ocean conditions are crucial to dictating whether productivity is sufficient to allow population rebound. Broadly dispersed individuals may lack the ability to find mates, further reducing realized productivity despite abundance being high enough to theoretically result in population persistence.

Species-level impacts from SSWS, both during the 2013-17 pandemic and on an ongoing basis, were identified as the major threat affecting the long-term persistence of sunflower sea stars on a global basis. SSWS directly affects abundance, distribution, productivity, and spatial connectivity at a scale that places the sunflower sea star at a moderate risk of extinction throughout its range. Productivity, population growth rate, and phenotypic and genetic diversity may also be affected, but data were not available to directly assess these impacts. Impacts from anthropogenic climate change were also identified as a substantial threat to species persistence, but all other threats were determined to likely have minimal effects on species viability over the next 30 years on a range-wide basis.

Distribution. Sunflower sea stars are native to marine waters along the Pacific Coast, from northern Baja California to the central Aleutian Islands (Figure 57). Although they can live in waters ranging from a few feet deep to greater than 1,400 ft deep, these sea stars are generally encountered in waters shallower than 120 ft in depth. This species has no clear associations with specific habitats and is considered a habitat generalist (Gravem et al. 2021). The large geographic and depth range of the sunflower sea star indicates this species is well adapted for a wide variety of environmental conditions and habitat types. They are found along the outer coasts and inside waters, which have complex geophysical features including glacial fjords, sounds, embayments, and tidewater glaciers. Preferring temperate waters, they inhabit kelp forests and rocky intertidal shoals (Hodin, Pearson-Lund et al. 2021), but are regularly found in eelgrass meadows as well (Dean and Jewett 2001, Gravem, Heady et al. 2021). Sunflower sea stars occupy a wide range of benthic substrates including mud, sand, shell, gravel, and rocky bottoms while roaming in search of prey (Lambert 2000, Konar, Mitchell et al. 2019). They dwell in the low intertidal and subtidal zones to a depth of 435 m (1,427 ft) but are most common at depths less than 25 m (82 ft) and rare in waters deeper than 120 m (394 ft) (Fisher 1928; Gravem et al. 2021; Lambert 2000).

Designated Critical Habitat. None designated.

Recovery Goals. A Recovery Plan has not been developed.

8.48 Queen Conch

Table 97. Queen conch; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Aliger gigas	Queen Conch	N/A	Proposed Threatened	2022	2022 87 FR 55200	N/A	None Designated

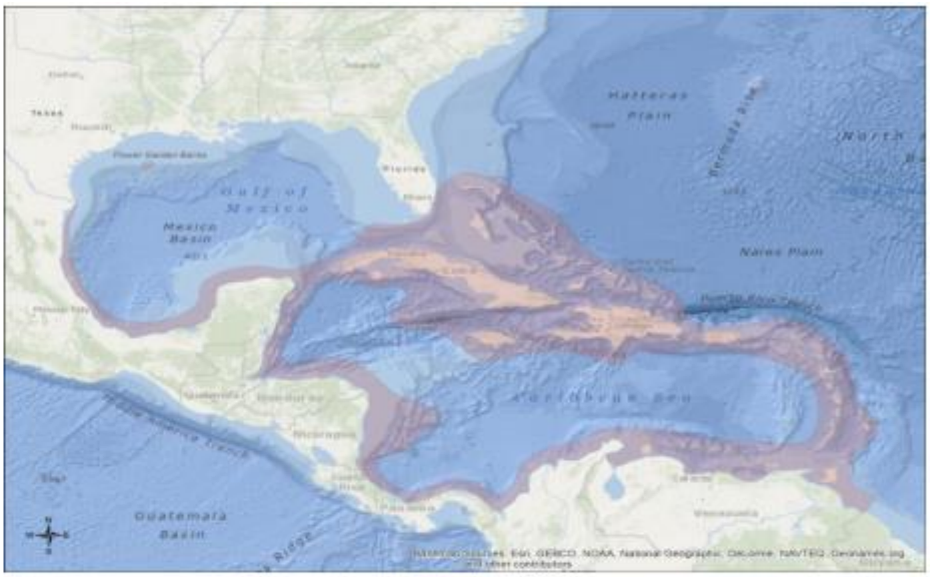


Figure 60. Queen Conch range

Species Description. The queen conch is a large gastropod belonging to the taxonomic group Mollusca. Queen conch are slow growing and late to mature, reaching up to 12 inches in length and living up to 30 years. Queen conch are highly sought after for their meat and are 1 of the most valuable species in the Caribbean.

Status. On September 17, 2022, NOAA proposed a rule to list the queen conch as threatened under the ESA (87 FR 55200). Queen conch numbers were severely reduced due to excessive harvest and habitat alterations. Estimates of adult queen conch population size are provided by jurisdiction in (Figure 61 and Figure 62). The median of the estimated population size in Cuba exceeded 400 million adult conch. Adult conch abundance was estimated to be between 10 and 100 million individuals in 6 jurisdictions, and 15 jurisdictions had median estimated abundances between 1 and 10 million adults. Estimated adult population size was less than 1 million adults in each of 20 jurisdictions, with 3 of those jurisdictions estimated to have populations of fewer than 100,000 adult queen conch. Total adult queen conch estimated abundance was 743 million individuals (90% confidence interval of 451 million to 1.49 billion). Seven jurisdictions (i.e., Cuba, Bahamas, Nicaragua, Jamaica, Honduras, the Turks and Caicos Islands, and Mexico) accounted for 95% of the population of adult queen conch.

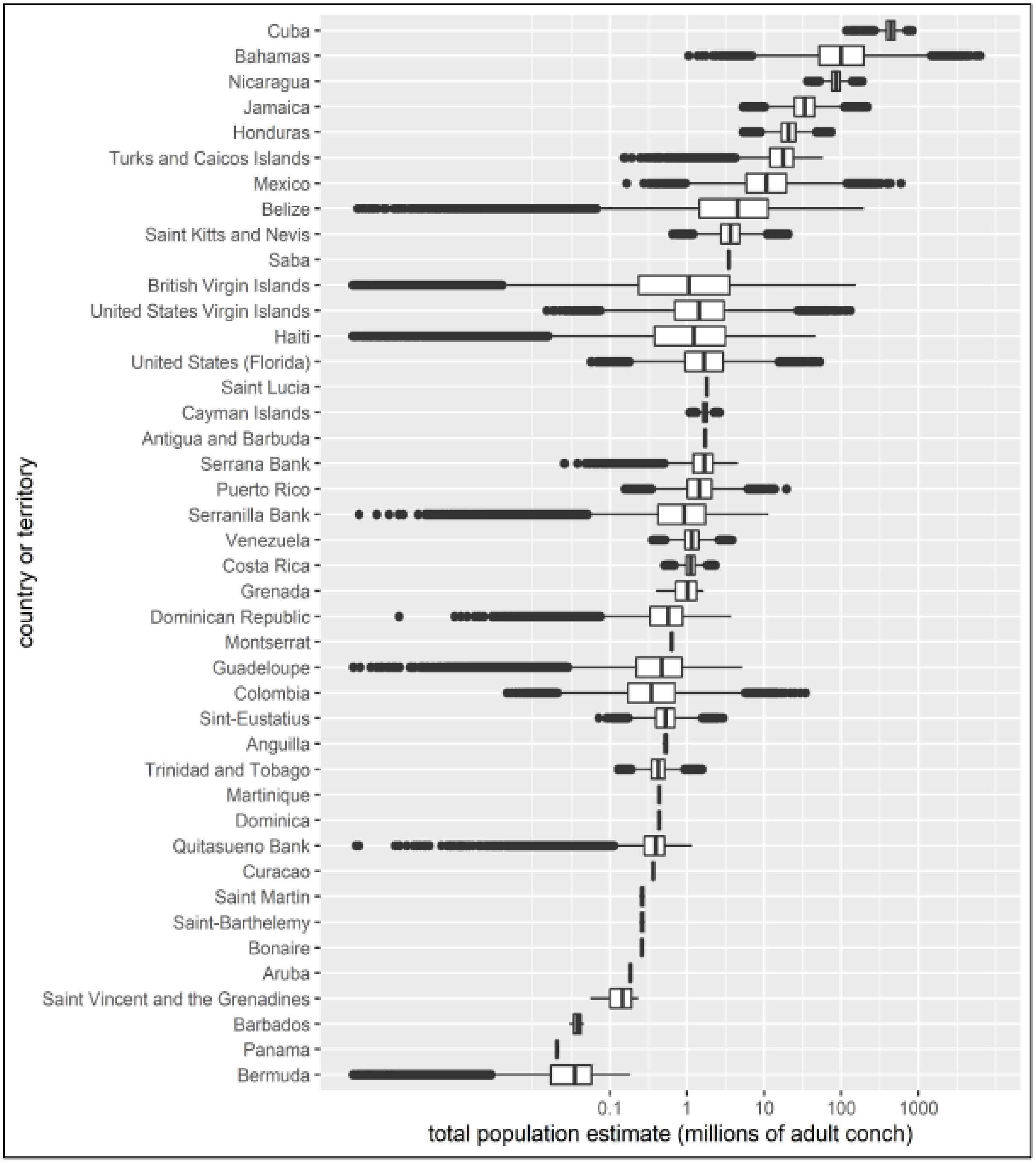


Figure 61**. The estimated queen conch adult population size (individuals) by jurisdiction. Distributional estimates as box and whisker plots; boxes denote interquartile range and points denote the full range of possible estimates. Note log scale.**

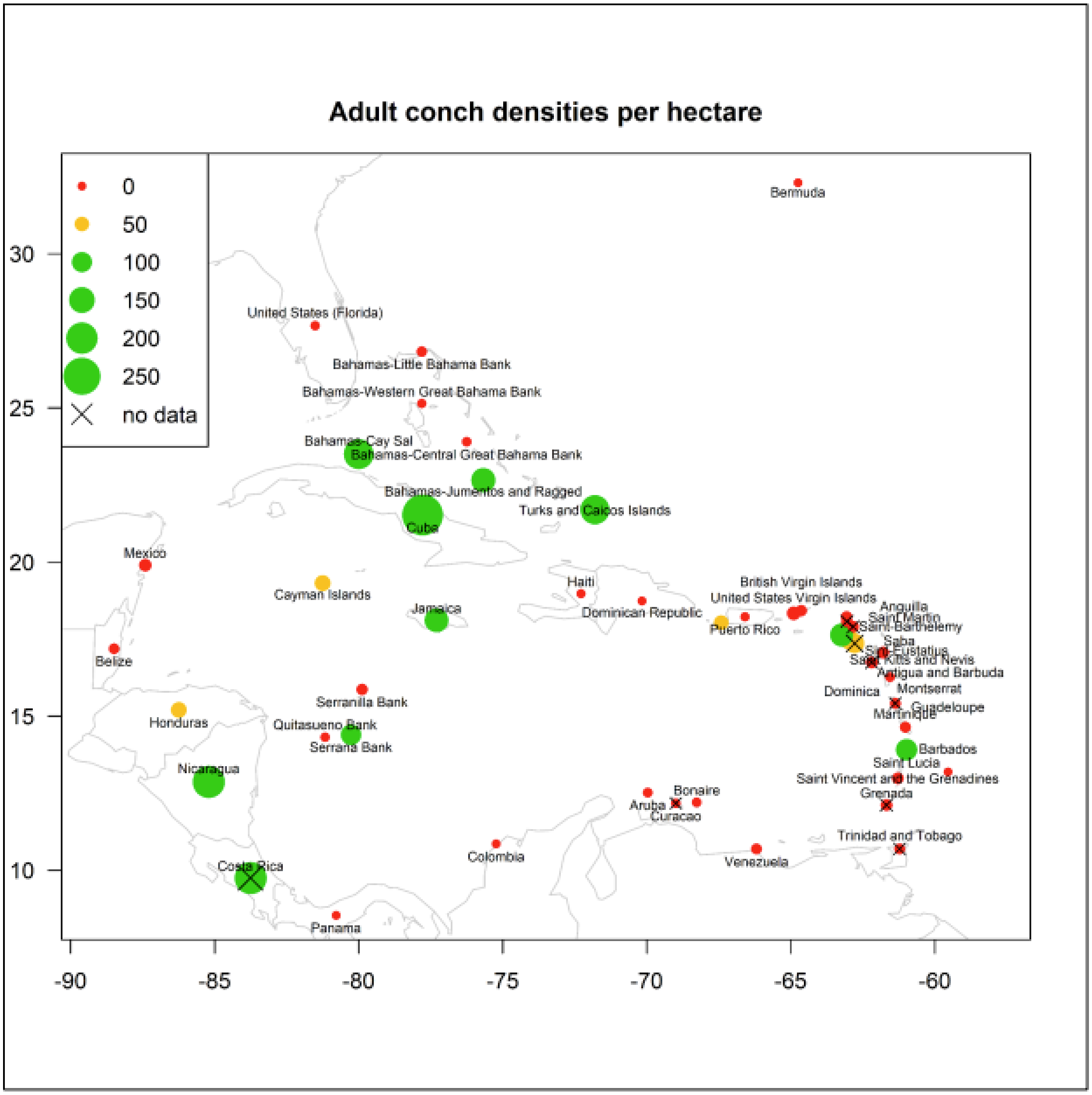


Figure 62. Estimated adult queen conch densities for jurisdictions within the species range. Data points are sized relative to densities.

Life History. Queen conch inhabit a range of habitat types during their life cycle. As conch develop, they use seagrass beds, sand flats, algal beds, and rubble areas from a few centimeters deep to approximately 30 meters (Brownell and Stevely 1981). Studies have suggested that adult conch move to different habitat types during their reproductive season, but afterwards return to feeding grounds (Glazer et al. 2003; Hesse 1979; Stoner and Sandt 1992). They are benthic-grazing herbivores that feed on diatoms, seagrass detritus, and various types of algae and epiphytes. Adult queen conch prefer sandy algal flats, but are also found on gravel, coral rubble, smooth hard coral, and beach rock bottom, while juveniles are primarily associated with seagrass beds. In general, adult conch do not move very far from their feeding grounds during their reproductive season (Stoner and Sandt 1992). Approximately 3 weeks after copulation the female lays a demersal egg mass on coarse sand of low organic content, completing deposition within 24-36 hours (D'Asaro 1965; Randall 1964). The egg mass consists of a long continuous egg-filled tube that folds and sticks together in a compact crescent shape, adhering to sand grains that provide camouflage and discourage predation. The eggs hatch after approximately 5 weeks. The veligers (larvae) drift in the water column up to 30 days depending on phytoplankton concentration, temperature, and the proximity of settlement habitat. These veligers are found primarily in the upper few meters of the water column (Paris et al. 2008; Posada and Appeldoorn 1994; Stoner 2003; Stoner and Davis 1997) where they feed on phytoplankton. When the veligers are morphologically and physiologically ready, they metamorphose into benthic animals in response to trophic cues from their seagrass habitat (Davis 2005). The key trophic cues shown to induce metamorphosis are epiphytes associated with macroalgae and sediment (Davis and Stoner 1994). Settlement locations are usually areas that have sufficient tidal circulation and high macroalgae production. Juvenile queen conch are primarily associated with native seagrass, such as *Thalassia testudinum*, in large parts of their range in the Caribbean and the southern Gulf of Mexico (Boman et al. 2019). However, juvenile queen conch can occur in a variety of habitat types. Randall (1964) reported that juvenile conch in the U.S. Virgin Islands were most abundant in shallow coral-rubble environments, with lower densities on bare sand and in seagrass beds. A similar association was reported from Puerto Rico, with high numbers in coral rubble compared with sand, seagrass, and hard bottom (Torres Rosado 1987). In Florida, juveniles are found in a variety of habitats, including reef rubble, algae-covered hard bottom, and secondarily in mixed beds of algae and seagrass, depending upon general location (Glazer and Berg Jr. 1994). In Cuba (Alcolado 1976), the Turks and Caicos Islands (Hesse 1979), Venezuela (Weil and Laughlin 1984), and the Bahamas, juvenile conch are associated primarily with native seagrass (Stoner 2003; Stoner et al. 1996; Stoner et al. 1994). In St. Croix, U.S. Virgin Islands, densities of juvenile and adult queen conch were the highest in habitats characterized as 50-90% and 10-50% patchy seagrass, respectively (Doerr and Hill 2018). After the veligers settle on the bottom, they bury into the sediment. This submerged life phase makes it difficult to survey and therefore they are often under-sampled (Appeldoorn 1987; Hesse 1979). They emerge about a year later (Stoner 1989a) as juveniles at around 60 mm shell length.

Most conch nursery areas occur primarily in back reef areas (i.e., shallow sheltered areas, lagoons, behind emergent reefs or cays) of medium seagrass density, depths between 2 to 4 m, with strong tidal currents (at least 50 cm/s; Stoner 1989b), and frequent tidal water exchanges (Stoner et al. 1996; Stoner and Waite 1991). Seagrass is thought to provide both nutrition and protection from predators (Ray and Stoner 1995; Stoner and Davis 2010).

Adult conch can be found in a wide range of environmental conditions (Stoner et al. 1994) such as in sand and algal or coral rubble (Acosta 2001; Stoner and Davis 2010). Adult queen conch are rarely, if ever, found on soft bottoms composed of silt and/or mud, or in areas with high coral cover (Acosta 2006). Adult conch are found in shallow, clear water of oceanic or near-oceanic salinities at depths generally less than 75 m, and are most often found in waters less than 30 meters (McCarthy 2007). It is believed that depth limitation is based mostly on light attenuation limiting their photosynthetic food source (McCarthy 2007; Randall 1964).

Population Dynamics

Abundance. The current total population of the queen conch is estimated to be in the millions throughout its range, but declining in several areas within its historic range.

Productivity / Population Growth Rate. The status review team defined the following thresholds to determine the status of queen conch populations throughout the greater Caribbean:

- Populations with densities above 100 adult conch/ha are considered to be at a density that supports reproductive activity resulting in population growth.
- Populations with densities between 50-99 adult conch/ha are considered to have reduced reproductive activity resulting in minimal population growth.
- Populations with densities below the 50 adult conch/ha threshold are considered to be not reproductively active due to low adult encounter rates or mate finding. This threshold is largely recognized as an absolute minimum required to support mate-finding and thus reproduction.

Genetic Diversity / Distribution. Early studies using allozymes to examine the genetic structure of queen conch implied high levels of gene flow but showed isolated genetic structure for populations either at isolated sites or at the microscale level (Mitton et al. 1989).

The queen conch occurs throughout the Caribbean Sea, the Gulf of Mexico, and around Bermuda (Figure 62) and includes the following jurisdictions: Anguilla, Antigua and Barbuda, Aruba, Barbados, Bahamas, Belize, Bermuda, Caribbean Netherlands, Colombia, Costa Rica, Cuba, Curaçao, Dominican Republic, French West Indies, Grenada, Haiti, Honduras, Jamaica, Mexico, Montserrat, Nicaragua, Panama, Puerto Rico, St. Kitts and Nevis, St. Lucia, St. Vincent and the Grenadines, Trinidad and Tobago, the Turks and Caicos, the United States (Florida, Puerto Rico, U.S. Virgin Islands, Flower Garden Banks National Marine Sanctuary), British Virgin Islands, and Venezuela (Theile 2001).

Designated Critical Habitat. The queen conch ESA listing status is proposed threatened. Critical habitat has not yet been designated.

Recovery Goals. The queen conch ESA listing status is proposed threatened. A Recovery Plan has not been developed.

8.49 Staghorn Coral

Table 98. Staghorn Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acropora cervicornis	Staghorn	N/A	Threatened	2022	2006 71 FR 26852	2015	2008 73 FR 72210

Figure 63. Staghorn Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. The staghorn coral is a cnidarian belonging to the taxonomic order of Scleractinia, a group of stony corals that secrete calcium carbonate to form hard exoskeletons. Staghorn coral occurs throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic. Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges from 0.25-5 centimeters in Lirman et al. (2010a), and linear branch growth rates have been reported to range between 3-11.5 centimeters per year (Acropora Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about 1.5 meters diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (Acropora Biological Review Team 2005). Staghorn corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, Staghorn coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

Status. The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. Although localized mortality events have continued to occur, percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable over its range since the mid-1980s. There is evidence of synergistic effects of threats for this species where the effects of increased nutrients are combined with acidification and sedimentation. Staghorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because staghorn coral is limited to areas with high, localized human impacts and predicted increasing threats. Staghorn coral commonly occurs in water ranging from 5-20 meters in depth, though it occurs in depths of 16-30 meters at the northern extent of its range, and has been rarely found to 60 meters in depth. It occurs in spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats. This habitat heterogeneity moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef and hard bottom environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these 2 locations due to the occurrence of the species in many other areas throughout its range. Staghorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate the species' vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing one of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is one previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. Relative to other corals, staghorn coral has a high growth rate that has allowed acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately 4 to 11 centimeters per year (*Acropora* Biological Review Team 2005). Annual linear extension has been found to be dependent on the size of the colony. New recruits and juveniles typically grow at slower rates. Stressed colonies and fragments may also exhibit slower growth.

Staghorn coral is a hermaphroditic broadcast spawning species. The spawning season occurs several nights after the full moon in July, August, or September depending on location and timing of the full moon and may be split over the course of more than 1 lunar cycle (Szmant 1986; Vargas-Angel et al. 2006). The estimated size at sexual maturity is approximately 17 centimeters branch length, and large colonies produce proportionally more gametes than small colonies (Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies. Laboratory studies have found that certain species of crustose-coralline algae produce exudates which facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Tunnicliffe 1981). The combination of branching morphology, asexual fragmentation, and fast growth rates, relative to other corals, can lead to persistence of large areas dominated by staghorn coral. The combination of rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition and can facilitate potential recovery from disturbances when environmental conditions permit. However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate spatially dispersed sites.

Population Dynamics

Abundance. Miller et al. (2013b) extrapolated population abundance of staghorn coral in the Florida Keys and Dry Tortugas from stratified random samples across habitat types. Population estimates of staghorn coral in the Florida Keys were 10.2 ± 4.6 (standard error [SE]) million colonies in 2005, 6.9 ± 2.4 (SE) million colonies in 2007 and 10.0 ± 3.1 (SE) million colonies in 2012. Population estimates in the Dry Tortugas were 0.4 ± 0.4 (SE) million colonies in 2006 and 3.5 ± 2.9 (SE) million colonies in 2008, though the authors note their sampling scheme in the Dry Tortugas was not optimized for staghorn coral. Because these population estimates were based on random sampling, differences in abundance estimates between years is more likely to be a function of sample design rather than population trends. In both the Florida Keys and Dry Tortugas, most of the population was dominated by small colonies less than 12 in (30 cm) diameter. Further, partial mortality was reported as highest in 2005 with up to 80% mortality observed and lowest in 2007 with a maximum of 30%. In 2012, partial mortality ranged from 20-50% across most size classes.

Based on population estimates, there are at least tens of millions of colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these 2 locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. There is no evidence of range constriction or extirpation at the island level. However the species is absent at the reef level. Populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Thickets are a prominent feature at only a few known locations. Across the Caribbean, percent cover appears to have remained relatively stable since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s. There are examples of increasing trends in some locations (Dry Tortugas and southeast Florida), but not over larger spatial scales or longer periods. Population model projections from Honduras at one of the only known remaining thickets indicate the retention of this dense stand under undisturbed conditions. If refuge populations are able to persist, it is unclear whether they would be able to repopulate nearby reefs as observed sexual recruitment is low. Thus, we conclude that the species has undergone substantial population decline and decreases in the extent of occurrence throughout its range. Percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable since the mid-1980s and since the listing of the species as threatened in 2006. We also conclude that population abundance is at least tens of millions of colonies, but likely to decrease in the future with increasing threats.

Productivity / Population Growth Rate. Staghorn coral historically was 1 of the dominant species on most Caribbean reefs, forming large, single-species thickets and giving rise to the nominal distinct zone in classical descriptions of Caribbean reef morphology (Goreau 1959b). Massive, Caribbean-wide mortality, apparently primarily from white band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011b). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events has added to the decline of staghorn coral (Brainard et al. 2011b). In locations where quantitative data are available (Florida, Jamaica, U.S.Virgin Islands, Belize), there was a reduction of approximately 92% to greater than 97% between the 1970s and early 2000s (*Acropora* Biological Review Team 2005).

Since the 2006 listing of staghorn coral as threatened, continued population declines have occurred in some locations with certain populations of both listed *Acropora* species (staghorn and elkhorn) decreasing up to an additional 50% or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008a). There are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and Dominican Republic (Lirman et al. 2010a). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold-water die-off of the 1970s. Cover of staghorn coral increased on a Jamaican reef from 0.6% in 1995 to 10.5% in 2004 (Idjadi et al. 2006).

Riegl et al.(2009) monitored staghorn coral in photo plots on the fringing reef near Roatan, Honduras from 1996 to 2005. Staghorn coral cover declined from 0.42% in 1996 to 0.14% in

1999 after the Caribbean bleaching event in 1998 and mortality from run-off associated with a Category 5 hurricane. Staghorn coral cover further declined to 0.09% in 2005. Staghorn coral colony frequency decreased 71% between 1997 and 1999. In sharp contrast, offshore bank reefs near Roatan had dense thickets of staghorn coral with 31% cover in photo-quadrats in 2005 and appeared to survive the 1998 bleaching event and hurricane, most likely due to bathymetric separation from land and greater flushing. Modeling showed that under undisturbed conditions, retention of the dense staghorn coral stands on the banks off Roatan is likely with a possible increased shift towards dominance by other coral species. However, the authors note that because their data and the literature seem to point to extrinsic factors as driving the decline of staghorn coral, it is unclear what the future may hold for this dense population (Riegl et al. 2009).

While cover of staghorn coral increased from 0.6% in 1995 to 10.5% in 2004 (Idjadi et al. 2006) and 44% in 2005 on a Jamaican reef, it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5% in 2006 (Quinn and Kojis 2008). A cold water die-off across the lower to upper Florida Keys in January 2010 resulted in the complete mortality of all staghorn coral colonies at 45 of the 74 reefs surveyed (61%) (Schopmeyer et al. 2012). Walker et al.(2012) report increasing size of 2 thickets (expansion of up to 7.5 times the original size of 1 of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50% highlighting the dynamic nature of staghorn coral distribution via fragmentation and re-attachment.

A report on the status and trends of Caribbean corals over the last century indicates that cover of staghorn coral has remained relatively stable (though much reduced) throughout the region since the large mortality events of the 1970s and 1980s. The frequency of reefs at which staghorn coral was described as the dominant coral has remained stable. The number of reefs with staghorn coral present declined during the 1980s (from approximately 50% to 30% of reefs), remained relatively stable at 30% through the 1990s, and decreased to approximately 20% of the reefs in 2000-2004 and approximately 10% in 2005-2011 (Jackson et al. 2014b).

Genetic Diversity. Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from 9 regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 500 kilometers apart are genetically different from each other with low gene flow across the greater Caribbean. Fine-scale genetic differences have been detected at reefs separated by as little as 2 kilometers, suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A.prolifera* with staghorn coral (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). Populations in Florida and Honduras are genetically distinct from each other and other populations in the U.S.Virgin Islands, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the U.S.Virgin Islands and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

Distribution. Staghorn coral is distributed throughout the Caribbean Sea, in the southwestern Gulf of Mexico, and in the western Atlantic Ocean. The fossil record indicates that during the Holocene epoch, staghorn coral was present as far north as Palm Beach County in southeast Florida (Lighty et al. 1978), which is also the northern extent of its current distribution (Goldberg 1973). Staghorn coral commonly occurs in water ranging from 5-20 meters in depth, though it occurs in depths of 16-30 meters at the northern extent of its range, and has been rarely found to 60 meters in depth.

Designated Critical Habitat. In 2008, critical habitat for staghorn and elkhorn corals was designated in areas in or around Southeast Florida and the Florida Keys, Puerto Rico, and the Virgin Islands. These 4 distinct areas comprise of approximately 2,959 square miles of marine habitat. The essential features chosen to select critical habitat was substrate of suitable quality and availability, in water depths from the mean high water (MHW) line to 30 meters to allow for successful sexual and asexual reproduction. Successful sexual and asexual reproduction includes flourishing larval settlement, recruitment, and reattachment of coral fragments (73 FR 72210). “Substrate of suitable quality and availability” means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover.

Recovery Goals. The 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan (NMFS 2015c) contains complete downlisting/delisting criteria for each of the 2 following recovery goals:

Ensure Population Viability
Specific criteria include: 1) preserving abundance; 2) maintaining genotypic diversity; and 3) properly observing and recording recruitment rates.

Eliminate or sufficiently abate global, regional, and local threats
Specific criteria include: 1) developing quantitative recovery criterion through research to identify, treat, and reduce outbreaks of coral disease; 2) controlling the local and global impacts of rising ocean temperature and acidification; 3) reducing the loss of recruitment habitat; 4) reducing sources of nutrients, sediments, and contaminants; 5) developing and adopting appropriate and effective regulatory mechanisms to abate threats; 6) reducing impacts of natural and anthropogenic abrasion and breakage; and 7) reducing impacts of predation.

8.50 Elkhorn Coral

Table 99. Elkhorn Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acropora palmata	Elkhorn	N/A	Threatened	2022	2006 71 FR 26852	2015	2008 73 FR 72210

Figure 64. Elkhorn Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation(Veron 2014).

Species Description. The elkhorn coral is a cnidarian belonging to the taxonomic order of scleractinia, a group of stony corals that secrete calcium carbonate to form hard exoskeletons. Elkhorn coral occurs throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic. Elkhorn corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, Elkhorn coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

Acropora palmata was listed as threatened under the ESA in 2006. In 2012, a proposal to change the listing to endangered was made, but in 2014 its threatened status was upheld. Along with staghorn coral, elkhorn coral is the only other large, branching species of coral to produce and occupy vast complex environments within the Caribbean Sea's reef system. In all, there appears to be 2 distinct populations of elkhorn coral, a western Caribbean population and an eastern (Baums et al. 2005).

Status. The decline in the total abundance of elkhorn coral has been attributed to a series of stressors consisting of disease, temperature-induced bleaching, excessive sedimentation, nitrification, pollution (i.e., oxybenzone from sunscreen), and large hurricanes/tropical storms (Brainard et al. 2011b; Downs et al. 2016; Hernandez-Delgado et al. 2011; Mayor et al. 2006; Rogers and Muller 2012). It is believed that these effects act synergistically with one another thereby increasing the overall damage to already-stressed *A. palmata* colonies that have undergone disturbance by another threat. The current population trend appears to be steady, although there are places where populations continue to decrease and others where there appears to be modest or contained recovery (Miller et al. 2013a). However, even if growth and recruitment end up surpassing mortality, this species requires prompt analysis and monitoring on a regional scale. Reasoning for this includes the current presence of areas with low genetic diversity and density within western Caribbean populations along with localized high rates of disease and bleaching (Miller et al. 2013a).

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing 1 of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is 1 previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. Elkhorn coral, like most stony corals, employ both sexual and asexual reproductive strategies to propagate. Sexual reproduction in corals includes gametogenesis, the process in which cells undergo meiosis to form gametes within the polyps near the base of the mesenteries. Since Acropora palmata is hermaphroditic, each polyp contains both sperm and egg cells that are released together in a ‘bundle’, causing the coral gametes to develop externally from the parental colony. Elkhorn coral reproduces sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (Acropora Biological Review Team 2005). Split spawning (spawning over a 2 month period) has been reported from the Florida Keys (Fogarty et al. 2012). The estimated size at sexual maturity is approximately 250 in2 (1,600 cm2), and growing edges and encrusting base areas are not fertile (Soong and Lang 1992). Larger colonies have higher fecundity per unit area, as do the upper branch surfaces (Soong and Lang 1992). Although self-fertilization is possible, elkhorn coral is largely self-incompatible (Baums et al. 2005a; Fogarty et al. 2012). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies in the field. Rates of post-settlement mortality after 9 months are high based on settlement experiments (Szmant and Miller 2005).

Reproduction occurs primarily through asexual reproduction, generating multiple colonies that are genetically identical. Elkhorn coral can quickly monopolize large spaces of shallow ocean floor through fragment dissemination. A branch of *A. palmata* can be carried by waves and currents away from the mother colony to distances that range from 0.1 – 100 meters, but fragments usually travel less than 30 meters (NMFS 2005).

Because large colonies of *A. palmata* contain several thousand partially autonomous polyps, growth rates for the species are conveyed through the measurement of linear extensions of the organisms' skeletal branches. Depending on the size and location of the colony, physical growth rates for elkhorn corals range from approximately 4-11 centimeters per year. Branches are up to approximately 50 centimeters wide and range in thickness of about 4-5 centimeters. Individual colonies can grow to at least 2 meters in height and 4 meters in diameter (NMFS 2005). Total lifespan for the species is unknown (NMFS 2014).

Population Dynamics

Abundance / Productivity. Colonial species present a special challenge in determining the appropriate unit to evaluate for abundance. However, the present population of elkhorn coral is continuing at a very low abundance due to large declines in the past several decades (NMFS 2005b). The western Caribbean is characterized by genetically depauperate populations with lower densities (0.13 ± 0.08 colonies per m^2). The eastern Caribbean populations are characterized by denser (0.30 ± 0.21 colonies per m^2), genotypically richer stands (Baums et al. 2006a).

Based on population estimates from both the Florida Keys and St. Croix, U.S. Virgin Islands, there are at least hundreds of thousands of elkhorn coral colonies. Absolute abundance is higher than estimates from these 2 locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. Across the Caribbean, percent cover appears to have remained relatively stable, albeit it at extremely low levels, since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s, indicating potential decreases in the extent of occurrence and effects on the species' range. However, the proportions of Caribbean sites where elkhorn coral is present and dominant have recently stabilized since the mid-2000s. There are locations such as the U.S. Virgin Islands where populations of elkhorn coral appear stable or possibly increasing in abundance and some such as the Florida Keys where population number appears to be decreasing.

Genetic Diversity. Genetic samples from 11 locations throughout the Caribbean indicate that elkhorn coral populations in the eastern Caribbean (St. Vincent and the Grenadines, U.S. Virgin Islands, Curaçao, and Bonaire) have had little or no genetic exchange with populations in the western Atlantic and western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Puerto Rico) (Baums et al. 2005). While Puerto Rico is more closely connected with the western Caribbean, it is an area of mixing with contributions from both regions (Baums et al. 2005). Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico promotes dispersion of larval and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006b).

Distribution. Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from 1 to 30 meters in depth. Historically, *A. palmata* inhabited most waters of the Caribbean between 1-5 meters depth. This included a diverse set of areas comprising of zones along Puerto Rico, Hispaniola, the Yucatan peninsula, the Bahamas, the southwestern Gulf of Mexico, the Florida Keys, the Southeastern Caribbean islands, and the northern coast of South America (Dustan and Halas 1987; Goreau 1959a; Jaap 1984; Kornicker and Boyd 1962; Scatterday 1974; Storr 1964). While the present-day spatial distribution of elkhorn coral is similar to its historic spatial distribution, its presence within its range has become increasingly sparse due to declines in the latter half of the 20th century from a variety of abiotic and biotic threats.

Designated Critical Habitat. Critical habitat for elkhorn and staghorn corals was designated in 2008. The PBF essential to the conservation of Atlantic *Acropora* species is substrate of suitable quality and availability in water depths from the mean high water line to 30 meters in order to support successful larval settlement, recruitment, and reattachment of fragments. “Substrate of suitable quality and availability” means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing this feature have been identified in four locations within the jurisdiction of the United States: the Florida area, which comprises approximately 1,329 mi^2 (3,442 km^2) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 mi^2 (3,582 km^2) of marine habitat; the St. John/St. Thomas area, which comprises approximately 121 mi^2 (313 km^2) of marine habitat; and the St. Croix area, which comprises approximately 126 mi^2 (326 km^2) of marine habitat. The total area covered by the designation is thus approximately 2,959 mi^2 (7,664 km^2).

As defined in the final rule, critical habitat does not include areas subject to the 2008 Naval Air Station Key West Integrated Natural Resources Management Plan; all areas containing existing (already constructed) federally authorized or permitted man-made structures such as aids-to-navigation (ATONS), artificial reefs, boat ramps, docks, pilings, maintained channels, or marinas; or 12 federal maintained harbors and channels.

The PBF can be found unevenly dispersed throughout the critical habitat units, interspersed with natural areas of loose sediment, fleshy or turf macroalgae covered hard substrate. Existing federally authorized or permitted man-made structures such as artificial reefs, boat ramps, docks, pilings, channels or marinas do not provide the PBF. The proximity of this habitat to coastal areas subjects this feature to impacts from multiple activities including dredging and disposal activities, stormwater run-off, coastal and maritime construction, land development, wastewater and sewage outflow discharges, point and non-point source pollutant discharges, fishing, placement of large vessel anchorages, and installation of submerged pipelines or cables. The impacts from these activities, combined with those from natural factors (i.e., major storm events), significantly affect the quality and quantity of available substrate for these threatened species to successfully sexually and asexually reproduce.

A shift in benthic community structure from coral-dominated to algae-dominated that has been documented since the 1980s means that the settlement of larvae or attachment of fragments is often unsuccessful (Hughes and Connell 1999). Sediment accumulation on suitable substrate also impedes sexual and asexual reproductive success by preempting available substrate and smothering coral recruits.

While algae, including crustose coralline algae and fleshy macroalgae, are natural components of healthy reef ecosystems, increased algal dominance since the 1980s has impeded coral recruitment. The overexploitation of grazers through fishing has also contributed to fleshy macroalgae persistence in reef and hard bottom areas formerly dominated by corals. Impacts to water quality associated with coastal development, in particular nutrient inputs, are also thought to enhance the growth of fleshy macroalgae by providing them with nutrient sources. Fleshy macroalgae are able to colonize dead coral skeleton and other hard substrate and some are able to overgrow living corals and crustose coralline algae. Because crustose coralline algae is thought to provide chemical cues to coral larvae indicating an area is appropriate for settlement, overgrowth by macroalgae may affect coral recruitment (Steneck 1986). Several studies show that coral recruitment tends to be greater when algal biomass is low (Birrell et al. 2005; Connell et al. 1997; Edmunds et al. 2004; Hughes 1985; Rogers et al. 1984; Vermeij 2006). In addition to preempting space for coral larval settlement, many fleshy macroalgae produce secondary metabolites with generalized toxicity, which also may inhibit settlement of coral larvae (Kuffner and Paul 2004). The rate of sediment input from natural and anthropogenic sources can affect reef distribution, structure, growth, and recruitment. Sediments can accumulate on dead and living corals and exposed hard bottom, thus reducing the available substrate for larval settlement and fragment attachment.

In addition to the amount of sedimentation, the source of sediments can affect coral growth. In a study of 3 sites in Puerto Rico, Torres (2001) found that low-density coral skeleton growth was correlated with increased re-suspended sediment rates and greater percentage composition of terrigenous sediment. In sites with higher carbonate percentages and corresponding low percentages of terrigenous sediments, growth rates were higher. This suggests that re-suspension of sediments and sediment production within the reef environment does not necessarily have a negative impact on coral growth while sediments from terrestrial sources increase the probability that coral growth will decrease, possibly because terrigenous sediments do not contain minerals that corals need to grow (Torres 2001).

Long-term monitoring of sites in the USVI indicate that coral cover has declined dramatically; coral diseases have become more numerous and prevalent; macroalgal cover has increased; fish of some species are smaller, less numerous, or rare; long-spined black sea urchins are not abundant; and sedimentation rates in nearshore waters have increased from 1–2 orders of magnitude over the past 15 to 25 years (Rogers et al. 2008). Thus, changes that have affected elkhorn and staghorn coral and led to significant decreases in the numbers and cover of these species have also affected the suitability and availability of habitat.

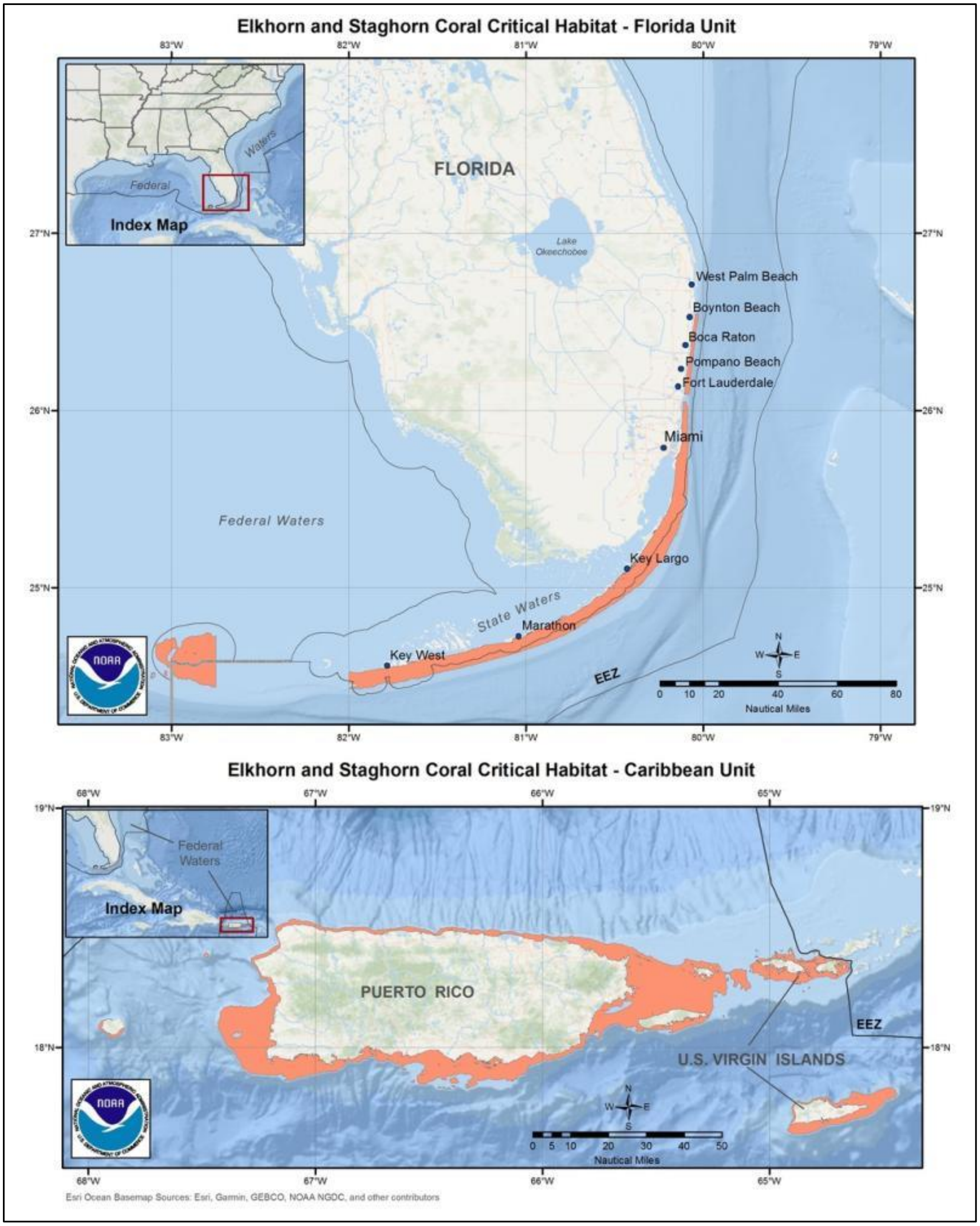


Figure 65**. Florida, Puerto Rico, and 2 USVI Critical Habitat. Units for Elkhorn and Staghorn Corals**

Elkhorn and staghorn corals require hard, consolidated substrate, including attached, dead coral skeleton, devoid of turf or fleshy macroalgae for their larvae to settle. The Atlantic and Gulf of Mexico Rapid Reef Assessment Program data from 1997-2004 indicate that although the historic range of both species remains intact, the number and size of colonies and percent cover by both species has declined dramatically in comparison to historic levels (Ginsburg and Lang 2003).

Long-term monitoring of marine habitats in natural reserves around Puerto Rico, begun in 1999 and now at full capacity indicates statistically significant declines in live coral cover (Garcia-Sais et al. 2008). The most pronounced declines in coral cover were observed between the 2005 and 2006 surveys, corresponding to the dramatic bleaching even that occurred because of high sea surface temperatures in 2005. Declines of up to 59% were measured in surveyed reefs and a proportional increase in turf algae was observed (Garcia-Sais et al. 2008). Together with bleaching-associated mortality, coral disease led to the recorded loss of 50% to 80% live coral cover from reefs in La Parguera, Culebra, Mona, and Desecheo, Puerto Rico, and other important reefs in the northeast and southern Caribbean between 2005 and 2011 (Bastidas et al. 2012; Bruckner and Hill 2009; Croquer and Weil 2009; Hernández-Pacheco et al. 2011; Weil et al. 2009). Thus, changes that have affected elkhorn and staghorn corals and led to significant decreases in their numbers and cover have also affected the suitability and availability of habitat for these species.

Recovery Goals. The 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan (NMFS 2015c) contains complete downlisting/delisting criteria for each of the 2 following recovery goals.

- **Ensure Population Viability**
  Specific criteria include: 1) preserving abundance; 2) maintaining genotypic diversity; and 3) properly observing and recording recruitment rates.

- **Eliminate or sufficiently abate global, regional, and local threats**
  Specific criteria include: 1) developing quantitative recovery criterion through research to identify, treat, and reduce outbreaks of coral disease; 2) controlling the local and global impacts of rising ocean temperature and acidification; 3) reducing the loss of recruitment habitat; 4) reducing sources of nutrients, sediments, and contaminants; 5) developing and adopting appropriate and effective regulatory mechanisms to abate threats; 6) reducing impacts of natural and anthropogenic abrasion and breakage; and 7) reducing impacts of predation.

8.51 Boulder Star Coral

Table 100. Boulder Star Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Orbicella franksi	Boulder Star	N/A	Threatened	2022	2014 79 FR 53851	2015 (Recovery Outline)	2023 88 FR 54026

Figure 66. Boulder Star coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. The boulder star coral is a cnidarian belonging to the taxonomic genus of Orbicella, a group of stony corals that secrete calcium carbonate to form hard exoskeletons. Boulder star coral occurs in the western Atlantic and throughout the Caribbean, including the Bahamas, Flower Garden Banks, and the entire Caribbean coastline. On September 10, 2014, NMFS listed boulder star coral as threatened (79 FR 53851). Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella* spp. complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species –*Montastraea annularis*– with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata*, and *Montastraea* (now *Orbicella*) *franksi*. The 3 species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into 3 species.

Boulder star corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, Boulder star coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

Some studies report on the star coral species complex rather than individual species because visual distinction can be difficult where colony structure cannot be discerned (e.g., small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. Information about boulder star coral published prior to 1994 will be attributed to the species complex, since it is dated prior to the split of *Orbicella annularis* into 3 separate species.
Boulder star coral is distinguished by large, unevenly arrayed polyps that give the colony its characteristic irregular surface. Colony form is variable, and the skeleton is dense with poorly developed annual bands. Colony diameter can reach up to 5 meters with a height of up to 2 meters.

Status. Boulder star coral has undergone declines most likely from disease and warming-induced bleaching. There is evidence of synergistic effects of threats for this species including increased disease severity with nutrient enrichment. Boulder star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction.

Despite declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes as has been observed in locations in its range. Its absolute population abundance has been estimated as at least tens of millions of colonies in both a portion of the U. S. Virgin Islands and the Dry Tortugas and is higher than the estimate from these 2 locations due to the occurrence of the species in many other areas throughout its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because boulder star coral is limited to a areas with high localized human impacts and predicted increasing threats. Its depth range of approximately 5-50 meters, possibly up to 90 meters, moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Boulder star coral occurs in most reef habitats, including both shallow and mesophotic reefs, although it has not been observed in mesophotic areas of the Florida Keys, Dry Tortugas, or Pulley Ridge in the Gulf of Mexico (NMFS 2022b). The occurrence in some multiple reef habitats moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable temperatures and ocean chemistry at any given point in time. Its abundance, life history characteristics, and depth distribution, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing one of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is one previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. All 3 species of the star coral complex are hermaphroditic broadcast spawners[7], with spawning concentrated on 6 to 8 nights following the full moon in late August, September, or early October, depending on timing of the full moon and location. Boulder star coral spawning is reported to be about 1–2 hours earlier than lobed star coral and mountainous star coral. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Fertilization success measured in the field was generally below 15% for all 3 species, as it was closely linked to the number of colonies concurrently spawning. In Puerto Rico, minimum size at reproduction for the star coral species complex was 83 square centimeters.

Successful recruitment by the star coral species complex appears to always have been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of approximately 12 square meters of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex. Of 351 colonies of boulder star coral tagged in Bocas del Toro, Panama, larger colonies were noted to spawn more frequently than smaller colonies between 2002 and 2009 (Levitan et al. 2011).

[7] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

Population Dynamics

Abundance. Boulder star coral is reported as common. In a 1995 survey of 16 reefs in the Florida Keys, boulder star coral had the highest percent cover of all species (Murdoch and Aronson 1999). In surveys throughout the Florida Keys, boulder star coral in 2005 ranked 26th most abundant out of 47 coral species, 32nd out of 43 in 2009, and 33rd out of 40 in 2012. Extrapolated population estimates from stratified random surveys were 8.0 ± 3.5 million (standard error [SE]) colonies in 2005, 0.3 ± 0.2 million (SE) colonies in 2009, and 0.4 ± 0.4 million (SE) colonies in 2012. The authors note that differences in extrapolated abundance between years were more likely a function of sampling design rather than an indication of population trends. In 2005, the greatest proportions of colonies were in the smaller size classes of approximately 10-20 centimeters and approximately 20-30 centimeter. Partial colony mortality ranged from 0% to approximately 73% and was generally higher in larger colonies (Miller et al. 2013a).

In the Dry Tortugas, Florida, boulder star coral ranked 4th highest in abundance out of 43 coral species in 2006 and 8th out of 40 in 2008. Extrapolated population estimates were 79 ± 19 million (SE) colonies in 2006 and 18.2 ± 4.1 million (SE) colonies in 2008. Miller et al. (2013a) notes the difference in estimates between years was more likely a function of sampling design rather than population decline. In the first year of the study (2006), the greatest proportion of colonies were in the size class approximately 20-30 centimeters with twice as many colonies as the next most numerous size class and a fair number of colonies in the largest size class of greater than 90 centimeters. Partial colony mortality ranged from approximately 10-55%. Two years later (2008), no size class was found to dominate, and proportion of colonies in the medium-to-large size classes (approximately 60-90 centimeters) appeared to be less than in 2006. The number of colonies in the largest size class of greater than 90 centimeters remained consistent. Partial colony mortality ranged from approximately 15-75% (Miller et al. 2013a).

Abundance in Curaçao and Puerto Rico appears to be stable over an 8 to 10 year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at 9 sites off Mona and Desecheo Islands, Puerto Rico. In 1998, 4% of all corals at 6 sites surveyed off Mona Island were boulder star coral colonies and approximately 5% in 2008; at Desecheo Island, about 2% of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

Based on population estimates, there are at least tens of millions of colonies present in both the Dry Tortugas and U. S. Virgin Islands. Absolute abundance is higher than the estimate from these 2 locations given the presence of this species in many other locations throughout its range. The frequency and extent of partial mortality, especially in larger colonies of boulder star coral, appear to be high in some locations such as Florida and Cuba, though other locations like the Flower Garden Banks appear to have lower amounts of partial mortality. A decrease in boulder star coral percent cover by 38% and a shift to smaller colony size across 5 countries suggest that population decline has occurred in some areas; colony abundance appears to be stable in other areas. We anticipate that while population decline has occurred, boulder star coral is still common with the number of colonies at least in the tens of millions. Additionally, we conclude that the buffering capacity of boulder star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Productivity / Population Growth Rate. The star coral species complex has growth rates ranging from 0.06-1.2 centimeters per year and averaging approximately 1-centimeter linear growth per year. Boulder star coral is reported to be the slowest of the 3 species in the complex (Brainard et al. 2011c). They grow slower in deep or murky waters.

In addition to low recruitment rates, lobed star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the lobed star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

Genetic Diversity. Of 351 boulder star coral colonies observed to spawn at a site off Bocas del Toro, Panama, 324 were unique genotypes. Over 90% of boulder star coral colonies on this reef were the product of sexual reproduction, and 19 genetic individuals had asexually propagated colonies made up of 2–4 spatially adjacent clones of each. Individuals within a genotype spawned more synchronously than individuals of different genotypes. Additionally, within 5 meters, colonies nearby spawned more synchronously than farther spaced colonies, regardless of genotype. At distances greater than 5 meters, spawning was random between colonies (Levitan et al. 2011).

Distribution Boulder star coral is found in the western Atlantic Ocean and throughout the Caribbean Sea including in the Bahamas, Bermuda, and the Flower Garden Banks. Boulder star coral tends to have a deeper distribution than the other 2 species in the *Orbicella* species complex. It occupies most reef environments and has been reported from water depths ranging from approximately 5-50 meters, with the species complex reported to 90 meters. *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e. g. > 30 meters), suggesting the potential for deep refugia for boulder star coral.

Designated Critical Habitat. See Status of 5 Caribbean Coral Critical Habitat below.

Recovery Goals. No final recovery plan currently exists for boulder star coral; however, a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.
- Long Term Goals:
- Cultivate and implement U. S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

8.52 Lobed Star Coral

Table 101. Lobed Star Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Orbicella annularis	Lobed Star	N/A	Threatened	2022	2014 79 FR 53852	2015 (Recovery Outline)	2023 88 FR 54026

Figure 67. Lobed Star Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. The lobed star coral is a cnidarian belonging to the taxonomic genus of Orbicella, a group of stony corals that secrete calcium carbonate to form hard exoskeletons. Lobed Star coral occurs in the western Atlantic and greater Caribbean as well as the Flower Garden Banksbut may be absent from Bermuda. On September 10, 2014, NMFS listed lobed star coral as threatened (79 FR 53851). Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella* spp. complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species – *Montastraea annularis*– with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata*, and *Montastraea* (now *Orbicella*) *franksi.* The 3 species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into 3 species.

Lobed star corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, lobed star coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b). Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other 2 star coral species, margins on the sides of columns are typically dead. Live colony surfaces usually lack ridges or bumps.

Lobed star coral is reported from most reef environments within the Caribbean (except for Bermuda) in depths of approximately 0.5-20 meters. The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >30 meters) reefs, suggesting the potential for deep refuge across a broader depth range, but lobed star coral is generally described with a shallower distribution.

Status. Lobed star coral has undergone major declines mostly due to warming-induced bleaching and disease. Several population projections indicate population decline in the future is likely at specific sites and that local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and increased disease severity with nutrient enrichment. Lobed star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes, as has been observed in locations in the species' range. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these 2 locations due to the occurrence of the species in many other areas throughout its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because lobed star coral is limited to an area with high localized human impacts and predicted increasing threats. Star coral occurs in most reef habitats 0.5-20 meters in depth which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience high temperature variation and ocean chemistry at any given point in time. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing one of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is one previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. The star coral species complex has growth rates ranging from 0.06-1.2 centimeters per year and averaging approximately 1 centimeter in linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 centimeters per year (Cruz-Piñón et al. 2003; Tomascik 1990). They grow slower in deep and murky waters.

All 3 species of the star coral complex are hermaphroditic broadcast spawners, with spawning concentrated on 6 to 8 nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Further, mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1–2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3 species, as it is closely linked to the number of colonies concurrently spawning. Lobed star coral is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction that the other 2 species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 83 square centimeters.

Successful recruitment by the star coral complex species has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 12 square meters of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

In addition to low recruitment rates, lobed star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the lobed star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies

Population Dynamics

Abundance. Lobed star corals are the third most abundant coral species by percent cover in permanent monitoring stations in the U.S.Virgin Islands. A decline of 60% was observed between 2001 and 2012 primarily due to bleaching in 2005. However, most of the mortality was partial mortality and colony density in monitoring stations did not change (Smith 2013). Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined to between 37% and 90% over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S.Virgin Islands, and St.Kitts and Nevis. Based on population estimates, there are at least tens of millions of lobed star coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these 2 locations given the presence of this species in many other locations throughout its range. Lobed star coral remains common in occurrence. Abundance has decreased in some areas to between 19% and 57%, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. We conclude that while substantial population decline has occurred in lobed star coral, it is still common throughout the Caribbean and remains one of the dominant species numbering at least in the tens of millions of colonies. We conclude that the buffering capacity of lobed star coral’s life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (standard error [SE]) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in 2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 90 centimeters size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than 10 centimeters (as low as approximately 5%) and up to approximately 70% in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013a).

Lobed star coral has been described as common overall, although new information indicates it is becoming less common in some locations (NMFS 2022b). Demographic data collected in Puerto Rico over 9 years before and after the 2005 bleaching event showed that population growth rates were stable in the pre-bleaching period (2001–2005) but declined 1 year after the bleaching event. Population growth rates declined even further 2 years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

Productivity / Population Growth Rate. Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area in Belize, lobed star coral cover declined significantly over a 10 year period (1998/99 to 2008/09) (Huntington et al. 2011). In a study of 10 sites inside and outside of a marine reserve in the Exuma Cays, Bahamas, cover of lobed star coral increased between 2004 and 2007 inside the protected area and decreased outside the protected area (Mumby and Harborne 2010). Between 1996 and 2006, lobed star coral declined in cover by 37% in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008a). Cover of lobed star coral declined 71% in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Cover of lobed star coral at Yawzi Point, St.John, U.S.Virgin Islands declined from 41% in 1988 to approximately 12% by 2003 as a rapid decline began with the aftermath of Hurricane Hugo in 1989 (Edmunds and Elahi 2007). This decline continued between 1994 and 1999 during a time of 2 hurricanes (1995) and a year of unusually high sea temperature (1998) but percent cover remained statistically unchanged between 1999 and 2003. Colony abundances declined from 47 to 20 colonies per approximately 1 square meter between 1988 and 2003, due mostly to the death and fission of medium-to-large colonies (≥ 151 square centimeters). Meanwhile, the population size class structure shifted between 1988 and 2003 to a higher proportion of smaller colonies in 2003 (60% less than 50 square centimeters in 1988 versus 70% in 2003) and lower proportion of large colonies (6% greater than 250 square centimeters in 1988 versus 3% in 2003. The changes in population size structure indicated a population decline coincident with the period of apparent stable coral cover. Population modeling forecasted the 1988 size structure would not be reestablished by recruitment and a strong likelihood of extirpation of lobed star coral at this site within 50 years (Edmunds and Elahi 2007).

Genetic Diversity. Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals is variable by location and is reported to range between 18% and 86% (thus, 14-82% are clones). Colonies in areas with higher disturbance from hurricanes tend to have more clonality. Genetic data indicate that there is some population structure in the eastern, central, and western Caribbean with population connectivity within but not across areas. Although lobed star coral is still abundant, it may exhibit high clonality in some locations, meaning that there may be low genetic diversity.

Distribution. Colony density varies by habitat and location, and ranges from less than 0.1 to greater than 1 colony per approximately 10 square meters. In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of lobed star coral ranged between 0.09 and 0.84 colonies per approximately 10 square meters and was highest on mid-channel reefs followed by inshore reefs, offshore patch reefs, and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami (0.34 colonies per approximately 10 square meters) compared to Palm Beach and Broward Counties (ten square meters; Burman et al. 2012). In surveys between 2005 and 2007 along the Florida reef tract from Martin County to the lower Florida Keys, density of lobed star coral was approximately 1.3 colonies per approximately 10 square meters (Wagner et al. 2010). Off southwest Cuba on remote reefs, lobed star coral density was 0.31 ± 0.46 (SD) per approximately 10 meters transect on 38 reef-crest sites and 1.58 ± 1.29 colonies per approximately 10 meters transect on 30 reef-front sites. Colonies with partial mortality were far more frequent than those with no partial mortality which only occurred in the size class less than 100 centimeters) (Alcolado et al. 2010).

Designated Critical Habitat. See Status of Caribbean Coral Critical Habitat below.

Recovery Goals. No final recovery plan currently exists for lobed star coral; however a recovery outline was published in 2014. The following contains the recovery goals listed in the document:

Short Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.
- Long Term Goals:
- Cultivate and implement U.S.and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

8.53 Mountainous Star Coral

Table 102. Mountainous Star Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Orbicella faveolata	Mountainous Star	N/A	Threatened	2022	2014 79 FR 53851	2015 (Recovery Outline)	2023 88 FR 54026

Figure 68. Mountainous Star Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. Mountainous star coral belongs to the taxonomic family of Merulinidae, a group of stony corals whose hard exoskeletons are highly fused and lack paliform lobes. Mountainous Star coral occurs in the western Atlantic and throughout the Caribbean, including the Bahamas, Flower Garden Banks, and the entire Caribbean coastline.

On September 10, 2014, NMFS listed mountainous star coral as threatened (79 FR 53851). Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella* spp. complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species –*Montastraea annularis*– with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata*, and *Montastraea* (now

Orbicella) *franksi.* The 3 species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into 3 species.

Mountainous star corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, mountainous star coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps. The skeleton is much less dense than in the other 2 star coral species. Colony diameters can reach up to 10 meters with heights of 4-5 meters.

As stated, mountainous star coral is found in the western Atlantic and throughout the Caribbean. There is conflicting information on whether or not it occurs in Bermuda. Mountainous star coral has been reported in most reef habitats and is often the most abundant coral at 10 to 20 meters in fore-reef environments. The depth range of mountainous star coral has been reported as approximately 0.5-40 meters, though the species complex has been reported to depths of 90 meters, although it has not been observed in mesophotic areas (e.g., > 30 meters) of the Florida Keys, Dry Tortugas, or Pulley Ridge in the Gulf of Mexico (NMFS 2022b). Star coral species are a common, often dominant component of Caribbean mesophotic reefs, suggesting the potential for deep refugia for mountainous star coral.

Status. Mountainous star coral has undergone major declines mostly due to warming-induced bleaching and disease (Manzello et al. 2015). There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and reduced thermal tolerance due to chronic local stressors stemming from land-based sources of pollution (Grottoli et al. 2014). Mountainous star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate its vulnerability to extinction(Grottoli et al. 2014). Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs (Smith 2013). Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. The buffering capacity of these life history characteristics, however, is expected to decrease as colonies shift to smaller size classes as has been observed in locations in its range. Its absolute population abundance has been estimated as at least tens of millions of colonies in each of several locations including the Florida Keys, Dry Tortugas, and the U.S. Virgin Islands and is higher than the estimate from these 3 locations due to the occurrence of the species in many other areas throughout its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because mountainous star coral is limited to an area with high, localized human impacts and predicted increasing threats. Its depth range of

0.5 meters to at least 40 meters, moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Mountainous star coral occurs in most reef habitats, including both shallow and mesophotic reefs, which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable temperatures and ocean chemistry at any given point in time. Its abundance, life history characteristics, and depth distribution, combined with spatial variability in ocean warming and acidification across the species' range, decreases its vulnerability to extinction because the threats are non-uniform (Smith 2013). Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing one of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is one previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. The star coral species complex has growth rates ranging from 0.06 - 1.2 centimeters per year and averaging approximately 1-centimeter linear growth per year. Mountainous star coral's growth rate is intermediate between the other star coral complex species (Szmant et al., 1997).

The star coral complex species are hermaphroditic broadcast spawners. Spawning is concentrated on 6 to 8 nights following the full moon in late August, September, or early October. All 3 species of star coral are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Fertilization success measured in the field was generally below 15% for all 3 species. In Puerto Rico, the minimum size at reproduction for a star coral species complex was 83 square centimeters.

Successful recruitment by star corals has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 12 square meters of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

Life history characteristics of mountainous star coral is considered intermediate between lobed star coral and boulder star coral especially regarding growth rates, tissue regeneration, and egg size. Spatial distribution may affect fecundity on the reef, with deeper colonies of mountainous star coral being less fecund due to greater polyp spacing. Reported growth rates of mountainous star coral range between 0.3 and 1.6 centimeters per year (Cruz-Piñón et al. 2003; Tomascik 1990; Villinski 2003; Waddell 2005). Graham and van Woesik (2013) report that 44% of small colony mountainous star coral in Puerto Morelos, Mexico that resulted from partial colony mortality produced eggs at sizes smaller than those typically characterized as being mature. The number of eggs produced per unit area of smaller fragments was significantly less than in larger size classes. Szmant and Miller (2005) reported low post-settlement survivorship for mountainous star coral transplanted to the field with only 3% to 15% remaining alive after 30 days. Post-settlement survivorship was much lower than the 29% observed for elkhorn coral after 7 months (Szmant and Miller 2005).

Mountainous star coral has slow growth rates, late reproductive maturity, and low recruitment rates. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, we conclude that the buffering capacity of this life history strategy has been reduced by recent population declines and partial mortality, particularly in large colonies.

Population Dynamics

Abundance / Productivity. Population trend data exists for several locations. At 9 sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over 10 years of monitoring between 1998 and 2008 (Bruckner and Hill 2009). Both mountainous star coral and lobed star coral sustained large losses during the period. The number of colonies of mountainous star coral decreased by 36% and 48% at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27% of all corals at 6 sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11% in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12% of all coral colonies were mountainous star coral in 2000, compared to 7% in 2008.

Extrapolated population estimates from stratified random samples in the Florida Keys were 39.7 ± 8 million (standard error [SE]) colonies in 2005, 21.9 ± 7 million (SE) colonies in 2009, and 47.3 ± 14.5 million (SE) colonies in 2012. The greatest proportion of colonies tended to fall in the 10-20 centimeter and 20-30 centimeter size classes in all survey years, but there was a fairly large proportion of colonies in the greater than 90 centimeter-size class. Partial mortality of the colonies was between 10% and 60% of the surface across all size classes. In the Dry Tortugas, Florida, mountainous star coral ranked 7th most abundant out of 43 coral species in 2006 and fifth most abundant out of 40 in 2008. Extrapolated population estimates were 36.1 ± 4.8 million

(SE) colonies in 2006 and 30 ± 3.3 million (SE) colonies in 2008. The size classes with the largest proportion of colonies were 10-20 centimeter and 20-30 centimeter, but there was a fairly large proportion of colonies in the greater-than-90 centimeter size class. Partial mortality of the colonies ranged between approximately 2% and 50%. Because these population abundance estimates are based on random surveys, differences between years may be attributed to sampling effort rather than population trends (Miller et al. 2013a).

In the U.S. Virgin Islands, the reproductive performance of O.faveolata was assessed over a 5-week period at 3 depth ranges 5–10 meters, 15–22 meters and 35–40 meters. The results showed that corals at the upper edge of the mesophotic zone 35–40 meters were more fecund and produced more eggs than those at shallower depths (Holstein et al. 2016).

Genetic Diversity. Information regarding population structure is limited. Observations of mountainous star coral from 182 sample sites in the upper and lower Florida Keys and Mexico showed 3 well-defined populations based on 5 genetic markers, but the populations were not stratified by geography, indicating they were shared among the 3 regions (Baums et al. 2010). Of 10 mountainous star coral colonies observed to spawn at a site off Bocas del Toro, Panama, there were only 3 genotypes (Levitan et al. 2011) potentially indicating 30% clonality.

Distribution. In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80% of the sites at 1% to 10% cover (Steiner 2003a). In a 1995 survey of 16 reefs in the Florida Keys, mountainous star coral ranked as the coral species with the second highest percent cover (Murdoch and Aronson 1999). On 84 patch reefs 3–5 meters depth spanning 149 miles (240 kilometers) in the Florida Keys, mountainous star coral was the third most abundant coral species comprising 7% of the 17,568 colonies encountered. It was present at 95% of surveyed reefs between 2001 and 2003 (Lirman and Fong 2007). In surveys of 280 sites in the upper Florida Keys in 2011, mountainous star coral was present at 87% of sites visited (Miller et al. 2011). In 2003 on the East Flower Garden Bank, mountainous star coral comprised 10% of the 76.5% coral cover on reefs 32-40 meters, and partial mortality due to bleaching, disease, and predation were rare at monitoring stations (Precht et al. 2005).

Colony density ranges from approximately 0.1-1.8 colonies per 10 square meters and varies by habitat and location. In surveys along the Florida reef tract from Martin County to the lower Florida Keys, density of mountainous star coral was approximately 1.6 colonies per 10 square meters (Wagner et al. 2010). On remote reefs off southwest Cuba, density of mountainous star coral was 0.12 ± 0.20 (SE) colonies per 10 meters transect on 38 reef-crest sites and 1.26 ± 1.06 (SE) colonies per 10 meters transect on 30 reef-front sites (Alcolado et al. 2010). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of mountainous star coral ranged between 0.17 and 1.75 colonies per 10 square meters and was highest on mid-channel reefs followed by offshore patch reefs and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami at 0.94 colonies per 10 square meters compared to 0.11 colonies per 10 square meters in Palm Beach and Broward Counties (Burman et al. 2012).

Designated Critical Habitat. See Status of Caribbean Coral Critical Habitat below.

Recovery Goals. No final recovery plans currently exists for mountainous star coral; however, a recovery outline for the species was published in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling;
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility;
- Decrease locally-manageable stress and mortality sources (e. g. , acute sedimentation, nutrients, contaminants, over-fishing); and
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.

Long Term Goals:

- Cultivate and implement U. S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats; and
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

8.54 Pillar Coral

Table 103. Pillar Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Dendrogyra cylindrus	Pillar Coral	N/A	Threatened Endangered (Proposed)	2022	2023 88 FR 59494 (Proposed)	2015 (Recovery Outline)	2023 88 FR 54026

Figure 69. Pillar Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. The pillar coral is a cnidarian belonging to the taxonomic order of Scleractinia, a group of stony corals that secrete calcium carbonate to form hard exoskeletons. Pillar coral has been described as a naturally rare species, and population status and trends have been difficult to discern due to low encounter rates. The species experienced a decrease in its spatial distribution with the mortality of wild colonies in the northernmost portion of its range in Florida; only two known healthy colonies remained in the Dry Tortugas in 2020, rendering the species functionally extinct in Florida (Neely et al. 2021). In other locations such as Mexico and the US Virgin Islands, local extirpation at specific sites has been reported, rendering the population more fragmented than it was previously (Alvarez-Filip et al. 2019; Brandt et al. 2021). On August 29, 2023, NOAA Fisheries issued a proposed rule to change the status of pillar coral from threatened to endangered (Proposed Rule 88 FR 59494). Pillar corals form tubular columns on top of encrusted foundations. Colonies are generally grey-brown in color and may reach approximately 3 meters in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance. Pillar corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, Pillar coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

Brainard et al. (2011b) identified a single known colony in Bermuda that is in poor condition. There is fossil evidence of the presence of the species off Panama less than 1,000 years ago, but it has been reported as absent today (Florida Fish and Wildlife Conservation Commission 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately 1–25 meters, but it is most common in water between approximately 5–15 meters deep (Acosta and Acevedo 2006; Cairns 1982; Goreau and Wells 1967).

Status. Pillar coral survival is susceptible to a number of threats, and there is evidence of population declines throughout most of its range as well as evidence of several disease impacts. New scientific and commercial data indicates there have been declines in the abundance and distribution in multiple locations with the most severe declines in the northern portions of its range (NMFS 2022b). In addition, the species is highly susceptible to stony coral tissue loss disease (SCTLD), which has emerged as a widespread and deadly new disease. In locations where SCTLD has been observed, pillar coral has experienced high rates of disease, fast disease progression, and high mortality from SCTLD. As a result of SCTLD, pillar coral has disappeared from individual sites in Florida, Mexico, and the U.S. Virgin Islands. In addition, no observed recruitment has been reported in the wild and reductions in population size and local extinctions with inhibit the species' ability to replenish populations through asexual and sexual reproduction.

SCTLD was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). Beginning in 2013, all known colonies of *D. cylindrus* in Florida were tracked in an effort to monitor colony health and status (Neely et al., 2021a). There were consecutive thermal bleaching events in 2014 and 2015, as well as ongoing and emerging disease events, which affected the monitored colonies. By the end of the monitoring period in 2020, there had been a 94% loss of coral tissue, 93% loss of colonies, and 86% loss of genotypes due primarily to disease. Only 2 genotypes remained unaffected and were located in the Dry Tortugas where SCTLD had not yet reached at the time of the study, but has now. Based on the extreme loss of colonies and live tissue, *D. cylindrus* is now considered functionally extinct along the Florida reef tract (Neely et al., 2021a). Coral colonies infected with SCTLD have been effectively treated to stop progression of the disease, but not to prevent new lesions from forming (Neely et al., 2020b; Shilling et al., 2021; Walker et al., 2021). A rescue effort was undertaken to collect fragments of live colonies and bring them under human care in both land-based and ocean-based nurseries for preservation and to aid in propagation and future restoration (Kabay, 2016; Neely et al., 2021b; O'Neil et al., 2021). The species has successfully reproduced in captivity and gametes from wild colonies have successfully transferred to captivity (Marhaver et al., 2015; Villalpando et al., 2021; O'Neil et al., 2021).

Pillar coral is susceptible to multiple threats including ocean warming, disease, acidification, nutrient enrichment, sedimentation, trophic effects of fishing, and inadequate existing regulatory mechanisms to address global threats. Geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because pillar coral is limited to an area with high, localized human impacts and predicted increasing threats. *Dendrogyra cylindrus* inhabits most reef environments in water depths ranging from 1–25 meters, but is naturally rare. It is observed to have low sexual recruitment, limiting its capacity for recovery. It has experienced population declines, resulting in a reduced geographic location and local extirpation. We anticipate that pillar coral is in danger of extinction throughout its range.

Life History. Reported average growth rates for pillar coral have been documented to be approximately 1.8-2.0 centimeters per year in linear extension within the Florida Keys, compared to 0.8 centimeters per year as reported in Colombia and Curaçao. Partial mortality rates are size-specific with larger colonies having greater rates. Frequency of partial mortality can be high (e.g., 65% of 185 colonies surveyed in Colombia), while the amount of partial mortality per colony is generally low (average of 3% of tissue area affected per colony).

Pillar coral is a species with relatively low annual egg production for its size and was previously only known to reproduce through gonochoric broadcast spawning[8]. New evidence of hermaphroditism and plasticity in reproductive mode has been observed in histological samples and in spawning colonies observed over several seasons in Florida (Kabay, 2016, Neely et al., 2018; Neely et al., 2020a; O'Neil et al., 2021). Histological samples from Florida revealed some most colonies produce either egg or sperm, while some hermaphroditic colonies produce both within the same polyp (Kabay, 2016). The species have also been observed to spawn as different genders on different nights of the same year, as different genders in different years, and as hermaphrodites spawning eggs and sperm simultaneously (Neely et al., 2018; Neely et al., 2020a; O'Neil et al., 2021). Spawning observations have also suggested that eggs may be fertilized within female colonies prior to release (Marhaver et al., 2015). The combination of gonochoric spawning with persistently low population densities is expected to yield low rates of successful fertilization and low larval supply. Sexual recruitment of this species is low, and reports indicate juvenile colonies are lacking in the Caribbean. The flexibility in reproductive mode may be a strategy to improve the chances of successful reproduction for a species that is naturally rare and whose potential mates are scarce (Neely et al., 2018).

Population Dynamics

Abundance / Productivity. Pillar coral is uncommon but conspicuous with scattered, isolated colonies and is rarely found in aggregations. In coral surveys, it generally has a rare encounter rate, low percent cover, and low density.

Information on pillar coral is most extensive for Florida. There were consecutive thermal bleaching events in 2014 and 2015, as well as ongoing and emerging disease events, which affected the monitored colonies. Recovery from bleaching was calculated to take 11 years (in the absence of additional severe stressors) based on colony growth rates observed after bleaching, but before disease (Neely et al., 2021a). Modeling of the species was conducted to examine the effects of thermally-induced bleaching stress events. Assuming 2 stress events per decade until

[8] Parents only contain one gamete (egg or sperm), which are released into the water column for fertilization by another parent's gamete.

2042 when thermal stress events are predicted to become annual, local extinction of *D. cylindrus* in Florida was predicted to occur in between 2039 and 2066 (Chan et al., 2019). These modeling predictions did not account for disease, which was observed to cause near extirpation from Florida much sooner than the model's predicted dates for local extinction (Neely et al., 2021a). Between 2013 and 2020, *D. cylindrus* colonies in Florida were monitored, with baseline surveys revealing a total of 542 colonies (533 alive), average tissue mortality of 30%, and recent low levels of mortality (2.2%) in 22% of colonies (Neely et al., 2021a). During the monitoring period, there were chronic stressors that occurred on about 1% of colonies and caused minor damage (on average less than 1% tissue loss), including damselfish gardens/nests, predation by the corallivorous snail (*Coralliophila abbreviata*), competition with other benthic organisms, and abrasion and burial. Acute stressors, including the 2014 and 2015 bleaching events, ongoing outbreaks of white plague and black band disease, and the outbreak of SCTLD, resulted in extremely high mortality (Lewis, 2018; Lewis et al., 2017; Neely et al., 2021a). By the end of the monitoring period in 2020, there had been a 94% loss of coral tissue and 93% loss of colonies, due primarily to disease. The species experienced severe loss of coral tissues and colonies from 2013 to 2020 and is now considered functionally extinct along the Florida reef tract (Neely et al., 2021a, Figure 70).

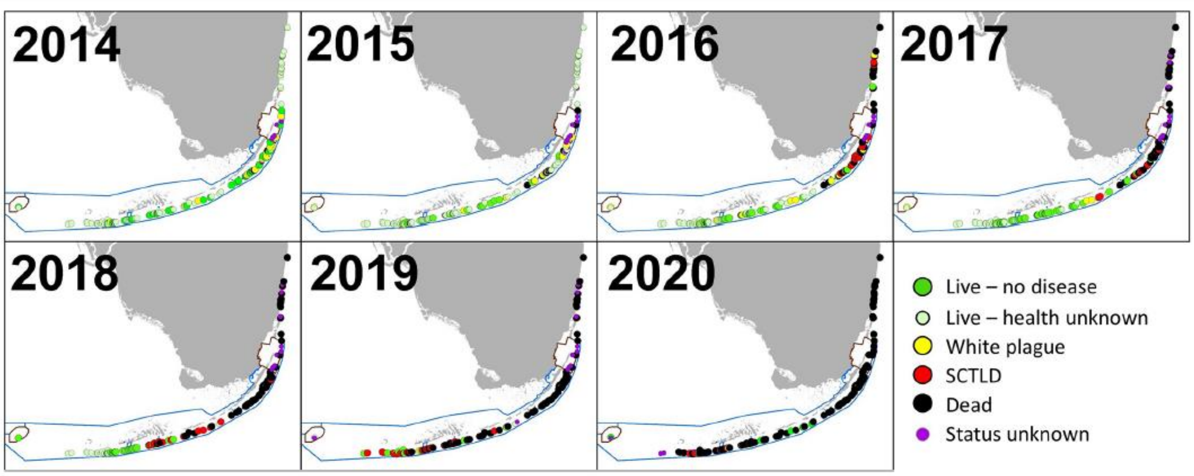


Figure 70. Status of known pillar coral colonies in Florida between 2014 and 2020 (Figure courtesy of K. Neely et al. 2021).

Surveys of *D. cylindrus* were conducted in 2002 and 2012 in Old Providence and St. Catalina Islands, which hosted more than 90% of the *D. cylindrus* population in Colombia, revealing in 2012 the species was present in 3 of the 4 reef areas where it was present in 2002 and occupied a small amount of the reef areas (Bernal-Sotelo et al., 2019). Average colony and fragment size was smaller in 2012, and the number of colonies with partial mortality and the amount of partial mortality were higher. The authors concluded that the reduced amount of living tissue, dominance of asexually produced fragments, and smaller fragment size limit the potential for population growth, making this population vulnerable and at risk of local extinction (Bernal-Sotelo et al., 2019).

Although quantitative population trend data are only available from Florida and Columbia, we assume the species is in decline throughout most of its range based on the evidence from these regions, which represent the northern and southwestern portions of its range, and the widespread evidence of severe disease (NMFS 2022b).

Genetic Diversity / Distribution. The monitoring of Florida *D. cylindrus* colonies from 2013 and 2020 included a total of 819 colonies of an assumed 190 genotypes based on genetic testing or colony distances from each other (Neely et al., 2021a). Distances between genotypes on average was about 1 kilometer (km), ranging from 2.5 m to 6.6 km. Half of the colonies represented clones of only 5 genotypes, with 62% of the genotypes represented by a single colony. Asexual reproduction accounted for 77% of the colonies. At the end of 2020, there had been an 86% loss of genotypes, with only 25 known genotypes remaining. Half of the remaining genotypes declined to less than 2% live tissue and the other half were actively experiencing rapid tissue loss due to SCTLD. Only 2 genotypes remained unaffected and were located in the Dry Tortugas where SCTLD had not yet reached at the time of the study, but has now.

Designated Critical Habitat. See Status of Caribbean Coral Critical Habitat below.

Recovery Goals. No final recovery plans currently exists for pillar coral, however a recovery outline was published in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling;
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility;
- Decrease locally-manageable stress and mortality sources (e. g. , acute sedimentation, nutrients, contaminants, over-fishing); and
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.

Long Term Goals:

- Cultivate and implement U. S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats; and
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

8.55 Rough Cactus Coral

Table 104. Rough Cactus Coral; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Mycetophylllia ferox	Rough Cactus	N/A	Threatened	2022	2014 79 FR 53851	2015 (Recovery Outline)	2023 88 FR 54026

Figure 71. Rough Cactus Coral distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. The rough cactus coral is a cnidarian belonging to the taxonomic genus of Mycetophyllia, a group of ridged corals that form colonies with a flat disc shape. Rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea.

Rough cactus coral forms a thin, encrusting plate that is weakly attached to substrate. Rough cactus coral is taxonomically distinct (i.e., separate species), though difficult to distinguish in the field from other *Mycetophyllia* species. The maximum colony size of the species is 50 centimeters in diameter. Rough cactus corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. As with most corals, in addition to being able to catch and eat their own food, rough cactus coral contains zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005b).

While rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea, it has not been reported in the Flower Garden Banks (Gulf of Mexico) or in

Bermuda. It inhabits reef environments in water depths of 5–90 meters, including shallow and mesophotic habitats (e.g., > 30 meters).

Status. Rough cactus coral has declined due to disease in at least a portion of its range and has low recruitment, which limits its capacity for recovery from mortality events and exacerbates vulnerability to extinction. Its depth range of 5 to 90 meters moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters. Acidification is predicted to accelerate most in deeper and cooler waters than those in which the species occurs. Its habitat includes shallow and mesophotic reefs which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Rough cactus coral is usually uncommon to rare throughout its range. Its abundance, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time.

Numerous diseases have been documented with increasing frequency since the first reports of coral disease in the Florida Keys emerged in the 1970s (Porter et al. 2001). The Florida Reef Tract is currently experiencing one of the most widespread and virulent disease outbreaks on record: stony coral tissue loss disease (Sharp & Maxwell, 2018). This disease is one previously unknown and its outbreak has resulted in the mortality of thousands of colonies of at least 20 species of scleractinians, including primary reef builders and ESA-listed species (Sharp and Maxwell 2018). The disease was first reported near Key Biscayne in 2014 (Precht et al. 2005) and progressed southward along the Florida Reef Tract, reaching Key West by December 2017 (Sharp and Maxwell 2018). The disease has since spread to St. Thomas in USVI, Bahamas, Jamaica, Mexico, and likely other locations throughout the Caribbean. A limited understanding of the disease outbreak, due to limited diagnostic capacity, and its mode and rate of transmission, has greatly hindered management efforts to control or prevent the spread of the disease (Sharp and Maxwell 2018).

Life History. Rough cactus coral is a hermaphroditic brooding species. Colony size at first reproduction is greater than 100 square centimeters. Recruitment of rough cactus coral appears to be very low, even in studies from the 1970s. Rough cactus coral has a lower fecundity compared to other species in its genus (Morales Tirado 2006). Over a 10 year period, no colonies of rough cactus coral were observed to recruit to an anchor-damaged site in the U.S. Virgin Islands, although adults were observed on the adjacent reef (Rogers and Garrison 2001). No other life history information appears to exist for rough cactus coral.

Population Dynamics

Abundance / Productivity. Rough cactus coral is usually uncommon or rare according to published and unpublished records, indicating that it constitutes < 0.1% species contribution (percent of all colonies counted) and occurs at densities < 0.8 colonies per 10 square meters in Florida and at 0.8 colonies per 100 meters transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (Veron 2002, Wagner el al., 2010, and AGRRA database as cited in Brainard et al. 2011d). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera Puerto Rico, and St. Croix (USVI/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show *Mycetophyllia ferox* cover to be consistently less occasional observations up to 2% and no apparent temporal trend (Brainard et al. 2011d).

Dustan (1977) proposes that rough cactus coral was much more abundant in the upper Florida Keys in the early mid- 1970s (the methods are not well described for that study) than current observations, but that it was highly affected by disease. This could be interpreted as a substantial decline. Long-term CREMP monitoring data in Florida on species presence/absence from fixed sites (stations) show a dramatic decline; for 97 stations in the main Florida Keys, occurrence had declined from 20 stations in 1996 to 4 stations in 2009; in Dry Tortugas occurrence had declined from 8 out of 21 stations in 2004 to 3 stations in 2009 (R. Ruzicka and M. Colella, Florida Marine Research Institute, St. Petersburg, FL. pers. comm., Oct 2010 cited in Brainard et al. 2011d).

Genetic Diversity / Distribution. According to the IUCN Species Account and the CITES species database, rough cactus coral occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al. 2008). The following areas include locations within federally protected waters where rough cactus coral has been observed and recorded (cited in Brainard et al. 2011d):Dry Tortugas National Park;Virgin Island National Park/Monument; Florida Keys; National Marine Sanctuary; Navassa Island National Wildlife Refuge; Biscayne National Park; Buck Island Reef National Monument.

On reefs where rough cactus coral is found, it generally occurs at abundances of less than 1 colony per approximately 10 square meters and percent cover of less than 0.1 (Burman et al. 2012). Based on population estimates, there are at least hundreds of thousands of rough cactus coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these 2 locations given the presence of this species in many other locations throughout its range. Low encounter rate and percent cover coupled with the tendency to include *Mycetophyllia spp.* at the genus level make it difficult to discern population trends of rough cactus coral from monitoring data. However, reported losses of rough cactus coral from monitoring stations in the Florida Keys and Dry Tortugas (63-80% loss) indicate population decline in these locations. Based on declines in Florida, we conclude rough cactus coral has likely declined throughout its range, and will continue to decline based on increasing threats. As a result it is presumed that genetic diversity for the species is low.

Designated Critical Habitat. See Status of Caribbean Coral Critical Habitat below.

Recovery Goals. No final recovery plan currently exists for rough cactus coral, however a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling;
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility;
- Decrease locally-manageable stress and mortality sources (e. g. , acute sedimentation, nutrients, contaminants, over-fishing); and
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.

Long Term Goals:

- Cultivate and implement U. S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats; and
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

8.56 Status of 5 Caribbean Coral Critical Habitats (lobed star coral, mountainous star coral, boulder star coral, pillar coral, and rough cactus coral)

In the final listing rule for lobed star coral, mountainous star coral, boulder star coral, pillar coral, and rough cactus coral, NMFS identified the major threats contributing to the species extinction risk as ocean warming, disease, ocean acidification, tropic effects of reef fishing, nutrient enrichment, and sedimentation. Of these threats, all but disease affect corals in part by changing coral habitat, making it unsuitable for corals to carry out the essential functions at all life stages. NMFS determined that protecting the essential features of coral habitat from these threats will facilitate recovery of these 5 species.

In 2023, 28 mostly overlapping specific occupied areas containing PBFs essential to the conservation of 5 species of ESA-listed corals (lobed star coral, mountainous star coral, boulder star coral, pillar coral, and rough cactus coral) were designated as critical habitat. These areas contain approximately 16,830 km^2 (6,500 nm^2) of marine habitat. The critical habitat boundaries are described in Table 105, which includes the locations of the critical habitat units for the 5 species of Caribbean corals. Depth contours or other identified boundaries form the boundaries of the critical habitat units. Specifically, the Convention on the International regulations for Preventing Collisions at Sea (COLREGS, 1972) Demarcation Lines (33 C.F.R. 80), the boundary between the SAFMC and Gulf Council (50 C.F.R. 600.105), the Florida Keys National Marine Sanctuary boundary (15 C.F.R. Part 922 Subpart P, Appendix I) and the Caribbean Islands Management Area (50 C.F.R. Part 622, Appendix E) create portions of the boundaries in several of the critical habitat units.

There are 5 or 6 specific areas per species within which the individual species' specific areas are largely overlapping. The difference between each of the areas is the particular depth contours used to create the boundaries. Overlaying the specific areas for each species results in the maximum geographic extent of the areas under consideration for designation, which covers 0.5-90 m (1.6-295 ft) water depth around all the islands of Puerto Rico, USVI, and Navassa, FGBNMS and 0.5-40 m (1.6-131.2 ft) from St. Lucie Inlet, Martin County to Dry Tortugas, Florida.

Within the geographic area occupied by these 5 ESA-listed coral species, designated critical habitat consists of specific areas where the PBFs essential to the conservation of each species are found. The PBF essential to the conservation of these 5 ESA-listed corals (lobed star coral, mountainous star coral, boulder star coral, pillar coral, and rough cactus coral) is reproductive, recruitment, growth, and maturation habitat found in the Caribbean, Florida, and Gulf of Mexico. Sites that support the normal function of all life stages of these 5 ESA-listed coral species are natural, consolidated hard substrate or dead coral skeleton, which is free of algae and sediment at the appropriate scale at the point of larval settlement or fragment reattachment, and the associated water column. Several attributes of these sites determine the quality of the area and influence the value of the associated feature to the conservation of the species:

- Substrate with the presence of crevices and holes that provide cryptic habitat, the presence of microbial biofilms, or the presence of crustose coralline algae;
- Reefscape with no more than a thin veneer of sediment and low occupancy by fleshy and turf macroalgae;

- Marine water with levels of temperature, aragonite saturation, nutrients, and water clarity that have been observed to support any demographic function; and
- Marine water with levels of anthropogenically-introduced (from humans) chemical contaminants that do not preclude or inhibit any demographic function.

Naval Air Station Key West, which includes the land and waters (generally out to 45.7 meters (50 yards) adjacent to the base for a total of approximately 800 in-water acres is excluded from the designated critical habitat designation. The Integrated Natural Resources Management Plan (INRMP) for the base was determined by NMFS to provide a benefit to the 4 ESA-listed coral species (pillar coral, lobed star, mountainous star, and boulder star) found within the in-water area of the base.

Table 105. Locations of the designated critical habitat units for 5 species of Caribbean corals.

Species	Critical Habitat Unit Name	Location	Geographic Extent	Water Depth Range (m)
Lobed Star Coral (*Orbicella annularis*)	OANN-1	Florida	Lake Worth Inlet, Palm Beach County to Government Cut, Miami-Dade County	2 to 20
	OANN-1	Florida	Government Cut, Miami-Dade County to Dry Tortugas	0.5 to 20
	OANN-2	Puerto Rico	All Islands	0.5 to 20
	OANN-3	U.S. Virgin Islands (USVI)	All Islands of St. Thomas and St. John	0.5 to 20
	OANN-4	USVI	All Islands of St. Croix	0.5 to 20
	OANN-5	Navassa	Navassa Island	05 to 20
	OANN-6	Flower Gardens Bank (FGB)	East and West FGB, Rankin, Geyer, and McGrail Banks	16 to 90

Mountainous Star Coral (*Orbicella faveolata*)	OFAV-1	Florida	St. Lucie Inlet, Martin County to Government Cut, Miami-Dade County	2 to 40
	OFAV-1	Florida	Government Cut, Miami-Dade County to Dry Tortugas	.5 to 40
	OFAV-2	Puerto Rico	All Islands of Puerto Rico	0.5 to 90
	OFAV-3	USVI	All Islands of St. Thomas and St. John	0.5 to 90
	OFAV-4	USVI	All Islands of St. Croix	0.5 to 90
	OFAV-5	Navassa	Navassa Island	0.5 to 90
	OFAV-6	FGB	East and West FGB, Rankin Geyer, and McGrail Banks	16 to 90
Boulder Star Coral (*Orbicella franksi*)	OFRA-1	Florida	St. Lucie Inlet, Martin County to Government Cut, Miami-Dade County	2 to 40
	OFRA-1	Florida	Government Cut, Miami-Dade County to Dry Tortugas, Monroe County	0.5 to 40
	OFRA-2	Puerto Rico	All Islands of Puerto Rico	0.5 to 90
	OFRA-3	USVI	All Islands of St. Thomas and St.	0.5 to 90

			John	
	OFRA-4	USVI	All Islands of St. Croix	0.5 to 90
	OFRA-5	Navassa	Navassa Island	0.5 to 90
	OFRA-6	FGB	East and West FGB, Rankin, Geye, and McGrail Banks	16 to 90
Pillar Coral (*Dendrogyra cylindrus*)	DCYL-1	Florida	Lake Worth Inlet, Palm Beach County to Government Cut, Miami-Dade County	2 to 25
	DCYL-1	Florida	Government Cut, Miami-Dade County to Dry Tortugas	1 to 25
	DCYL-2	Puerto Rico	All Islands	1 to 25
	DCYL-3	USVI	All Islands of St. Thomas and St. John	1 to 25
	DCYL-4	USVI	All Island of St. Croix	1 to 25
	DCYL-5	Navassa	Navassa Island	1 to 25
Rough Cactus Coral (*Mycetophyllia ferox*)	MFER-1	Florida	Broward County to Dry Tortugas	5 to 40
	MFER-2	Puerto Rico	All Islands of Puerto Rico	5 to 90
	MFER-3	USVI	All Islands of St. Thomas and St. John	5 to 90
	MFER-4	USVI	All Islands of St. Croix	2 to 40

	MFER-5	Navassa	Navassa Island	5 to 40

m=meter, USVI=U.S. Virgin Islands, FGB=Flower Garden Banks

8.57 Coral species: *Acropora retusa*

Table 106. *Acropora retusa*; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acropora retusa	Not Available	NA	Threatened	2021 (Initiated)	2014 79 FR 53852	2015 (Recovery Outline)	2023 88 FR 83644 (Proposed)

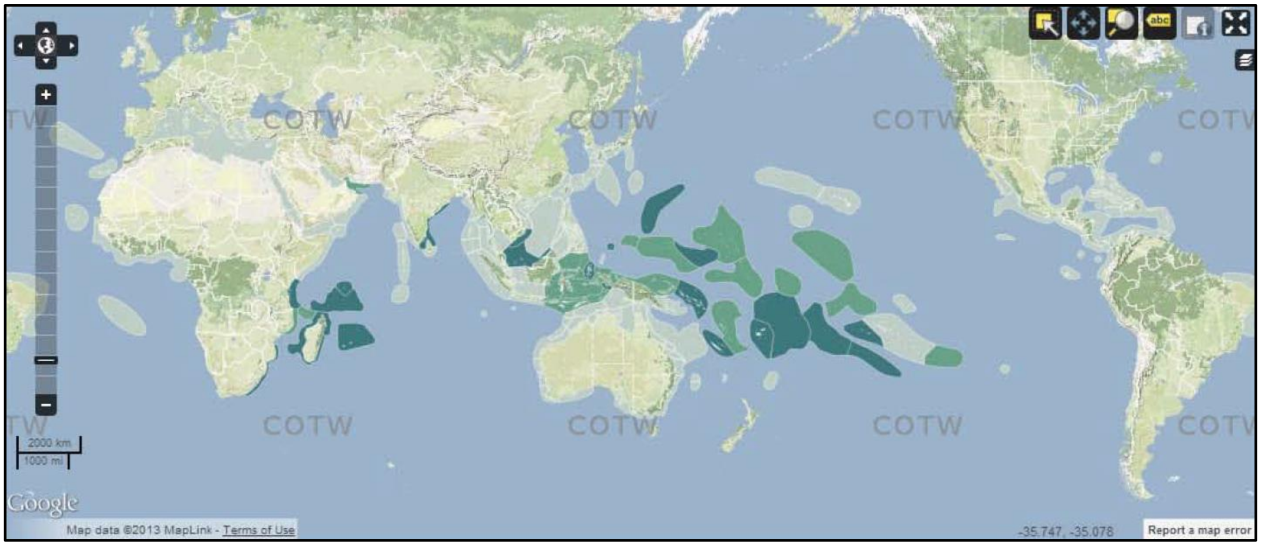


Figure 72. *Acropora retusa* distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation(Veron 2014).

Species Description. Colonies of *Acropora retusa* are flat plates with short thick digitate branchlets. Corallites have thick rounded walls and wide openings. Axial corallites are indistinct. Radial corallites are laying flat against each other, becoming nariform near branch ends. Colonies are brown in color. (Veron, 2000; Veron and Wallace, 1984). *Acropora retusa* is distributed from the Red Sea and the Indian Ocean to the central Pacific.

Status. *Acropora retusa* is highly susceptible to ocean warming, disease, ocean acidification, trophic effects of fishing, predation, and nutrients. These threats are expected to continue and increase into the future. In addition, existing regulatory mechanisms addressing global threats that contribute to extinction risk for this species inadequate. *Acropora retusa* is restricted to shallow habitat (0–5 meters), where many global and local threats may be more severe, especially near populated areas. Shallow reef areas are often subjected to highly variable environmental conditions, extremes, high irradiance, and simultaneous effects from multiple stressors, both local and global in nature. A limited depth range also reduces the absolute area in which the species may occur throughout its geographic range, and indicates that a large proportion of the population is likely to be exposed to threats that are worse in shallow habitats, such as simultaneously elevated irradiance and seawater temperatures, as well as localized impacts. *Acropora retusa' s* abundance is considered rare overall. This level of abundance, combined with its restricted depth distribution where impacts are more severe, leaves the species vulnerable to becoming of such low abundance within the foreseeable future that it may be at risk from depensatory processes, environmental stochasticity, or catastrophic events. The combination of these characteristics and future projections of threats indicates that the species is likely to be in danger of extinction within the foreseeable future throughout its range**.**

Life History. *Acropora* are sessile colonies that spawn their gametes into the water column, and the azooxanthellate larvae can survive in the planktonic stage from 4 to 209 days (Graham et al., 2008). This has allowed many *Acropora* species to have very wide geographic ranges, both longitudinally and latitudinally (Wallace, 1999). However, sessile colonies must be within a few meters of each other to have reasonable success in fertilization (Coma and Lasker, 1997). Vollmer and Palumbi (2007), using DNA sequence data, determined that *Acropora cervicornis* in the Caribbean have limited realized gene flow despite long-distance dispersal potential. Although spawners with long larval lives can eventually become distributed over broad geographic areas, as is typical for *Acropora*, the year-by-year replenishment of populations requires local source populations. All species of the genus *Acropora* studied to date are simultaneous hermaphrodites (Baird et al., 2009), with a gametogenic cycle in which eggs develop over a period of about 9 months and testes over about 10 weeks (Babcock et al., 1986; Szmant, 1986; Wallace, 1985). Fecundity in *Acropora* colonies is generally described as ranging from 3.6 to 15.8 eggs per polyp (Kenyon, 2008; Wallace, 1999). Mature eggs of species of *Acropora* are large when compared with those of other corals, ranging from 0.53 to 0.90 mm in mean diameter (Wallace, 1999). For 5 *Acropora* species examined by Wallace (1985), the minimum reproductive size ranged from 4 to 7 cm, and the estimated ages ranged from 3 to 5 years.

Acropora spp. release gametes as egg-sperm bundles that float to the sea surface, each polyp releasing all its eggs and sperm in 1 bundle. Fertilization takes place after the bundles break open at the sea surface. Sperm concentrations of 106 ml-1 have been found to be optimal for fertilization in the laboratory, and concentrations of this order have been recorded in the field during mass spawning events. Self-fertilization, although possible, is infrequent. Gametes remain viable and achieve high fertilization rates for up to 8 hours after spawning (Kenyon, 1994). Embryogenesis takes place over several hours, and further development leads to a planula that is competent to settle in 4 to 5 days after fertilization. *Acropora spp*. can show a high degree of hybridization (Kenyon, 1994; Richards et al., 2008b; Van Oppen et al., 2002; Van Oppen et al., 2000), which can complicate taxonomic classification but allow persistence of the genus if the hybrids are reproductively viable.

As sessile spawners with planktonic larvae, the Critical Risk Threshold assessments for *Acropora* species must weigh the broad distributions that provide replicated opportunities for potential escape from local disturbances against the necessity to have colonies in close enough proximity to have successful fertilization of enough eggs to replenish the attrition of the spawning stock. If the effective population size (i.e., the number of genotypes [might be substantially less than the number of colonies in highly clonal species] close enough for successful fertilization) becomes too low to replenish the population, then the positive-feedback depensatory processes begin. It is worth noting that Edinger and Risk (1995) concluded that brooding corals survived the harsh environmental conditions better than did the spawners in the western Atlantic during the major extinctions of the Oligocene-Miocene transition period. Many *Acropora* have branching morphologies, making them potentially susceptible to fragmentation. Fragment survival can increase coral abundance in the short-term but does not contribute new genotypes (or evolutionary opportunities) to the population.

Population Dynamics

Abundance. Veron (2014) reports that A. retusa occupied 0.5% of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.21 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as "rare." Overall abundance was described as "common in South Africa, rare elsewhere." The absolute abundance of this species is likely at least millions of colonies (Richards et al. 2008; Veron 2014).

Productivity. The overall decline in abundance ("Percent Population Reduction") was estimated at 49%, and the decline in abundance before the 1998 bleaching event ("Back-cast Percent Population Reduction") was estimated at 18%. However, live coral cover trends are highly variable both spatially and temporally, producing patterns on small scales that can be easily taken out of context, thus quantitative inferences to species-specific trends should be interpreted with caution. At the same time, an extensive body of literature documents broad declines in live coral cover and shifts to reef communities dominated by hardier coral species or algae over the past 50 to 100 years (Birkeland, 2004; Fenner, 2012; Pandolfi et al., 2003; Sale and Szmant, 2012). These changes have likely occurred, and are occurring, from a combination of global and local threats. Given that A. retusa occurs in many areas affected by these broad changes, and that it has some susceptibility to both global and local threats, we conclude that it is likely to have declined in abundance over the past 50 to 100 years, but a precise quantification is not possible due to the limited amount of species-specific information.

Genetic Diversity. Although spawners with long larval lives can eventually become distributed over broad geographic areas, as is typical for *Acropora*, the year-by-year replenishment of populations requires local source populations. For example, Vollmer and Palumbi (2007), using DNA sequence data, determined that *Acropora cervicornis* in the Caribbean have limited realized gene flow despite long-distance dispersal potential.

Distribution. *Acropora retusa* has been reported to occupy upper reef slopes and tidal pools (Veron, 2000; Veron and Wallace, 1984). *Acropora retusa* has been reported in water depths ranging from 1 meter to 5 meters (Carpenter et al., 2008).

Designated Critical Habitat. See Status of Proposed Indo-Pacific Coral Critical Habitat below.

Recovery Goals. A recovery plan has not yet been developed for this species. However, a recovery outline has been developed (NMFS 2015a).

8.58 Coral species: *Acropora globiceps*

Table 107. *Acropora globiceps*; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acropora globiceps	Not Available	NA	Threatened	2021 (Initiated)	2014 79 FR 53852	2015 (Recovery Outline)	2023 88 FR 83644 (Proposed)

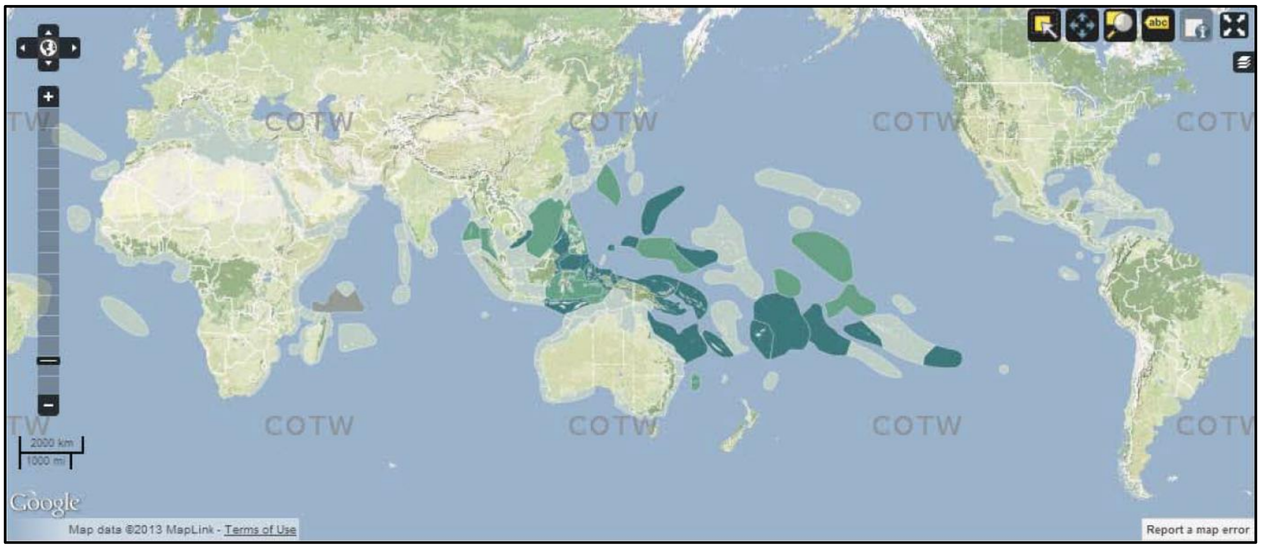


Figure 73. *Acropora globiceps* distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. Colonies of *Acropora globiceps* are digitate and usually small. The size and appearance of branches depend on degree of exposure to wave action but are always short and closely compacted. Colonies exposed to strong wave action have pyramid-shaped branchlets. Corallites are irregular in size, those on colonies on reef slopes are tubular, and those on reef flat colonies are more immersed. Axial corallites are small and sometimes indistinguishable. Radial corallites are irregular in size and are sometimes arranged in rows down the sides of branches. Colonies are uniform blue (which may photograph purple) or cream in color (Veron, 2000).

Status. *Acropora globiceps* is highly susceptible to ocean warming, disease, ocean acidification, trophic effects of fishing, nutrients, and predation. These threats are expected to continue and increase into the future. In addition, existing regulatory mechanisms to address global threats that contribute to extinction risk for this species are inadequate. Acropora globiceps occurs primarily in depths of 0 to 8 meters which can be considered a shallow depth range compared to the overall depth of occurrence for reef building corals in general. Shallow reef areas are often subjected to highly variable environmental conditions, extremes, high irradiance, and simultaneous effects from multiple stressors, both local and global in nature. A limited depth range reduces the absolute area in which the species may occur throughout its geographic range and indicates that a large proportion of the population is likely to be exposed to threats that are worse in shallow habitats, such as simultaneously elevated irradiance and seawater temperatures, as well as localized impacts. The combination of these characteristics and future projections of threats indicates that the species is likely to be in danger of extinction within the foreseeable future throughout its range.

Life History. *Acropora* are sessile colonies that spawn their gametes into the water column, and the azooxanthellate larvae can survive in the planktonic stage from 4 to 209 days (Graham et al., 2008). This has allowed many *Acropora* species to have very wide geographic ranges, both longitudinally and latitudinally (Wallace, 1999). However, sessile colonies must be within a few meters of each other to have reasonable success in fertilization (Coma and Lasker, 1997). Vollmer and Palumbi (2007), using DNA sequence data, determined that *Acropora cervicornis* in the Caribbean have limited realized gene flow despite long-distance dispersal potential. Although spawners with long larval lives can eventually become distributed over broad geographic areas, as is typical for *Acropora*, the year-by-year replenishment of populations requires local source populations. All species of the genus *Acropora* studied to date are simultaneous hermaphrodites (Baird et al., 2009), with a gametogenic cycle in which eggs develop over a period of about 9 months and testes over about 10 weeks (Babcock et al., 1986; Szmant, 1986; Wallace, 1985). Fecundity in *Acropora* colonies is generally described as ranging from 3.6 to 15.8 eggs per polyp (Kenyon, 2008; Wallace, 1999). Mature eggs of species of *Acropora* are large when compared with those of other corals, ranging from 0.53 to 0.90 mm in mean diameter (Wallace, 1999). For 5 *Acropora* species examined by Wallace (1985), the minimum reproductive size ranged from 4 to 7 cm, and the estimated ages ranged from 3 to 5 years.

Acropora spp. release gametes as egg-sperm bundles that float to the sea surface, each polyp releasing all its eggs and sperm in 1 bundle. Fertilization takes place after the bundles break open at the sea surface. Sperm concentrations of 106 ml-1 have been found to be optimal for fertilization in the laboratory, and concentrations of this order have been recorded in the field during mass spawning events. Self-fertilization, although possible, is infrequent. Gametes remain viable and achieve high fertilization rates for up to 8 hours after spawning (Kenyon, 1994). Embryogenesis takes place over several hours, and further development leads to a planula that is competent to settle in 4 to 5 days after fertilization. *Acropora spp.* can show a high degree of hybridization (Kenyon, 1994; Richards et al., 2008b; Van Oppen et al., 2002; Van Oppen et al., 2000), which can complicate taxonomic classification but allow persistence of the genus if the hybrids are reproductively viable.

As sessile spawners with planktonic larvae, the Critical Risk Threshold assessments for *Acropora* species must weigh the broad distributions that provide replicated opportunities for potential escape from local disturbances against the necessity to have colonies in close enough proximity to have successful fertilization of enough eggs to replenish the attrition of the spawning stock. If the effective population size (i.e., the number of genotypes [might be substantially less than the number of colonies in highly clonal species] close enough for successful fertilization) becomes too low to replenish the population, then the positive-feedback depensatory processes begin. It is worth noting that Edinger and Risk (1995) concluded that brooding corals survived the harsh environmental conditions better than did the spawners in the western Atlantic during the major extinctions of the Oligocene-Miocene transition period. Many *Acropora* have branching morphologies, making them potentially susceptible to fragmentation. Fragment survival can increase coral abundance in the short-term but does not contribute new genotypes (or evolutionary opportunities) to the population.

Population Dynamics

Abundance. Veron (2014) reports that A. globiceps occupied 3.2% of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.95 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as “uncommon.” Overall abundance was described as “sometimes common.” Veron did not infer trends in abundance from these data. As described in the Indo-Pacific Species Determinations introduction above, based on results from Richards et al. (2008) and Veron (2014), the absolute abundance of this species is likely at least tens of millions of colonies.

Productivity. The overall decline in abundance (“Percent Population Reduction”) was estimated at 35%, and the decline in abundance before the 1998 bleaching event (“Back-cast Percent Population Reduction”) was estimated at 14% (Carpenter et al., 2008). However, live coral cover trends are highly variable both spatially and temporally, producing patterns on small scales that can be easily taken out of context, thus quantitative inferences to species-specific trends should be interpreted with caution. At the same time, an extensive body of literature documents broad declines in live coral cover and shifts to reef communities dominated by hardier coral species or algae over the past 50 to 100 years (Birkeland, 2004; Fenner, 2012; Pandolfi et al., 2003; Sale and Szmant, 2012). These changes have likely occurred, and are occurring, from a combination of global and local threats. Given that A. globiceps occurs in many areas affected by these broad changes, and that it has some susceptibility to both global and local threats, we conclude that it is likely to have declined in abundance over the past 50 to 100 years, but a precise quantification is not possible due to the limited species-specific information.

Genetic Diversity. Although spawners with long larval lives can eventually become distributed over broad geographic areas, as is typical for *Acropora*, the year-by-year replenishment of populations requires local source populations. For example, Vollmer and Palumbi (2007), using DNA sequence data, determined that *Acropora cervicornis* in the Caribbean have limited realized gene flow despite long-distance dispersal potential.

Distribution. *Acropora globiceps* is distributed from the oceanic west Pacific to the central Pacific as far east as the Pitcairn Islands. In the US *Acropora globiceps* occurs in American Samoa, the Northern Mariana Islands, and the U.S. minor outlying islands. *Acropora globiceps* has been reported from intertidal, upper reef slopes and reef flats (Veron, 2000). *Acropora globiceps* has been reported in water depths ranging from 0 meters to 8 meters (Veron, 2000).

Designated Critical Habitat. See Status of Proposed Indo-Pacific Coral Critical Habitat below.

Recovery Goals. A recovery plan has not yet been developed for this species. However, a recovery outline has been developed (NMFS 2015a).

8.59 Coral species: *Acropora speciosa*

Table 108. *Acropora speciosa*; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acropora speciosa	Not Available	NA	Threatened	2021 (Initiated)	2014 79 FR 53852	2015 (Recovery Outline)	2023 88 FR 83644 (Proposed)

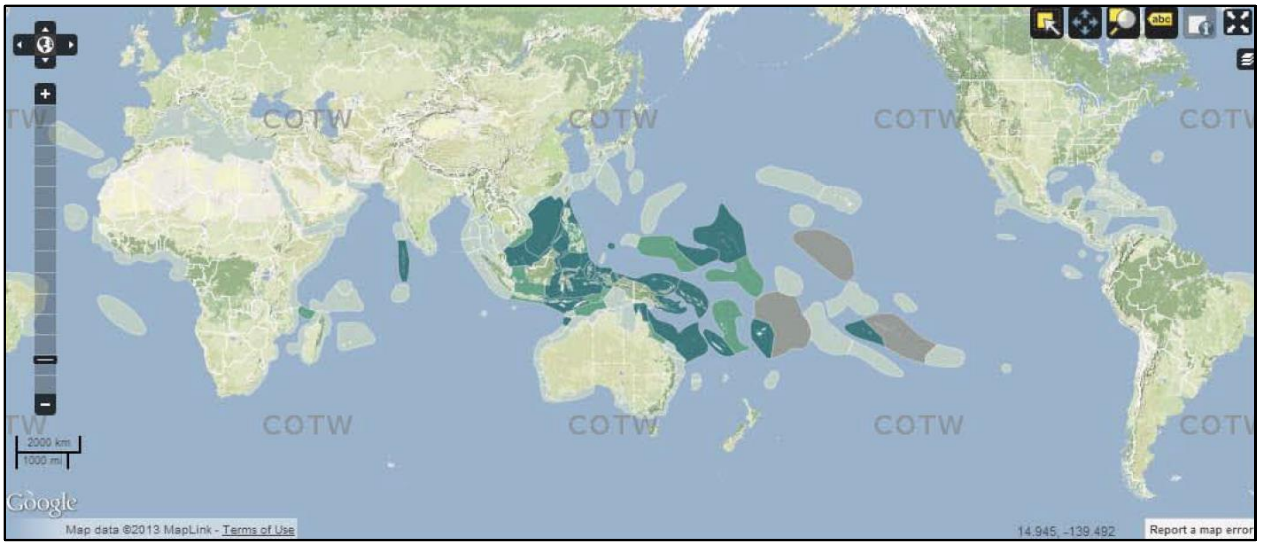


Figure 74. *Acropora speciosa* distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. Colonies of *Acropora speciosa* form thick cushions or bottlebrush branches. They have large and elongate axial corallites; radial corallites are small and tubular or pocketed. Colonies are cream in color with delicately colored branch tips (Veron, 2000). *Acropora speciosa* is distributed from Indonesia to French Polynesia. The IUCN database lists it in American Samoa, and U.S. minor outlying islands.

Status. *Acropora speciosa* is highly susceptible to ocean warming, disease, ocean acidification, trophic effects of fishing, predation, and nutrient enrichment. These threats are expected to continue and increase into the future. In addition, existing regulatory mechanisms to address global threats that contribute to extinction risk for this species are inadequate. Although *A. speciosa' s* habitat includes mesophotic depths which may provide some buffering capacity against threats that are more severe in shallower reef environments such as warming, its habitat is quite specialized, which may limit buffering capacity if threats are more pronounced within the type of habitat where the species occurs within. *Acropora speciosa's* effective population size of 1.2 million genetically distinct colonies could increase vulnerability to extinction if a high proportion of the effective population occurs within the parts of its range most affected by threats, potentially causing the species to decline to such low abundance within the foreseeable future that it may be at risk from depensatory processes, environmental stochasticity, or catastrophic events. The combination of these characteristics and projections of future threats indicates that the species is likely to be in danger of extinction within the foreseeable future throughout its range.

Life History. *Acropora* are sessile colonies that spawn their gametes into the water column, and the azooxanthellate larvae can survive in the planktonic stage from 4 to 209 days (Graham et al., 2008). This has allowed many *Acropora* species to have very wide geographic ranges, both longitudinally and latitudinally (Wallace, 1999). However, sessile colonies must be within a few meters of each other to have reasonable success in fertilization (Coma and Lasker, 1997). Vollmer and Palumbi (2007), using DNA sequence data, determined that *Acropora cervicornis* in the Caribbean have limited realized gene flow despite long-distance dispersal potential. Although spawners with long larval lives can eventually become distributed over broad geographic areas, as is typical for *Acropora*, the year-by-year replenishment of populations requires local source populations. All species of the genus *Acropora* studied to date are simultaneous hermaphrodites (Baird et al., 2009), with a gametogenic cycle in which eggs develop over a period of about 9 months and testes over about 10 weeks (Babcock et al., 1986; Szmant, 1986; Wallace, 1985). Fecundity in *Acropora* colonies is generally described as ranging from 3.6 to 15.8 eggs per polyp (Kenyon, 2008; Wallace, 1999). Mature eggs of species of *Acropora* are large when compared with those of other corals, ranging from 0.53 to 0.90 mm in mean diameter (Wallace, 1999). For 5 *Acropora* species examined by Wallace (1985), the minimum reproductive size ranged from 4 to 7 cm, and the estimated ages ranged from 3 to 5 years.

Acropora spp. release gametes as egg-sperm bundles that float to the sea surface, each polyp releasing all its eggs and sperm in 1 bundle. Fertilization takes place after the bundles break open at the sea surface. Sperm concentrations of 106 ml-1 have been found to be optimal for fertilization in the laboratory, and concentrations of this order have been recorded in the field during mass spawning events. Self-fertilization, although possible, is infrequent. Gametes remain viable and achieve high fertilization rates for up to 8 hours after spawning (Kenyon, 1994). Embryogenesis takes place over several hours, and further development leads to a planula that is competent to settle in 4 to 5 days after fertilization. *Acropora spp.* can show a high degree of hybridization (Kenyon, 1994; Richards et al., 2008b; Van Oppen et al., 2002; Van Oppen et al., 2000), which can complicate taxonomic classification but allow persistence of the genus if the hybrids are reproductively viable.

As sessile spawners with planktonic larvae, the Critical Risk Threshold assessments for *Acropora* species must weigh the broad distributions that provide replicated opportunities for potential escape from local disturbances against the necessity to have colonies in close enough proximity to have successful fertilization of enough eggs to replenish the attrition of the spawning stock. If the effective population size (i.e., the number of genotypes [might be substantially less than the number of colonies in highly clonal species] close enough for successful fertilization) becomes too low to replenish the population, then the positive-feedback depensatory processes begin. It is worth noting that Edinger and Risk (1995) concluded that brooding corals survived the harsh environmental conditions better than did the spawners in the western Atlantic during the major extinctions of the Oligocene-Miocene transition period. Many *Acropora* have branching morphologies, making them potentially susceptible to fragmentation. Fragment survival can increase coral abundance in the short-term but does not contribute new genotypes (or evolutionary opportunities) to the population.

Population Dynamics

Abundance. The total world population of this species has been estimated at 10,942,000 colonies, with an effective population size of 1,204,000 colonies (Richards et al. 2008; Veron 2014).

Productivity. The overall decline in abundance ("Percent Population Reduction") was estimated at 35%, and the decline in abundance before the 1998 bleaching event ("Back-cast Percent Population Reduction") was estimated at 14% (Carpenter et al. 2008). However, live coral cover trends are highly variable both spatially and temporally, producing patterns on small scales that can be easily taken out of context, thus quantitative inferences to species-specific trends should be interpreted with caution. At the same time, an extensive body of literature documents broad declines in live coral cover and shifts to reef communities dominated by hardier coral species or algae over the past 50 to 100 years (Birkeland, 2004; Fenner, 2012; Pandolfi et al., 2003; Sale and Szmant, 2012). These changes have likely occurred, and are occurring, from a combination of global and local threats. Given that A. speciosa occurs in many areas affected by these broad changes, and likely has some susceptibility to both global and local threats, we conclude that it is likely to have declined in abundance over the past 50 to 100 years, but a precise quantification is not possible based on the limited species-specific information.

Genetic Diversity. There is little information available regarding the genetic diversity of this species.

Distribution. *Acropora speciosa* has been reported to occupy protected environments with clear water and high diversity of *Acropora* (Veron, 2000) and steep slopes or deep, shaded waters (IUCN, 2010). *Acropora speciosa* has been reported in water depths ranging from 12 meters to 30 meters (Carpenter et al., 2008) and 15 meters to 40 meters (Richards, 2009). It is found in mesophotic assemblages in American Samoa (Bare et al., 2010), suggesting the potential for deep refugia.

Designated Critical Habitat. See Status of Proposed Indo-Pacific Coral Critical Habitat below.

Recovery Goals. A recovery plan has not yet been developed for this species. However, a recovery outline has been developed (NMFS 2015a).

8.60 Coral species: *Euphyllia paradivisa*

Table 109. *Euphyllia pardivisa*; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Euphyllia paradivisa	Not Available	NA	Threatened	2021 (Initiated)	79 FR 53852	2015 (Recovery Outline)	2023 88 FR 83644 (Proposed)

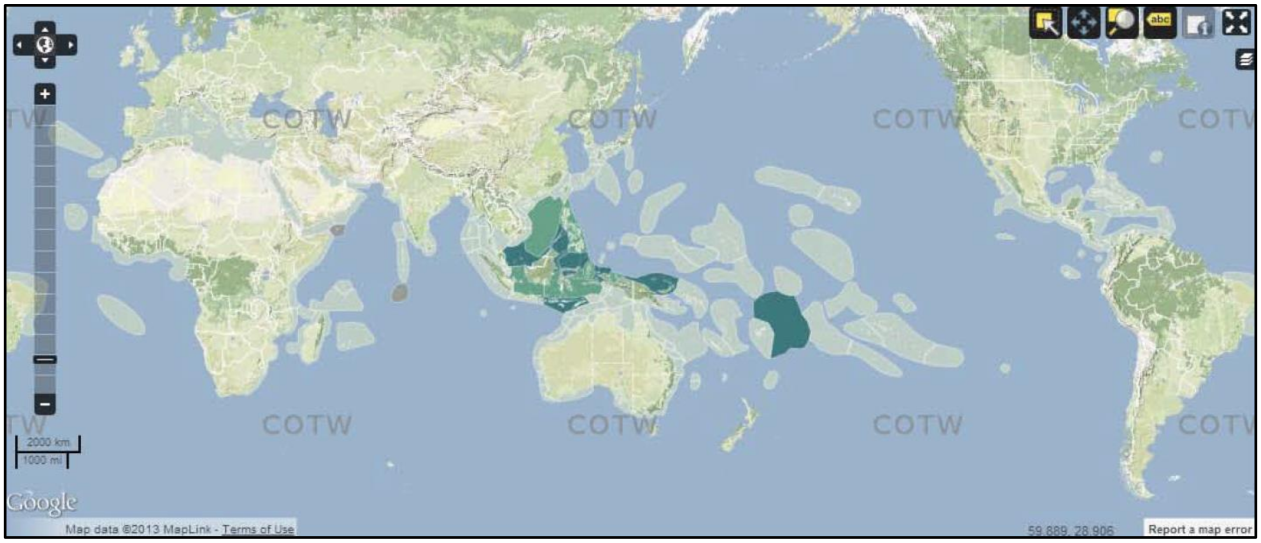


Figure 75. *Isopora crateriformis* distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. Colonies are phaceloid, made up of branching separate corallites. Several species in this genus (including *Euphyllia glabrescens*, *Euphyllia paraglabrescens*, and *Euphyllia paraancora*) cannot be distinguished based on skeletal characters, but only by the characters of polyp tentacles. Polyps have branching tentacles almost identical to those of *Euphyllia divisa*. Color is pale greenish-grey with lighter tentacle tips (Veron, 2000). *Euphyllia paradivisa* has a restricted range, existing only in the highly disturbed Coral Triangle Region. According to the IUCN Species Account, *Euphyllia paradivisa* occurs in American Samoa.

Status. *Euphyllia paradivisa* is susceptible to warming-induced bleaching, disease, ocean acidification, trophic effects of fishing, nutrients, predation, and collection and trade. These threats are expected to continue and worsen into the future. In addition, the species has inadequate existing regulatory mechanisms for global threats. *Euphyllia paradivisa's* distribution is limited mostly to the Coral Triangle, which is projected to have the most rapid and severe impacts from climate change and localized human impacts for coral reefs over the 21st century. Multiple ocean warming events have already occurred within the Coral Triangle that suggest future ocean warming events may be more severe than average in this part of the world. A range constrained to this particular geographic area that is likely to experience severe and increasing threats indicates that a high proportion of the population of this species is likely to be exposed to those threats over the foreseeable future. Considering the limited range of this species in an area where severe and increasing impacts are predicted, this level of abundance leaves the species vulnerable to becoming of such low abundance within the foreseeable future that it may be at risk from depensatory processes, environmental stochasticity, or catastrophic events. The combination of these characteristics and projections of future threats indicates that the species is likely to be in danger of extinction within the foreseeable future throughout its range.

Life History. Reproductive mode is not known. One congener (*Euphyllia ancora*) is a gonochoric spawner (Guest et al., 2005a; Willis et al., 1985) while another congener (*Euphyllia glabrescens*) is reported to be a hermaphroditic brooder in southern Taiwan (Fan et al., 2006). No other information regarding its ecology or life history is available.

Population Dynamics

Abundance. Veron (2014) reports that E. paradivisa occupied 0.2% of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.5 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as “rare,” and overall abundance was described as “uncommon.” Veron did not infer trends in abundance from these data. The absolute abundance of this species is likely at least tens of millions of colonies (Richards et al. 2008; Veron 2014).

Productivity. The overall decline in abundance was estimated at 38%, and the decline in abundance before the 1998 bleaching event (“Back-cast Percent Population Reduction”) was estimated at 15% (Carpenter et al. 2008). However, live coral cover trends are highly variable both spatially and temporally, producing patterns on small scales that can be easily taken out of context. Thus, quantitative inferences to species-specific trends should be interpreted with caution. At the same time, an extensive body of literature documents broad declines in live coral cover and shifts to reef communities dominated by hardier coral species or algae over the past 50 to 100 years (Birkeland, 2004; Fenner, 2012; Pandolfi et al., 2003; Sale and Szmant, 2012). These changes have likely occurred, and are occurring, from a combination of global and local threats. Given that E. paradivisa occurs in many areas affected by these broad changes, and likely has some susceptibility to both global and local threats, we conclude that it is likely to have declined in abundance over the past 50 to 100 years, but a precise quantification is not possible due to the limited species-specific information.

Genetic Diversity. There is little information available regarding the genetic diversity of this species.

Distribution. *Euphyllia paradivisa* has been reported from shallow or mid-slope reef environments protected from wave action (Veron, 2000). *Euphyllia paradivisa* occurs at depths of 5 meters to 20 meters (IUCN Species Account).

Designated Critical Habitat. See Status of Proposed Indo-Pacific Coral Critical Habitat below.

Recovery Goals. A recovery plan has not yet been developed for this species. However, a recovery outline has been developed (NMFS 2015a).

8.61 Coral species: *Isopora crateriformis*

Table 110. *Isopora crateriformis*; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Isopora crateriformis	Not Available	NA	Threatened	2021 (Initiated)	79 FR 53852	2015 (Recovery Outline)	2023 88 FR 83644 (Proposed)

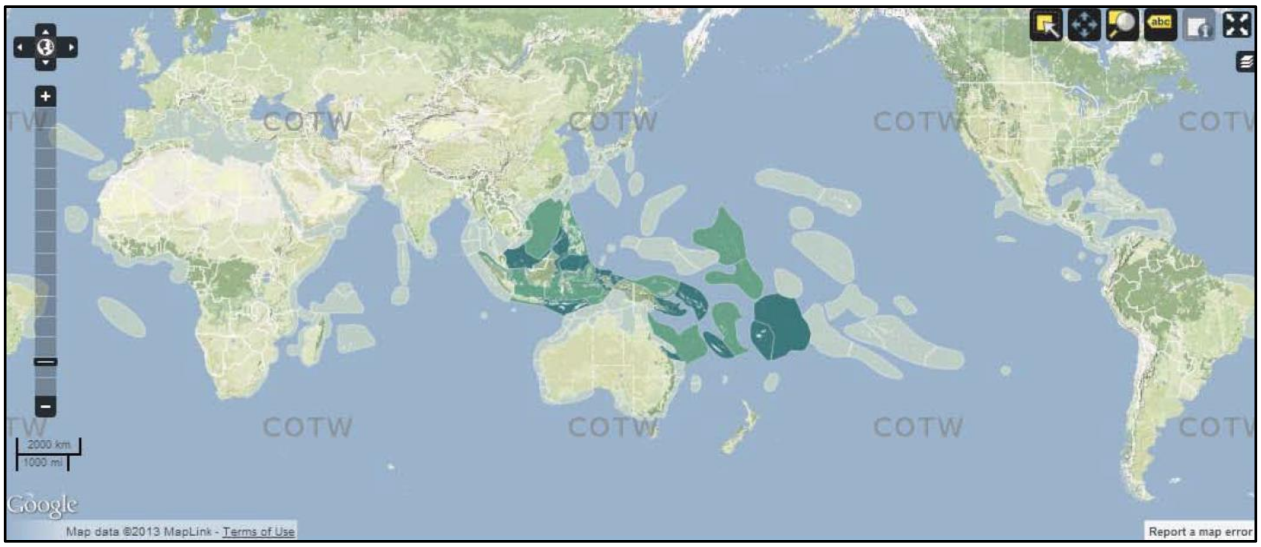


Figure 76. *Isopora crateriformis* distribution. Off-white = no record, dark green = confirmed record, pale green = predicted record, tan = published record that needs further investigation (Veron 2014).

Species Description. *Isopora crateriformis* forms flattened solid encrusting plates sometimes referred to as "cowpies." They can sometimes be over a meter in diameter. Colonies are brown in color (Veron, 2000).

Status. *Isopora crateriformis* is highly susceptible to ocean warming, disease, acidification, trophic effects of fishing, and nutrients, and predation. In addition, existing regulatory mechanisms to address global threats that contribute to extinction risk for this species are inadequate. The majority of *Isopora crateriformis'* distribution is within the Coral Triangle and western equatorial Pacific, which is projected to have the most rapid and severe impacts from climate change and localized human impacts for coral reefs over the 21st century. Multiple ocean warming events have already occurred within the western equatorial Pacific that suggest future ocean warming events may be more severe than average in this part of the world. A range constrained to this particular geographic area that is likely to experience severe and increasing threats indicates that a high proportion of the population of this species is likely to be exposed to those threats over the foreseeable future. *Isopora crateriformis'* qualitative abundance is rare overall. Considering that much of the range of this species includes areas where severe and increasing impacts are predicted, this level of abundance combined with its restricted depth distribution, leaves the species vulnerable to becoming of such low abundance within the foreseeable future that it may be at risk from depensatory processes, environmental stochasticity, or catastrophic events. The combination of these biological and environmental characteristics and future projections of threats indicates that the species is likely to be in danger of extinction within the foreseeable future throughout its range.

Life History. *Isopora crateriformis* is most likely a simultaneous hermaphroditic brooder as is the closely related *Isopora cuneata* (Bothwell, 1981). *Isopora cuneata* planulae lack zooxanthellae, and in some areas the species can undergo several seasonal cycles of larval production (Kojis, 1986). Its brooding life history allows *Isopora spp.* to locally dominate recruitment at Lord Howe Island, Australia; colonies of this genus also dominate the adult population there, suggesting brooding may drive community structure in remote areas (Harriott, 1992; 1995). *Isopora cuneata* is not prone to asexual reproduction via fragmentation, based on its semi-encrusting morphology (Bothwell, 1981). The species shows moderate gene flow (Mackenzie et al., 2004) but little potential for large-scale dispersal (Ayre and Hughes, 2004).

Population Dynamics

Abundance. Veron (2014) reports that I. crateriformis occupied 0.3% of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.4 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as “rare.” Overall abundance was described as “occasionally common on reef flats.” The absolute abundance of this species is likely at least millions of colonies (Richards et al. 2008; Veron 2014).

Productivity. The overall decline in abundance was estimated at 38%, and the decline in abundance before the 1998 bleaching event (“Back-cast Percent Population Reduction”) was estimated at 14%. However, live coral cover trends are highly variable both spatially and temporally, producing patterns on small scales that can be easily taken out of context, thus quantitative inferences of species-specific trends should be interpreted with caution. At the same time, an extensive body of literature documents broad declines in live coral cover and shifts to reef communities dominated by hardier coral species or algae over the past 50 to 100 years (Birkeland, 2004; Fenner, 2012; Pandolfi et al., 2003; Sale and Szmant, 2012). These changes have likely occurred, and are occurring, from a combination of global and local threats. Given that I. crateriformis occurs in many areas affected by these broad changes, and likely has some susceptibility to both global and local threats, we conclude that it is likely to have declined in abundance over the past 50 to 100 years, but a precise quantification is not possible based on the limited species-specific information.

Genetic Diversity. The species shows moderate gene flow (Mackenzie et al., 2004) but little potential for large-scale dispersal (Ayre and Hughes, 2004).

Distribution. *Isopora crateriformis'* distribution is from Sumatra (Indonesia) to American Samoa, and there are reports from the western and central Indian Ocean that need confirmation. According to both the IUCN Species Account and the CITES species database, *Isopora* *crateriformis* occurs in American Samoa. *Isopora crateriformis* is found most commonly in shallow, high-wave energy environments. *Isopora craterformis* has been reported in water depths ranging from low tide commonly to at least 12 meters (Birkeland, 1987). The species was recently reported (as *Acropora crateriformis*) on mesophotic reefs (< 50 meters depth) in American Samoa (Bare et al., 2010).

Designated Critical Habitat. See Status of Proposed Indo-Pacific Coral Critical Habitat below.

Recovery Goals. A recovery plan has not yet been developed for this species. However, a recovery outline has been developed (NMFS 2015a).

8.62 Status of Proposed Indo-Pacific Coral Critical Habitat

Reef-building corals, including the 5 listed Indo-Pacific species that can be found in U.S. waters in the action area, have specific habitat requirements including hard substrate, narrow mean temperature range, adequate light, and adequate water flow, among others. These habitat requirements are most commonly found in shallow tropical and subtropical coral reef ecosystems, but can also be found in non-reef and mesophotic areas (NMFS 2019). Proposed critical habitat includes 251 km2 (97 m2) of marine habitat in 5 U.S. Pacific Islands jurisdictions, encompassing 16 island units in Tutuila and Offshore Banks, Ofu-Olosega, Ta'u, Rose Atoll, Guam, Rota, Aguijan, Tinian, Saipan, FDM, Alamagan, Pagan, Maug Islands, Uracas, Palmyra Atoll, Johnston Atoll, Wake Atoll, and French Frigate Shoals. Several areas are ineligible for critical habitat because of final Department of Defense INRMPs that we have determined will benefit the listed corals.

For A. globiceps, specific areas around all 16 islands are proposed, including 4 in American Samoa, 1 in Guam, 9 in CNMI, 3 in PRIA, and 1 in Hawaii. The depth ranges of the specific areas for A. globiceps are 0–20 m (3 islands), 0–12 m (9 islands), and 0–10 m (4 islands). For A. retusa, specific areas around three islands are proposed, all of which are in American Samoa. The depth ranges of the specific areas for A. retusa are 0–20 m on all three islands. For A. speciosa and E. paradivisa, specific areas around Tutuila and its offshore banks in American Samoa are proposed. The depth ranges of the specific areas for A. speciosa and E. paradivisa are 20–50 m. For I. crateriformis, specific areas around three islands are proposed, all of which are in American Samoa. The depth ranges of the specific areas for I. crateriformis are 0–20 m on all three islands. The 4(a)(3)(B)(i) INRMP analyses found that the entire areas around FDM and Wake Atoll, several areas off of Guam, and most of Tinian are ineligible for proposed coral critical habitat.

The PBFs identified as essential to the conservation of each species is reproductive, recruitment, growth, and maturation habitat. Sites that support the normal function of all life stages of the corals are natural, consolidated hard substrate or dead coral skeleton free of algae and sediment at the appropriate scale at the point of larval settlement of fragment reattachment, and the associated water column. Several attributes of these sites determine the quality of the area and influence the value of the associated feature of the conservation of the species (88 FR 83644):

- Substrate with presence of crevices and holes that provide cryptic habitat, the presence of microbial biofilms, or presence of crustose coralline algae;
- Reefscape with no more than a thin veneer of sediment and low occupancy by fleshy and turf macroalgae;
- Marine waters with levels of temperature, aragonite saturation, nutrients, and water clarity that have been observed to support any demographic function; and
- Marine water with levels of anthropogenically-introduced (from humans) chemical contaminants that do not preclude or inhibit any demographic function.

The Navy's Joint Region Marianas INRMP and the Air Force's Wake Island Air Field, Wake Atoll, Kokee Air Force Station, Kuia, Hawaii, and Mt. Kaala Air Force Station, Oahu, Hawaii (Wake INRMP) includes marine areas around Guam, Tinian, FDM, and Wake that are excluded from the proposed critical habitat designation. The INRMPs for these areas were determined by

NMFS to provide a benefit to the ESA-listed coral species found within the in-water area of the base.

In addition to the excepted areas subject to the 2023 Wake Island and 2019 Joint Region Marianas INRMP (see paragraph (d) of 88 FR 83644 for more detailed information), proposed critical habitat does not include areas where the essential feature does not occur and the following particular locations:

- Pursuant to ESA section 3(5)(A)(i)(I), all managed areas that may contain natural hard substrate but do not provide the quality of substrate essential for the conservation of threatened corals. Managed areas that do not provide the quality of substrate essential for the conservation of the 5 Indo-Pacific corals are defined as particular areas whose consistently disturbed nature renders them poor habitat for coral growth and survival over time. These managed areas include specific areas where the substrate has been disturbed by planned management authorized by local, territorial, state, or Federal governmental entities at the time of critical habitat designation, and will continue to be periodically disturbed by such management. Examples include, but are not necessarily limited to, dredged navigation channels, shipping basins, vessel berths, and active anchorages; and
- Pursuant to ESA section 3(5)(A)(i), artificial substrates including but not limited to: Fixed and floating structures, such as ATONs, seawalls, wharves, boat ramps, fishpond walls, pipes, submarine cables, wrecks, mooring balls, docks, aquaculture cages.

As discussed in other sections of this opinion, the percentage of live cover of all reef-building coral species combined has declined across much of the Indo-Pacific since the 1970s, and likely many decades before then in some locations (NMFS 2014a; NMFS 2020a). Furthermore, from 2014 to 2017, an unprecedented series of bleaching events impacted most of the Indo-Pacific's coral reefs (Eakin et al. 2019), further reducing overall habitats with high mean coral cover, especially of relatively sensitive species. While coral bleaching patterns are complex, there is general agreement that thermal stress has led to accelerated bleaching and mass mortality during the past several decades. During the years 1983, 1987, 1995, 1996, 1998, 2002, 2004, 2005, 2014, 2015, 2016, and 2017 widespread warming-induced coral bleaching and mortality was documented in many Indo-Pacific reef coral communities (Brainard et al. 2011a; Hughes et al. 2017; Jokiel and Brown 2004). The series of coral bleachings in 2014-2017 are considered a single 3-year event by NOAA's Coral Reef Watch (Eakin et al. 2019). It was the longest, most widespread, and likely the most damaging coral bleaching event on record. It affected more coral reefs than any previous global bleaching event, and was worse in some locales than ever recorded before (e.g., Great Barrier Reef). Heat stress during this event also caused mass bleaching in several reefs where bleaching had never been recorded before, such as in the uninhabited atolls of the central Pacific (Eakin et al. 2019).

In addition to bleaching, impacts from ocean acidification is causing numerous adverse effects to coral habitat in the Pacific. Ocean acidification reduces the aragonite saturation state (Ω_{arg}) in seawater by lowering the supersaturation of carbonate minerals including aragonite, which requires marine calcifiers like reef-building corals to expend more energy to calcify their skeletons. The effects of the lower Ω_{arg} projected for Indo-Pacific coral reef waters on coral calcification and growth, reef erosion, and coral reproduction have been extensively studied via laboratory experiments, modeling efforts, and at field sites with naturally low Ω_{arg} representative of projected conditions. The ocean acidification projected for the foreseeable future is expected to result in erosion outpacing accretion on many Indo-Pacific reefs, just as it has already done on eastern Pacific reefs (Brainard et al. 2011a). An analysis of 22 coral reef sites, including 19 in the Indo-Pacific, and the resulting model projected that 17 of the 19 sites would fall below Ωarg levels of 2.92 by 2100, the threshold below which dissolution of reef sediments would exceed accumulation of reef sediments, thus demonstrating that reef erosion is outpacing reef accretion (Eyre et al. 2018). Field studies at Indo-Pacific sites with naturally acidic seawater show that reef erosion exceeds reef accretion at a pH of approximately 7.8 (Enochs et al. 2016), and that very high rates of reef erosion characterize such sites (Barkley Hannah et al. 2015). In addition to effects on coral calcification and reef erosion, the ocean acidification projected for the foreseeable future is also expected to lower the fertilization, settlement, and recruitment of some Indo-Pacific reef-building corals (Brainard et al. 2011a).

Because of the above effects of projected ocean acidification on coral calcification, reef erosion, and coral reproduction, Indo-Pacific reef-building coral communities are expected to experience reductions in complexity and resilience, loss of reef corals, increases in macroalgae, simplification, and overall degradation. For example, within Indo-Pacific communities where naturally acidic seawater roughly approximates pH levels projected by 2100 (8.1 to 7.8), there is lower reef coral diversity, recruitment, and abundances than in other Indo-Pacific reef coral communities, suggesting that projected ocean acidification in the foreseeable future will reduce the complexity and resilience of these communities (Fabricius et al. 2011) by affecting coral colonies and their calcium carbonate substrate.

Since the 2014 listing of *Acropora retusa, Acropora speciosa, Euphyllia paradivisa,* and *Isopora crateriformis* the threats to these species and their habitat have worsened, especially the most important threat to the ESA-listed species, global warming. All threats are projected to further worsen, based on current information (NMFS 2021). Recovery of the 5 species is not possible unless the worsening trends are at least stabilized, especially for the 2 most important threats, ocean warming and ocean acidification, both of which are caused by global climate change (NMFS 2021). In order for adverse impacts on Pacific coral habitat to subside, a viable recovery strategy must be based on controlling global climate change.

There are several protected areas within the proposed critical habitat designation where habitat conditions are better because of the lack of human activities, although these areas are still subject to stressors associated with climate change and ocean acidification. Howland and Jarvis Islands were designated as a National Wildlife Refuge in 1974 and expanded to include submerged lands out to 12 nautical miles in 2009. In 2009, the islands were also included in the designation of the Pacific Remote Islands Marine National Monument. USFWS and NOAA conduct occasional ship-based research and monitoring every 3 years but there no structures or other activities requiring special management in this area. Similarly, Kingman Reef in the Pacific Remote Islands Marine National Monument (designated in 2009) is visited every 3 years to conduct surveys of the reef area and on rare occasions a research vessel may visit the area to conduct other studies of the marine environment. Rose Atoll is a National Wildlife Refuge and was designated as a Marine National Monument in 2009. Rose Atoll is visited approximately 3 times per year for inventory and monitoring, and sea turtle and other research. Maug Island in the northern CNMI is included in the islands area of the Marianas Trench Marine National

Monument established in 2009. Fishing and diving, as well as research cruises, occur infrequently at Maug.

Pala Lagoon and Pago Pago Harbor, Tutuila Island were excluded from the proposed coral critical habitat because of the amount of artificial substrate associated with the construction and management of shoreline protection and beach erosion control structures, small boat harbors and other channels, turning basins and berthing areas. Similarly, the Ofu Small Boat Harbor; Ta'u Small Boat Harbor and Faleasao Small Boat Harbor; Rota Harbor; Tinian Harbor; CNMI Ports Authority harbors, basins, and navigation channels; breakwaters; areas around Apra Harbor (outside the Naval area discussed previously); ATONs; small boat ramps; shoreline protection and erosion control structures, and other artificial structures are not included in the proposed designation due to the alteration of habitat in this area associated with the construction and management of artificial structures.

8.63 Killer Whale, Southern Resident DPS

Table 111. Killer Whale, Southern Resident DPS; overview table

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Orcinus orca	Killer Whale	Southern Resident	Endangered	2021	2005 70 FR 69903	2008	2021 86 FR 41668

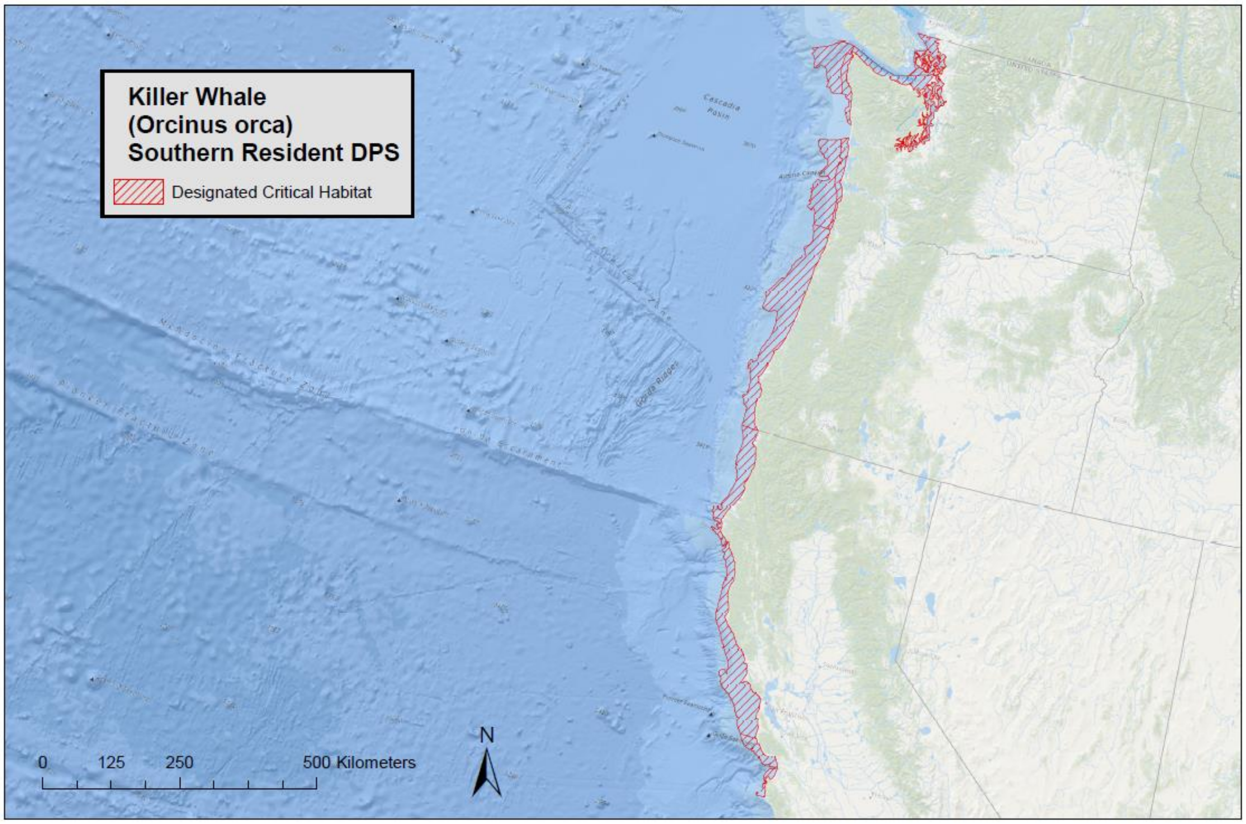


Figure 77. Killer Whale, Southern Resident DPS designated critical habitat

Species Description. Killer whales are distributed worldwide, but populations are isolated by region and ecotype. Killer whales have been divided into distinct population segments on the basis of differences in genetics, ecology, morphology and behavior. The Southern Resident killer whale distinct population segment can be found along the Pacific Coast of the United States and Canada, and in the Salish Sea, Strait of Juan de Fuca and Puget Sound, north to the Chatham Strait in southeast Alaska.

Killer whales are odontocetes and the largest delphinid species with black coloration on their dorsal side and white undersides and patches near the eyes. They also have a highly variable gray or white saddle behind the dorsal fin. The Southern Resident DPS of killer whales was listed as endangered under the ESA on November 18, 2005 (70 FR 69903).

Status. The Southern Resident killer whale DPS was listed as endangered in 2005 in response to the population decline from 1996 to 2001, small population size, and reproductive limitations (i.e., few reproductive males and delayed calving). Since listing, there have been no signs of recovery. Current threats to its survival and recovery include: contaminants, vessel traffic, and reduction in prey availability. Chinook salmon populations have declined due to degradation of habitat, hydrology issues, harvest, and hatchery introgression; such reductions may require an increase in foraging effort. In addition, these prey contain environmental pollutants. These contaminants become concentrated at higher trophic levels and may lead to immune suppression or reproductive impairment (Wasser, 2017). The inland waters of Washington and British Columbia support a large whale watch industry, commercial shipping, and recreational boating; these activities generate underwater noise, which may mask whales' communication or interrupt foraging. The factors that originally endangered the species persist throughout its habitat: contaminants, vessel traffic, and reduced prey. The DPS's resilience to future perturbation is reduced as a result of its small population size. The recent decline, unstable population status, and population structure (e.g., few reproductive age males and non-calving adult females) continue to be causes for concern. The relatively low number of individuals (74 as of September 2021) in this population makes it difficult to resist or recover from natural spikes in mortality, including disease and fluctuations in prey availability.

Life History. Southern Resident killer whales (SRKW) are geographically, matrilineally, and behaviorally distinct from other killer whale populations (70 FR 69903). The DPS includes 3 large, stable pods (J, K, and L), which occasionally interact (Parsons et al. 2009). Some mating occurs outside natal pods, during temporary associations of pods, or as a result of the temporary dispersal of males (Pilot et al. 2010). However, based on an updated pedigree from new genetic data, most of the offspring in recent years were sired by 2 fathers, meaning that less than 30 individuals make up the effective reproducing portion of the population. Because a small number of males were identified as the fathers of many offspring, a smaller number may be sufficient to support population growth than was previously thought (Ford et al. 2011, NWFSC unpublished data). In addition many offspring were the result of matings within the same pod raising questions and concerns about inbreeding effects. Research into the relationship between genetic diversity, effective breeding population size, and health is currently underway to determine how this metric can inform us about extinction risk and inform recovery (NWFSC unpublished data). Males become sexually mature at 10 to 17 years of age. Females reach maturity at 12 to 16 years of age and produce an average of 5.4 surviving calves during a reproductive life span of approximately 25 years. Mothers and offspring maintain highly stable, life-long social bonds, and this natal relationship is the basis for a matrilineal social structure. They prey upon salmonids, especially Chinook salmon (Hanson et al. 2010).

Population Dynamics

Abundance. At the time of listing in 2005, the SRKW population included 88 whales. As of the official summer census in 2021, there were 74 whales in the population, with an additional whale (K21) presumed dead at the time of this report.

Productivity / Population Growth Rate. Population growth has varied since listing, with both increasing and decreasing years, but the whales are currently experiencing a downward trend.

Genetic Diversity. After thorough genetic study, the Biological Review Team concluded that Southern Resident killer whales were discrete from other killer whale groups (NMFS 2008, Parsons et al. 2013, Morin et al. 2015). Despite the fact that their ranges overlap, Southern Resident killer whales do not intermix with Northern Resident killer whales. Southern Resident killer whales consist of 3 pods, called J, K, and L. Low genetic diversity within a population is believed to be in part due to the matrilineal social structure (NMFS 2008d, Parsons et al. 2013, Morin et al. 2015).

Distribution. Southern Resident killer whales occur in the inland waterways of Puget Sound, Strait of Juan de Fuca, and Southern Georgia Strait during the spring, summer and fall. During the winter, they move to coastal waters primarily off Oregon, Washington, California, and British Columbia, and have been documented as far south as central California and as far north as Southeast Alaska (Black et al. 2001, Hilborn et al. 2012).

Designated Critical Habitat. On November 29, 2006, NMFS designated critical habitat for the Southern Resident killer whale (71 FR 69054). The critical habitat consists of approximately 6,630 km^2 in 3 areas: the Summer Core Area in Haro Strait and waters around the San Juan Islands; Puget Sound; and the Strait of Juan de Fuca. It provides the following physical and biological features essential to the conservation of Southern Resident killer whales: water quality to support growth and development; prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and inter-area passage conditions to allow for migration, resting, and foraging.

On January 21, 2014, NMFS received a petition to revise critical habitat for SRKW, which cited recent information on the SRKW habitat use along the West Coast of the United States. The petitioner, the Center for Biological Diversity, requested that the critical habitat designation be revised and expanded to include areas of the Pacific Ocean between Cape Flattery, WA, and Point Reyes, CA, extending approximately 47 miles (76 km) offshore. NMFS published a 90-day finding on April 25, 2014 (79 FR 22933) that the petition contained substantial information to support the proposed measure and that NMFS would further consider the action and also solicited information from the public.

NMFS revised the critical habitat designation for Southern Resident killer whales on August 2, 2021. The final rule maintained the previously designated critical habitat in inland waters of Washington and expands it to include coastal waters off Washington, Oregon, and California. The revision adds to Southern Resident killer whales critical habitat approximately 15,910 square miles of marine waters between the 6.1-meter and 200-meter depth contours from the U.S.-Canada border to Point Sur, California.

Recovery Goals. See the 2008 Final Recovery Plan (NMFS 2008c) for the Southern Resident killer whale for complete down listing/delisting criteria for each of the following recovery goals:

- Prey Availability: Support salmon restoration efforts in the region including habitat, harvest and hatchery management considerations and continued use of existing NMFS authorities under the ESA and Magnuson-Stevens Fishery Conservation and Management Act to ensure an adequate prey base.
- Pollution/Contamination: Clean up existing contaminated sites, minimize continuing inputs of contaminants harmful to killer whales, and monitor emerging contaminants.
- Vessel Effects: Continue with evaluation and improvement of guidelines for vessel activity near Southern Resident killer whales and evaluate the need for regulations or protected areas.
- Oil Spills: Prevent oil spills and improve response preparation to minimize effects on Southern Residents and their habitat in the event of a spill.
- Acoustic Effects: Continue agency coordination and use of existing ESA and Marine Mammal Protection Act (MMPA) mechanisms to minimize potential impacts from anthropogenic sound.
- Education and Outreach: Enhance public awareness, educate the public on actions they can participate in to conserve killer whales and improve reporting of Southern Resident killer whale sightings and strandings.
- Response to Sick, Stranded, Injured Killer Whales: Improve responses to live and dead killer whales to implement rescues, conduct health assessments, and determine causes of death to learn more about threats and guide overall conservation efforts.
- Transboundary and Interagency Coordination: Coordinate monitoring, research, enforcement, and complementary recovery planning with Canadian agencies, and Federal and State partners.
- Research and Monitoring: Conduct research to facilitate and enhance conservation efforts. Continue the annual census to monitor trends in the population, identify individual animals, and track demographic parameters.

9 ENVIRONMENTAL BASELINE

Section Contents

9.1 Introduction

The environmental baseline refers to the condition of the ESA-listed species or its designated critical habitat in the action area, without the consequences to the ESA-listed species or designated critical habitat caused by the action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process. The consequences to ESA-listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR §402.02).

The key purpose of the environmental baseline is to describe the natural and anthropogenic factors influencing the status and condition of ESA-listed species and designated critical habitat in the action area. Because this is a consultation on a program with a large geographic scope, this environmental baseline focuses more generally on the status and trends of the aquatic ecosystems in the U.S. and the consequences of that status for ESA resources that may be adversely affected by the action.

Activities that negatively impact water quality also threaten aquatic species. The deterioration of water quality is a contributing factor that has led to the reduction in populations of some ESA-listed aquatic species under the NMFS jurisdiction. Declines in populations of these species leave them vulnerable to a multitude of threats. Due to the combined effects of reduced abundance, low or highly variable growth capacity, and the loss of essential habitat, these species are less resilient to additional disturbances. In larger populations, stressors that affect only a limited number of individuals could once be tolerated by the species without resulting in population level impacts; in smaller populations, the same stressors are more likely to reduce the likelihood of survival. In addition, populations that have ongoing stressors already present in the environment are less likely to be resilient to additional stressors resulting from the action.

The quality of the biophysical components within aquatic ecosystems is affected by natural events as well as human activities conducted within and around coastal waters, estuarine and riparian zones, as well as those conducted more remotely in the upland portion of the watershed. Industrial activities can result in discharge of pollutants, changes in water temperature and levels of dissolved oxygen, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow.

The environmental baseline sections that follow are organized by region (e.g., Pacific island region). Within each region, discussions of land-use, water quality, and other components of the baseline are presented at the sub region level (hydrologic unit code 2) when applicable.

9.2 General Factors

9.2.1 Climate Variability and Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Effects of climate change include sea level rise, increased frequency and magnitude of severe weather events, changes in air and water temperatures, and changes in precipitation patterns, all of which are likely to impact ESA resources. The NOAA climate information portal provides basic background information on these and other measured or anticipated climate change effects (see https://www.climate.gov).

In order to evaluate the implications of different climate outcomes and associated impacts throughout the 21st century, many factors have to be considered. The amount of future greenhouse gas (GHG) emissions is a key variable. Developments in technology, changes in energy generation and land use, global and regional economic circumstances, and population growth must also be considered.

A set of 5 scenarios were developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections of future emissions or concentrations of GHGs are employed consistently across the various branches of climate science while taking into consideration assumptions of how socio-economic systems could evolve over the course of the 21st century. Scenario uncertainty is explored by assessing alternative socio-economic futures; the 5 scenarios described in IPCC (2021) are based on Shared Socio-economic Pathways (SSPs). These 5 new SSP scenarios cover a broader range of GHG and air pollutant futures than did the previous IPCC scenarios, or representative concentration pathways (RCPs) described in IPCC (2014). Although not directly comparable, both sets of scenarios are labelled by the level of radiative forcing that will be reached in 2100, and modelling studies relying on RCPs complement the assessment based on SSP scenarios (IPCC 2021).

The 5 SSP scenarios start in 2015 and project to either 2100 or 2300, and they include both high-CO2 emissions pathways without climate change mitigation as well as new low-CO2 emissions pathways. Differences in the level of climate change mitigation and air pollution control strongly affect anthropogenic emissions trajectories of short-lived climate forcers (SLCFs) (IPCC 2021). Scenario SSP1-1.9 represents the low end of future emissions pathways, leading to warming below 1.5°C in 2100 and limited temperature overshoot of 1.5°C over the 21st century; it is characterized by very low GHG emissions. Scenario SSP1-2.6 is characterized by low GHG emissions and, along with SSP1-1.9, CO_2 emissions declining to net zero around or after 2050 followed by varying levels of net negative CO_2 emissions. Scenario SSP2-4.5 is characterized by intermediate GHG emissions and CO_2 emissions remaining around current levels until the middle of the century. Scenario SSP3-7.0 is characterized by high GHG and CO_2 emissions that will approximately double from the current levels by 2100. Scenario SSP5-8.5 represents the very high warming end of the range of future emissions pathways that have been presented in the literature, and is characterized by very high GHG and CO_2 emissions that will approximately double from the current levels by 2050 (IPCC 2021). SSP2-4.5 and SSP1-2.6 represent scenarios with stronger climate change mitigation and thus lower GHG emissions, but only SSP1-2.6 was designed to limit warming to below 2°C. The Paris Agreement aims to limit the future rise in global average temperature to 2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios (Hayhoe et al. 2018).

Global mean surface temperature, calculated by merging sea surface temperature over the ocean and air temperature 2 meters over land and sea ice areas, is used in most paleo, historical, and present-day observational estimates of global warming (IPCC 2021). Warming greater than the global average has already been experienced in many regions and seasons, with most land regions experiencing greater warming than over the ocean and with the Arctic region warming most rapidly (Allen et al. 2018; IPCC 2021). Global warming has led to more frequent heatwaves in most land regions and an increase in the frequency and duration of marine heatwaves (Allen et al. 2018). Global mean surface temperature has increased by 1.09 (0.95 to 1.20)°C from pre-industrial times to 2011-2020; global temperatures have risen at an unprecedented rate since 2012, with the period from 2016-2020 being the hottest 5-year period between 1850 and 2020 (IPCC 2021). Average global warming up to 1.5°C as compared to pre-industrial levels is expected to lead to regional changes in extreme temperatures, and increases in the frequency and intensity of precipitation and drought (Allen et al. 2018). Projections show that the average global surface temperature during the period from 2081-2100 is very likely to be higher by 1.0°C to 1.8°C under the low CO_2 emissions scenario SSP1-1.9 and by 3.3°C to 5.7°C under the high CO_2 emissions scenario SSP5-8.5 as compared to 1850-1900 (IPCC 2021).

Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (Evans and Bjørge 2013; Kintisch 2006; Learmonth et al. 2006; MacLeod et al. 2005; McMahon and Hays 2006; Pachauri and Meyer 2014; Robinson et al. 2005). Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring.

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). (Hazen et al. 2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses.

These changes will not be spatially homogeneous. The largest hydrologic responses are expected to occur in basins with significant snow accumulation, where warming decreases snow pack, increases winter flows, and advances the timing of spring melt (Mote 2016; Mote et al. 2014). Rain-dominated watersheds and those with significant contributions from groundwater may be less sensitive to predicted changes in climate (Mote et al. 2014; Tague et al. 2013).

Decreases in summer precipitation of as much as 30% by the end of the century are consistently predicted across climate models (Abatzoglou et al. 2014). Precipitation is more likely to occur during October through March and less during summer months. More winter precipitation will be rain than snow (ISAB 2007; Mote et al. 2013; Mote et al. 2014). Earlier snowmelt will cause lower stream flows in late spring, summer, and fall, and water temperatures will be warmer (ISAB 2007; Mote et al. 2014). Models consistently predict increases in the frequency of severe winter precipitation events (i.e., 20-year and 50-year events), in the western United States (Dominguez et al. 2012). The largest increases in winter flood frequency and magnitude are predicted in mixed rain-snow watersheds (Mote et al. 2014).

The combined effects of increasing air temperatures and decreasing spring through fall flows are expected to cause increasing stream temperatures; in 2015 this resulted in 3.5-5.3 degree increases in Columbia Basin streams and a peak temperature of 26 °C in the Willamette (NWFSC 2015c). Overall, about one-third of the current cold-water salmonid habitat in the Pacific Northwest is likely to exceed key water temperature thresholds by the end of this century (Mantua et al. 2009).

Higher temperatures will reduce the quality of available salmonid habitat for most freshwater life stages (ISAB 2007). Reduced flows will make it more difficult for migrating fish to pass physical and thermal obstructions, limiting their access to available habitat (Isaak et al. 2012; Mantua and Hamlet 2010). Temperature increases shift timing of key life cycle events for salmonids and species forming the base of their aquatic foodwebs (Crozier et al. 2008; Tillmann and Siemann 2011; Winder and Schindler 2004). Higher stream temperatures will also cause decreases in dissolved oxygen and may also cause earlier onset of stratification and reduced mixing between layers in lakes and reservoirs, which can also result in reduced oxygen (Meyer et al. 1999; Raymondi et al. 2013; Winder and Schindler 2004). Higher temperatures are likely to cause several species to become more susceptible to parasites, disease, and higher predation rates (Crozier et al. 2008; Raymondi et al. 2013; Wainwright and Weitkamp 2013).

As more basins become rain-dominated and prone to more severe winter storms, higher winter stream flows may increase the risk that winter or spring floods in sensitive watersheds will damage spawning redds and wash away incubating eggs (Goode et al. 2013). Earlier peak stream flows will also alter migration timing for salmon smolts, and may flush some young salmon and steelhead from rivers to estuaries before they are physically mature, increasing stress and reducing smolt survival (Lawson et al. 2004; McMahon and Hartman 1989). In addition to changes in freshwater conditions, predicted changes for coastal waters in the Pacific Northwest as a result of climate change include increasing surface water temperature, increasing but highly variable acidity, and increasing storm frequency and magnitude (Mote et al. 2014). Habitat loss, shifts in species' ranges and abundances, and altered marine food webs could have substantial consequences to anadromous, coastal, and marine species in the Pacific Northwest (Reeder et al. 2013; Tillmann and Siemann 2011). In a recent modeling study that investigated impacts to a variety of salmon populations across multiple life stages using a stochastic, age-structured life-cycle model with both density-dependent and density-independent climate effects, it was found that salmon populations rapidly declined in response to climate change (Crozier et al. 2021). In the freshwater life stages, migration timing shifted earlier in response to warmer freshwater conditions, but this did not prevent population declines under RCP climate change projections. The marine life stages were the most vulnerable to warming, with rising sea surface temperatures across diverse model assumptions and climate scenarios leading to a 90% decline in survival (Crozier et al. 2021).

9.2.2 Oceanographic Factors

As atmospheric carbon emissions increase, increasing levels of carbon are absorbed by the oceans, changing the pH of the water. A 38% to 109% increase in acidity is projected by the end of this century in all but the most stringent CO2 mitigation scenarios, and is essentially irreversible over a time scale of centuries (IPCC 2014b). Regional factors appear to be amplifying acidification in Northwest ocean waters, which is occurring earlier and more acutely than in other regions and is already impacting important local marine species (Barton et al. 2012; Feely et al. 2012). Acidification also affects sensitive estuary habitats, where organic matter and nutrient inputs further reduce pH and produce conditions more corrosive than those in offshore waters (Feely et al. 2012; Sunda and Cai 2012).

Global sea levels are expected to continue rising throughout this century, reaching likely predicted increases of 10-32 inches by 2081-2100 (IPCC 2014b). These changes will likely result in increased erosion and more frequent and severe coastal flooding, and shifts in the composition of nearshore habitats (Reeder et al. 2013; Tillmann and Siemann 2011). Estuarine-dependent salmonids such as chum and Chinook salmon are predicted to be impacted by significant reductions in rearing habitat in some Pacific Northwest coastal areas (Glick et al. 2007), for example. Historically, warm periods in the coastal Pacific Ocean have coincided with relatively low abundances of salmon and steelhead, while cooler ocean periods have coincided with relatively high abundances, and therefore these species are predicted to fare poorly in warming ocean conditions (Scheuerell and Williams 2005; Zabel et al. 2006). This is supported by the recent observation that anomalously warm sea surface temperatures off the coast of Washington from 2013 to 2016 resulted in poor coho and Chinook salmon body condition for juveniles caught in those waters (NWFSC 2015c). Changes to estuarine and coastal conditions, as well as the timing of seasonal shifts in these habitats, have the potential to impact a wide range of listed aquatic species (Reeder et al. 2013; Tillmann and Siemann 2011).

Oceanographic features of the action area may influence prey availability and habitat for listed species. These features comprise climate regimes which may suffer regime shifts due to climate changes or other unknown influences. The action area includes important spawning and rearing grounds and physical or biological features essential to the conservation of listed Pacific salmonids - i.e., water quality, prey, and passage conditions. These Pacific oceanographic conditions, climatic variability, and climate change may affect salmonids in the action area. There is evidence that Pacific salmon abundance may have fluctuated for centuries as a consequence of dynamic oceanographic conditions (Beamish and Bouillon 1993; Beamish et al. 2009; Finney et al. 2002). Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes (Finney et al. 2002). The long-term pattern of the Aleutian Low pressure system has corresponded to the trends in salmon catch, to copepod production, and to other climate indices, indicating that climate and the marine environment may play an important role in salmon production. Pacific salmon abundance and corresponding worldwide catches tend to be large during naturally-occurring periods of strong Aleutian low pressure causing stormier winters and upwelling, positive Pacific Decadal Oscillation (PDO), and an above average Pacific circulation index (Beamish et al. 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmosphere and ocean climate (Francis and Hare 1994).

Over the past century, regime shifts have occurred as a result of the North Pacific’s natural climate regime. Reversals in the prevailing polarity of the PDO occurred around 1925, 1947, 1977, and 1989 (Hare and Mantua. 2000; Mantua et al. 1997). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean (Mantua et al. 1997). During the pre-1977 climate regime, the productivity of salmon populations from the Snake River exceeded expectations (residuals were positive) when values of the PDO were negative (Levin 2003). During the post-1977 regime when ocean productivity was generally lower (residuals were negative), the PDO was negative (Levin 2003).

A smaller, less pervasive regime shift occurred in 1989 (Hare and Mantua. 2000). Beamish et al. (2000) analyzed this shift and found a decrease in marine survival of coho salmon in Puget Sound and off the coast of California to Washington. Trends in coho salmon survival were linked over the southern area of their distribution in the Northeast Pacific to a common climatic event. The Aleutian Low Pressure Index and the April flows from the Fraser River also changed abruptly about this time (Beamish et al. 2000).

In 2014-2019, the North Pacific experienced a warming event and an anomalously low Aleutian Low Pressure Index (Litzow et al. 2020). Compared to earlier regime shifts, in 2014-2019 the PDO predicted weaker atmospheric forcing associated with the Aleutian Low, warmer sea surface temperature in the Gulf of Alaska, weaker alongshore transport, and reduced wind mixing. These climate conditions resulted in a novel negative relationship between PDO and salmon production; as the PDO becomes more positive, salmon production decreases. According to , the changes in atmospheric forcing associated with the PDO and the resulting changes in ocean variables mapping onto the PDO provide an explanation for changing PDO-salmon relationships, namely: reductions in wind mixing and alongshore transport may impact salmon production by reducing upwelling and coastal nutrients which would impact prey availability, increasing temperature may impact salmon directly, and novel wind conditions could disrupt the conditions that made the positive PDO previously beneficial for salmon production (Litzow et al. 2020).

Poor environmental conditions for salmon survival and growth may be more prevalent with projected warming increases and ocean acidification. Increasing climate temperatures can influence smolt development which is limited by time and temperature (McCormick et al. 2009). Food availability and water temperature may affect proper maturation and smoltification and feeding behavior (Mangel 1994). Climate change may also have profound effects on seawater entry and marine performance of anadromous fish, including increased salinity intrusion in estuaries due to higher sea levels, as well as a projected decrease of seawater pH (Orr et al. 2005). There is evidence that Chinook salmon survival in the Pacific during climate anomalies and El Niño events changes as a result of a shift from predation- to competition-based mortality in response to declines in predator and prey abundances and increases in pink salmon abundance (Ruggerone and Goetz 2004). If climate change leads to an overall decrease in the availability of food, then returning fish will likely be smaller (Mangel 1994). In more recent years, it has been shown through modeling using long-term data from both wild and hatchery populations of salmon that some populations of North American Chinook salmon are becoming smaller and younger throughout the majority of the Pacific coast which could be due to changes in climate, fishing practices, or species interactions including predation (Ohlberger et al. 2018). Finally, future climatic warming could lead to alterations of river temperature regimes, which could further reduce available fish habitat (Yates et al. 2008); as of 2021, water temperatures in many rivers have exceeded the upper tolerance limit for salmon and many salmon populations are encountering water temperatures near or above the lethal limit (Thorstad et al. 2021).

9.2.3 Pesticides

9.2.3.1 Monitoring Data – General Overview

The following discussion is a general overview of monitoring information. Details specific to each region are provided below.

During the years 2012-2014 the USEPA and USGS conducted an assessment of targeted-chemical composition and cumulative biochemical activity of water samples collected from streams across the United States. Eight of the 10 most-frequently detected anthropogenic organics were pesticides with frequencies ranging 66-84% of all sites (Bradley et al. 2017). The USGS NAWQA program assessed trends in pesticide concentration at 59 sites across the U.S. for 3 overlapping periods: 1992-2001, 1997-2006, and 2001-2010. Trends in reported agriculture use intensity were assessed for the same periods at 57 sites (Ryberg et al. 2014). The report found widespread agreement between trends in concentration and use for agricultural pesticides. Additionally, the report found that trends between concentration and use for pesticides with both agricultural and urban use could be explained by taking into consideration concentration trends in urban streams (Ryberg et al. 2014).

In a 2014 study, the authors found pesticide concentrations were detected at concentrations which exceeded aquatic-life benchmarks in many rivers and streams throughout the 20-year sampling period (Stone et al. 2014). In a more recent decade sampled (2002 – 2011), 61% of streams and rivers which drain agricultural watersheds contained pesticides at concentrations which exceeded thresholds. In addition, 46% of mixed-land and 90% of urban streams were found to have pesticides in exceedance of aquatic-life benchmarks. According to (Stone et al. 2014) a number of important pesticides were not included in the sampling protocol and thus the potential for adverse effect is likely greater than is suggested by the percent of streams with exceedances. In a study conducted in 2021, 12 -24 samples were collected each year from 2013-2017 at each of 74 river sites across 5 regions of the U.S. (Stackpoole et al. 2021). The authors found that at least 50% of the sites within each of the 5 regions had at least 1 chronic benchmark exceedance. Most of these exceedances were for aquatic invertebrate benchmarks, however 16% of sites in the Midwest region and 4% of sites in the South region displayed chronic fish benchmark exceedances.

When pesticides are released into the environment, they frequently end up as contaminants in aquatic environments. Depending on their physical properties some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds, known as degradates, or transformation products. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence. In a recent study conducted in 2021, pesticide parents and transformation products were measured in 442 small streams across 5 major regions of the U.S. (sampled from 2013-2017) within basins with differing land uses, therefore representing the most spatially extensive study to date of pesticide degradates in surface waters (Mahler et al. 2021). Parent pesticide compounds were detected at 95% of the 442 sites sampled, while degradates occurred at 90% of the 442 sites sampled across the 5 regions of the U.S., indicating that pesticide degradates are almost as widespread as pesticide parent compounds, although the parent compounds exceeded aquatic life benchmarks more often than the degradates did. There were 100 unique degradates found in the Midwest, 77 unique degradates found in the Pacific Northwest and Northeast, 74 in the Southeast, and 68 in Coastal California.

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom et al. 2006). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural waterbodies as mixtures than as individual compounds. In a study conducted in 2006, mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. More than 90% of the time, water from streams in these developed land use settings had detections of 2 or more pesticides or degradates. About 70% and 20% of the time, streams had 5 or more and 10 or more pesticides or degradates, respectively (Gilliom et al. 2006). NAWQA analysis of all detections indicated that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams and that the number of unique mixtures varied with land use (Gilliom et al. 2006).

Similarly, a 2018 study measured dissolved pesticide concentrations at 100 sampling sites throughout the Midwestern United States, covering portions of 11 states (Nowell et al. 2018). Ninety-nine of the 100 sampling sites had drainage basins ranging from 2 – 2870 km^2 in size, while the remaining site had a basin size of 6350 km^2. Of the 100 drainage basins, 88 were considered agricultural and 12 had a large urban influence. Water samples were taken at each of these sites approximately once per week over the course of 14 weeks (May 7 – August 9) in 2013. A total of 1197 water samples were collected over this time period, and in 99.9% of the samples (in all samples but 1), mixtures of 2 or more pesticides were observed. The pesticides detected in these Midwestern streams represent more complex mixtures than have been previously reported. The number of pesticide compounds detected per sample ranged from 1-62, with a median of 25 compounds detected. The number of pesticide compounds detected per site ranged from 24-79, with a median of 54 compounds detected. The composition of the various mixtures varied with the land use of the basin; the least developed sites with the lowest level of agricultural and urban use had the lowest number of pesticide compounds while the sites with a medium to high level of agricultural use had the highest median number of herbicide compounds and the most urban sites had the highest median numbers of insecticide and fungicide compounds.

In a study conducted in 2021, data collected from 5 major regions of the U.S. from 2013-2017 (Midwest in 2013, Southeast in 2014, Pacific Northwest in 2015, Northeast in 2016, and the Central California Coast in 2017) as part of USGS's Regional Stream Quality Assessment studies was used to investigate the effect of region and urbanization on the occurrence and potential toxicity of dissolved pesticide mixtures (Nowell et al. 2021). The authors sampled 271 streams across the 5 U.S. regions and analyzed 225 pesticide compounds. They found that 16 pesticides were consistently detected across the 5 regions and accounted for 83% of pesticides in the 20 most frequently occurring 2-compound mixtures at urban sites. They also found that the concentrations of these 16 urban signature pesticides, the complexity of the pesticide mixture, and the potential toxicity to organisms all increased with increasing basin urbanization. The potential toxicity was highest for invertebrates (with benchmarks exceeded in 51% of urban streams) and lowest for plants and fish.

Pollution originating from a discrete location such as a pipe discharge or wastewater treatment outfall is known as a point source. Point sources of pollution require a National Pollutant Discharge Elimination System (NPDES) permit. These permits are issued for aquaculture, concentrated animal feeding operations, industrial wastewater treatment plants, biosolids (sewer/sludge), pre-treatment and stormwater overflows. The EPA administers the NPDES permit program and states certify that NPDES permit holders comply with state water quality standards. Nonpoint source discharges do not originate from discrete points; thus, nonpoint sources are difficult to identify, quantify, and are not regulated. Examples of nonpoint source pollution include, but are not limited to, urban runoff from impervious surfaces, areas of fertilizer and pesticide application, sedimentation, and manure.

According to EPA's database of NPDES permits, about 243 NPDES individual permits are co-located with listed Pacific salmonids in California. Collectively, the total number of EPA-recorded NPDES permits in Idaho, Oregon, and Washington, that are co-located with listed Pacific salmonids is 1,978.

On November 27, 2006, EPA issued a final rule which exempted pesticides from the NPDES permit process, provided that application was approved under FIFRA. The NPDES permits, then, did not include any point source application of pesticides to waterways in accordance with FIFRA labels. On January 7, 2009, the Sixth Circuit Court of Appeals vacated this rule (National

Cotton Council v. EPA, 553 F.3d 927 (6th Cir. 2009)). The result of the vacatur, according to the Sixth Circuit, is that "discharges of pesticide pollutants are subject to the NPDES permitting program" under the CWA. In response, EPA has developed a Pesticide General Permit through the NPDES permitting program to regulate such discharges.

9.2.3.2 Baseline Pesticide Consultations

NMFS has consulted with EPA on the registration of numerous pesticides. In 2008, NMFS determined that EPA was unable to insure that the use of chlorpyrifos, diazinon, and malathion was not likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPS (NMFS 2008b). The Fourth Circuit Court of Appeals remanded this opinion on February 21, 2013. The 2017 and 2022 opinions subsequently addressed these issues. NMFS (NMFS 2009b) further determined that EPA was unable to insure that both the uses of carbaryl and carbofuran were not likely to jeopardize the continued existence of 22 salmonid ESUs/DPSs and the uses of methomyl were not likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids. NMFS also published conclusions regarding the registration of 12 different a.i.s (NMFS 2010b). NMFS concluded that EPA was able to insure that pesticide products containing azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that EPA was unable to insure that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are not likely to jeopardize the continued existence of some listed Pacific Salmonids and to destroy or adversely modify designated habitat of some listed salmonids. NMFS issued a biological opinion on the effects of 4 herbicides and 2 fungicides (NMFS 2011b). NMFS concluded that EPA was unable to insure that the products containing 2,4-D are not likely to jeopardize the existence of all listed salmonids, and not likely to destroy or adversely modify the critical habitat of some ESU / DPSs. EPA was unable to insure that products containing chlorothalonil or diuron were not likely to destroy or adversely modify critical habitat, but was able to insure that they were not likely to jeopardize listed salmonids. NMFS also concluded that EPA was able to insure that products containing captan, linuron, or triclopyr BEE were not likely to jeopardize the continued existence of any ESUs/DPSs of listed Pacific salmonids or destroy or adversely modify designated critical habitat. NMFS still found; however, that an ITS was necessary for each of these chemicals to minimize the effects of the take of individuals of listed species. In 2012, NMFS completed 2 additional biological opinions covering 4 more pesticides. In May, 2012 NMFS issued a biological opinion on oryzalin, pendimethalin, and trifluralin concluding that EPA was unable to insure that each of these chemicals were not likely to jeopardize the continued existence of some listed Pacific salmonids and not likely to destroy or adversely modify designated critical habitat of some listed salmonids (NMFS 2012b). In July 2012, NMFS issued a biological opinion on thiobencarb, an herbicide authorized for use only on rice. California is the only state within the range of listed Pacific salmonids that has approved the use of thiobencarb and is the only state among the action area states that grows rice. The thiobencarb biological opinion focused on 3 listed Pacific salmon ESUs/DPSs in California's Central Valley where rice is grown. NMFS concluded EPAs registration of thiobencarb would harm ESA-listed species, but that EPA was still able to insure that their action would not jeopardize the continued existence of these 3 species and would not destroy or adversely modify their designated critical habitat. In 2013, NMFS issued a biological opinion on the registration of

3 pesticides: diflubenzuron, fenbutatin oxide, and propargite. NMFS concluded that EPA was unable to insure that products containing diflubenzuron, fenbutatin oxide, and propargite were not likely to jeopardize the existence of many listed salmonids and are not likely to destroy or adversely modify the critical habitat of many ESU / DPSs. In 2021 NMFS issued biological opinions on the registration of 4 pesticides: bromoxynil, prometryn, 1,3-Dichloropropene, and metolachlor. NMFS found that EPA was able to insure that their registration of these 4 pesticides was not likely to jeopardize species or destroy or adversely modify designated critical habitat. In 2022 NMFS issued a revised conference and biological opinion on the registration of chlorpyrifos, malathion, and diazinon (original completed in 2017). NMFS coordinated with EPA, USDA and the pesticide registrants to reach agreement on additional protections or “conservation measures” that are to be incorporated into pesticide labels to reduce pesticide exposure to habitats of listed species. NMFS determined that these measures would avoid jeopardy and minimize incidental take of ESA-listed species. All of NMFS previous biological opinions on pesticides can be found at https://www.fisheries.noaa.gov/national/consultations/pesticide-consultations.

9.2.3.3 Pesticide Usage

Pesticide environmental mixtures may increase risk to listed-species because of additive or synergist effects. Accepted methodologies for calculating mixture toxicity indicate that additivity is the appropriate initial assumption (Cedergreen 2014) unless available data suggest antagonism (less than additive toxicity) or synergism (greater than additive toxicity) is more appropriate.

To assess pesticide environmental mixtures, we examined land use categories within each species range by performing an overlap analysis with the NLCD information (NLCD, 2019) (e.g., Table 113). USGS surface water monitoring e.g., (Ryberg et al. 2014) suggest predictable associations between pesticide detections and land use for both agricultural and urban applications. As such, we used land use categories such as “cultivated crops”, “pasture/hay”, and “developed land” as proxies for areas with an increased potential for environmental mixtures. Additional sources of information available to characterize the occurrence of pesticide environmental mixtures include: species recovery plans, status updates, listing documents, pesticide monitoring data, pesticide usage information, and incident data. We also consider existing consultations on pesticide use within the species range.

Pesticide usage information was considered in the environmental baseline. Note that pesticide usage information is just 1 of numerous types of information qualitatively considered when evaluating pesticide environmental mixtures within species habitats.

The term “use” describes the authorized parameters (e.g., application rate, frequency, crop type, etc.) of pesticide application as described on the FIFRA label. EPA authorizes the FIFRA label that describe when, where, and how pesticide products can legally be applied. Therefore, the label defines the Federal action and is the subject of the analysis in the “Effects of the Action” portion of this opinion. A related concept is that of “usage” which describes parameters (e.g., rate, frequency, percent treated) related to the ways in which a particular pesticide has been applied in the past. In short, use describes how pesticides are authorized to be applied whereas usage describes how pesticides have been applied in the past. Both use and usage can change over time. While use of carbaryl and methomyl defines the action being evaluated in this opinion, the usage of all pesticides and other stressors that occur in the action area from past and present actions are also evaluated in the environmental baseline section. Ultimately, the conclusions regarding the species and designated critical habitat are derived through an integration of the information presented in the Status, Environmental Baseline, Effects of the Action, and Cumulative Effects sections of the opinion.

EPA has provided NMFS with national and state use and usage summaries for carbaryl and methomyl. The use information (i.e., registered use sites and application rates) comes from approved product labels and summarizes the maximum permitted usage. The usage information within these reports comes from both direct pesticide usage reporting (e.g., California Department of Pesticide Regulation) as well as usage estimates from proprietary surveys (e.g., the AgroTrak Study from Kynetec USA, Inc). This and other pesticide usage information is considered as part of the environmental baseline i.e., “past and present impacts of all Federal, State, or private actions” as defined in 50 CFR 4.

9.2.4 Reports of Ecological Incidents

FIFRA section 6(a)(2) requires pesticide product registrants to report adverse effects information, such as incident data involving fish and wildlife. Criteria require reporting of large-scale incidents. For example, pesticide registrants are required to report the following (40 CFR part 159):

- Fish – Affecting 1,000 or more individuals of a schooling species or 50 or more individuals of a non-schooling species.
- Birds – Affecting 200 or more individuals of a flocking species, or 50 or more individuals of a songbird species, or 5 or more individuals of a predatory species.
- Mammals, reptiles, amphibians – Affecting 50 or more individuals of a relatively common or herding species or 5 or more individuals of a rare or solitary species.

The number of documented incidents is believed to be a very small fraction of total incidents caused by pesticides for a variety of reasons. Incident reports for non-target organisms typically provide information only on mortality events and plant damage. Sub-lethal effects in organisms such as abnormal behavior, reduced growth and/or impaired reproduction are rarely reported, except for phytotoxic effects in terrestrial plants. An absence of reports does not necessarily equate to an absence of incidents given the nature of the incident reporting.

Information on the potential effects of pesticides on non-target plants and animals is compiled in the Ecological Incident Information System (EIIS). The EIIS is a database containing adverse effect (typically mortality) reports on non-target organisms where such effects have been associated with the use of pesticides. Other Ecological Incident databases used are the Incident Data System (IDS), Aggregated Incident Database, and Avian Information Monitoring System (AIMS).

Each incident record indicates whether the incident occurred due to a misuse, registered use, or whether it is undetermined. Each incident is additionally classified with a certainty of the association with the identified active ingredient and are classified as: “highly probable,” “probable,” “possible,” and “unlikely.”

9.2.4.1 Incidents Involving Carbaryl

The following summary of ecological incidents was provided in EPA’s 2017 BE for carbaryl.

A review of the aggregate ecological incidents involving carbaryl was completed on December 23, 2019. The Aggregate Incident Report database contains information on 18 “minor” wildlife incidents. The database also includes 12 incidents associated with carbaryl for “other non-target” species (unspecified) that are also classified as “minor.”

With respect to ecological incidents involving fish reported in the Incident Database System (IDS), a total of 6 fish-kill incidents were reported for carbaryl. Only 1 of those incidents, report #B0000-501-92, could be credibly associated with a specific carbaryl use; i.e., to control gypsy moth in New Jersey in 1980. No data on residues were provided.

In an incident (I000910-001) in Louisiana, a fish kill was reported to have occurred in early June 1992. A number of pesticides (carbaryl, MSMA, atrazine, iprodione, dimethylamine, dicamba with 2,4-D, and chlorpyrifos) had been applied to area lawns and golf courses prior to the incident, which followed a high rain event. No chemical residues were reported; however, carbaryl had not been applied in the area since late April, while chlorpyrifos (bluegill LC_{50} = 0.0018 mg/L) (USEPA 2009a) and iprodione (Channel catfish LC_{50} = 3.1 mg/L)(USEPA 2009b) had been applied less than a week before the incident. It is unlikely that carbaryl residues would have been sufficiently high to result in a fish kill if the chemical had been applied 2 months prior. Both chlorpyrifos and iprodione are more likely candidates for being responsible for this fish kill.

A number of pesticides (toxaphene, carbaryl, endrin, methyl parathion and DDT) were associated with a fish kill in Oklahoma where approximately 22,000 catfish died (B0000-246-01). No residue data were provided; however, given that toxaphene and endrin are both classified as very highly toxic to catfish with LC_{50} values of 0.0027 mg/L and 0.013 mg/L (NIH 2019), respectively, it is likely that they are more credible candidates for having caused the fish kill than carbaryl.

In 2001, a large incident (several thousand fish) occurred in the San Joaquin River in California (I013436-001). The fish were primarily threadfin shad and small catfish (< 3 in). A variety of pesticides were found in the river water and in discharges to the river, including demeton-S, diazinon, naled (dibrom), disulfoton and azinphos methyl. Dioxathion, carbaryl, carbofuran, fenuron, methomyl, and monuron were found in the gill tissue of the fish. Carbaryl was found only in the fish tissue at 1.75 mg/kg. Azinphos methyl was found at 0.016 mg/L in water from an agricultural drain entering the river, and 0.002-0.008 mg/L in the San Joaquin River itself. It is possible that azinphos methyl was the cause of the fish kill rather than carbaryl.

For 2 other incidents in Texas in 1994 (I001297-011) and 2004 (I015419-664), insufficient information was provided in the report to allow any evaluation of a cause and effect relationship with carbaryl.

No incidents specific for effects to aquatic invertebrates were available. The Aggregate Incident Reports database identified 18 incidents linked to carbaryl use as aggregated counts of minor wildlife incidents (W-B) and 12 reported for other non-target (ONT). Because limited details about these incidents were reported, no information was available on the use site, the certainty level, or on the types of organisms that were involved. See EPA's carbaryl BE for more details.

9.2.4.2 Incidents Involving Methomyl

The following summary of ecological incidents was provided in EPA's 2021 BE for methomyl.

There are currently (as of Jan. 22, 2020) 2 aquatic animal incident reports in the Incident Data System (IDS) with a certainty index of 'possible', 'probable' or 'highly probable'. For these 2 incidents, the legality of use was undetermined. The following discussion only includes those incident reports with a certainty index of 'possible', 'probable' or 'highly probable' and a legality classification of 'registered' and 'undetermined' (the incidents that were caused by a misuse are not reported further).

The dates of the fish-kill incident reports range from 1992 to 2001 and both are fairly large (from approximately 125 dead fish to "several thousand"). The incidents involve a variety of fish species (bluegill, bowfin, carp, catfish and threadfin shad) and in 1, methomyl residues of 5.08 ppm were reported in composited gill samples. One of the incidents is associated with the corn use, but for the other, the use site is not reported or unknown. The methomyl product involved is Lannate LV for 1 incident but not reported for the other incident; both incidents involve at least 1 pesticide in addition to methomyl. Overall, the incident data that are available indicate that exposure pathways for methomyl are complete and that exposure levels are sufficient to result in field-observable effects.

Reports contained in the database must be interpreted in the context that 1) not all incidents are expected to be reported and 2) in many instances it is difficult to establish a direct cause-effect relationship. Generally, if there are a significant number of incidents associated with the use of a certain pesticide, it is an indication that the pesticide may pose a higher environmental risk. However, the lack of reported incidents does not necessarily indicate a lack of incidents.
In addition to the non-aggregated aquatic incident reports available in IDS, there have also been a total of 12 aggregate wildlife incidents and 1 other non-target incident reported to the Agency. Of these 13, 7 are associated with active registrations (6 involve products either no longer registered or no registration numbers reported).

Since 1998, incidents that are allowed to be reported aggregately by registrants [under FIFRA 6(a)(2)] include those that are associated with an alleged effect to wildlife (birds, mammals, or fish) without differentiation between species or terrestrial and aquatic environments. Typically, the only information available for aggregate incidents is the date (i.e., the quarter) that the incident(s) occurred, the number of aggregate incidents that occurred in the quarter, and the PC code of the pesticide and the registration number of the product involved in the incident. Because of the limited amount of data available on aggregate incidents it is not possible to assign certainty indices or legality of use classifications to the specific incidents. Therefore, the incidents associated with currently registered products are assumed to be from registered uses unless additional information becomes available to support a change in that assumption.

There are currently (as of January 22, 2020) 2 aquatic animal incident reports in IDS with a certainty index of 'possible', 'probable' or 'highly probable'. None of the reported incidents involved aquatic invertebrates (although 1 incident in the IDS aggregate database was classified as "other non-target" and could have involved aquatic invertebrates); however, absence of reported incidents does not ensure that none occurred. Overall, the incident data that are available indicate that exposure pathways for methomyl are complete and that exposure levels are sufficient to result in field-observable effects to aquatic organisms, in general. See EPA's methomyl BE for more details.

9.2.5 Non-Native Species

Plants and animals that are introduced into habitats where they do not naturally occur are called non-native species. They are also known as non-indigenous, exotic, introduced, or invasive species, and have been known to affect ecosystems. Non-native species are introduced to the environment in a variety of ways, including through ballast water contamination and release, intentional or accidental releases of aquaculture or aquarium species, and releases of live bait. The Aquatic Nuisance Species Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. In 1989, non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the previous 100 years (Miller et al. 1989). In 2018, a systemic review was conducted on the interactions between invasive species and ESA federally-listed species in the U.S. and 175 unique case studies containing listed species were analyzed (Dueñas et al. 2018). Within these case studies, 116 ESA-listed species were mentioned, and of these, 85 listed species were impacted negatively, making up 6.2% of the entire ESA list. The authors found that the proportion of ESA-listed species that were negatively impacted was higher for marine species than it was for terrestrial species. These negatively impacted marine animals included 4 species of sea turtles (impacted on the terrestrial portion of their range) and 3 species of anadromous fish impacted in freshwater habitat (Dueñas et al. 2018).

Non-native species can interact with native species both directly, as competitors and predators, or indirectly, as vectors of new disease pathogens or as plants that alter the aquatic habitats in which the native species live (Thorstad et al. 2021). Non-native species also may be affected by increased temperatures resulting from climate change which could lead new and/or different combinations of species to invade native waterways (Thorstad et al. 2021). By competing with native species for food and habitat, preying on them, infecting them with new diseases, or altering their habitats, non-native species can reduce or eliminate populations of native species.

9.2.6 Impaired Water Bodies

Under the authority of the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses including drinking, recreation, aquatic habitat, and industrial uses are impaired by pollutants. This process is in accordance with section 303(d) of the CWA. Estuaries, lakes, and streams listed under 303(d) are those that are considered impaired or threatened by pollution, and are not expected to improve within the next 2 years.

Each state has unique 303(d) listing criteria and processes. Generally, a water body is listed separately for each standard it exceeds, so it may appear on the list more than once. If a water body is not on the 303(d) list, it is not necessarily contaminant-free; rather it may not have been tested. Therefore, the 303(d) list is a minimum list for each state regarding polluted water bodies by parameter. After states develop their lists of impaired waters, they are required to prioritize and submit their lists to EPA for review and approval. Each state establishes a priority ranking for such waters, considering the severity of the pollution and the uses to be made of such waters. States are expected to identify high priority waters targeted for TMDL development within 2 years of the 303(d) listing process.

Elevated temperature is considered a pollutant in most states with approved Water Quality Standards under the federal CWA. Temperature is significant for the health of aquatic life. Water temperatures affects dissolved oxygen saturation levels. Temperature and dissolved oxygen are inversely related. Increases in stream temperatures can be caused by many factors. Land-use and water withdrawals are primary causes, and increases in global temperatures with climate change will exacerbate the local conditions found in our Nation's watersheds. This, in turn, affects aquatic life. The local and cumulative conditions within watersheds affect the distribution, health, and survival of native cold-blooded listed fish species distributed over the action area. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. For the various listed fish species, water temperature tolerance varies between species and life stages. For various listed salmon species, optimal temperatures for rearing salmonids range from 10ºC to 16ºC. Warm temperatures can reduce fecundity, reduce egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, decrease the ability of young salmon and trout to compete with other species for food, and to avoid predation (McCullough 1999; Spence et al. 1996). Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease (Colgrove and Wood 1966) or elevating metabolic demand (Brett 1995). Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6ºC (McCullough 1999). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures (Crossin et al. 2008). Ambient stream temperatures may also negatively affect incubating and rearing salmonids when too high (Gregory and Bisson 1997). In a recent modeling study that investigated impacts to a variety of salmon populations across multiple life stages, it was found that salmon populations rapidly declined in response to rising sea surface temperatures across diverse model assumptions and climate scenarios, with the marine life stage showing a 90% decline (Crozier et al. 2021). Sturgeon are also dependent on temperature; generally, sturgeon overwinter until temperatures reach about 7 ºC in the spring. They get a little more active and begin foraging and spawning. For Atlantic sturgeon, summer temps above 27 ºC cause a sharp drop in growth and increase in stress. While the distribution range of North

American Sturgeon extends over a zone with the temperature variation up to 30 ºC, they generally prefer and perform optimally under cool (e.g., under 25 ºC) temperature conditions (Haxton et al. 2016). Due to the sensitivity of listed fish species to temperature, states have established lower temperature thresholds for habitat as part of their water quality standards.

Other water quality impairments found in waterbodies that can lead to adverse effects to listed aquatic organisms include: dissolved oxygen, nutrients, biological oxygen demand (BOD), toxics and other chemical inputs, and total suspended solids, to name a few. In general, the abundance of aquatic life and other fish is markedly reduced with the increased exposure to stressful water temperatures, depressed DO levels, and/or exposures to toxics. The reduction of suitable habitat caused by drought conditions exacerbate the response to other stressful variables and further reduce the abundance of fish.

Excessive stream temperatures affect dissolved oxygen (DO) concentrations in the water column. DO is the requirement for aquatic life respiration, and high DO levels contribute to the diversity of aquatic life that would benefit Pacific salmon and other listed species. In addition to the inverse association with temperature, DO concentrations can be compressed by other aquatic inputs that can lead to exceeding standards. Nutrients from agricultural and urban inputs allow phytoplankton and aquatic plants to flourish causing diurnal pulses in DO concentrations. As the season progresses into the warmer months, DO consumption is greater than what is produced. Plant senescence, along with low flows can cause excessive demand on DO. Decay requires DO to complete its process; use of oxygen for this purpose is referred to as biological oxygen demand (BOD). The higher the decay needs, the higher the BOD in the system. High BOD is also inversely related to DO. High temperatures and BOD combined can suppress DO enough to cause lethal conditions for fish and other aquatic life. Thus water bodies that exceed standards for nutrients and BOD levels can affect the distribution of aquatic life including Pacific salmonids and other fish. As stated above, DO is a requirement for aquatic life respiration. High DO levels contribute to the diversity of aquatic life that would benefit Pacific salmon and other species.

Toxics and other chemical inputs also contribute to adverse responses in the aquatic environment. In addition to causing acute or chronic harm to aquatic species, chemical inputs can indirectly affect them by contributing to reductions in DO through their decay known as the chemical oxygen demand (COD). Areas where these inputs occur can greatly affect the distribution of fish and other organisms. Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, polybrominated diphenyl ethers (PBDEs) compounds, PAHs, nutrients (phosphorus and nitrogen), and sediment (Table 112). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt (Moore et al. 2017; Paul and Meyer 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 112. Examples of Water Quality Contaminants in Residential and Urban Areas.

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
Fertilizers	Nutrients	Phosphorus Nitrogen	lawns, golf courses, urban landscaping
Heavy Metals	Pb, Zn, Cr, Cu, Cd, Ni, Hg, Mg	Cu	brake pad dust, highway and parking lot runoff, rooftops
Pesticides including- Insecticides (I) Herbicides (H) Fungicides (F) Wood Treatment chemicals (WT) Legacy Pesticides (LP) Other ingredients in pesticide formulations (OI)	Organophosphates (I) Carbamates (I) Organochlorines (I) Pyrethroids (I) Triazines (H) Chloroacetanilides (H) Chlorophenoxy acids (H) Triazoles (F) Copper containing fungicides (F) Organochlorines (LP) Surfactants/adjuvants (OI)	Chlorpyrifos (I) Diazinon (I) Carbaryl (I) Atrazine (H) Esfenvalerate (I) Creosote (WT) DDT (LP) Copper sulfate (F) Metalaxyl (F) Nonylphenol (OI)	golf courses, right-of-ways, lawn and plant care products, pilings, bulkheads, fences
Pharmaceuticals and personal care products	Natural and synthetic hormones soaps and detergents	Ethinyl estradiol Nonylphenol	hospitals, dental facilities, residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricyclic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs	Penta-PBDE	utility infrastructure, flame retardants,

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
	Dioxins		electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Wheeler et al. 2005). The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50 - 7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

PAH compounds also have distinct and specific effects on fish at early life history stages (Incardona et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson et al. 2002). In a recent study, the sources of PAHs to streambed sediment were investigated in 10 urban watersheds in 3 regions of the United States, and it was found that total PAH sediment concentrations were significantly higher in the Northeast and Southeast than they were in the Northwest; the Northeast and Southeast commonly use coal-tar pavement sealant (CTS) where the Northwest doesn't, so CTS can be considered a source of PAHs to the aquatic environment (Van Metre et al. 2022). Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate species (Honda and Suzuki 2020; Meador et al. 1995; Varanasi et al. 1989; Varanasi et al. 1992). PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (Johnson et al. 2002). Although PAHs do not bioaccumulate in vertebrate tissues, PAHs cause a variety of deleterious effects in exposed animals. Some PAHs are known to be immunotoxic and to have adverse effects on reproduction and development. Studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals (Honda and Suzuki 2020). There have been several toxicological studies on the early development, bone metabolism, liver metabolism, and reproduction of fish in response to exposure to PAHs (Honda and Suzuki 2020).

Total suspended solids can affect aquatic life by limiting photosynthetic activities (also affecting DO). Excessive sedimentation can blanket substrates needed for spawning, incubation, and emergence of Pacific and Atlantic salmon and suppress inter-gravel DO concentrations.

For each region discussed below, we used GIS layers made publically available through USEPA's Assessment and Total Maximum Daily Load Tracking and Implementation System (ATTAINS) to determine the number of km on the 303(d) list that were impaired for any reason; some waterbodies were impaired for more than 1 reason, or "Parent Cause." We then grouped

"Parent Causes" by similarity into 9 separate groups, and calculated the number of km that were impaired for those more specific reasons. Finally, we calculated the percent of total stream impairment that each group contributed to (Table 117, Table 122, Table 123, Table 128). Because the 303(d) list is limited to the subset of rivers tested, the values in the Tables should be regarded as lower-end estimates. While some ESU/DPS ranges do not contain any 303(d) listed waterbodies, others show considerable overlap. These comparisons demonstrate the relative significance of various water quality impairments among ESUs/DPSs.

9.3 Pacific Island Region

The Pacific island region includes the Hawaiian Islands, American Samoa, Guam, and the Northern Mariana Islands (Figure 78).

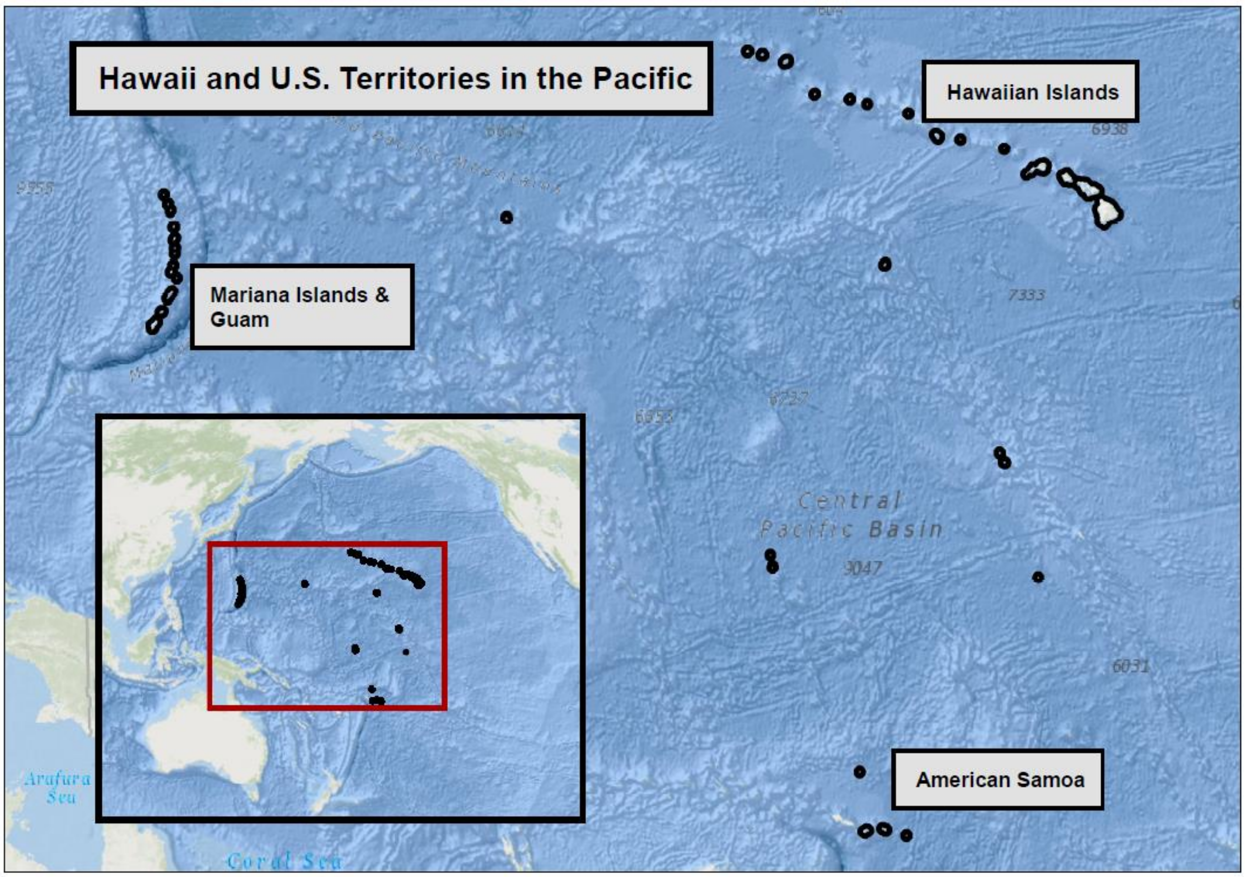


Figure 78. Hawaiian Islands and US territories in the Pacific

Five of the 59 species addressed in this opinion occur in this subregion. They are the coral species: *Acropora globiceps*, *Acropora retusa*, *Acropora speciosa*, *Euphyllia pardivisa*, and *Isopora crateriformis*.

9.3.1 Coral species in the Pacific

Climate Change and Acidification. The reefs of American Samoa are not immune to the impacts from the global phenomenon of climate change. The global mean temperature has risen by 0.76 °C over the last 150 years, and much of that increase has occurred over the past 50 years (Solomon et al. 2007). The incidence of climate-related events to corals in American Samoa have been minimal compared to many areas around the world. Mass coral bleaching events happened in American Samoa in 1994 (Goreau and Hayes 1994), 2002, 2003 (Fenner et al., 2008), and 2015 (Fenner, personal comm.). There was mortality from some of these events; however, it was not massive. In several backreef pools on Tutuila, some corals bleach every summer, but mortality has been very minimal (Fenner and Heron, 2009). A diverse assemblage of corals living around the Samoan island of Ofu have demonstrated thermal resistance to bleaching despite being continually exposed to temperatures exceeding the regional coral bleaching threshold of 32°C (Barker 2018). There is no evidence yet of effects of acidification on reefs in American Samoa. In 2023, extreme marine heatwaves engulfed much of the eastern tropical Pacific and wider Caribbean and anomalously high sea surface tempertatures in the eastern tropical Pacific and wider Caribbean were more extensive than any other year in the satellite record, which started in the early 1980s, putting coral reefs at increased exposure to heat stress (Hoegh-Guldberg 2023). Historical data suggests the 2023 marine heatwaves throughout the eastern tropical Pacific and wider Caribbean will likely be the precursor to a global mass coral bleaching and mortality event (Hoegh-Guldberg 2023).

Disease. In the Pacific, coral populations continue to be spared from epidemic disease outbreaks. However, a 2003 survey (N=73) of the Northwestern Hawaiian Islands found 10 types of coral diseases (Aeby et al. 2011). The coral diseases were found at most of the survey sites (68.5%) but at low levels of occurrence with an average of 0.5% colonies showing signs of infection (Aeby, 2006). Additional surveys in 2004 and 2005 identified 12 coral diseases across the Hawaiian Archipelago (Aeby et al. 2011). These diseases included Porites growth anomalies, Porites trematodiasis, Porites multi-focal tissue loss, Porites discolored tissue thinning syndrome, Porites brown necrotizing disease, Porites bleaching with tissue loss, Montipora multifoci tissue loss syndrome, Montipora white syndrome, Montipora patchy tissue loss, Montipora growth anomaly, Acropora white syndrome, and Acropora growth anomaly (Aeby et al. 2011). Later, a 2004 disease outbreak around Kauai was determined to be black band disease (Aeby et al. 2015). Montipora white syndrome, which causes acute tissue loss, has been documented throughout the main Hawaiian Islands; however, prevalence of this disease is approximately 4 times higher in Kaneohe Bay, Oahu (average prevalence=0.27 + 0.08% SE) than in the other main islands (Friedlander et al. 2008).

Coral diseases in American Samoa are widespread and present on regularly monitored coral reefs, though only a very small proportion (0.14%) are affected, with the most common disease being the white syndrome, similar to that found on the Great Barrier Reef (Fenner 2019). Likewise, white syndrome appears to be the most prevalent disease in Guam (observed in 9 out of 10 sites) and the source of greatest tissue mortality, though black band disease, brown band disease, ulcerative white spots, and multiple growth anomalies are also present on Guam reefs (Turgeon 2008).

Fisheries Interactions. Fishing in the American Samoa could lead to direct impacts on coral reefs in the harvest area. Fishing pressure in the 1970s in American Samoa was reported as among the highest in the world (Dalzell 1996), although increasing prosperity since then has led to a shift to purchasing food in stores and decreasing fishing pressures (Sabater and Carroll 2009). When fishing does occur, harvest on reef flats is fairly common at the lowest tides, and some other forms of fishing such as hook and line, and throw net are also carried out on reef flats at times. This leads fishermen to sometimes have to walk on corals which directly impacts them; this walking on fragile branching *Acropora* staghorn corals can lead to the branches breaking.

Land-based Contaminants. Runoff carries nutrients from on land, including from piggeries and septic systems. In most areas, the nutrients are probably carried quickly to the ocean with sediment. However, in narrow bays such as Pago Pago Harbor and Vatia Bay, circulation is limited and water residence times are greatly increased, and so runoff nutrients accumulate in the water.

Construction Activities. Construction activities have done considerable damage to some areas of the reefs around Tutuila in the past. Material has been dredged from inner reef flats in several areas to provide material to add to village land, and in the largest such project, to build over the reef flat to construct the airport runways.

Predation. Crown-of-thorns starfish (COTS) eat the tissue off of coral skeletons. They are normally quite rare on reefs, but periodically they reach outbreak proportions and kill almost all of the corals. Outbreaks occurred on Tutuila in 1938 and 1978, with the 1978 outbreak involving millions of COTS that resulted in an estimated 90% or more loss of all corals in the area.

9.4 West Coast Region

The West coastal region includes rocky coasts, estuaries, bays, sub-estuaries and city harbors. In total the west coast contains 2,200 square miles of estuaries, over 60% of which is part of 3 major estuarine systems: the San Francisco Estuary, Columbia River Estuary, and Puget Sound (USEPA 2015). The coastal counties of the West Coast are home to 19% of the US population, and 63% of the total population of the West Coast states. The population in these coastal counties has nearly doubled since 1970 and is currently estimated to be around 40 million people (USEPA 2015).

9.4.1 Coastal Condition Assessment

Figure 79 shows a summary of findings from the EPA's 2010 National Coastal Condition Assessment (NCCA) Report for the West Coast Region (USEPA 2015). A total of 134 sites were sampled to assess approximately 2,200 square miles of West Coast coastal waters. Biological quality was rated as good in 71% of the West coast region based on the benthic index.

According to the more recent 2015 NCCA Report, the biological condition was good in 85% of West Coast estuaries according to the M-AMBI marine benthic index. This is better than the overall national estimate of estuaries in good biological condition of 71% and is a statistically significant increase over the proportion of West Coast area rated good in 2010. About 67% of West Coast estuarine area had good sediment quality based on measures of chemical contaminants found in sediments and laboratory tests of toxicity. While this is lower than the overall estuarine area in the continental US in 2015, it shows statistically significant increase in area (by 34%) over the NCCA results for sediment quality in the West Coast in 2010. Ecological fish tissue contamination was degraded in estuaries of the West Coast in 2015 with only 25% of the area rated good and area rated fair and poor both at 24%. The eutrophication index, which examines the potential for estuarine area to undergo social eutrophication based upon measurements of nutrients, chlorophyll a, dissolved oxygen and water clarity, found that 76% of West Coast estuarine area was in good condition, 19% of area was in fair condition and 5% in poor condition. While this is a slight improvement, the condition in 2015 doesn't represent a statistically significant change since 2010. For more information, see EPA's website (https://www.epa.gov/national-aquatic-resource-surveys/west-coast-estuaries-national-coastal-condition-assessment-2015).

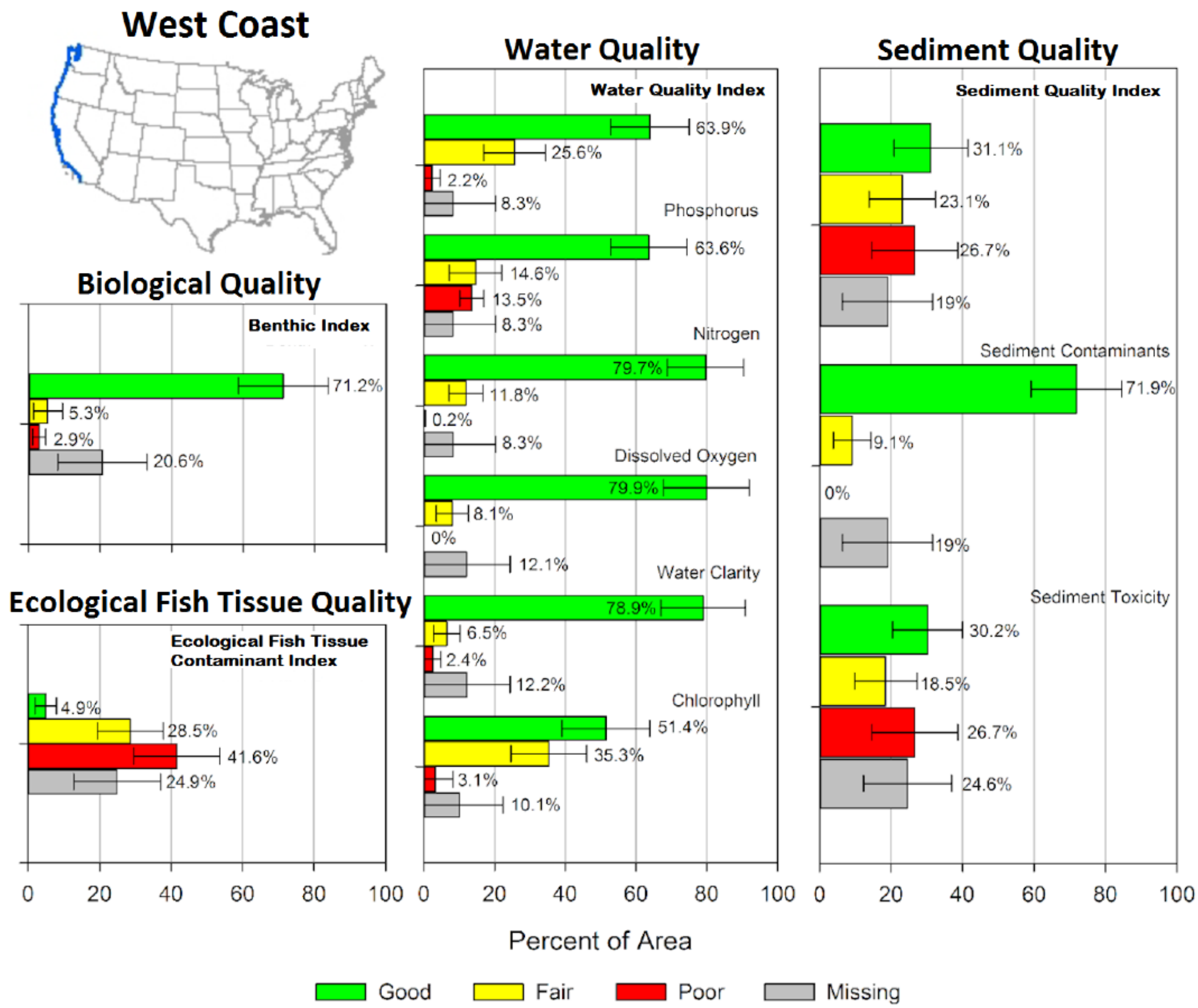


Figure 79. National Coastal Condition Assessment 2010 Report findings for the West Coast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 134 sites sampled). Error bars represent 95% confidence levels (USEPA 2015).

9.4.2 West Coast Salmonids

Baseline Habitat Condition. As noted in the status of the species section, the riparian zones for many of the ESUs/ DPSs are degraded. Riparian zones are the areas of land adjacent to rivers and streams. These systems serve as the interface between the aquatic and terrestrial environments. Riparian vegetation is characterized by emergent aquatic plants and species that thrive in close proximity to water, such as willows. This vegetation maintains a healthy river system by reducing erosion, stabilizing main channels, and providing shade. Leaf litter that enters the river becomes an important source of nutrients for invertebrates (Bisson and Bilby 2001). Riparian zones are also the major source of large woody debris (LWD). When trees fall and enter the water, they become an important part of the ecosystem. The LWD alters the flow, creating the pools of slower moving water preferred by salmon (Bilby et al. 2001). While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle (Bilby and Bisson 2001).

Bilby and Bisson (2001) discuss several studies that associate increased LWD with increased pools, and both pools and LWD with salmonid productivity. Their review also includes documented decreases in salmonid productivity following the removal of LWD. Other benefits of LWD include deeper pools, increased sediment retention, and channel stabilization. Floodplains are relatively flat areas adjacent to streams and rivers that stretch from the banks of the channel to the base of the enclosing valley walls. They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. The floodplain includes the floodway, which consists of the stream channel, and adjacent areas that actively carry flood flows downstream; and the flood fringe, which are areas that are inundated, but which do not experience a strong current. Water stored in the floodplain is later released during periods of low flow. This process ensures adequate flows for salmonids during the summer months, and reduces the possibility of high-energy flood events destroying salmonid redds (Smith 2005).

Periodic flooding of these areas creates habitat used by salmonids. Thus, floodplain areas vary in depth and widths and may be intermittent or seasonal. Storms also wash sediment and LWD into the main stem river, often resulting in blockages. These blockages may force the water to take an alternate path and result in the formation of side channels and sloughs (Benda et al. 2001), which are important spawning and rearing habitat for salmonids. The degree to which these off-channel habitats are linked to the main channel via surface water connections is referred to as connectivity (PNERC 2002). As river height increases with heavier flows, more side channels form and connectivity increases. Juvenile salmonids migrate to and rear in these channels for a certain period of time before swimming out to the open sea.

Healthy riparian habitat and floodplain connectivity are vital for supporting a salmonid population. Chinook salmon and steelhead have life history strategies that rely on floodplains during their juvenile life stages. Chum salmon use adjacent floodplain areas for spawning. Soon after their emergence, chum salmon use the riverine system to rapidly reach the estuary where they mature, rear, and migrate to the ocean. Coho salmon use the floodplain landscape extensively for rearing. Estuarine floodplains can provide value to juveniles of all species once they reach the salt water interface.

Once floodplain areas have been disturbed, it can take decades for their recovery (Smith 2005). Consequently, most land use practices cause some degree of impairment. Development leads to construction of levees and dikes, which isolate the mainstem river from the floodplain. Agricultural development and grazing in riparian areas also significantly change the landscape. Riparian areas managed for logging, or logged in the past, are often impaired by a change in species composition. Most areas in the Pacific Northwest were historically dominated by conifers. Logging results in recruitment of deciduous trees, decreasing the quality of LWD in the rivers. Deciduous trees have smaller diameters than conifers; they decompose faster and are more likely to be displaced (Smith 2005).

Without a properly functioning riparian zone, salmonids contend with a number of limiting factors. They face reductions in quantity and quality of both off-channel and pool habitats. Also, when seasonal flows are not moderated, both higher and lower flow conditions exist. Higher flows can displace fish and destroy redds, while lower flows cut off access to parts of their habitat. Finally, decreased vegetation limits the available shade and cover, exposing individuals to higher temperatures and increased predation.

Parasites and/or Disease. Most young fish are highly susceptible to disease during the first 2 months of life. The cumulative mortality in young animals can reach 90 to 95%. Although fish disease organisms occur naturally in the water, native fish have co-evolved with them. Fish can carry these diseases at less than lethal levels (Foott et al. 2003; Kier Associates 1991; Walker and Foott 1993). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Guillen 2003; Spence et al. 1996). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (Spence et al. 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). Examples of parasites and disease for salmonids include whirling disease, infectious hematopoietic necrosis (IHN), sea-lice (e.g., *Lepeophtheirus salmonis*, various *Caligus* species *Henneguya salminicola*, or Ich (*Ichthyopthirius multifiliis)* and Columnaris *(Flavobacterium columnare*)).

Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The disease occurs in the wild and in hatcheries and results in losses to fry and fingerling salmonids, especially rainbow trout. The disease is transmitted by infected fish, fish parts and birds.

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. Infection results in a variety of conditions including anemia, abnormal behavior, and hemorrhages, often resulting in death. This disease affects rainbow/steelhead trout, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook, sockeye, chum, and coho salmon. The virus is triggered by low water temperatures and is shed in the feces, urine, sexual fluids, and external mucus of salmonids. Transmission is mainly from fish to fish, primarily by direct contact and through the water.

Sea lice is a marine ectoparasite found in coastal waters that can also cause deadly infestations of farm-grown salmon and may affect wild salmon. *Henneguya salminicola*, a protozoan parasite, is commonly found in the flesh of salmonids, particularly in British Columbia. The fish responds by walling off the parasitic infection into a number of cysts that contain milky fluid. This fluid is an accumulation of a large number of parasites. Fish with the longest freshwater residence time as juveniles have the most noticeable infection. The order of prevalence for infection is coho followed by sockeye, Chinook, chum, and pink salmon. The *Henneguya* infestation does not appear to cause disease in the host salmon – even heavily infected fish tend to return to spawn successfully. Additionally, ich (a protozoan) and *Columnaris* (a bacterium) are 2 common fish diseases that were implicated in the massive kill of adult salmon in the Lower Klamath River in September 2002 (Belchik 2004; Guillen 2003).

Predation. Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. Concentrations of juvenile salmon in the coastal zone experience high rates of predation. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations may have reduced the survival of some salmon ESUs/DPSs. Threatened Puget Sound Chinook adults are preferred prey of endangered Southern Resident killer whales.

Wildland Fire. Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and loss of LWD (Buchwalter et al. 2004; Rinne 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (Greswell 1999; Rinne 2004). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (USFS 2000). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Greswell 1999).

The presence of ash also has indirect effects on aquatic species depending on the amount of ash entry into the water. All ESA-listed salmonids rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or the water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year with long-term effects lasting 10 years or more (Buchwalter et al. 2003; Buchwalter et al. 2004; Minshall et al. 2001). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to 4 months after forest fires (Buchwalter et al. 2003).

9.4.3 Artificial Propogation

For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction. These programs were instituted under federal law to lessen the effects of lost natural salmon production within the basin from the dams. Anadromous fish hatcheries have existed in California since establishment of the McCloud River hatchery in 1872. Federal, state, and tribal managers operate the hatcheries. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid populations (*e.g.*, Redfish Lake sockeye salmon, and White River Spring Chinook).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive as first described by(Hard et al. 1992). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-basins. The impacts of these hatchery practices on ESA-listed salmonid populations are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations as described earlier in (Busack 1990; Hard et al. 1992; Reisenbichler 1997; Riggs 1990). These and other studies also raised the concern for disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Fresh 1997; Hard et al. 1992; Steward and Bjornn 1990). Species with extended freshwater residence may face higher risk of domestication, predation, or altered migration than species that spend only a brief time in freshwater (Hard et al. 1992). Nonetheless, artificial propagation may also contribute to the conservation of listed salmon and steelhead. However, it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural populations (Hard et al. 1992).

A range of additional concerns have been identified about the genetic (Fraser 2008) and ecological consequences hatcheries may have for natural fish populations (Rand et al. 2012). Hatchery-raised fish may introduce density-dependent effects on natural populations (Bohlin et al. 2002) and can also negatively affect natural fish through inter-breeding (Bourret et al. 2011). If hatchery-origin fish have reduced fitness, allowing hatchery individuals to breed with natural fish can have negative impacts on those natural populations (Araki et al. 2007; Hindar et al. 1991; Tillotson et al. 2019). Further, recent research suggests that high abundances of hatchery-raised salmon may alter life-history strategies of wild populations, possibly through increased competition (Cline et al. 2019).

To address these concerns, hatcheries must have approved hatchery management plans called Hatchery Genetic Management Plans (HGMPs). HGMPs are technical documents that thoroughly describe the composition and operation of each individual hatchery program. The primary goal of an HGMP is to describe biologically-based artificial propagation management strategies that ensure the conservation and recovery of ESA-listed salmon and steelhead populations. NMFS uses the information provided by HGMPs to evaluate impacts on salmon and steelhead listed under the ESA. Completed HGMPs may also be used for regional fish production and management planning by federal, state and tribal resource managers.

9.4.4 Commercial, Recreational and Subsistence Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. For example, NMFS has issued permits under Section 10 that have allowed these activities to be exempted from Section 9 prohibitions. Section 4(d) rules issued by NMFS provide exemptions from take for resource, harvest, and hatchery management plans. Furthermore, there are several treaties that have reserved the right of fishing to tribes in the Northwest Region.

Management of salmon fisheries in the Washington-Oregon-Northern California drainages are a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from 3 to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out 3 miles from state coastlines. The states of Oregon, Idaho, California and Washington issue salmon fishing licenses for these areas.

Management of salmon fisheries in the Columbia River Basin is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries regulations in federal waters from 3 to 200 miles off the coasts of Washington, Oregon, and California. Salmon and steelhead fisheries in the Columbia River and its tributaries are co-managed by the states of Washington, Oregon, Idaho, 4 treaty tribes, and other tribes that traditionally have fished in those waters. A federal court oversees Columbia River harvest management through the U.S. v. Oregon proceedings. Inland fisheries are those in waters within state boundaries, including those extending out 3 miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for these areas.

There are Treaty Indian and non-Treaty fisheries which are managed subject to state and tribal regulation, consistent with provisions of a U.S. v. Oregon 2008 agreement. Treaty Indian fisheries are managed subject to the regulation of the Columbia River Treaty Tribes. They include all mainstem Columbia River fisheries between Bonneville Dam and McNary Dam, and any fishery impacts from tribal fishing that occurs below Bonneville Dam. Tribal fisheries within specified tributaries to the Columbia River are included. Non-Treaty fisheries are managed under the jurisdiction of the states.

Management of salmon fisheries in the Puget Sound Region is a cooperative process involving federal, state, tribal, and Canadian representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from 3 to 200 miles off the coasts of Washington, Oregon, and California. The annual North of Falcon process sets salmon fishing seasons in waters such as Puget Sound, Willapa Bay, Grays Harbor, and Washington State rivers. Inland fisheries are those in waters within state boundaries, including those extending out 3 miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for these areas. Adult salmon returning to Washington migrate through both U.S. and Canadian waters and are harvested by fishermen from both countries. The 1985 Pacific Salmon Treaty helps fulfill conservation goals for all members and is implemented by the 8-member bilateral Pacific Salmon Commission. The Commission does not regulate salmon fisheries, but provides regulatory advice.

Management of salmon fisheries in the Southwest Coast Region is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from 3 to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out 3 miles from state coastlines. The states of Oregon, Idaho, California, and Washington issue salmon fishing licenses for inland fisheries. The California Fish and Game Commission (CFGC) establish the salmon seasons and issues permits for all California waters and the Oregon Department of Fish and Game sets the salmon seasons and issues permits for all Oregon waters.

9.4.5 Southern Resident Killer Whale

Natural Mortality. As apex predators, sources of natural mortality in SR killer whales are likely limited. Possible sources can still include disease and parasitism. While disease is not known to limit any killer whale population and no epidemics are known in the SR killer whale DPS, killer whales may be vulnerable to disease outbreaks given their distribution patterns and strong social networks (Altizer et al. 2003; Guimarães Jr et al. 2007). A variety of pathogens have been identified in killer whales, and there are other pathogens in sympatric marine mammal species that could be transmittable to killer whales (Gaydos et al. 2004).

Prey Availability. SR killer whales predominantly prey upon salmonids, particularly Chinook salmon. Maintaining a robust prey resource is essential to SR killer whale recovery; the U.S. recovery goal of 2.3% annual growth over 28 years would imply a 75% increase in energetic requirements (Williams et al. 2011). Limited prey availability can have detrimental effects for SR killer whales, including requiring the whales to spend more time and energy foraging, possibly causing negative effects on reproductive rates and morality. Inadequate prey is a source of stress for SR killer whales, and a comparatively greater one than vessel traffic (Ayres et al. 2012). Nutritional stress has also been thought to be a contributing factor to slower growth rates in SR killer whales (Fearnbach et al. 2011). Prey availability is also a possible influencing factor in the interconnectivity of SR killer whale social network (Foster et al. 2012).

Pollution and Contaminants. Persistent organic pollutants (POPs) is a collective term for environmental contaminants like dioxins, furans, PCBs, etc. These chemicals are used (or have previously been used) in pesticides, industrial manufacturing, and pharmaceutical production, to name a few applications. The relative contribution of any 1 source in contaminating killer whales with POPs is poorly understood (NMFS 2008). As a long-lived, top marine predator, SR killer whales bioaccumulate POPs in their tissues and blubber, potentially leading to numerous adverse health effects such as skeletal deformity, reproductive dysfunction, impaired immune function, and enzyme disruption (Krahn et al. 2009). Levels of contaminants in wild individuals are much higher than those found in captive killer whales (Bennett et al. 2009). Numerous factors can affect concentrations of POPs in marine mammals, such as age, sex and birth order, diet, and habitat use (Mongillo et al. 2012). In marine mammals, POP contaminant load for males increases with age, whereas females pass on contaminants to offspring during pregnancy and lactation (Addison and Brodie 1987; Borrell et al. 1995). POPs can be transferred from mothers to juveniles at a time when their bodies are undergoing rapid development, putting juveniles at risk for immune and endocrine system dysfunction later in life (Krahn et al. 2009).

Oil Spills. Exposure to petroleum hydrocarbons released into the environment via oil spills and other discharge sources represents a serious and potentially catastrophic risk for SR killer whales. The substantial volume of shipping traffic and the presence of refineries in the action area creates the risk of a catastrophic oil spill that could affect SR killer whales and their prey. Washington state is home to 5 oil refineries, all located within the Puget Sound region and representing 3.5% of the United States' refining capacity; in 2011, the 5 refineries processed 536,000 barrels of crude oil per day (WRC 2012).

Vessel Strikes. Ship strikes of SR killer whales do occur and can result in serious injury and mortality. Scheffer and Slipp (1948) noted several collisions between killer whales and boats, but gave no information on effects to the whales from these encounters. One killer whale mortality from a ship strike was reported for Washington and British Columbia from 1960-1990 (Baird 2001). More recently, in British Columbia, there were 10 known killer whale ship strikes from 1995- 2007, 2 of them fatal, and with 1 individual struck and died the following year (Williams and O'Hara 2010). These 10 ship strikes were not on whales from the listed SR DPS.). In 2005, a

Southern Resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. In 2006, an adult male Southern Resident killer whale, L98, was killed in a collision with a tug boat; given the gender imbalances in the Southern Resident killer whale population, we assume that the death of this adult male would have reduced the demographic health of this population. In fall 2016 another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that the whale died of blunt force trauma to the head "the animal had injuries consistent with blunt trauma to the dorsal side, and a hematoma indicating that it was alive at the time of injury and would have survived the initial trauma for a period of time prior to death" (Fisheries and Oceans Canada 2019). The injuries are consistent with those incurred during a vessel strike.

Vessel Presence and Whale Watching. Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS 2008). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity (see 76 FR 20870 for a review).

Noise. Transportation, including commercial and recreational vessel traffic, airplanes and helicopters, all contribute to sound in the ocean (NRC 2003). The military uses sound to test the construction of new vessels, as well as for naval training and testing activities involving sonar and explosives. In some areas where oil and gas production takes place, noise originates from the drilling and production platforms, tankers, vessel and aircraft support, seismic surveys, and the explosive removal of platforms (NRC 2003).

Researchers have described behavioral responses from marine mammals due to these noises, which included cessation of feeding, resting, or social interactions. Many contend that anthropogenic sources of noise have increased ambient noise levels in the ocean over the last 50 years (NRC 2003; NRC 2005). Much of this increase is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003).

Anthropogenic sound can drown out the clicks, calls, and whistles killer whales use to communicate with one another during foraging and the echolocation signals used to navigate (Bain et al. 1993; Erbe 2002; Gordon and Moscrop 1996; Holt et al. 2009; NMFS 2008; Williams et al. 2002a; Williams et al. 2002b). Killer whales have a wide frequency range of hearing (from 1-100 kHz) (Szymanski et al. 1999), and although large vessels emit predominantly low frequency sound, studies report broadband noise from large cargo ships with significant levels above 2 kHz, and thus may interfere with important biological functions of killer whales (Holt 2008; NMFS 2008).

In addition to the disturbance associated with the presence of vessels, vessel traffic affects the acoustic ecology of Southern Resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of Southern Resident killer whales that were made in the presence or absence of boat noise in Puget Sound during 3 time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about

15% during the last of the 3 time periods (2001 to 2003). At the same time, Holt et al. (2009) reported that Southern Resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales need to increase the duration of their vocalization to avoid masking by the boat noise. With the disruption of feeding behavior that has been observed, it is estimated that the presence of vessels could result in an 18% decrease in Southern Resident killer whale energy intake, a consequence that could have a significant negative effect on an already prey-limited species (Lusseau et al. 2009; Williams et al. 2006).

Scientific Research. SR killer whales have been the subject of scientific research activities in the action area, as authorized by NMFS permits. After the listing of SR killer whales as endangered under the ESA, NMFS issued 3 new scientific research permits, amended 3 existing permits and renewed 1 additional permit to authorize a variety of research activities targeting these whales (NMFS 2006). In subsequent years, additional research permits have authorized take of SR killer whales. No mortalities or serious injuries are authorized for SR killer whales under these permits.

Conservation and Management Efforts. In 2011, NMFS established regulations prohibiting vessels from approaching killer whales within 200 yds (189.2 meters) and from parking in the path of whales when in inland waters of Washington State (76 FR 20870). Certain exceptions to these regulations apply, such as to government vessels engaged in official business, cargo vessels in shipping lanes, fishing vessels actively fishing, and vessel maneuvers necessary for safety reasons.

9.4.6 Sea Star Wasting Syndrome

The following is as described in the 2022 sunflower sea star status review report (Lowry 2022). Beginning in 2013, sea star wasting syndrome (SSWS) caused ~72-100% declines in locally monitored populations of *P. helianthoides* across its range. The global *P. helianthoides* population declined by an estimated 90.6% due to SSWS (Gravem et al. 2021). Some local populations were functionally extirpated within a matter of weeks, such as those in the northern Channel Islands. Recent laboratory studies suggest that P. helianthoides die as quickly as 2-4 days after exposure to SSWS. Throughout the range some populations have undergone significant declines and there are concerns that remaining *P. helianthoides* are too widely dispersed for successful reproduction. There is considerable variation in the degree of impact, however, associated with depth, latitude, and recent temperature regime, in some cases. Small increases in temperature have been shown to increase susceptibility of *Pisaster ochraceus* to SSWS (Bates et al. 2009), and decreased temperature has been demonstrated to slow progression of the disease, though the end results is still death (Kohl et al. 2016).

The causative agent of SSWS is currently unknown and various hypotheses regarding transmission dynamics and the lethality of SSWS under diverse physiochemical circumstances exist. Initially, SSWS was thought to be caused by a densovirus or suite of densoviruses (Hewson et al. 2018). Subsequent studies, however, have determined that the disease is more complex. A number of factors ranging from environmental stressors to the microbiome in the sea stars may play a role (Aquino et al. 2021; Konar et al. 2019). Ocean warming has also been linked to outbreaks, hastening disease progression and severity (Harvell et al. 2019; Aalto et al. 2020). Regardless of the pathogen's unknown etiology to date, stress and rapid degeneration ultimately result with symptomatic sea stars suffering from abnormally twisted arms, white lesions, loss of body tissue, arm loss, melting, and death.

The SSWS has been, and continues to be, the primary stressor threatening the continued existence of *P. helianthoides*. The disease has caused mass mortality and local extirpation of some populations, especially in the southern portion of its range. The major concern about the potential for SSWS outbreaks to recur on the West Coast, resulting in rapid loss of the remaining fraction of the population that survived the 2013-17 pandemic. If recent SSWS-associated population declines continue extinction is all but certain throughout the range. If population growth rates are able to return to pre-pandemic levels in coming years, however, likelihood of population persistence is moderate in the Alaska Region and the British Columbia and Salish Sea Region, but lower in the West Coast Region.

9.5 Pacific Northwest Subregion

9.5.1 Land Use

The Pacific Northwest subregion includes all of Washington and parts of California, Idaho, Montana, Nevada, Oregon, Utah, and Wyoming. The subregion totals roughly 700,000 km^2 of which about 600,000 km^2 is classified as undeveloped, 30,000 km^2 is classified as developed and about 70,000 km^2 is classified as agriculture (Figure 80.)

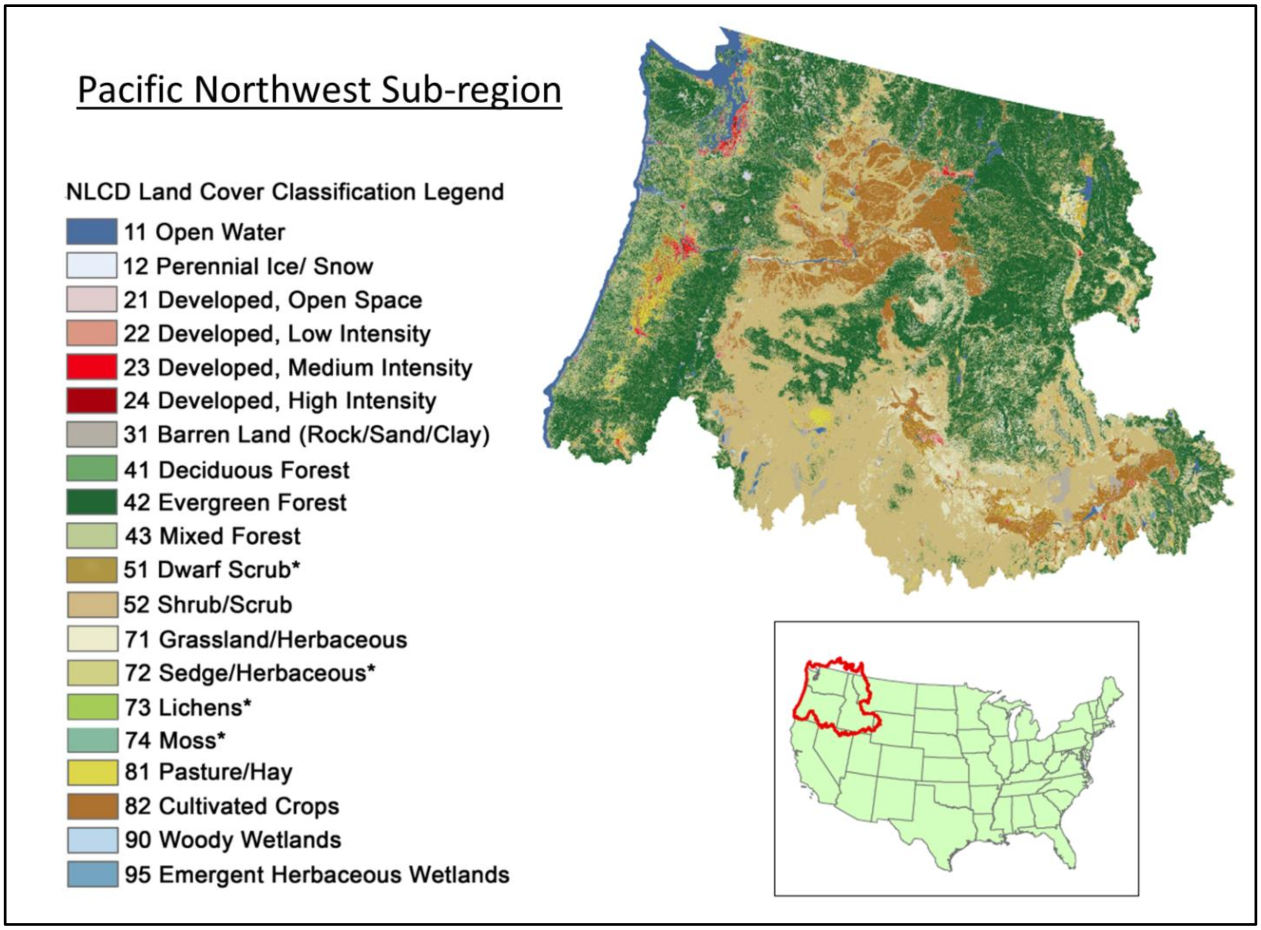


Figure 80. Land use in the Pacific Northwest sub-region. Data from the NLCD (www.mrlc.gov).

Twenty-four of the 59 species addressed in the conference and biological opinion occur in this subregion. They are: bococcio rockfish, yelloweye rockfish, green sturgeon Southern DPS, chinook salmon (ESUs: Snake River spring/summer-run, Snake River fall-run, Puget Sound, Upper Columbia River spring-run, Lower Columbia River, and Upper Willamette River), chum salmon (ESUs: Columbia River, and Hood Canal summer-run), coho salmon (ESUs: Oregon coast, Southern Oregon/Northern California coast, Lower Columbia River), sockeye salmon (ESUs: Ozette Lake, and Snake River), steelhead (DPSs: Upper Columbia River, Upper Willamette River, Middle Columbia River, Lower Columbia River, Snake River basin, Puget Sound), southern DPS eulachon, and southern resident killer whale. Table 113, Table 114, Table 115, and Table 116 show the types and areas of land use within each of the species' ranges.

Table 113. Area of land use categories within Pacific Northwest subregion selected Chinook salmon ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover	Chinook salmon					
NLCD Sub category	Snake River spring/ summer	Snake River fall	Puget Sound	Upper Columbia River spring	Lower Columbia River	Upper Willamette River
Water	**1,661**	**1,028**	**724**	**1,396**	**951**	**321**
Open Water	1,620	1,028	493	1,372	911	312
Perennial Ice/Snow	42	0	232	24	40	9
Developed Land	**3,660**	**2,098**	**3,621**	**2,706**	**2,040**	**2,181**
Open Space	1,688	796	1,202	959	864	635
Low Intensity	1,054	624	1,270	895	589	707
Medium Intensity	699	498	832	646	411	590
High Intensity	218	181	317	206	175	250
Undeveloped Land	**177,306**	**24,857**	**24,215**	**36,111**	**18,882**	**17,869**
Barren Land	416	19	1,037	358	224	131
Deciduous Forest	448	250	662	272	516	231
Evergreen Forest	51,220	6,128	16,638	14,205	14,008	14,039
Mixed Forest	545	422	2,153	441	1,002	731
Shrub/Scrub	75,687	6,964	2,089	11,137	1,571	1,389
Grassland/Herbaceous	47,144	10,574	798	8,843	735	662
Woody Wetlands	778	177	584	391	443	321
Emergent Wetlands	1,068	324	253	465	384	365
Agriculture	**14,755**	**8,146**	**1,366**	**5,864**	**1,167**	**4,482**
Pasture/Hay	2,428	850	973	619	1,015	2,844
Cultivated Crops	12,327	7,296	393	5,244	152	1,638
TOTAL (inc. open water)	**197,382**	**36,129**	**29,926**	**46,077**	**23,039**	**24,854**
TOTAL (w/o open water)	**195,720**	**35,101**	**29,202**	**44,681**	**22,088**	**24,533**

1. Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

Table 114. Area of land use categories within Pacific Northwest subregion selected fish and killer whale ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover NLCD Sub category	Bocaccio rockfish	Yelloweye rockfish	Eulachon	Southern Resident Killer Whale[2]
Water	**6,170**	**6,170**	**720**	**16,206**
Open Water	6,170	6,170	720	16,206
Perennial Ice/Snow	0	0	0	0
Developed Land	**60**	**60**	**737**	**9**
Open Space	20	20	281	1
Low Intensity	21	21	201	1
Medium Intensity	12	12	170	3
High Intensity	7	7	85	3
Undeveloped Land	**237**	**237**	**3,883**	**121**
Barren Land	45	45	23	39
Deciduous Forest	17	17	269	1
Evergreen Forest	46	46	2,039	7
Mixed Forest	29	29	490	1
Shrub/Scrub	5	5	281	1
Grassland/Herbaceous	7	7	195	3
Woody Wetlands	8	8	245	2
Emergent Wetlands	81	81	342	69
Agriculture	**7**	**7**	**488**	**1**
Pasture/Hay	6	6	435	0
Cultivated Crops	0	0	53	0
TOTAL (inc. open water)	**6,474**	**6,474**	**5,828**	**16,337**
TOTAL (w/o open water)	**304**	**304**	**5,108**	**131**

Table 115. Area of land use categories within Pacific Northwest subregion selected chum, coho and sockeye species' ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover	Chum		Coho			Sockeye	
NLCD Sub category	Columbia River	Hood Canal summer-run	Oregon Coast	Southern Oregon/ Northern California	Lower Columbia River	Ozette Lake	Snake River
Water	**977**	**34**	**278**	**341**	**951**	**30**	**832**
Open Water	952	31	278	331	911	30	820
Perennial Ice/Snow	25	3	0	10	40	0	12
Developed Land	**2,201**	**304**	**1,330**	**2,586**	**2,040**	**7**	**1,548**
Open Space	931	153	989	1,975	864	6	506
Low Intensity	654	102	200	349	589	1	475
Medium Intensity	436	39	104	201	411	0	408
High Intensity	181	10	37	61	175	0	159
Undeveloped Land	**20,756**	**3,373**	**26,096**	**48,496**	**18,882**	**194**	**18,735**
Barren Land	199	152	113	240	224	0	39
Deciduous Forest	515	72	329	499	516	3	241
Evergreen Forest	13,871	2,506	17,350	30,267	14,008	143	6,121
Mixed Forest	991	170	3,820	2,043	1,002	2	415
Shrub/Scrub	2,657	288	2,339	9,842	1,571	26	6,211
Grassland/Herbaceous	1,658	99	1,461	5,242	735	7	5,191
Woody Wetlands	452	60	237	143	443	12	188
Emergent Wetlands	413	27	446	220	384	2	330
Agriculture	**1,812**	**88**	**992**	**1,225**	**1,167**	**1**	**3,700**
Pasture/Hay	1,026	87	986	937	1,015	1	539
Cultivated Crops	786	1	6	288	152	0	3,161
TOTAL (inc. open water)	**25,746**	**3,799**	**28,696**	**52,648**	**23,039**	**232**	**24,815**
TOTAL (w/o open water)	**24,769**	**3,765**	**28,418**	**52,307**	**22,088**	**202**	**23,984**

[1.] *Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed*

Table 116. Area of land use categories within Pacific Northwest subregion selected steelhead species' ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover	Steelhead salmon DPS					
NLCD Sub category	Upper Columbia River	Upper Willamette River	Middle Columbia River	Lower Columbia River	Snake River Basin	Puget Sound
Water	**1,530**	**200**	**856**	**598**	**1,665**	**761**
Open Water	1,505	197	826	563	1,623	529
Perennial Ice/Snow	24	3	30	35	42	232
Developed Land	**3,019**	**2,367**	**3,551**	**1,908**	**3,720**	**4,419**
Open Space	1,035	668	1,695	771	1,708	1,399
Low Intensity	1,005	785	1,015	559	1,072	1,523
Medium Intensity	750	646	636	405	716	1,078
High Intensity	228	269	205	173	224	419
Undeveloped Land	**38,729**	**10,682**	**71,002**	**16,813**	**177,494**	**25,852**
Barren Land	359	31	245	197	416	1,056
Deciduous Forest	273	267	287	469	448	765
Evergreen Forest	14,492	7,230	22,805	12,647	51,220	17,657
Mixed Forest	442	1,047	455	840	545	2,385
Shrub/Scrub	12,738	989	24,880	1,291	75,719	2,198
Grassland/Herbaceous	9,447	462	21,170	612	47,300	864
Woody Wetlands	415	307	477	410	778	639
Emergent Wetlands	563	350	683	347	1,068	290
Agriculture	**9,391**	**4,559**	**14,069**	**1,111**	**14,984**	**1,480**
Pasture/Hay	681	2,718	1,333	968	2,438	1,081
Cultivated Crops	8,711	1,841	12,736	143	12,546	398
TOTAL (inc. open water)	**52,668**	**17,808**	**89,478**	**20,431**	**197,862**	**32,513**
TOTAL (w/o open water)	**51,139**	**17,609**	**88,622**	**19,833**	**196,197**	**31,751**

[1] Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

9.5.2 Water Quality

As described in *General Factors* Section 9.2.6, impaired baseline water temperature, DO, nutrients, BOD, COD, toxics and other 303(d) impairments are significant detriments to the health, diversity, and distribution of aquatic life affecting the survival of native listed species within the action area. Figure 81 and Table 117 depict waterbodies that exceed 303(d) standards and give us insights into which ESUs and DPSs are affected within the Pacific Northwest subregion.

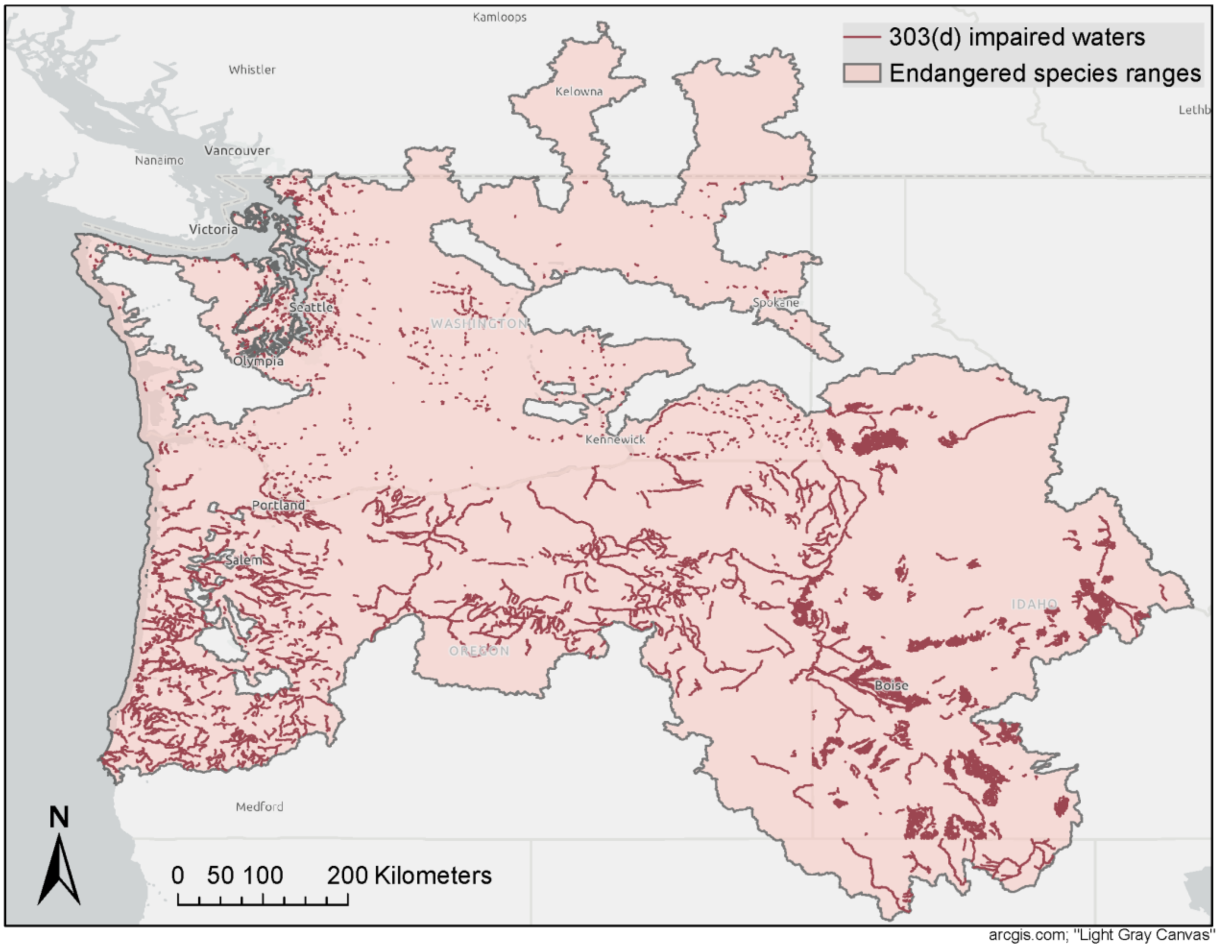


Figure 81. 303(d) impairments within the combined species ranges for the Pacific Northwest subregion. Data downloaded from USEPA ATTAINS website in September 2022.

Of the total number of kilometers of impaired waterbodies in the Pacific Northwest subregion, 2.9% were due to background pesticides and 59.5% were due to elevated temperature (Table 117). The background pesticides reported in this subregion (found via the "Detailed Cause" associated with a "Parent Cause" as defined by USEPA's ATTAINS database) include: 4, 4'-DDD/DDE/DDT, aldrin, alpha-BHC, chlordane, chlorpyrifos, DDE/DDT, diazinon, dieldrin, endosulfan, guthion (azinphos-methyl), heptachlor, heptachlor epoxide, malathion, and methyl parathion.

Table 117. Number of kilometers of river, stream and estuaries included in ATTAINS 303(d) lists that are located within species ranges in the Pacific Northwest region, along

with the percent of impaired waters[1] that each group[2] represents. Data were taken from USEPA ATTAINS website in September 2022.

Group [2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
All reasons	**NA**	**24,887.0**	**NA**
Organics	**TOTAL**	**2,351.6**	**9.4**
	Toxic Organics	60.4	0.2
	PCBs	145.3	0.6
	Dioxins	28.5	0.1
	Oil and Grease	560.5	2.3
	Organic Enrichment/Oxygen Depletion	1,556.9	6.3
Inorganics/Metals	**TOTAL**	**2,713.5**	**10.9**
	Toxic Inorganics	8.7	> 0
	Mercury	925.1	3.7
	Metals (other than mercury)	1,779.6	7.2
Nutrients/pathogens/algal growth	**TOTAL**	**6,255.2**	**25.1**
	Ammonia	584.0	2.3
	Nutrients	1,229.8	4.9
	Pathogens	4,168.8	16.8
	Algal Growth	272.6	1.1
	Biotoxins	0.0	0.0
Sediment/Turbidity	**TOTAL**	**5,254.8**	**21.1**
	Turbidity	1,105.4	4.4
	Sediment	4,149.4	16.7

Group [2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
Pesticides	**TOTAL**	**722.8**	**2.9**
	Pesticides	722.8	2.9
Water Quality	**TOTAL**	**831.7**	**3.3**
	Ph/Acidity/Caustic	725.4	2.9
	Salinity/Total Dissolved Solids/Chlorides/ Sulfates	103.7	0.4
	Chlorine	2.7	> 0
Temperature	**TOTAL**	**14,819.9**	**59.5**
	Temperature	14,819.9	59.5
Other	**TOTAL**	**4,302.5**	**17.3**
	Other Cause	85.3	0.3
	Cause Unknown	846.4	3.4
	Radiation	0.0	0.0
	Taste, Color, Odor	0.0	0.0
	Noxious Aquatic Plants	0.0	0.0
	Nuisance Native Species	0.0	0.0
	Nuisance Exotic Species	0.0	0.0
	Trash	0.0	0.0
	Cause Unknown - Impaired Biota	3,365.9	13.5
	Cause Unknown - Fish Kills	0.0	0.0
	Total Toxics	4.9	> 0
	Fish Consumption Advisory	0.0	0.0

Group [2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]

1. The percentages may not add up to 100%, because certain water bodies have more than 1 303(d) impairment and therefore are accounted for more than once on occasion (i.e., in different "Parent Causes").

2. "Parent Causes", as described by 303(d), were grouped into 9 categories and the group totals are shown in bold.

9.5.3 Monitoring Data

9.5.3.1 Washington Data

The Washington State Department of Agriculture – Natural Resources Assessment Section (NRAS) program focuses on monitoring and evaluating the impacts of agriculture chemicals on Washington State's natural resources, including ESA-listed endangered species. Several programs at NRAS have high relevance to this consultation including: 1) the agricultural land use mapping geodatabase; 2) the surface and groundwater monitoring program; and 3) the development of crop-based typical use profiles which describe factors including rate, application timing, percent crop treated, and application method.

The WSDA agricultural land use geodatabase combines targeted fieldwork, expertise in agricultural practice/crop identification, and existing land use data to provide high quality crop mapping data. The crop data is classified by several categories: 1) general crop group (berry, cereal grain, orchard, vegetable, etc.); 2) crop types (blueberry, wheat, apple, potato, etc.), and 3) irrigation method (center pivot, drip, rill, none, etc.). Additional information on WSDA's agricultural land use mapping program, including an interactive land use web map, are available at https://agr.wa.gov/departments/land-and-water/natural-resources/agricultural-land-use.

The WSDA has monitored surface water throughout the state since 2003. The program adds and removes sampling sites and sub basins based on pesticide detection history, changing pesticide use practices, site conditions, land use patterns, and the presence of listed threatened or endangered species (Tuttle et al. 2017). Currently, the program is monitoring waters at 16 locations including 3 locations in urban settings. The complete set of surface water monitoring reports, as well as an interactive surface water monitoring web map, are available at https://agr.wa.gov/departments/land-and-water/natural-resources.

Washington State also has a voluntary program that assists growers in addressing water rights issues within a watershed. Several watersheds have elected to participate, forming Comprehensive Irrigation District Management Plans (CIDMPs). The CIDMP is a collaborative process between government and landowners and growers; the parties determine how they will ensure growers get the necessary volume of water while also guarding water quality. This structure allows for greater flexibility in implementing mitigation measures to comply with both the CWA and the ESA.

9.5.3.2 Oregon Data

In Oregon, water quality policies related to pesticides is handled by several state agencies. An interagency team was thus formed: the Water Quality Pesticide Management Team (WQPMT). WQPMT facilitates and coordinates water quality activities such as monitoring, analysis and interpretation of data, effective response measures, and management solutions. The initial goal of the WQPMT was to develop and implement a statewide pesticide management plan (PMP), which was approved by EPA in 2011. The overall objectives of the program are: 1) to identify and characterize pesticides that may pose a risk to water resources; 2) actively manage them by facilitating efforts to reduce or prevent contamination below the reference point (an established benchmark or standard); and 3) demonstrate how management efforts are keeping concentrations at acceptable levels.

The Oregon Pesticide Stewardship Partnership (PSP) Program is a cooperative, voluntary process that is designed to identify potential concerns regarding surface and groundwater affected by pesticide use within Oregon. The PSP Program began with a small number of pilot projects in north Mid-Columbia watersheds in the late 1990s and early 2000s as an alternative to regulatory approaches for achieving reductions in current use pesticides from application activities. Since 2013, the Oregon Legislature has supported the implementation and expansion of the PSP Program, that now addresses pesticides applied in watersheds that encompass applications from urban, forested, agricultural and mixed land uses (taken from the Pesticide Stewardship Partnership Program 2015 – 2017 Biennial Report; Cook and Masterson, 2018).

Between 2015 and 2017 the PSP surface water monitoring program collected samples across 9 watersheds and 2 additional pilot studies. The program analyzes for 89 registered pesticides, 26 non-registered pesticides, and 18 pesticide metabolites. Ground water monitoring is conducted by the Oregon Department of Environmental Quality in the Walla Walla and Middle Rogue watersheds. The PSP also maintains a Waste Pesticide Collection program which, between 2015 and 2017 resulted in the removal of 152,679 pounds of unused or unusable pesticides from sensitive watersheds (Cook and Masterdon 2018). NMFS sees high potential in programs like this in aiding the recovery of listed aquatic species. Additional information on the PSP, including biennial summaries can be found at https://www.oregon.gov/ODA/programs/Pesticides/Water/Pages/PesticideStewardship.aspx.

9.5.3.3 Idaho Data

The Idaho State Department of Agriculture (ISDA) has developed regional and local agricultural ground and surface water monitoring programs. The goal of these programs are to conduct monitoring to fill data and information gaps to effectively and efficiently monitor pesticides. ISDA conducts monitoring in partnership with the Idaho Department of Environmental Quality (DEQ), Idaho Department of Water Resources (IDWR), and many other state, local, and private agencies, organizations, businesses, and individuals. Every year, about 400 monitoring sites are sampled. Most sites are sampled once every 5 years. Water quality results include: bacteria, nutrients, common ions (e.g., calcium, magnesium), trace elements (e.g., iron, arsenic, lead), pesticides, volatile organic compounds, and radioactivity. Additional information on the statewide groundwater quality monitoring program, including reports, maps, and publications, can be found at https://idwr.idaho.gov/water-data/groundwater-quality/.

The Idaho State Department of Agriculture has published a Best Management Practices (BMP) guide for pesticide use. The BMPs include "core" voluntary measures that will prevent pesticides from leaching into soil and groundwater. These measures include applying pest-specific controls, being aware of the depth to ground water, and developing an Irrigation Water Management Plan.

9.5.3.4 National-Level Data

Both carbaryl and methomyl data were obtained by the EPA via a download from the Water Quality Portal (http://www.waterqualitydata.us/). EPA's carbaryl BE and methomyl BE reported the methods with which they extracted data from the Water Quality Portal: Surface water and groundwater carbaryl data were obtained for every HUC2 region in February 2021, and surface water and groundwater methomyl data were obtained for every HUC2 region in 2019. A significant portion of the data obtained from the water quality portal has been supplied through NAWQA, a national-scale ambient water quality monitoring program that contains monitoring data for pesticides in streams. The database includes an extensive amount of data for methomyl; however, the NAWQA monitoring program was not designed to specifically target methomyl use. Specifically, the sample timing and frequency were not designed to correspond with methomyl applications.

For carbaryl, sampling occurred both from 1973 to 2006 at over 6,367 sites with a maximum detected concentration of 335 µg/L (surface water sample from 1973 from a creek in Pennsylvania), and from 2007 to 2021 at over 4,452 sites with a maximum detected concentration of 14.1 µg/L. Some important regulatory changes occurred from 2005 to 2008 with implementation of the re-registration mitigations for carbaryl. Applications to wheat were cancelled, aerial applications of some formulations were no longer allowed, broadcast applications of liquid formulations were cancelled in residential settings, and dust applications in agricultural settings were cancelled (USEPA 2007; USEPA 2008). These registration changes have the potential to impact concentrations that may be observed in monitoring. The extent to which historical values represent current agronomic or labeled use instructions is uncertain.
For methomyl, sampling occurred from 1982 to 2019 at over 4,310 sites with a maximum detected concentration of 12 µg/L.

9.5.3.5 Compilation and Spatial Analysis

For each compound, data were compiled from the EPA's data obtained from the Water Quality Portal, and from updated data (from 2003-2020) sent by WSDA. Maximum pesticide concentrations from the monitoring data were overlaid on the ESA-listed species ranges for the Pacific Northwest subregion and were visualized with graduated symbols; the larger symbols represent higher maximum pesticide concentrations for either carbaryl or methomyl (Figure 82). Carbaryl was detected at roughly 175 sites, with maximum concentrations ranging from 0.001 to 33.5 µg/L (Figure 82). These sites were sampled over the course of many years, and of the samples collected over time at any given site, carbaryl was detected between 0.4 - 100% of the time, with the majority of locations showing detections over 30% of the time. Methomyl was detected at 48 sites with maximum concentrations ranging from 0.0022 to 1.33 µg/L (Figure 82). Of the samples collected at these sites over time, methomyl was detected between 0.28 – 100% of the time with the majority of locations showing methomyl detections over 30% of the time.

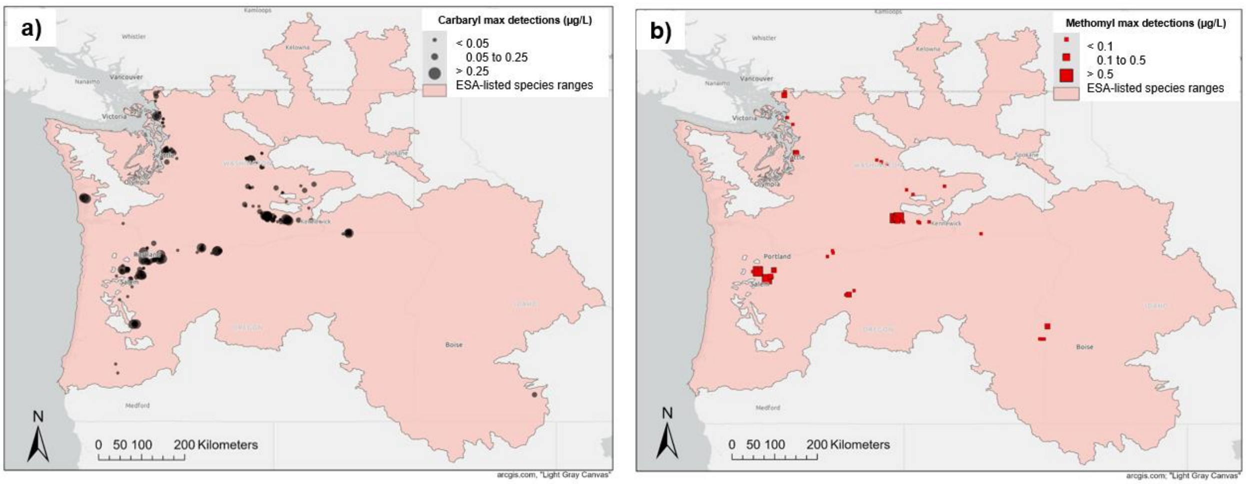


Figure 82. Monitoring data from EPA's carbaryl BE (panel a) and methomyl BE (panel b), taken from the Water Quality portal in 2021 and 2019 respectively, combined with regional data from WSDA ranging from 2003-2020. Detections are overlaid on ESA-listed species ranges in the Pacific Northwest subregion.

9.5.3.6 Additional Highlighted Programs

The Columbia Gorge Fruit Growers Association is a non-profit organization dedicated to the needs of growers in the mid-Columbia area. The association brings together over 440 growers and 20 shippers of fruit from Oregon and Washington. It has issued a BMP handbook for pesticide use, including information on alternative methods of pest control. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

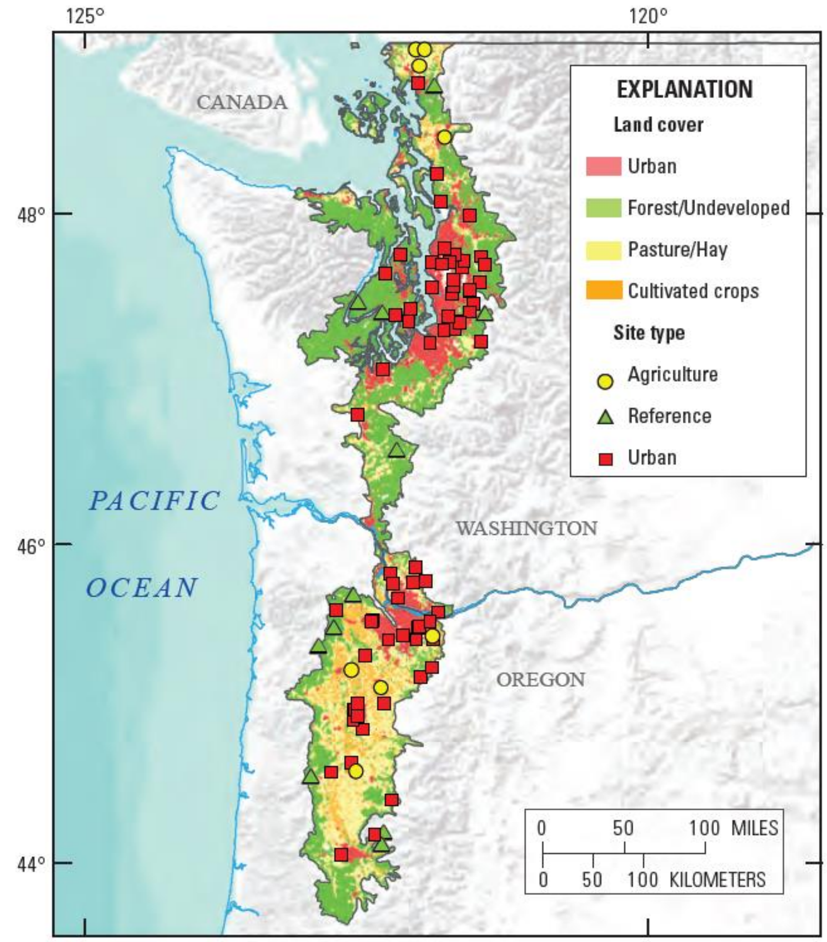
Stewardship Partners is a non-profit organization in Washington State that works to build partnerships between landowners, government, and non-profit organizations. In large part, its work focuses on helping landowners to restore fish and wildlife habitat while maintaining the economic viability of their farmland. Projects include restoring riparian areas, reestablishing floodplain connectivity, and removing blocks to fish passage. Another current project is to promote rain gardens as a method of reducing surface water runoff from developed areas. Rain gardens mimic natural hydrology, allowing water to collect and infiltrate the soil.

Stewardship Partners also collaborates with the Oregon-based Salmon-Safe certification program (www.salmonsafe.org). Salmon-Safe is an independent eco-label recognizing organizations who have adopted conservation practices that help restore native salmon habitat in Pacific Northwest, California, and British Columbia. These practices protect water quality, fish and wildlife habitat, and overall watershed health. While the program began with a focus on agriculture, it has since expanded to include industrial and urban sites as well. The certification process includes pesticide restrictions. Salmon-Safe has produced a list of "high risk" pesticides which, if used, would prevent a site from becoming certified. If a grower wants an exception, they must provide written documentation that demonstrates a clear need for use of the pesticide, that no safer alternatives exist, and that the method of application (such as timing, location, and amount used) represents a negligible risk to water quality and fish habitat. Over 300 farms, 250 vineyards, and 240 parks currently have the Salmon-Safe certification. Salmon-Safe has also worked with over 20 corporate / industrial sites and is beginning programs that focus on golf courses and nurseries.

9.5.3.7 USGS NAWQA Regional Stream Quality Assessment

Analysis of surface and ground water contaminants were conducted for a number of basins within the Pacific Northwest Region by the NAWQA program. The USGS has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River. Many of the water quality sampling sites have been in place for decades. Water volumes, crop rotation patterns, crop type, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. In addition to current use-chemicals, legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck et al. 2004).

In 2015, the USGA sampled 88 sites as part of the Pacific Northwest Stream Quality Assessment (Figure 83). Water samples were analyzed for about 230 dissolved pesticides and pesticide degradates. Results from the 2015 water quality assessment were considered and are available at https://webapps.usgs.gov/rsqa/#!/region/PNSQA.



 Base modified from Esri digital data, 1:70,000 (Esri, 2009) Land cover from National Land Cover Database, 2006, Web Mercator Projection, World Geodetic System of 1984 (WGS 84)

Figure 83. The Pacific Northwest Stream Quality Assessment study area. Taken from Van Metre et al. 2017: Figure 1: "Study area boundary is based on the Willamette Valley and Puget Lowlands level 3 ecological regions (ecoregions) of the United States."

9.5.4 Habitat Modification

This section briefly describes how anthropogenic land use has altered aquatic habitat conditions for salmonids in the Pacific Northwest Region. Basin wide, critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. Dams have flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (Good et al. 2005b). The mainstem habitats of the lower Columbia and Willamette rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of LWD in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control, and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. Consequently, estuary dynamics have changed substantially.

Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants within the Columbia River Estuary (Anderson et al. 2007). About 77% of swamps, 57% of marshes, and over 20% of tree cover have been lost to development and industry. Twenty-four threatened and endangered species occur in the estuary, some of which are recovering while others (i.e., Chinook salmon) are not.

Stream habitat degradation in Columbia Central Plateau is relatively high (Williamson et al. 1998). In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation (Williamson et al. 1998). Streams in this area have an average of 20% canopy cover and 70% bank erosion. These factors have severely affected the quality of habitat available to salmonids. The Palouse subunit of the Lower Snake River exceeds temperature levels for the protection of aquatic life (Williamson et al. 1998).

The Willamette Basin Valley has been dramatically changed by modern settlement. The complexity of the mainstem river and extent of riparian forest have both been reduced by 80% (PNERC 2002). About 75% of what was formerly prairie and 60% of what was wetland have been converted to agricultural purposes. These actions, combined with urban development, extensive (96 miles) bank stabilization, and in-river and nearshore gravel mining, have resulted in a loss of floodplain connectivity and off-channel habitat (PNERC 2002).

Much of the estuarine wetlands in Puget Sound have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996). Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands also experience increasing effects from industrial and urban causes. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound (Bortleson et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound (Bortleson et al. 1980). As of 2005, tidal wetlands in Puget Sound amount to about 17 - 19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from salt-water and rearing habitat. The land conversions and losses of Pacific Northwest wetlands constitute a major impact. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats (Brennan et al. 2004).

About 800 miles of Puget Sound’s shorelines are hardened or dredged (PSAT 2004; Ruckelshaus and McClure 2007). The area most intensely modified is the urban corridor (eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% of the shoreline has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks

(Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Urbanization has caused direct loss of riparian vegetation and soils and has significantly altered hydrologic and erosion rates. Watershed development and associated urbanization throughout the Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased LWD recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and filled estuarine rearing areas (Bishop and Morgan 1996 in (NMFS 2008f)). Large areas of the lower rivers have been channelized and diked for flood control and to protect agricultural, industrial, and residential development.

The principal factor for decline of Puget Sound steelhead is the destruction, modification, and curtailment of its habitat and range. Barriers to fish passage and adverse effects on water quality and quantity resulting from dams, the loss of wetland and riparian habitats, and agricultural and urban development activities have contributed and continue to contribute to the loss and degradation of steelhead habitats in Puget Sound (NMFS 2008f).

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. According to the State of the Sound Report (PSAT 2007), in 2004 more than 1,400 fresh and marine waters in the region were listed as “impaired.” Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans. More than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs; flame retardants), and roughly one-third (180,000 acres) of submerged lands within Puget Sound are considered moderately contaminated. In 2005 the Puget Sound Action Team (PSAT) identified the primary pollutants of concern in Puget Sound and their sources listed below in Table 118.

Table 118. Pollutants of Concern in Puget Sound (PSAT 2005).

Pollutant	Sources
Heavy Metals: Pb, Hg, Cu, and others	vehicles, batteries, paints, dyes, stormwater runoff, spills, pipes.
Organic Compounds: Polycyclic aromatic hydrocarbons (PAHs)	Burning of petroleum, coal, oil spills, leaking underground fuel tanks, creosote, asphalt.
Polychlorinated biphenyls (PCBs)	Solvents electrical coolants and lubricants, pesticides, herbicides, treated wood.
Dioxins, Furans	Byproducts of industrial processes.
Dichloro-diphenyl-trichloroethane (DDTs)	Chlorinated pesticides.

Phthalates	Plastic materials, soaps, and other personal care products. Many of these compounds are in wastewater from sewage treatment plants.
Polybrominated diphenyl ethers (PBDEs)	PBDEs are added to a wide range of textiles and plastics as a flame retardant. They easily leach from these materials and have been found throughout the environment and in human breast milk.

While much of the coastal region is forested, it has still been impacted by land use practices. As of 2000, less than 3% of the Oregon coastal forest was old growth conifers (Gregory 2000). The lack of mature conifers indicates high levels of habitat modification. As such, overall salmonid habitat quality is poor, though it varies by watershed. The amount of remaining high quality habitat ranges from 0% in the Sixes River to 74% in the Siltcoos River (ODFW 2005). Approximately 14% of freshwater winter habitat available to juvenile coho is of high quality. Much of the winter habitat is unsuitable due to high temperatures. For example, 77% of coho salmon habitat in the Umpqua basin exceeds temperature standards.

Reduction in stream complexity is the most significant limiting factor in the Oregon coastal region. An analysis of the Oregon coastal range determined the primary and secondary life cycle bottlenecks for the 21 populations of coastal coho salmon (Nicholas et al. 2005). Nicholas et al. (2005) determined that stream complexity is either the primary (13) or secondary (7) bottleneck for every population. Stream complexity has been reduced through past practices such as splash damming, removing riparian vegetation, removing LWD, diking tidelands, filling floodplains, and channelizing rivers.

Habitat loss through wetland fills is also a significant factor. Table 119 summarizes the change in area of tidal wetlands for several Oregon estuaries between 1870 and 1970 (Good 2000).

Table 119. Change in total area (acres2) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Necanicum	15	10
Nehalem	1,571	75
Tillamook	3,274	79
Netarts	16	7
Sand Lake	9	2

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Nestucca	2,160	91
Salmon	313	57
Siletz	401	59
Yaquina	1,493	71
Alsea	665	59
Siuslaw	1,256	63
Umpqua	1,218	50
Coos Bay	3,360	66
Coquille	4,600	94
Rogue	30	41
Chetco	5	56
Total	20,386	72

The only listed salmonid population in coastal Washington is the Ozette Lake sockeye. The range of this ESU is small, including only 1 lake (31 km^2) and 71 km of stream. Like the Oregon Coastal drainages, the Ozette Lake area has been heavily managed for logging. Logging resulted in road building and the removal of LWD, which affected the nearshore ecosystem (NMFS Salmon Recovery Division 2008). LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation (NMFS Salmon Recovery Division 2008). Aerial photograph analysis shows near-shore vegetation has increased significantly over the past 50 years (Ritchie 2005). Further, there is strong evidence that water levels in Ozette Lake have dropped between 1.5 and 3.3 ft from historic levels [Herrera 2005 *in* (NMFS Salmon Recovery Division 2008)]. The impact of this water level drop is unknown. Possible effects include increased desiccation of sockeye redds and loss of spawning habitat. Loss of LWD has also contributed to an increase in silt deposition, which impairs the quality and quantity of spawning habitat. Very little is known about the relative health of the Ozette Lake tributaries and their impact on the sockeye salmon population.

9.5.4.1 Urban and Industrial Development

The largest urban area in the Columbia River basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top 3 permitted sources of contaminants within the lower Columbia River basin according to discharge volumes and concentrations (Rosetta and Borys 1996). Rosetta and Borys (1996) review of 1993 data indicate that 52% of the point source waste water discharge volume is from sewage treatment plants, 39%

from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, the paper and allied products industry are the primary sources of the suspended sediment load (71%). Additionally, 26% of the point source waste water discharge volume comes from sewage treatment plants and 1% is from the chemical and allied products industry. Nonpoint source discharges (urban stormwater runoff) account for significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural nonpoint sources contributions were not calculated, Rosetta and Borys (1996) surmised that in some areas and for some contaminants, rural areas may contribute a large portion of the nonpoint source discharge. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary. Fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high and present a persistent and long lasting effect on estuary biology. Health advisories have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides.

In the 1930s, all of western Washington contained about 15.5 million acres of "harvestable" forestland. By 2004, the total acreage was nearly half that originally surveyed (PSAT 2007). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s. About a decade later, the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland. In this area, forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (Ruckelshaus and McClure 2007). Coniferous forests are also projected to decline at an alarming rate as urban uses increase.

According to the 2001 State of the Sound report (PSAT 2007), impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 ft elevation) covered by impervious surfaces. From 1991 to 2001, the amount of impervious surfaces increased to 10.4% region wide. Consequently, changes in rainfall delivery to streams alter stream flow regimes. Peak flows are increased and subsequent base flows are decreased and alter in-stream habitat. Stream channels are widened and deepened and riparian vegetation is typically removed which can cause increases in water temperature and will reduce the amounts of woody debris and organic matter to the stream system.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence et al. 1996). O'Neill et al. (2006) found that Chinook salmon returning to Puget Sound had significantly higher concentrations of PCBs and PBDEs compared to other Pacific coast salmon populations. Furthermore, Chinook salmon that resided in Puget Sound in the winter rather than migrate to the Pacific Ocean (residents) had the highest concentrations of persistent organic pollutants (POPs), followed by Puget Sound fish populations believed to be more ocean-reared. Fall-run Chinook salmon from Puget Sound have a more localized marine distribution in Puget Sound and the

Georgia Basin than other populations of Chinook salmon from the west coast of North America. This ESU is more contaminated with PCBs (2 to 6 times) and PBDEs (5 to 17 times). O'Neill et al. (2006) concluded that regional body burdens of contaminants in Pacific salmon, and Chinook salmon in particular, could contribute to the higher levels of contaminants in federally-listed endangered southern resident killer whales.

Endocrine disrupting compounds are chemicals that mimic natural hormones, inhibit the action of hormones and/or alter normal regulatory functions of the immune, nervous and endocrine systems and can be discharged with treated effluent (King County 2002). Endocrine disruption has been attributed to DDT and other organochlorine pesticides, dioxins, PAHs, alkylphenolic compounds, phthalate plasticizers, naturally occurring compounds, synthetic hormones and metals. Natural mammalian hormones such as 17β-estradiol are also classified as endocrine disruptors. Both natural and synthetic mammalian hormones are excreted through the urine and are known to be present in wastewater discharges. Jobling et al. (1995) reported that chemicals commonly detected in sewage effluent interacted with the fish estrogen receptor by reducing binding of 17β-estradiol to its receptor, stimulating transcriptional activity of the estrogen receptor or inhibiting transcription activity. Binding of these chemicals with the fish endocrine receptor indicates that the chemicals could be endocrine disruptors and forms the basis of concern about effluent and fish endocrine disruption.

Fish communities are impacted by urbanization (Wheeler et al. 2005). Urban stream fish communities have lower overall abundance, diversity, taxa richness and are dominated by pollution tolerant species. Lead content in fish tissue is higher in urban areas. Furthermore, the proximity of urban streams to humans increases the risk of non-native species introduction and establishment. Thirty-nine non-native species were collected in Puget Sound during the 1998 Puget Sound Expedition Rapid Assessment Survey (Brennan et al. 2004).

9.5.4.2 Mining

Mining has a long history in Washington. In 2004, the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (NMA 2007; Palmisano et al. 1993). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined in both Oregon and Washington. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Most of the mining in the Columbia River basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate. Ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho. More than half of the nation's silver output has come from a few select silver deposits.

Many of the streams and river reaches in the Columbia River basin are impaired from mining. Several abandoned and former mining sites are also designated as Superfund cleanup areas (Anderson et al. 2007; Stanford et al. 2005). According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment [Quigley, 1997 *in* (Hinck et al. 2004)]. Contaminants detected in the water include lead and other trace metals.

Oregon was ranked 35th nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (NMA 2007; Palmisano et al. 1993). Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano et al. 1993).

9.5.4.3 Hydromodification Projects

More than 400 dams exist in the Columbia River basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. The Federal Columbia River Power System (FCRPS) encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers. These dams and reservoirs operate as a coordinated system. The Corps operates 9 of 10 major federal projects on the Columbia and Snake rivers, and the Dworshak, Libby and Albeni Falls dams. The Bureau of Reclamation (BOR) operates the Grand Coulee and Hungry Horse dams. These federal projects are a major source of power in the region. These same projects provide flood control, navigation, recreation, municipal and industrial water supply, and irrigation benefits.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. Approximately 80% of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Damming has cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well.

Both upstream and downstream migrating fish are impeded by the dams. Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delay in passage, exposure to high concentrations of dissolved gases, warm water, and increased predation. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning.

Qualitatively, several hydromodification projects have improved the productivity of naturally produced SR Fall-run Chinook salmon. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers [USBR 1998 *in* (NMFS 2008d)]; providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River [see (Corps et al. 2007), Appendix 1 in (NMFS 2008d)]. Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the expression of productive “yearling” life history strategy that was previously unavailable to SR Fall-run Chinook salmon.

The mainstem FCRPS corridor has also improved safe passage through the hydro-system for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements (Corps et al. 2007).

For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The FCRPS action agencies recently implemented 18 estuary habitat projects that removed passage barriers. These activities provide fish access to good quality habitat.

The Corps et al. (2007) estimated that hydropower configuration and operational improvements implemented from 2000 to 2006 have resulted in an 11.3% increase in survival for yearling juvenile LCR Chinook salmon from populations that pass Bonneville Dam. Improvements during this period included the installation of a corner collector at Powerhouse II (PH2) and the partial installation of minimum gap runners at Powerhouse 1 (PH1) and of structures that improve fish guidance efficiency at PH2. Spill operations have been improved and PH2 is used as the first priority powerhouse for power production because bypass survival is higher than at PH1. Additionally, drawing water towards PH2 moves fish toward the corner collector. The bypass system screen was removed from PH1 because tests showed that turbine survival was higher than through the bypass system at that location.

More than 20 dams occur within the Puget Sound region’s rivers and overlap with the distribution of salmonids. A number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been modified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, as well as several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

In 1990, only one-third of the water withdrawn in the Pacific Northwest was returned to the streams and lakes (NRC 1996). Water that returns to a stream from an agricultural irrigation is often substantially degraded. Problems associated with return flows include increased water temperature, which can alter patterns of adult and smolt migration; increased toxicant concentrations associated with pesticides and fertilizers; increased salinity; increased pathogen populations; decreased dissolved oxygen concentration; and increased sedimentation (NRC 1996). Water-level fluctuations and flow alterations due to water storage and withdrawal can affect substrate availability and quality, temperature, and other habitat requirements of salmon. Indirect effects include reduction of food sources; loss of spawning, rearing, and adult habitat; increased susceptibility of juveniles to predation; delay in adult spawning migration; increased egg and alevin mortalities; stranding of fry; and delays in downstream migration of smolts (NRC 1996).

Several hydroelectric projects in Puget Sound and elsewhere have been relicensed by the West Coast Federal Energy Regulatory Commission (FERC) in “recent years” (a relative term because a FERC license is generally valid for 50 years). Relicensing that has occurred post ESA-listings has generally improved conditions for listed salmonids (see: West Coast Federal Energy Regulatory Commission (FERC) Licensed Hydroelectric Projects: Puget Sound)

For example, Morse Creek is the largest of the independent drainages to salt water between the Dungeness and Elwha rivers, entering the Strait of Juan de Fuca approximately 2 miles East of Port Angeles. The Hydro project, built in 1985 and brought online in 1987, provided about 0.3% of the city of Port Angeles electric usage. The Morse Creek Hydroelectric facility affected stream flow below the dam for ESA-listed fish including Puget Sound Chinook and steelhead. FERC adopted an amendment to increase streamflow in 2008. The licensee has increased instream flow which enhanced protection of ESA listed fish.

In the Elwha River 2 large hydroelectric dams were removed. After 2 decades of planning, dam removal began on September 17, 2011. Six months later the Elwha Dam was gone, followed by the Glines Canyon Dam in 2014. Today, the Elwha River once again flows freely from its headwaters in the Olympic Mountains to the Strait of Juan de Fuca opening over 70 miles of salmon habitat.

While the Elwha River project was the largest dam removal endeavor in U.S. history, it will be dwarfed in comparison to the removal of 4 large dams in the Kalamath River system. These dams are scheduled for removal beginning in 2023. When the project is complete it will open over 400 hundreds of miles of listed steelhead, Coho and Chinook salmon habitat.

Other dams removed on the Columbia River System include the Condit Dam on the White Salmon River (2011) which opened 33 miles of steelhead habitat and 14 miles of salmon habitat;

and the Marmot Dam (2007) on the Sandy River opening dozens of miles of free-flowing habitat for steelhead and salmon.

Elsewhere in Oregon, the Brownsville Dam on the Calapooia River, Chiloquin Dam on the Sprague River, Savage Rapids Dam on the Rogue River, and Sodom Dam on the Calapooia River have all been removed from 2006 – 2016. Each of these removals opened miles of habitat benefitted steelhead and salmon.

Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (e.g., Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano et al. (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, and filling the pond with logs. When the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While the dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. This practice stopped long ago, but the effects from the channel scouring and loss of channel complexity resulted in the long-term loss of salmon habitat (NRC 1996).

9.6 California Subregion

9.6.1 Land Use

The California subregion includes parts of California, Nevada, and Oregon. The subregion totals roughly 430,000 km^2 of which about 320,000 km^2 is classified as undeveloped, 50,000 km^2 is classified as developed and about 50,000 km^2 is classified as agriculture (Figure 84).

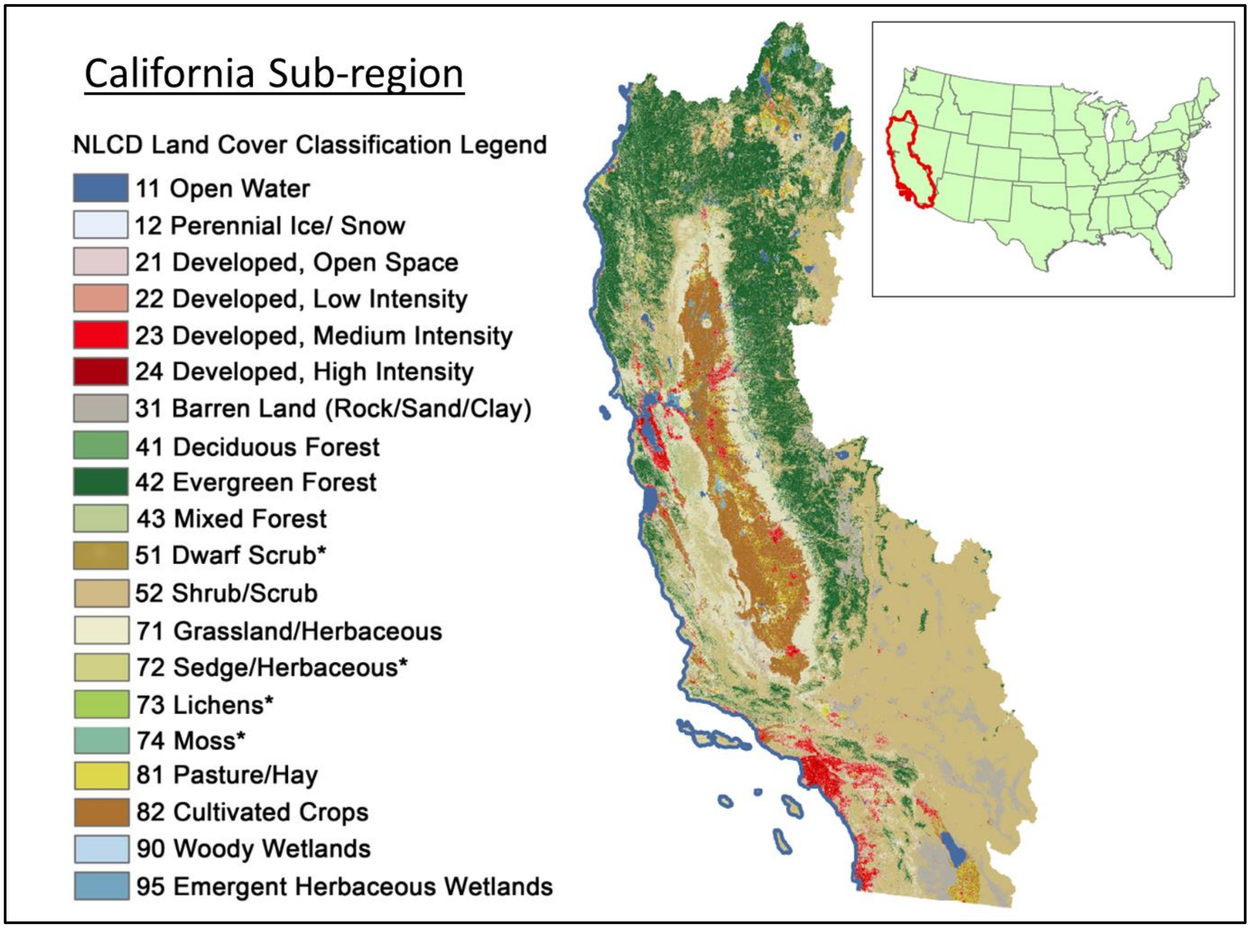


Figure 84. Land use in the California sub-region. Data from the NLCD (www.mrlc.gov).

Fifteen of the 61 species addressed in the opinion occur in this subregion. They are: Chinook salmon (ESUs: Central Valley spring-run, California coastal, Sacramento River winter-run), coho salmon (ESUs: southern Oregon/northern California coastal, central California coast), steelhead salmon (DPSs: northern California, south-central California coast, central California coast, California Central Valley, southern California), southern DPS eulachon, southern DPS green sturgeon, southern resident killer whale, black abalone, and white abalone. Table 120 and Table 121 show the types and areas of land use within each of the species' ranges.

Table 120. Area of land use categories within California subregion selected salmonid ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover	Chinook			Coho	Steelhead	
NLCD Sub category	Central Valley spring	California Coastal	Sacramento River winter	Central California Coast	Northern California	South-Central California Coast
Water	**1,541**	**168**	**418**	**1,319**	**139**	**67**
Open Water	1,496	168	409	1,319	139	67
Perennial Ice/Snow	44	0	9	0	0	0
Developed Land	**5,759**	**1,378**	**1,419**	**3,621**	**961**	**1,084**
Open Space	2,392	982	510	1,104	792	559
Low Intensity	1,314	195	363	797	98	247
Medium Intensity	1,530	158	414	1,242	59	220
High Intensity	523	43	132	478	12	59
Undeveloped Land	**74,900**	**19,790**	**17,831**	**14,491**	**16,637**	**15,809**
Barren Land	1,060	64	118	35	60	39
Deciduous Forest	617	350	37	112	301	0
Evergreen Forest	33,856	11,652	8,071	5,795	10,649	1,607
Mixed Forest	843	1,418	99	2,138	1,063	1,544
Shrub/Scrub	22,070	4,113	5,393	3,534	3,039	6,098
Grassland/Herbaceous	15,031	2,015	3,536	2,531	1,367	6,295
Woody Wetlands	368	77	158	86	60	104
Emergent Wetlands	1,056	102	419	260	97	123
Agriculture	**18,314**	**463**	**6,205**	**568**	**210**	**1,586**
Pasture/Hay	862	272	502	99	208	101
Cultivated Crops	17,452	190	5,704	468	3	1,485
TOTAL (inc. open water)	**100,514**	**21,799**	**25,873**	**19,999**	**17,948**	**18,547**
TOTAL (w/o open water)	**98,973**	**21,631**	**25,455**	**18,679**	**17,809**	**18,479**

[1.] Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

Table 121. Area of land use categories within California subregion selected steelhead and sturgeon ranges in km². The total area for each category is given in bold.[1] Land

cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover	Steelhead DPS			Sturgeon
NLCD Sub category	Central California Coast	California Central Valley	Southern California	Green Sturgeon Southern DPS
Water	**1,485**	**1,903**	**219**	**15,966**
Open Water	1,485	1,859	219	15,966
Perennial Ice/Snow	0	44	0	0
Developed Land	**3,608**	**6,676**	**9,155**	**11,791**
Open Space	948	2,865	1,970	2,998
Low Intensity	827	1,539	2,171	3,064
Medium Intensity	1,323	1,703	3,719	4,028
High Intensity	510	570	1,296	1,701
Undeveloped Land	**11,220**	**88,128**	**24,561**	**32,739**
Barren Land	29	1,109	64	394
Deciduous Forest	106	669	2	778
Evergreen Forest	2,656	35,127	1,978	13,535
Mixed Forest	2,027	1,474	936	3,607
Shrub/Scrub	3,168	25,878	16,418	6,154
Grassland/Herbaceous	2,704	22,108	4,735	5,560
Woody Wetlands	50	391	176	940
Emergent Wetlands	479	1,372	253	1,771
Agriculture	**628**	**24,185**	**979**	**9,716**
Pasture/Hay	97	950	235	1,298
Cultivated Crops	531	23,236	744	8,418
TOTAL (inc. open water)	**16,941**	**120,893**	**34,914**	**70,212**
TOTAL (w/o open water)	**15,456**	**118,990**	**34,695**	**54,246**

[1.] *Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed*

9.6.2 Water Quality

As described in General Factors, Section 9.2.6, impaired baseline water temperature, DO, nutrients, BOD, COD, toxics and other 303(d) impairments are significant detriments to the health, diversity, and distribution of aquatic life affecting the survival of native listed species within the action area. Figure 85 and Table 122 depict water bodies that exceed 303(d) standards and give us insights into which ESUs and DPSs are affected within the California subregion.

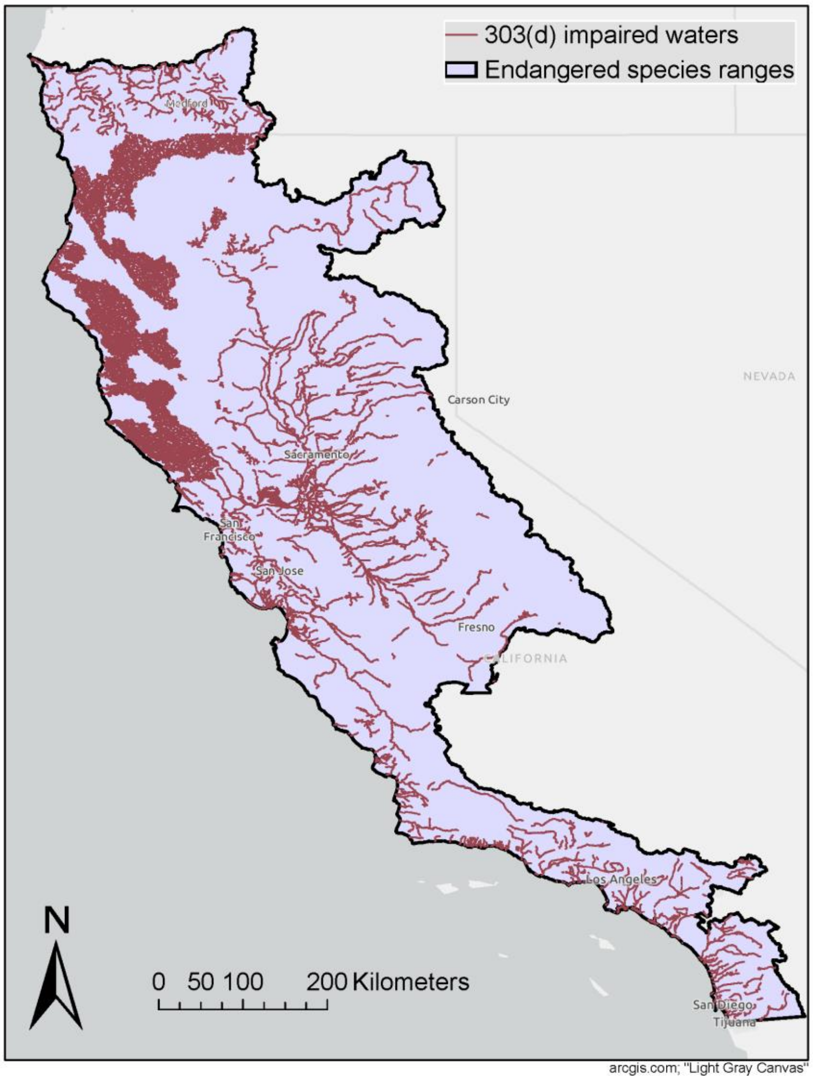


Figure 85. 303(d) impairments within the combined species ranges for the California subregion. Data downloaded from USEPA ATTAINS website in September 2022.

Of the total number of kilometers of impaired waterbodies in the California subregion, 41.7% were due to background pesticides and 251.1% were due to elevated temperature (Table 122). The background pesticides found in this subregion (found via the "Detailed Cause" associated with a "Parent Cause" as defined by USEPA's ATTAINS database) include: 2-methylnaphthalene, aldicarb, alpha-BHS, bifenthrin, carbofuran, chlordane (which also appears in tissue and sediment), chlorpyrifos, DDT, DDE, DDT (which also appears in tissue and sediment), diazinon, dichlorvos, dieldrin (which also appears in tissue and sediment), dimethoate, disulfotron, diuron, endrin, group A pesticides (generally), guthion (azinphos-methyl), heptachlor epoxide, hexachlorobenzene, lindane, malathion, organophosphorus pesticides (generally), oxyfluorfen, permethrin, pyrethroids, simazine, toxaphene, and trifluralin.

Table 122. Number of kilometers of river, stream and estuaries included in ATTAINS 303(d) lists that are located within species ranges in the California subregion, along

with the percent of impaired waters[1] that each group[2] represents. Data were taken from USEPA ATTAINS website in September 2022.

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
All reasons	**NA**	**35,073.2**	**NA**
Organics	**TOTAL**	**45,017.3**	**128.4**
	Toxic Organics	430.9	1.2
	PCBs	2,709.0	7.7
	Dioxins	1,460.6	4.2
	Oil and Grease	37.6	0.1
	Organic Enrichment/Oxygen Depletion	40,379.2	115.1
Inorganics/Metals	**TOTAL**	**19,822.6**	**56.5**
	Toxic Inorganics	2,826.1	8.1
	Mercury	7,640.6	21.8
	Metals (other than mercury)	9,355.9	26.7
Nutrients/pathogens/algal growth	**TOTAL**	**82,140.2**	**234.2**
	Ammonia	750.6	2.1
	Nutrients	36,055.4	102.8

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
	Pathogens	16,139.5	46.0
	Algal Growth	3,507.1	10.0
	Biotoxins	25,687.5	73.2
Sediment/Turbidity	**TOTAL**	**76,812.6**	**219.0**
	Turbidity	1,918.6	5.5
	Sediment	74,894.0	213.5
Pesticides	**TOTAL**	**14,635.2**	**41.7**
	Pesticides	14,635.2	41.7
Water Quality	**TOTAL**	**16,644.8**	**47.5**
	Ph/Acidity/Caustic	4,156.6	11.9
	Salinity/Total Dissolved Solids/Chlorides/ Sulfates	12,488.2	35.6
	Chlorine	0.0	0.0
Temperature	**TOTAL**	**88,056.5**	**251.1**
	Temperature	88,056.5	251.1
Habitat/Flow Alterations	**TOTAL**	**117.1**	**0.3**
	Habitat Alterations	43.3	0.1

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
	Flow Alterations	73.8	0.2
Other	**TOTAL**	**13,062.7**	**37.2**
	Other Cause	55.7	0.2
	Cause Unknown	0.0	0.0
	Radiation	0.0	0.0
	Taste, Color, Odor	141.6	0.4
	Noxious Aquatic Plants	0.0	0.0
	Nuisance Native Species	0.0	0.0
	Nuisance Exotic Species	1,915.4	5.5
	Trash	1,520.2	4.3
	Cause Unknown - Impaired Biota	250.7	0.7
	Cause Unknown - Fish Kills	0.6	> 0
	Total Toxics	9,171.2	26.1
	Fish Consumption Advisory	7.4	> 0

1. The percentages may not add up to 100%, because certain water bodies have more than 1 303(d) impairment and therefore are accounted for more than once on occasion (i.e., in different "Parent Causes").

2. "Parent Causes", as described by 303(d), were grouped into 9 categories and the group totals are shown in bold.

9.6.3 Monitoring Data

9.6.3.1 California Data

The California Department of Pesticide Regulation (CADPR) has developed and maintained a number of excellent programs with the overall mission to "protect human health and the environment by regulating pesticide sales and use, and by fostering reduced-risk pest management". As further described on the CADPR website - The Environmental Monitoring Branch monitors the environment to determine the fate of pesticides, protecting the public and the environment from pesticide contamination through analyzing hazards and developing pollution prevention strategies. The Branch provides environmental monitoring data required for emergency eradication projects, environmental contamination assessments, pesticide registration, pesticide use enforcement, and human exposure evaluations. It also takes the lead in implementing many of DPR's environmental protection programs (https://www.cdpr.ca.gov/). The CADPR surface water database (SURF) was developed in 1997 and currently contains data representing 58 counties, over 4,000 sample sites, and over 760,000 chemical analysis records from water samples. Access to SURF is available at: (https://www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm).

9.6.3.2 National-Level Data

National-level data were taken from EPA's carbaryl and methomyl BEs and downloads from NAWQA as described in Section 9.5.3.4.

9.6.3.3 Compilation and spatial analysis

For each compound, monitoring data obtained by the EPA from the Water Quality Portal was used to plot maximum pesticide concentrations which were overlaid on the ESA-listed species ranges for the California subregion in GIS. The concentrations were visualized with graduated symbols; the larger symbols represent higher maximum pesticide concentrations for either carbaryl or methomyl (Figure 86).

Carbaryl was detected at 117 sites, with maximum concentrations ranging from 0.003 to 13 µg/L (Figure 86). These sites were sampled over the course of many years, and of the samples collected over time at any given site, carbaryl was detected between 4.2 - 100% of the time, with the majority of locations showing detections over 30% of the time. Methomyl was detected at 53 sites with maximum concentrations ranging from 0.008 to 12 µg/L (Figure 86). Of the samples collected at these sites over time, methomyl was detected between 1.38 – 100% of the time with the majority of locations showing methomyl detections over 40% of the time.

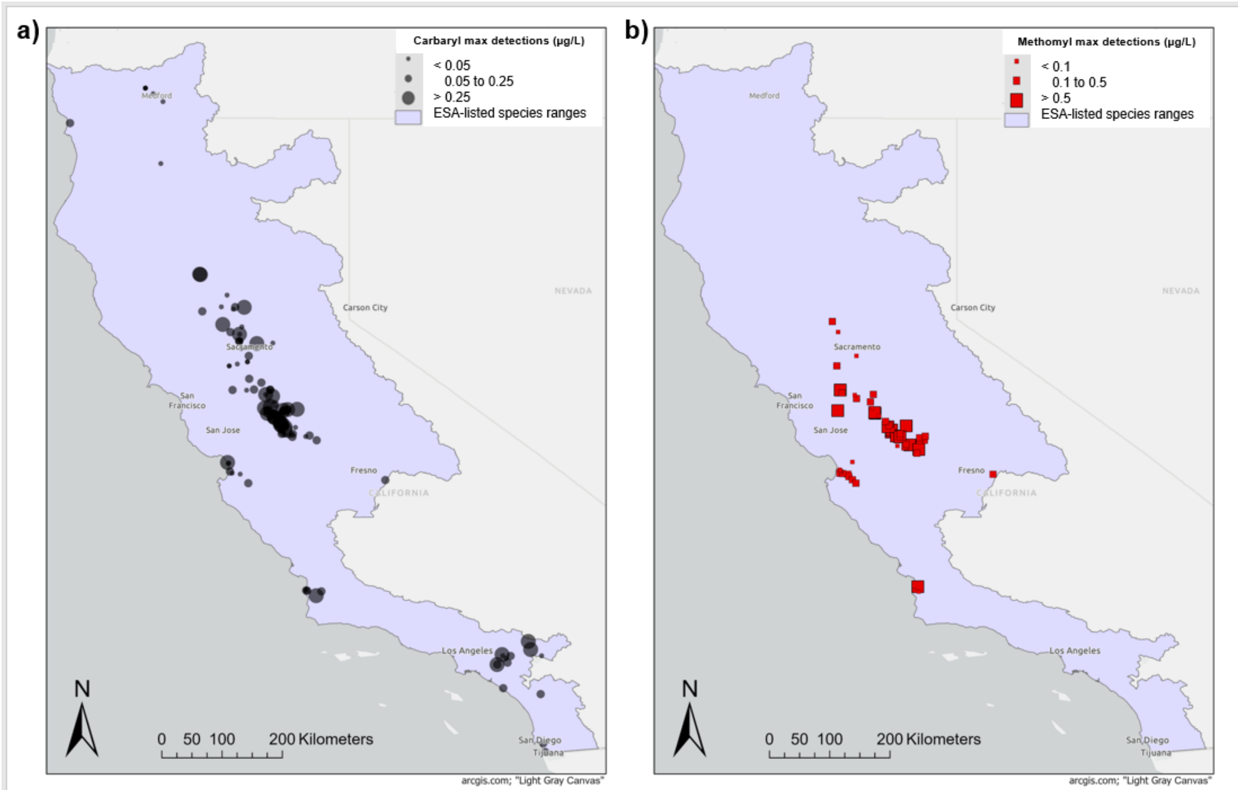


Figure 86. Monitoring data from EPA's carbaryl BE (panel a) and methomyl BE (panel b), taken from the Water Quality portal in 2021 and 2019 respectively, overlaid on ESA-listed species ranges in the California subregion.

9.6.3.4 Pesticide Reduction Programs

When using these 2 a.i.s, growers must adhere to the court-ordered injunctive relief, requiring buffers of 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all 4 states, pending completion of consultation.

California State Code does not include specific limitations on pesticide application aside from human health protections. It only includes statements advising that applicators are required to follow all federal, state, and local regulations.

Additionally, pesticide reduction programs already exist in California to minimize levels of carbaryl and methomyl into the aquatic environment. Monitoring of water resources is handled by the California State Water Resources Control Boards. Each Regional Board makes water quality decisions for its region including setting standards and determining waste discharge requirements. The Central Valley Regional Water Quality Control Board (CVRWQCB) addresses issues in the Sacramento and San Joaquin River Basins. These river basins are characterized by crop land, specifically orchards, which historically rely heavily on organophosphates for pest control.

In 2003, the CVRWQCB adopted the Irrigated Lands Waiver Program (ILWP). Participation was required for all growers with irrigated lands that discharge waste which may degrade water quality. However, the ILWP allowed growers to select 1 of 3 methods for regulatory coverage (Markle et al. 2005). These options included: 1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation (Markle et al. 2005). Many growers opted to join a Coalition as the other options were more costly. Coalition Groups were charged with completing 2 reports – a Watershed Evaluation Report and a Monitoring and Reporting Plan. The Watershed Evaluation Report included information on crop patterns and pesticide/nutrient use, as well as mitigation measures that would prevent orchard runoff from impairing water quality. Similar programs are in development in other agricultural areas of California.

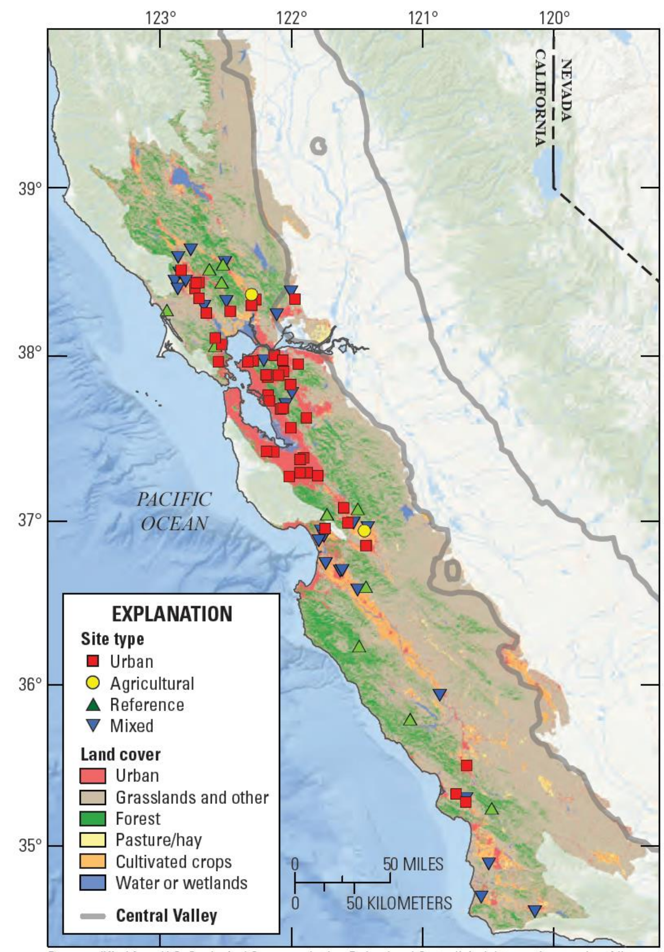
As a part of the Waiver program, the Central Valley Coalitions undertook monitoring of "agriculture dominated waterways." Some of the monitored waterways are small agricultural streams and sloughs that carry farm drainage to larger waterways. The coalition was also required to develop a management plan to address exceedance of State water quality standards. Currently, the Coalitions monitor toxicity to test organisms, stream parameters (*e.g.*, flow, temperature, etc.), nutrient levels, and pesticides used in the region. The Coalitions were charged with developing and implementing management and monitoring plans to address the TMDL and reduce diazinon runoff.

The Coalition for Urban/Rural Environmental Stewardship (CURES) is a non-profit organization that was founded in 1997 to support educational efforts for agricultural and urban communities focusing on the proper and judicious use of pest control products. CURES educates growers on methods to decrease pesticide surface water contamination in the Sacramento River Basin. The organization has developed best-practice literature for pesticide use in both urban and agricultural settings (www.curesworks.org). CURES also works with California's Watershed Coalitions to standardize their Watershed Evaluation Reports and to keep the Coalitions informed. The organization has worked with local organizations, such as the California Dried Plum Board and the Almond Board of California, to address concerns about diazinon, pyrethroids, and sulfur. The CURES site discusses alternatives to organophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. The CURES literature does not specifically address the a.i.s discussed in this opinion.

California also has PURS legislation whereby all agricultural uses of registered pesticides must be reported. In this case "agricultural" use includes applications to parks, golf courses, and most livestock uses. The CDPR publishes voluntary interim measures for mitigating the potential impacts of pesticide usage to ESA-listed species. These measures are available online as county bulletins.

9.6.3.5 USGS NAWQA Regional Stream Quality Assessment

In 2017, the USGA sampled 85 sites as part of the California Stream Quality Assessment (Figure 87). Water samples were analyzed for about 230 dissolved pesticides and pesticide degradates. Results from the 2017 water quality assessment were considered and are available at https://webapps.usgs.gov/rsqa/#!/region/CSQA.



Base modified from U.S. Geological Survey and other Federal and State digital data, various scales; Albers Equal-Area Conic projection, Standard parallels are 29° 30" and 45° 30", Central meridian -121°; North American Datum of 1983. Land cover modified from National Land Cover Database 2011. Base-map image is the intellectual property of Esri and is used herein under license. Copyright © 2014 Esri and its licensors. All rights reserved.

Figure 87. The California Stream Quality Assessment study area. Taken from Van Metre et al. 2017: Figure 1: "California Stream Quality Assessment study area and provisionally selected sampling sites; the boundary is based on the U.S. Environmental Protection Agency level III ecoregions of the United States"

9.6.4 Habitat Modification

Salmon habitat in California has experienced anthropogenic stressors for nearly 200 years (Munsch et al. 2022) The Central Valley area, including San Francisco Bay and the Sacramento and San Joaquin River Basins, has been drastically changed by development. Salmonid habitat has been reduced to 300 miles from historic estimates of 6,000 miles (CDFG 1993). In the San Joaquin Basin alone, the historic floodplain covered 1.5 million acres with 2 million acres of riparian vegetation (CDFG 1993). Roughly 5% of the Sacramento River Basin's riparian forests remain. Impacts of development include loss of LWD, increased bank erosion and bed scour, changes in sediment loadings, elevated stream temperature, and decreased base flow. Thus, lower quantity and quality of LWD and modified hydrology reduce and degrade salmonid rearing habitat.

The Klamath Basin in Northern California has been heavily modified as well. Water diversions have reduced spring flows to 10% of historical rates in the Shasta River, and dams block access to 22% of historical salmonid habitat. The Scott and Trinity Rivers have similar histories. Agricultural development has reduced riparian cover and diverted water for irrigation (NRC 2003). Riparian habitat has decreased due to extensive logging and grazing. Dams and water diversions are also common. These physical changes resulted in water temperatures too high to sustain salmonid populations. The Salmon River, however, is comparatively pristine; some reaches are designated as Wild and Scenic Rivers. The main cause of riparian loss in the Salmon River basin is likely wild fires – the effects of which have been exacerbated by salvage logging (NRC 2003).

9.6.4.1 Hydromodification Projects

Several of the rivers within California have been modified by dams, water diversions, drainage systems for agriculture and drinking water, and some of the most drastic channelization projects in the nation. There are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995). In general, the southern basins have a warmer and drier climate and the more northern, coastal-influenced basins are cooler and wetter. About 75% of the runoff occurs in basins in the northern half of California, while 80% of the water demand is in the southern half. Two water diversion projects meet these demands—the federal Central Valley Project (CVP) and the California State Water Project (CSWP). The CVP is one of the world's largest water storage and transport systems. The CVP has more than 20 reservoirs and delivers about 7 million acre-ft per year to southern California. The CSWP has 20 major reservoirs and holds nearly 6 million acre-ft of water. The CSWP delivers about 3 million acre-ft of water for human use. Together, both diversions irrigate about 4 million acres of farmland and deliver drinking water to roughly 22 million residents.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. The Rogue, Russian, and Santa Ana rivers each have more than 50 dams, and the Eel, Salinas, and the Klamath Rivers have between 14 and 24 dams each. The Santa Margarita is considered one of the last free flowing rivers in coastal southern California with 9 dams occurring in its watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state. Alteration of the temperature and sediment transport regimes had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent increases in suitable habitat for non-native species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin. A run of the spring-run Chinook salmon once produced about 300,000 to 500,000 fish (Carter and Resh 2005).

9.7 Northeast Region

The Northeast coastal region includes rocky coasts, drowned river valleys, estuaries, salt marshes, and city harbors. The Northeast is the most populous coastal region in the U.S. In 2010, the region was home to 54.2 million people, representing about a third of the nation's total coastal population (USEPA 2015). The population in this area has increased by 10 million residents (~ 23%) since 1970. The coast from Cape Cod to the Chesapeake Bay consists of larger watersheds that are drained by major riverine systems that empty into relatively shallow and poorly flushed estuaries. These estuaries are more susceptible to the pressures of a highly populated and industrialized coastal region.

9.7.1 National Coastal Condition Assessment

Figure 88 shows a summary of findings from the EPA's 2010 Report for the Northeast Region (USEPA 2015). A total of 238 sites were sampled to assess approximately 10,700 square miles of Northeast coastal waters. Biological quality was rated as good in 62% of the Northeast coast region based on the benthic index.

According to the more recent 2015 NCCA Report, the biological condition was good in 75% of the estuaries in the Northeast according to the M-AMBI marine benthic index. This is slightly better than the overall national estuarine biological condition and a statistically significant increase of 10% in area rated good from 2010 to 2015. About 76% of the Northeast estuarine area had good sediment quality based on measures of chemical contaminants found in sediments and laboratory tests of toxicity. This is on par with the rest of the estuarine area in the continental US in 2015 and represents a 20% increase in area rated good from 2010 to 2015. Ecological fish tissue contamination is degraded in the Northeast with 51% of waters in poor condition and 15% in fair. Only 18% of the area is rated good. This represents statistical increases in area rated poor and fair from 2010 to 2015; however when evaluating this change readers should be aware that there was a corresponding decrease in area that wasn't assessed (from 43% to 16%) from 2010 to 2015. The eutrophication Index, which examines the potential for estuarine area to undergo social eutrophication based upon measurements of nutrients, chlorophyll a, dissolved oxygen and water clarity, found that 48% of Northeast estuarine area was in good condition and 52% of area was in fair and poor conditions combined. For more information, see EPA's website (https://www.epa.gov/national-aquatic-resource-surveys/northeast-coast-estuaries-national-coastal-condition-assessment).

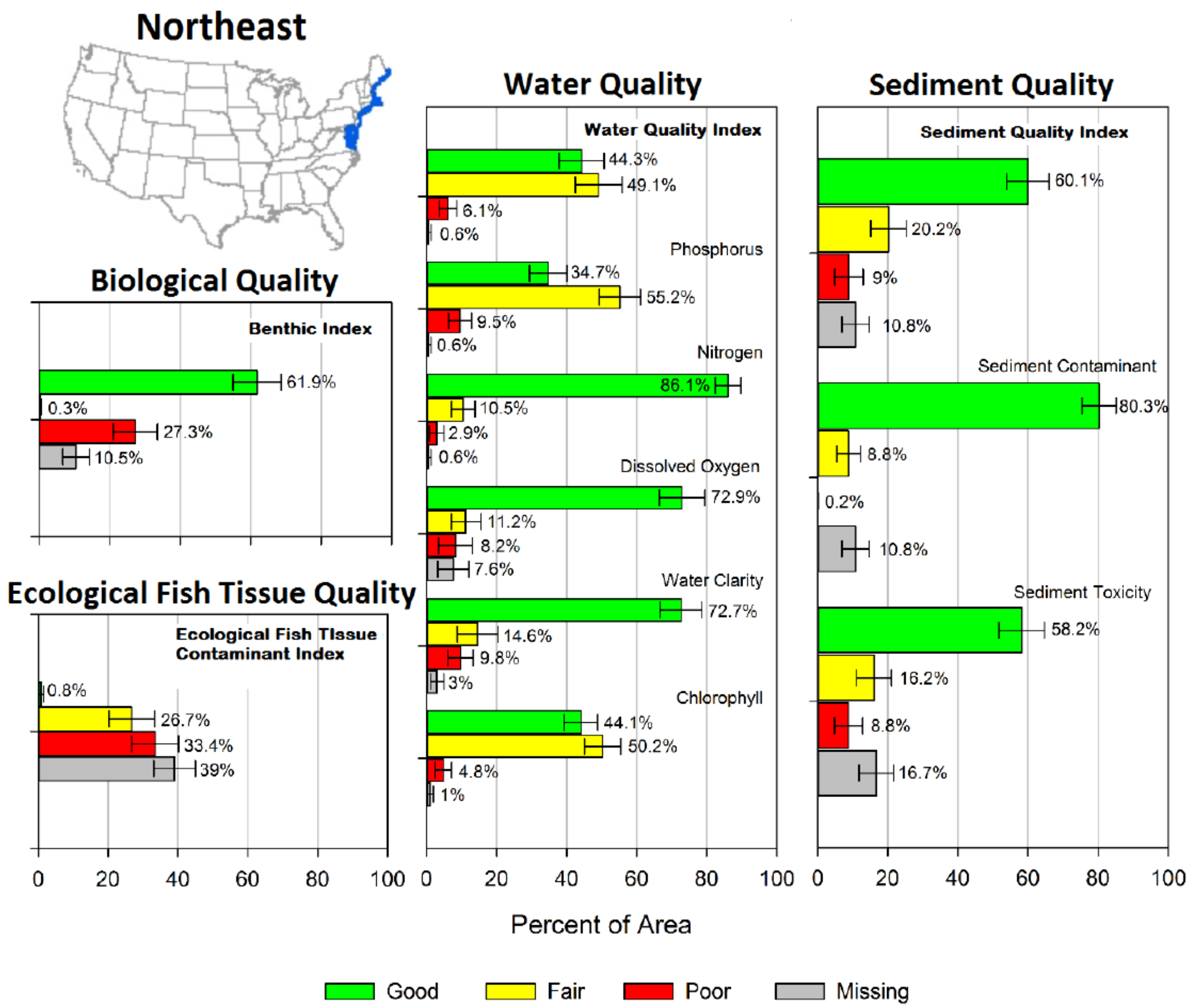


Figure 88. National Coastal Condition Assessment 2010 Report findings for the Northeast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95% confidence levels (USEPA 2015).

9.7.2 Water Quality

As described in General Factors, Section 9.2.6, impaired baseline water temperature, DO, nutrients, BOD, COD, toxics and other 303(d) impairments are significant detriments to the health, diversity, and distribution of aquatic life affecting the survival of native listed species within the action area. Figure 88 and

Table 123 depict water bodies that exceed 303(d) standards and give us insights into which ESUs and DPSs are affected within the Northeast region.

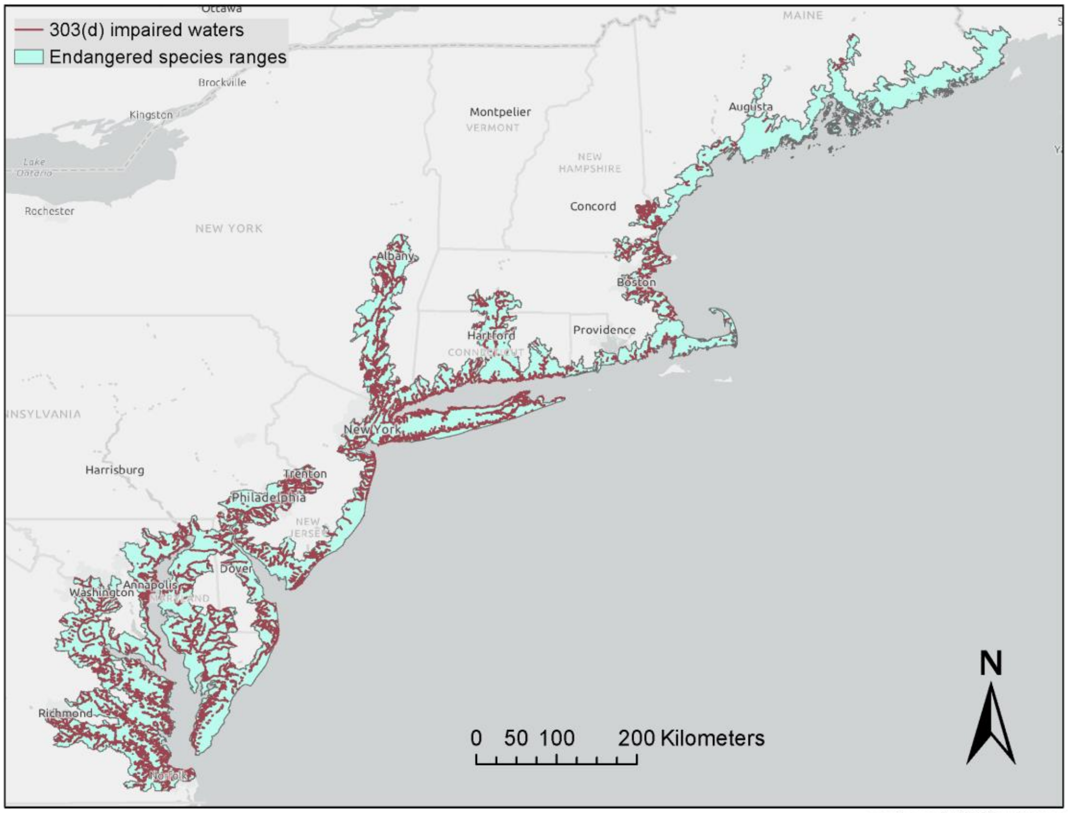


Figure 89. 303(d) impairments within the combined species ranges for the Northeast region. Data downloaded from USEPA ATTAINS website in September 2022.

Of the total number of kilometers of impaired waterbodies in the Northeast region, 33.2% were due to background pesticides and 2.9% were due to elevated temperature (
Table 123, Table 122). The background pesticides found in this subregion (found via the "Detailed Cause" associated with a "Parent Cause" as defined by USEPA's ATTAINS database) include: aldicarb, aldrin, chlordane (which also appears in fish tissue), chlorpyrifos, DDT, DDE, DDT, dieldrin, heptachlor epoxide, hexachlorobenzene, and mirex.

Table 123. Number of kilometers of river, stream and estuaries included in ATTAINS 303(d) lists that are located within species ranges in the Northeast region, along with the percent of impaired waters[1] that each group[2] represents. Data were taken from USEPA ATTAINS website in September 2022.

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
All reasons	**NA**	**19,251.7**	**NA**

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
Organics	**TOTAL**	**16,651.0**	**86.5**
	Toxic Organics	552.8	2.9
	PCBs	7,229.6	37.6
	Dioxins	658.8	3.4
	Oil and Grease	436.3	2.3
	Organic Enrichment/Oxygen Depletion	7,773.5	40.4
Inorganics/Metals	**TOTAL**	**4,589.0**	**23.8**
	Toxic Inorganics	0.0	0.0
	Mercury	1,910.9	9.9
	Metals (other than mercury)	2,678.2	13.9
Nutrients/pathogens/algal growth	**TOTAL**	**20,432.8**	**106.1**
	Ammonia	319.8	1.7
	Nutrients	7,184.3	37.3
	Pathogens	12,226.1	63.5
	Algal Growth	702.7	3.6
	Biotoxins	0.0	0.0

Group[2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
Sediment/Turbidity	**TOTAL**	**2,779.9**	**14.4**
	Turbidity	717.5	3.7
	Sediment	2,062.4	10.7
Pesticides	**TOTAL**	**6,396.1**	**33.2**
	Pesticides	6,396.1	33.2
Water Quality	**TOTAL**	**1,513.3**	**7.9**
	Ph/Acidity/Caustic	1,151.9	6.0
	Salinity/Total Dissolved Solids/Chlorides/ Sulfates	306.0	1.6
	Chlorine	55.4	0.3
Temperature	**TOTAL**	**562.8**	**2.9**
	Temperature	562.8	2.9
Habitat/Flow Alterations	**TOTAL**	**362.4**	**1.9**
	Habitat Alterations	206.7	1.1
	Flow Alterations	155.6	0.8
Other	**TOTAL**	**5,762.2**	**29.9**
	Other Cause	1,007.3	5.2

Group[2]	**Parent Causes - as described by 303(d)**	**Sum of impaired waters (km)**	**Percent of total km impaired waters** [1]
	Cause Unknown	1,055.6	5.5
	Radiation	0.0	0.0
	Taste, Color, Odor	223.5	1.2
	Noxious Aquatic Plants	606.3	3.1
	Nuisance Native Species	0.0	0.0
	Nuisance Exotic Species	341.8	1.8
	Trash	27.0	0.1
	Cause Unknown - Impaired Biota	2,050.6	10.7
	Cause Unknown - Fish Kills	0.0	0.0
	Total Toxics	450.0	2.3
	Fish Consumption Advisory	0.0	0.0

1. The percentages may not add up to 100%, because certain water bodies have more than 1 303(d) impairment and therefore are accounted for more than once on occasion (i.e., in different "Parent Causes").

2. "Parent Causes", as described by 303(d), were grouped into 9 categories and the group totals are shown in bold.

9.7.3 Monitoring Data

9.7.3.1 National-Level Data

National-level data were taken from EPA's carbaryl and methomyl BEs and downloads from NAWQA as described in Section 9.5.3.4.

9.7.3.2 Compilation and Spatial Analysis

For each compound, monitoring data obtained by the EPA from the Water Quality Portal was used to plot maximum pesticide concentrations which were overlaid on the ESA-listed species ranges for the Northeast region in GIS. The concentrations were visualized with graduated symbols; the larger symbols represent higher maximum pesticide concentrations for either carbaryl or methomyl (Figure 90).

Carbaryl was detected at 93 sites, with maximum concentrations ranging from 0.001 to 3.2 µg/L (Figure 90). These sites were sampled over the course of many years, and of the samples collected over time at any given site, carbaryl was detected between 1.7 - 100% of the time, with the majority of locations showing detections over 40% of the time. Methomyl was detected at 5 sites with maximum concentrations ranging from approximately 0.003 to 0.008 µg/L (Figure 90). Of the samples collected at these sites over time, methomyl was detected between 0.5 - 50% of the time with the majority of locations showing detections over 3% of the time.

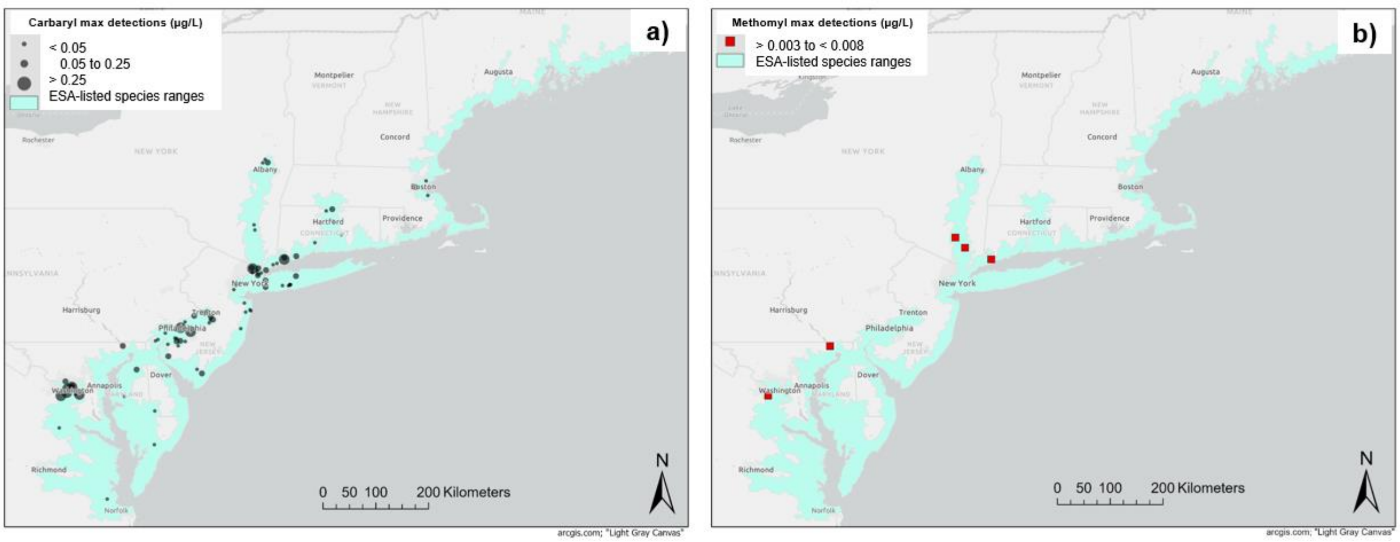


Figure 90. Monitoring data from EPA's carbaryl BE (panel a) and methomyl BE (panel b), taken from the Water Quality portal in 2021 and 2019 respectively, overlaid on ESA-listed species ranges in the Northeast region.

New England Subregion

9.8.1 Land Use

The New England subregion includes all of Maine, New Hampshire and Rhode Island and parts of Connecticut, Massachusetts, New York, and Vermont. The subregion totals roughly 160,000 km^2 of which about 130,000 km^2 is classified as undeveloped, 15,000 km^2 is classified as developed and about 7,000 km^2 is classified as agriculture (Figure 91).

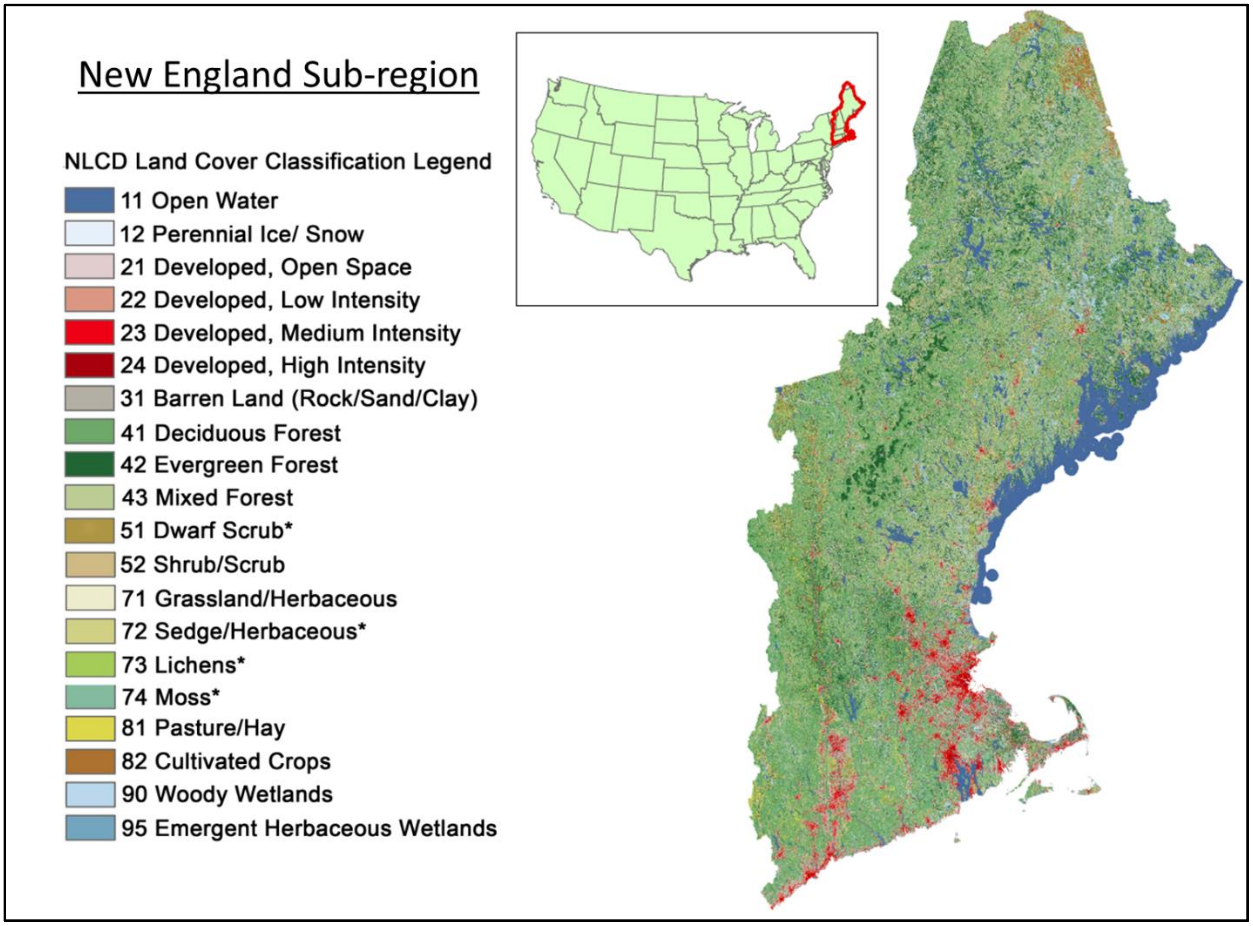


Figure 91. Land use in the New England subregion. Data from the NLCD (www.mrlc.gov).

Four of the 61 species addressed in the opinion occur in this subregion. They are: Atlantic salmon, shortnose sturgeon, Gulf of Maine Atlantic sturgeon, and the New York bight Atlantic sturgeon. Table 124 show the types and areas of land use within selected species' ranges.

Table 124. Area of land use categories within Atlantic salmon and Atlantic sturgeon species ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover NLCD Sub category	Atlantic Salmon	Atlantic Sturgeon, Gulf of Maine DPS	Atlantic Sturgeon, New York Bight DPS
Water	**887**	**925**	**3,904**
Open Water	887	925	3,904
Perennial Ice/Snow	0	0	0
Developed Land	**497**	**2,798**	**10,967**
Open Space	208	729	3,184
Low Intensity	180	873	3,213
Medium Intensity	86	816	2,949
High Intensity	22	380	1,621
Undeveloped Land	**4,514**	**8,339**	**12,083**
Barren Land	40	106	166
Deciduous Forest	505	1,194	5,358
Evergreen Forest	1,389	1,940	712
Mixed Forest	1,467	2,839	1,674
Shrub/Scrub	102	144	118
Grassland/Herbaceous	168	247	232
Woody Wetlands	613	1,373	2,386
Emergent Wetlands	230	497	1,436
Agriculture	**194**	**513**	**2,477**
Pasture/Hay	144	431	927
Cultivated Crops	50	81	1,550
TOTAL (inc. open water)	**6,092**	**12,575**	**29,431**
TOTAL (w/o open water)	**5,205**	**11,649**	**25,527**

1. Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

9.9 Mid-Atlantic subregion

Land Use

The mid-Atlantic subregion includes all of Delaware and New Jersey and the District of Columbia, and parts of Connecticut, Maryland, Massachusetts, New York, Pennsylvania, Vermont, Virginia, and West Virginia. The subregion totals roughly 300,000 km^2 of which about 170,000 km^2 is classified as undeveloped, 40,000 km^2 is classified as developed and about 60,000 km^2 is classified as agriculture (Figure 92).

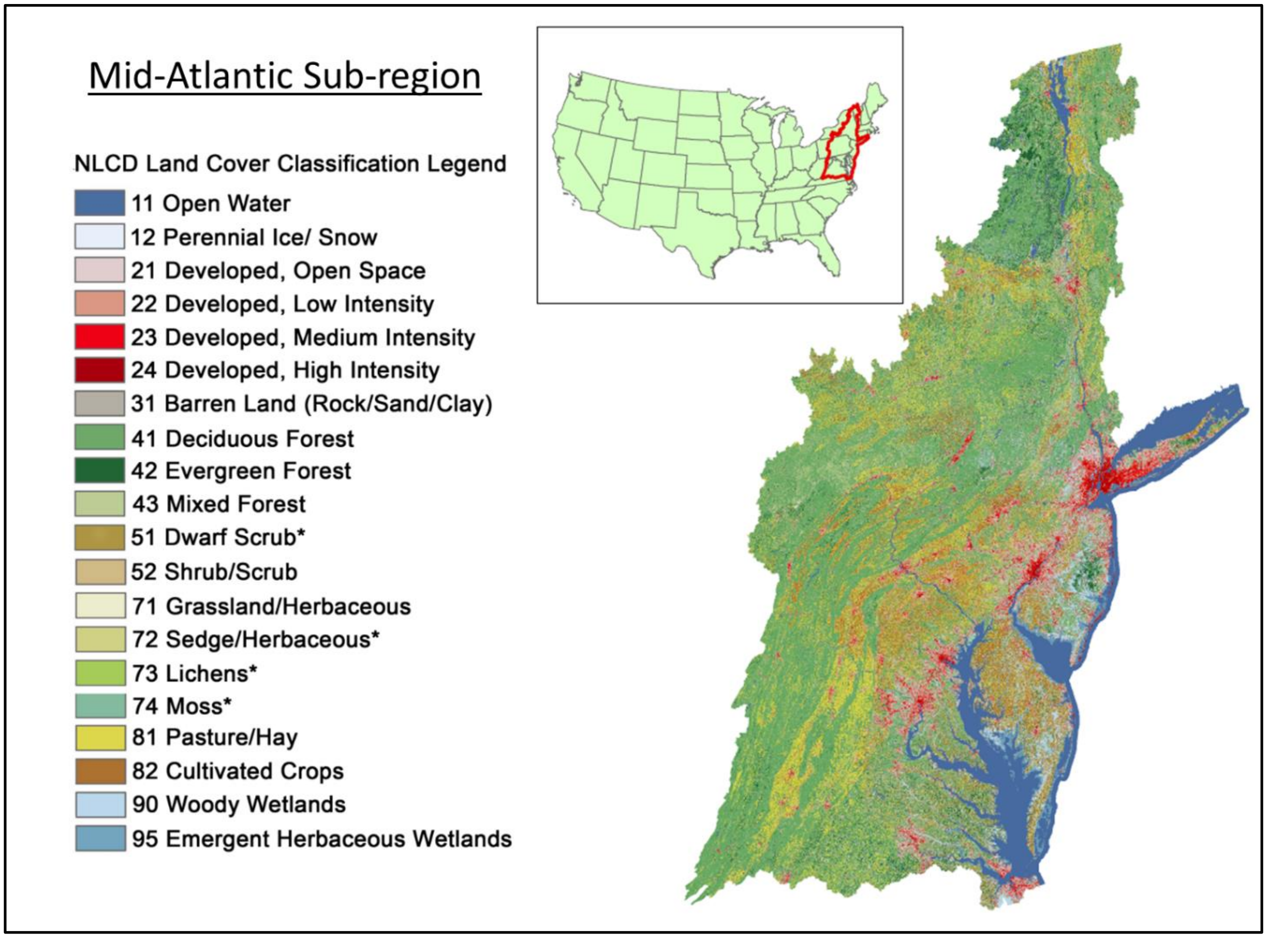


Figure 92. Land use in the Mid-Atlantic subregion. **Data from the NLCD (www.mrlc.gov).**

Three of the 61 species addressed in the opinion occur in this subregion. They are: shortnose sturgeon, Atlantic sturgeon (DPSs: New York Bight, Chesapeake Bay). Table 125 shows the types and areas of land use within selected species' ranges. Note that not all species known to occur in this region are discussed in this section. Species not discussed here are discussed in the other regional reviews.

Table 125. Area of land use categories within selected sturgeon ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover NLCD Sub category	Shortnose Sturgeon	Atlantic Sturgeon, Chesapeake Bay DPS
Water	**35,106**	**6,813**
Open Water	35,106	6,813
Perennial Ice/Snow	0	0
Developed Land	**15,161**	**5,794**
Open Space	5,868	2,356
Low Intensity	4,565	1,851
Medium Intensity	3,158	1,116
High Intensity	1,570	471
Undeveloped Land	**56,262**	**13,494**
Barren Land	367	77
Deciduous Forest	7,877	2,407
Evergreen Forest	13,341	2,003
Mixed Forest	6,170	3,004
Shrub/Scrub	2,100	282
Grassland/Herbaceous	1,911	275
Woody Wetlands	19,287	3,875
Emergent Wetlands	5,209	1,571
Agriculture	**11,964**	**5,615**
Pasture/Hay	2,612	485
Cultivated Crops	9,352	5,131
TOTAL (inc. open water)	**118,493**	**31,716**
TOTAL (w/o open water)	**83,387**	**24,903**

[1.] Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

9.9.1 Factors Affecting Sturgeon throughout the U.S. East Coast

Dams and Diversions. Dams are used to impound water for water resource projects such as hydropower generation, irrigation, navigation, flood control, industrial and municipal water supply, and recreation. Most modern reservoirs are designed for 2 or more of these purposes (Baxter 1977). Dams can have profound effects on diadromous fishes by fragmenting populations, eliminating or impeding access to historic habitat, modifying free-flowing rivers to reservoirs and altering downstream flows and water temperatures. Direct physical damage and mortality can occur to diadromous fishes that migrate through the turbines of traditional hydropower facilities or as they attempt to move upstream using passage devices.

Perhaps the biggest impact dams have on sturgeon is the loss of upriver spawning and rearing habitat. Migrations of sturgeon in rivers without barriers are wide-ranging with total distances sometimes exceeding 200 km or more depending on the river system (Kynard 1997). The construction of dams has blocked upriver passage for the majority of sturgeon populations. Dams have restricted spawning activities to areas below the impoundment, often in close proximity to the dam (Cooke and Leach 2004; Duncan et al. 2004; Kynard 1997).

The suitability of riverine habitat for sturgeon spawning and rearing depends on annual fluctuations in flow, which can be greatly altered or reduced by the presence and operation of dams (Cooke et al. 2004; Jager et al. 2001). Effects on spawning and rearing may be most dramatic in hydropower facilities that operate in peaking mode (Auer 1996; Secor 2002). Daily peaking operations store water above the dam when demand is low and release water for electricity generation when demand is high, creating substantial, daily fluctuations in flow and temperature regimes. Kynard et al. (2012) have documented that flow fluctuations for hydroelectric power generation affected access to spawning habitat and possibly deterred spawning of shortnose sturgeon on the Connecticut River.

Dredging. Many rivers and estuaries are periodically dredged for flood control or to support commercial shipping and recreational boating. Dredging also aids in construction of infrastructure and in marine mining. Dredging may have significant impacts on aquatic ecosystems including the direct removal/burial of organisms; turbidity/siltation effects; contaminant resuspension; noise/disturbance; alterations to hydrodynamic regime and physical habitat and actual loss of riparian habitat (Chytalo 1996; Winger et al. 2000).

The impacts of dredging operations on sturgeon are often difficult to assess. Hydraulic dredges can lethally take sturgeon by entraining sturgeon in dredge drag arms and impeller pumps (NMFS 2022a). Mechanical dredges have also been documented to lethally take shortnose sturgeon (Dickerson 2006). In addition to direct effects, indirect effects from either mechanical or hydraulic dredging include destruction of benthic feeding areas, disruption of spawning migrations, and deposition of resuspended fine sediments in spawning habitat (Chapman et al. 2019).

Dickerson (2006) summarized observed takes of sturgeon from dredging activities conducted by the ACOE; overall 24 sturgeon (11 shortnose sturgeon, 11 Atlantic sturgeon and 2 Gulf sturgeon) were observed during the years of 1990-2005. Of the 24 sturgeon observed, 15 (62.5%) were reported as dead. Dickerson (2006) noted that the largest take of sturgeon species was observed in the Delaware (n=6) and Kennebec (n=6) rivers. To reduce the impacts of dredging on sturgeon, NMFS imposes seasonal restrictions through ESA Section 7 consultations.

Blasting and Pile Driving. Bridge demolition and other projects require blasting with powerful explosives. Fishes are particularly susceptible to the effects of underwater explosions. Unless appropriate precautions are made to mitigate the potentially harmful effects of shock wave transmission, internal damage and/or death may result (Keevin and Hempen 1997). Additionally, in-water pile driving for bridge construction has resulted in high underwater sound pressures that have proved lethal to fishes (Rodkin and Reyff 2008). The impacts from pile driving vary with the methods used and the species tested.

Water Quality and Contaminants. The quality of water in river/estuary systems is affected by human activities conducted directly in the riparian zone and those conducted upland. Industrial activities can result in discharges of pollutants, changes in water temperature and levels of DO, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow. Coastal and riparian areas are also heavily impacted by real estate development and urbanization that result in storm water discharges, non-point source pollution, and erosion. The water quality over the range of sturgeon varies by watershed.

Life history characteristics of shortnose sturgeon (i.e., long lifespan, developmental delay, extended residence in estuarine habitats, benthic foraging, long-distance migration) predispose the species to long-term and repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979; Xu et al. 2022). However, there has been little work on the effects of contaminants on sturgeon to date.

Chemicals and metals such as chlordane, DDE, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders, such as macroinvertebrates, and then work their way higher into the food web (e.g., to sturgeon). Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing DO, altering pH, and altering other physical properties of the water body. Shortnose sturgeon collected from the Delaware and Kennebec Rivers had total toxicity equivalent concentrations of PCDDs, PCDFs, PCBs, DDE, aluminum, cadmium, and copper above adverse effect concentration levels reported in the literature (ERC 2002; ERC 2003). Six individuals collected from the Hudson River have been tested over the past 37 years; most carried very high burden load of PCBs, or one of its derivatives.

Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but the long-term effects are not known (Layshock et al. 2022; Ruelle and Keenlyne 1993; Wirgin and Chambers 2022). Elevated levels of contaminants in several other fish species are associated with reproductive impairment (Arcand-Hoy and Benson 1998; Billsson et al. 1998; Crump and Trudeau 2009; Hammerschmidt et al. 2002), reduced survival of larval fishes (Giesy et al. 1986; Jezierska et al. 2009), delayed maturity (Jørgensen et al. 2004) and posterior malformations (Billsson et al. 1998). Pesticide exposure in fishes may affect anti-predator and homing behavior, reproductive function, physiological development, and swimming speed and distance (Beauvais et al. 2001; Moore and Waring 2001; Scholz 2000; Waring and Moore 2004).

Sensitivity to environmental contaminants also varies across life stage. Early life stages of fishes appear to be more susceptible to environmental and pollutant stress than older life stages (Rosenthal and Alderdice 1976). Dwyer et al. (2005) compared relative sensitivities of common surrogate species used in contaminant studies to 17 ESA-listed species including shortnose and Atlantic sturgeon during a 96-hour acute water exposure to carbaryl, copper, 4-nonphenol, PCP and permethrin using early life stages with mortality as the endpoint. Atlantic and shortnose sturgeon were ranked the 2 most sensitive species of the 17 tested (Dwyer et al. 2005). Additionally, a study examining the effects of coal tar, a byproduct of the process of destructive distillation of bituminous coal, indicated that components of coal tar are toxic to shortnose sturgeon embryos and larvae in whole sediment flow-through and coal tar elutrtraite static renewal (Kocan et al. 1996).

Climate Change. Rising sea level may result in the salt wedge moving upstream, possibly affecting the survival of drifting larvae and young-of-the-year (YOY) sturgeon that are sensitive to elevated salinity. Similarly, for river systems with dams, YOY may experience a habitat squeeze between a shifting (upriver) salt wedge and a dam causing loss of available habitat for this life stage.

The increased rainfall predicted by some models in some areas may increase runoff and scour spawning areas and flooding events could cause temporary water quality issues. Rising temperatures predicted for all of the U.S. will likely exacerbate existing water quality problems with DO and temperature. While this occurs primarily in rivers in the southeast U.S. and the Chesapeake Bay, it may start to occur more commonly in the northern rivers. One might expect range extensions to shift northward (i.e., into the St. Lawrence River, Canada) while truncating the southern distribution.

Increased droughts (and water withdrawal for human use) predicted by some models in some areas may cause loss of habitat including loss of access to spawning habitat. Drought conditions in the spring may also expose eggs and larvae in rearing habitats. If a river becomes too dry all sturgeon life stages, including adults, may become susceptible to strandings. Low flow and drought conditions are also expected to cause additional water quality issues.

Any of the conditions associated with climate change are likely to disrupt river ecology causing shifts in community structure and the type and abundance of prey. Additionally, cues for spawning migration and spawning could occur earlier in the season causing a mismatch in prey that are currently available to developing sturgeon in rearing habitat.

9.10 Southeast Region

The extent of the Southeast coastal region along the Carolinas, Georgia, and Florida encompasses about 4,500 square miles and includes salt marshes, barrier islands, tidal rivers, coastal lagoons, bays and sounds with busy ports and resort areas. Between 1980 and 2006, the coastal counties of the Southeast Coast region showed the largest rate of population increase of any coastal region in the conterminous U.S. The population grew from 7.15 million to 12.8 million people, a 79% increase, and continues to grow with over 15 million people living in the region as of 2010. The aggregated populations of coastal shoreline counties of the Southeast region have grown at rate exceeding about 12% since 2010.

Figure 93 shows a summary of findings from the EPA's 2010 NCCA Report for the Southeast Region (USEPA 2015). Biological quality was rated as good in 77% of the Southeast coast region based on the benthic index. A total of 87 sites were sampled within waters of the Southeast coastal region.

According to the more recent 2015 NCCA Report, the biological condition was good in 62% of the estuaries in the Southeast, according to the M-AMBI marine benthic index. About 84% of the Southeast estuarine area had good sediment quality based on measures of chemical contaminants found in sediments and laboratory tests of toxicity. This is better than the rest of the estuarine area in the continental US in 2015 but is not significantly different from the Southeast sediment quality in 2010. Ecological fish tissue contamination is degraded in the Southeast with 38% of waters in poor condition and 35% in fair. Only 15% of the area is rated good. These estimates do not represent statistically significant change in the proportion of area rate good, fair or poor from 2010 to 2015. The eutrophication index, which examines the potential for estuarine area to undergo social eutrophication based upon measurements of nutrients, chlorophyll a, dissolved oxygen and water clarity, found that 17% of Southeast estuarine area was in good condition, 71% of area was in fair condition and 10% in poor condition. While this indicates that the Southeast is more likely to experience eutrophication than the country as a whole, they are not significantly different than the conditions in the Southeast in 2010. For more information, see EPA's website (https://www.epa.gov/national-aquatic-resource-surveys/southeast-coast-estuaries-national-coastal-condition-assessment).

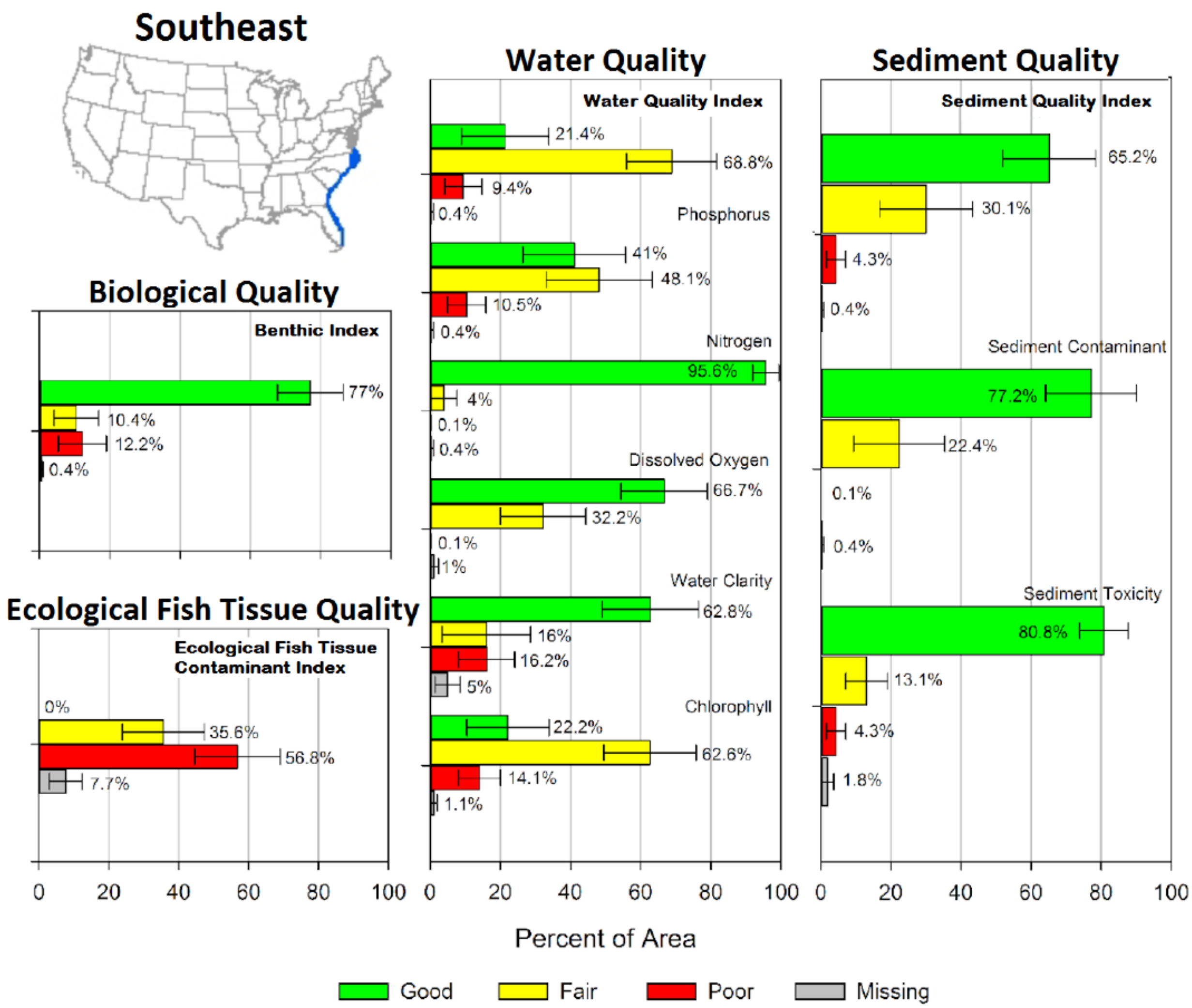


Figure 93. National Coastal Condition Assessment 2010 Report findings for the Southeast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 87 sites sampled). Error bars represent 95% confidence levels (USEPA 2015).

9.11 South Atlantic – Gulf subregion

9.11.1 Land Use

The South Atlantic-Gulf subregion includes all of Florida and South Carolina, and parts of Alabama, Georgia, Louisiana, Mississippi, North Carolina, Tennessee, and Virginia. The subregion totals roughly 700,000 km^2 of which about 470,000 km^2 is classified as undeveloped, 73,000 km^2 is classified as developed and about 118,000 km^2 is classified as agriculture (Figure 94).

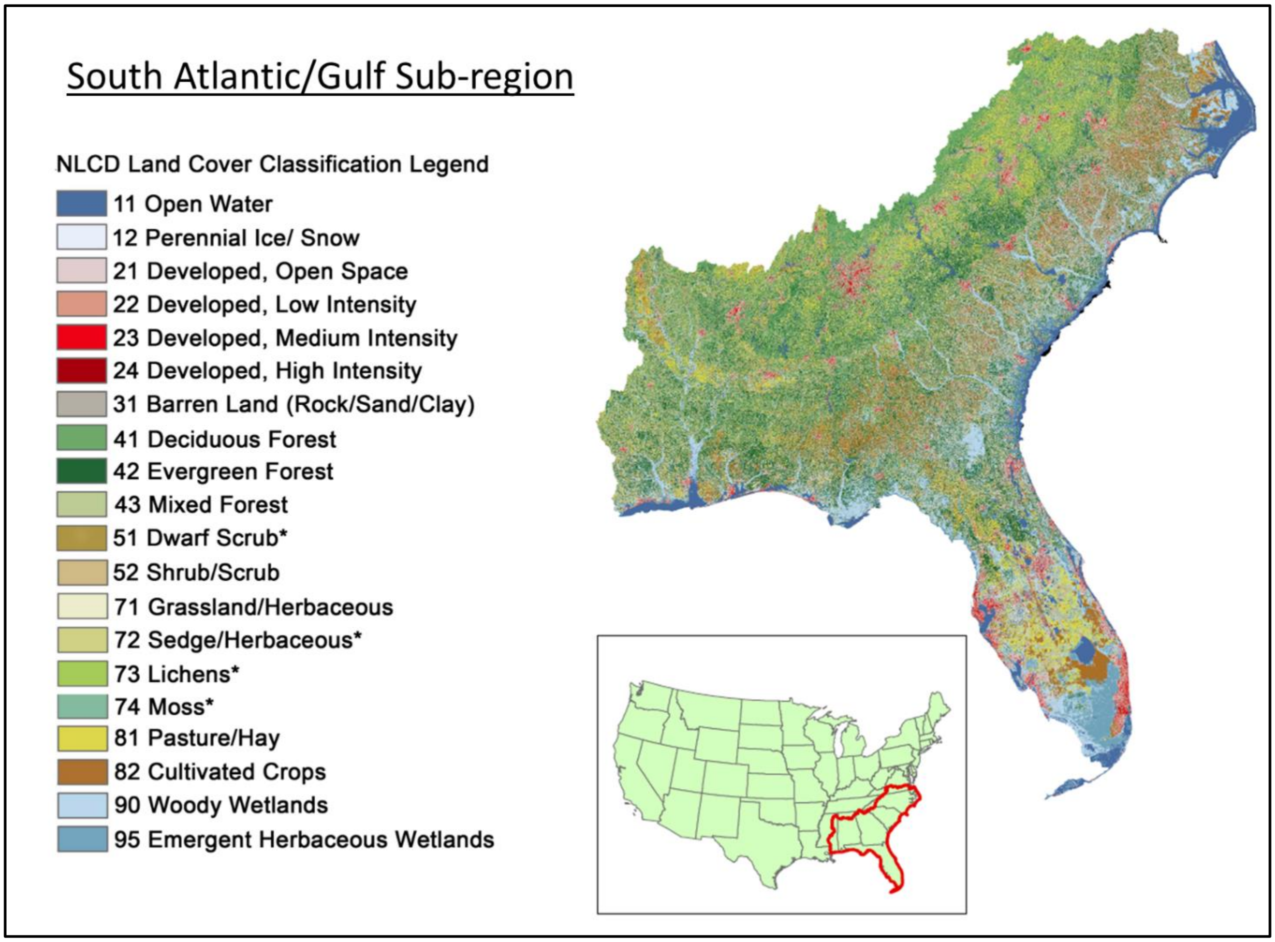


Figure 94. Land use in the South Atlantic/Gulf subregion. Data from the NLCD (www.mrlc.gov).

Thirteen of the 61 species addressed in the opinion occur in this subregion. They are: shortnose sturgeon, Atlantic sturgeon (DPSs: Carolina, South Atlantic), Gulf sturgeon, smalltooth sawfish, Nassau grouper, staghorn coral, elkhorn coral, pillar coral, lobed star coral, mountainous star coral, rough cactus coral, and boulder star coral. Table 126 and Table 127 show the types and areas of land use within selected species' ranges.

Table 126. Area of land use categories within selected sturgeon species ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover NLCD Sub category	Atlantic Sturgeon, Carolina DPS	Atlantic Sturgeon, South Atlantic DPS
Water	**7,395**	**4,058**
Open Water	7,395	4,058
Perennial Ice/Snow	0	0
Developed Land	**3,289**	**4,041**
Open Space	1,512	1,982
Low Intensity	1,035	1,242
Medium Intensity	561	591
High Intensity	182	226
Undeveloped Land	**18,694**	**31,445**
Barren Land	185	158
Deciduous Forest	292	1,051
Evergreen Forest	5,056	10,675
Mixed Forest	599	888
Shrub/Scrub	759	2,200
Grassland/Herbaceous	721	2,121
Woody Wetlands	9,054	10,857
Emergent Wetlands	2,029	3,495
Agriculture	**4,903**	**4,415**
Pasture/Hay	253	1,165
Cultivated Crops	4,650	3,250
TOTAL (inc. open water)	**34,281**	**43,958**
TOTAL (w/o open water)	**26,886**	**39,901**

Table 127. Area of land use categories within selected south Atlantic/Gulf species ranges in km². The total area for each category is given in bold.[1] Land cover was determined via the NLCD 2019. Land cover class definitions are available at: https://www.mrlc.gov/data/nlcd-2019-land-cover-conus

Land Cover / NLCD Sub category	Smalltooth Sawfish	Gulf Sturgeon	Florida Coast Coral Species[2]
Water	**9,481**	**2,360**	**2,122**
Open Water	9,481	2,360	2,122
Perennial Ice/Snow	0	0	0
Developed Land	**9,127**	**3,234**	**656**
Open Space	2,428	1,355	84
Low Intensity	3,248	1,036	200
Medium Intensity	2,564	628	263
High Intensity	887	214	109
Undeveloped Land	**18,042**	**15,091**	**559**
Barren Land	212	210	6
Deciduous Forest	18	15	1
Evergreen Forest	2,145	4,073	11
Mixed Forest	153	83	1
Shrub/Scrub	407	891	4
Grassland/Herbaceous	321	700	5
Woody Wetlands	8,444	6,648	277
Emergent Wetlands	6,343	2,472	254
Agriculture	**1,309**	**797**	**13**
Pasture/Hay	818	432	9
Cultivated Crops	491	366	4
TOTAL (inc. open water)	**37,958**	**21,482**	**3,350**
TOTAL (w/o open water)	**28,477**	**19,122**	**1,227**

1. Note that values for sub-categories have been rounded, and their rounded values may not sum to the total value as displayed

2. The range of 7 species of coral addressed in this opinion have overlap with the Florida coast, the values listed here represent only the Florida portion of these species ranges. The species are: staghorn, elkhorn, pillar, lobed star, mountainous star, rough cactus, and boulder star.

9.11.2 Water Quality

As described in General Factors, Section 9.2.6, impaired baseline water temperature, DO, nutrients, BOD, COD, toxics and other 303(d) impairments are significant detriments to the health, diversity, and distribution of aquatic life affecting the survival of native listed species within the action area. Figure 94 and Table 128 depict water bodies that exceed 303(d) standards and give us insights into which ESUs and DPSs are affected within the South Atlantic/Gulf subregion.

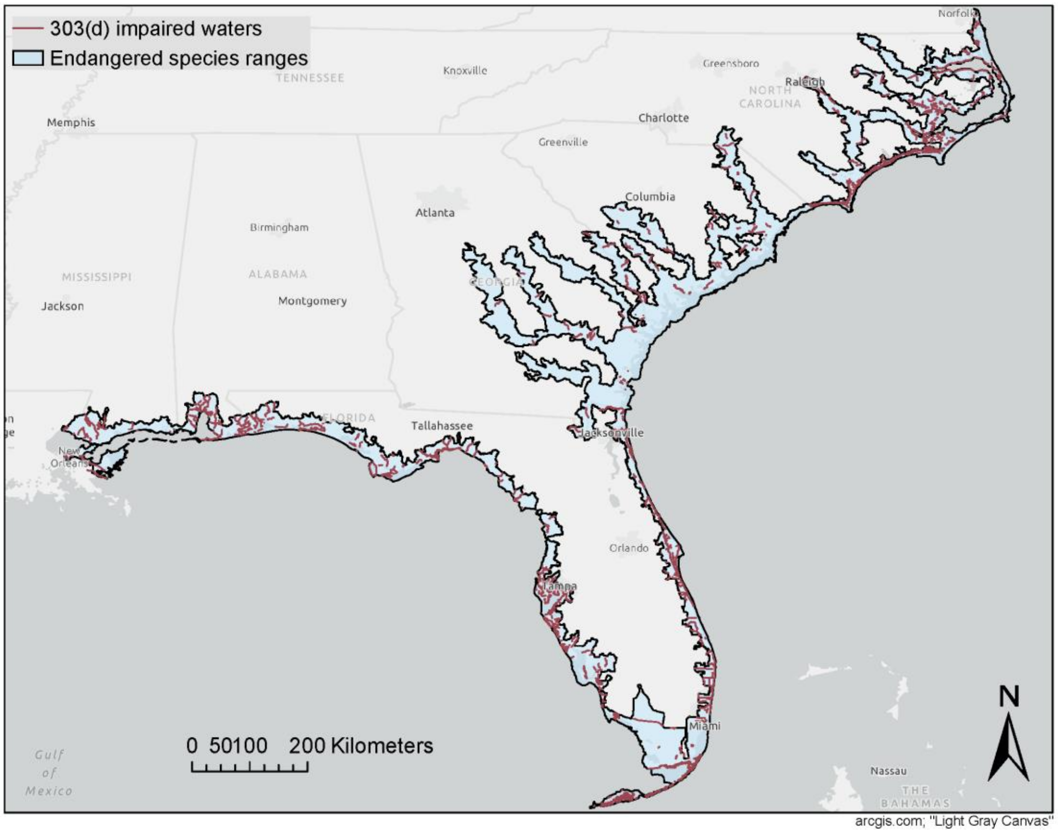


Figure 95. 303(d) impairments within the combined species ranges for South Atlantic/Gulf subregion. Data downloaded from USEPA ATTAINS website in September 2022.

Of the total number of kilometers of impaired waterbodies in the South Atlantic/Gulf subregion, 0.1% were due to background pesticides and 0.6% were due to elevated temperature (Table 128). The background pesticides found in this subregion, found via the “Detailed Cause” associated with a “Parent Cause” as defined by USEPA’s ATTAINS database, include only toxaphene.

Table 128. Number of kilometers of river, stream and estuaries included in ATTAINS 303(d) lists that are located within species ranges in the South Atlantic/Gulf subregion, along with the percent of impaired waters[1] that each group[2] represents. Data were taken from USEPA ATTAINS website in September 2022.

Group [2]	Parent Causes - as described by 303(d)	Sum of impaired waters (km)	Percent of total km impaired waters [1]
All reasons	**NA**	**10,272.0**	**NA**
Organics	**TOTAL**	**3,663.5**	**35.7**
	Toxic Organics	0.0	0.0
	PCBs	61.3	0.6
	Dioxins	32.8	0.3
	Oil and Grease	0.0	0.0
	Organic Enrichment/Oxygen Depletion	3,569.3	34.7
Inorganics/Metals	**TOTAL**	**3,164.8**	**30.8**
	Toxic Inorganics	6.6	0.1
	Mercury	1,517.0	14.8
	Metals (other than mercury)	1,641.2	16.0
Nutrients/pathogens/algal growth	**TOTAL**	**8,697.1**	**84.7**
	Ammonia	51.9	0.5

	Nutrients	3,614.1	35.2
	Pathogens	4,969.4	48.4
	Algal Growth	61.7	0.6
	Biotoxins	0.0	0.0
Sediment/Turbidity	**TOTAL**	**951.7**	**9.3**
	Turbidity	917.9	8.9
	Sediment	33.8	0.3
Pesticides	**TOTAL**	**9.4**	**0.1**
	Pesticides	9.4	0.1
Water Quality	**TOTAL**	**567.3**	**5.5**
	Ph/Acidity/Caustic	119.3	1.2
	Salinity/Total Dissolved Solids/Chlorides/ Sulfates	448.0	4.4
	Chlorine	0.0	0.0
Temperature	**TOTAL**	**61.1**	**0.6**
	Temperature	61.1	0.6
Habitat/Flow Alterations	**TOTAL**	**0.0**	**0.0**
	Habitat Alterations	0.0	0.0
	Flow Alterations	0.0	0.0

Other	**TOTAL**	**714.7**	**7.0**
	Other Cause	32.0	0.3
	Cause Unknown	39.8	0.4
	Radiation	0.0	0.0
	Taste, Color, Odor	1.1	> 0
	Noxious Aquatic Plants	0.0	0.0
	Nuisance Native Species	0.0	0.0
	Nuisance Exotic Species	0.0	0.0
	Trash	0.0	0.0
	Cause Unknown - Impaired Biota	641.8	6.2
	Cause Unknown - Fish Kills	0.0	0.0
	Total Toxics	0.0	0.0
	Fish Consumption Advisory	0.0	0.0

1. The percentages may not add up to 100%, because certain water bodies have more than 1 303(d) impairment and therefore are accounted for more than once on occasion (i.e., in different "Parent Causes").

2. "Parent Causes", as described by 303(d), were grouped into 9 categories and the group totals are shown in bold.

9.11.3 Monitoring Data

9.11.3.1 National-Level Data

National-level data were taken from EPA's carbaryl and methomyl BEs and downloads from NAWQA as described in Section 9.5.3.4.

9.11.3.2 Compilation and Spatial Analysis

For each compound, monitoring data obtained by the EPA from the Water Quality Portal was used to plot maximum pesticide concentrations which were overlaid on the ESA-listed species ranges for the South Atlantic/Gulf subregion in GIS. The concentrations were visualized with graduated symbols; the larger symbols represent higher maximum pesticide concentrations for either carbaryl or methomyl (Figure 96). Carbaryl was detected at 25 sites, with maximum concentrations ranging from 0.005 to 0.66 µg/L (Figure 96). These sites were sampled over the course of many years, and of the samples collected over time at any given site, carbaryl was detected between 1.04 - 100% of the time, with the majority of locations showing detections over 40% of the time. Methomyl was detected at 2 sites with maximum concentrations of 0.003 to 0.021 µg/L (Figure 96). Of the samples collected at these 2 sites over time, methomyl was detected between 2.2 – 6.9% of the time.

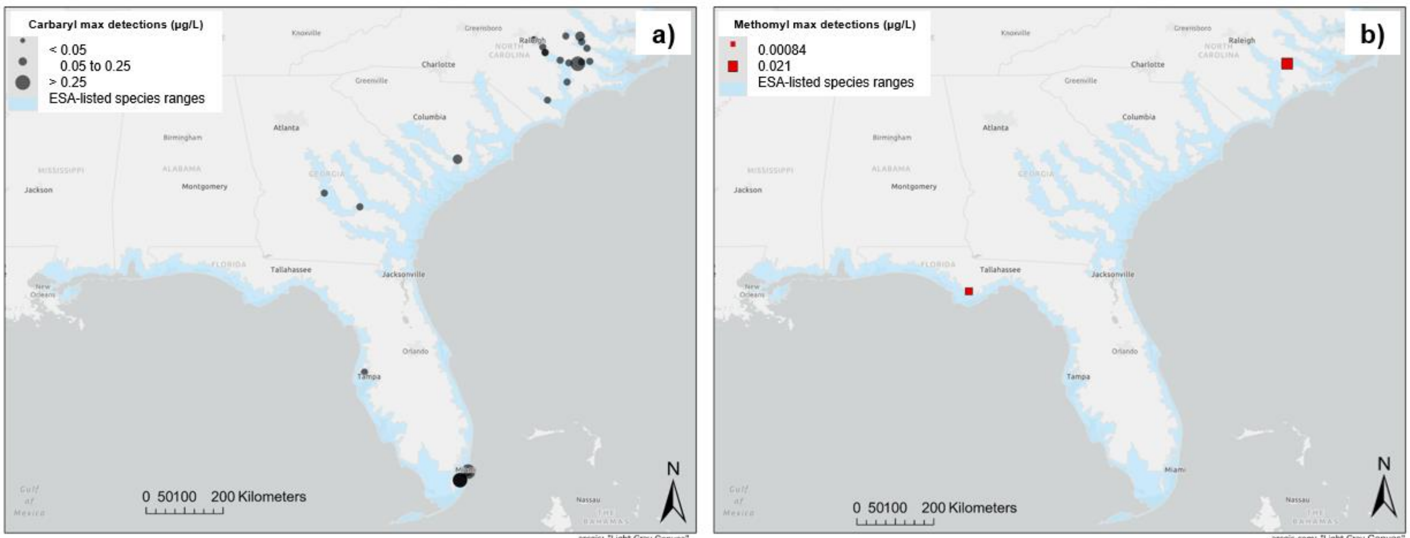


Figure 96. Monitoring data from EPA's carbaryl BE (panel a) and methomyl BE (panel b), taken from the Water Quality portal in 2021 and 2019 respectively, overlaid on ESA-listed species ranges in the South Atlantic/Gulf subregion.

9.12 South Florida and the U.S. Caribbean Subregion

The South Florida and US Caribbean Subregion includes southern Florida, Puerto Rico and the US Virgin Islands (Figure 97).

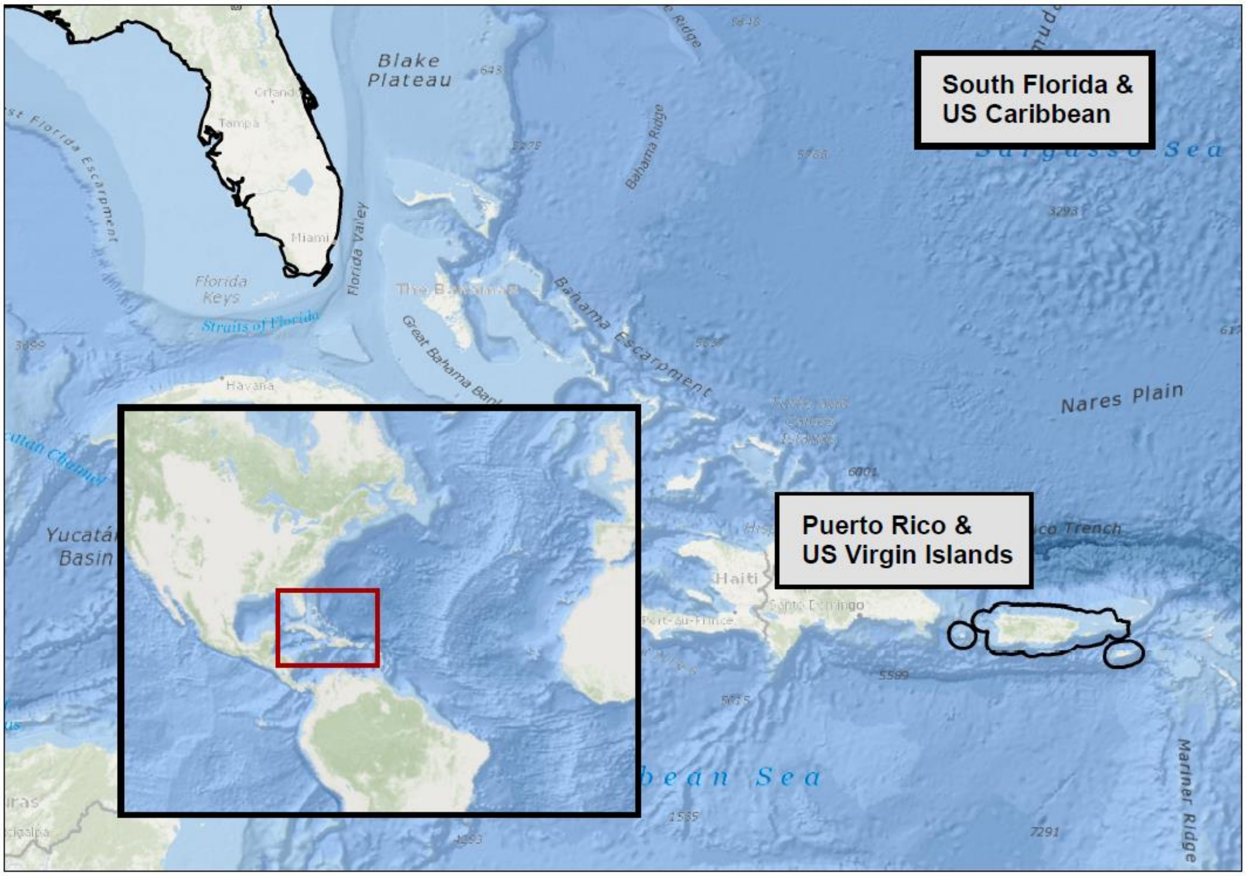


Figure 97. Southern Florida and US Territories in the Caribbean

Nine of the 61 species addressed in the conference and biological opinion occur in this subregion. They are: smalltooth sawfish, Nassau grouper, staghorn coral, elkhorn coral, pillar coral, lobed star coral, mountainous star coral, rough cactus coral, and boulder star coral.

9.12.1 Corals in Caribbean

Coral reefs face an increasing number of threats, including pollution, unsustainable fishing practices, and global climate change. More than 60% of the world's reefs are under threat from local stressors, like fishing and land-based pollution (Burke et al. 2011). That number jumps to 75% when local stressors to reefs are combined with the threat of thermal stress from a changing climate (Burke et al. 2011). Widespread acute and chronic threats to coral habitats adversely affect their ecosystem functions and services, and in certain circumstances, lead to the mortality of coral reef ecosystems.

Coastal Development. In 2010, more than 163 million people (approximately 52% of the U.S. population) lived in coastal counties, and this number is expected to increase to 178 million by the year 2020. With increased coastal development comes increased vessel traffic and increased land-based pollution. Commercial and recreational vessel traffic can adversely affect listed corals through propeller scarring, propeller wash, and accidental groundings. In 1988, anchor damage from the 440-foot cruise ship Wind Spirit destroyed a 300-yd^2 area of coral reef in Francis Bay, St. John, in one of the worst documented cases of anchor impacts within the Virgin Islands National Park (Allen 1992). In 2005 and 2006, a coral reef in Florida was imaged using video mosaic techniques and it was determined that the damage from a 49-foot vessel grounding covered an area of 150 square meters; the damaged coral took over 3 years to show minimal recovery (Lirman et al. 2010b).

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local, or private action, may indirectly affect corals in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs, storm water runoff from coastal towns, and runoff into rivers that empty into bays and groundwater. Point and nonpoint source pollutants, such as excess sediment, nutrients, metals, and pesticides, are particularly relevant due to the potential impacts to habitats such as mangroves, seagrasses, and coral reefs. In addition, sediments and other forms of terrestrial runoff correspond with increased coral degradation and disease (Waddell and Clarke 2008c). Factors affecting stormwater runoff and nonpoint source pollution include rainfall intensity, preceding wet and dry days, pervious and impervious surfaces, land use, and drainage. Runoff from more developed and urban areas includes greater concentrations of soluble metals and fuel-related contaminants (Young et al. 2018). Runoff from agricultural areas is dominated by sediments but, like urban runoff, may also include pathogens, nutrients, pesticides, and metals.

Fisheries. Reef-related commercial, recreational, and subsistence fisheries are economically important. Healthy coral reefs are important for sustainable fisheries production. Commercial fisheries are those that target wild stocks of species with the intent to sell their catch at market. The most recent NOAA data indicate the commercial fishing industry employs around 1 million people (about 1,029,000 in 2009) and contributes $116 billion to the nation's economy, although commercial fisheries landings in coral reef areas are not as large as those in temperate waters. Recreational and charter fisheries in coral reef areas are typically driven by tourism, with patrons hiring a local guide with knowledge of preferred fishing areas to maximize their success. Approximately 8 million individuals participated in coastal recreational fishing along the Atlantic and Gulf of Mexico coasts each year between 2009 and 2014 (NOAA 2019). For many coral reef-associated communities, subsistence harvest of coral reef fishes is a primary source of dietary protein. It is also regarded as less wasteful than commercial harvests due to its reef-to-table nature (Martin et al. 2017). The harvest of fish and invertebrates for the aquarium industry is more prevalent outside of U.S. waters, but Hawaii and Puerto Rico have some active fisheries for the aquarium trade (LeGore et al. 2008; Miyasaka 1997). However, the U.S. is the top importer of these organisms (Bruckner 2005; Rhyne et al. 2017). Demand is driven by aquaria hobbyists, ornamental shell and coral skeleton collectors, as well as a live reef food fish trade, which targets the larger reef fish for consumption abroad.

Various types of fishing gear can interact with corals. Available information suggests hooks and lines from other types of hook-and- line gear and longline gear can become entangled in reefs, resulting in breakage and abrasion of corals but impacts are expected to be minor. Traps have been found to be the most damaging. A study of the trap fishery in the USVI found that, while most fishers deployed traps in seagrass or algae, sand, or coral rubble, a few fishers targeted corals (Sheridan et al. 2006), resulting in habitat impacts. In 2014, it was found that coral reefs off the coast of the Florida Keys contained the highest amount of trap debris compared to noncoral habitats, despite the fishermen claims that they avoid coral reefs while fishing (Uhrin et al. 2014).

For decades, participants in the U.S. Caribbean reef fish fishery (both in the EEZ and USVI and Puerto Rico waters) have targeted species of all trophic levels. Amendments implemented in the past have altered gear construction and usage, closed seasons and areas, changed fishery management units, implemented size limits, placed prohibitions on the use of some fishing practices, and the harvest of some species (e.g., Nassau and goliath grouper). However, the FMP has never set catch quotas. Global reef fisheries are considered unsustainable based on several studies(McClanahan et al. 2011; Newton et al. 2007; Teh et al. 2013).

Climate Change. Corals have already survived multiple major extinction events, which can be attributed to changes in global temperatures and ocean circulation patterns (Hallock 1997; Kiessling 2001; Pandolfi et al. 2011). Global mean greenhouse gases have increased since the advent of the Industrial Era, leading to an increase in atmospheric and ocean temperatures, sea level, and ocean acidification (IPCC 2021). These increasing trends for temperature, sea level, and ocean acidification have only accelerated in recent years and with them so have the frequency and severity of coral bleaching events (Gattuso et al. 2014; Glynn 1993; Hoegh-Guldberg et al. 2014; Hughes et al. 2018), resulting in significant mortality worldwide. Earth's current climate is comparable to the Paleocene/Eocene Thermal Maximum, which was also characterized by a rapid rise in atmospheric greenhouse gas concentrations, increased temperatures, sea-level height, and ocean acidification, resulting in a global reduction of coral (Pandolfi et al. 2011). The impact of this acceleration is most visible for ocean temperatures, which triggered a 3-year, global bleaching event from 2014-2017 that was without precedent in recorded history (Eakin et al. 2018). This 36-month event brought bleaching-level thermal stress to 75% and mortality-level stress to 30% of the world's shallow-water coral reefs, with much more severe levels locally (Eakin et al. 2018). Ocean temperatures have a substantial impact on coral health. During periods of thermal stress, the corals expel their symbiotic zooxanthellae (known as "coral bleaching"), which removes a primary energy and oxygen source for the coral. Although coral can recover from a bleaching episode, bleaching leaves corals vulnerable to disease and other stressors and can also lead to mortality (Brandt and McManus 2009; Brandt et al. 2013). In 2023, extreme marine heatwaves engulfed much of the eastern tropical Pacific and wider Caribbean and anomalously high sea surface tempertatures in the eastern tropical Pacific and wider Caribbean were more extensive than any other year in the satellite record, which started in the early 1980s, putting coral reefs at increased exposure to heat stress (Hoegh-Guldberg 2023). Historical data suggests the 2023 marine heatwaves throughout the eastern tropical Pacific and wider Caribbean will likely be the precursor to a global mass coral bleaching and mortality event (Hoegh-Guldberg 2023).

Disease. Corals are affected by an array of diseases, which can cause mortality on an ocean-basin scale. In the Caribbean, coral diseases include white plague-II, yellow band disease, white band, black band, white pox, red band, Caribbean ciliate infection, dark spots disease, fungal aspergillosis, and tumors. These diseases are especially prevalent during times of stress (e.g., bleaching), as demonstrated in 2005 when a bleaching event coincided with a 2,530% increase in disease lesions, a 770% increase in denuded skeletons, and a loss of 51.5% live coral cover in the

USVI (Miller et al. 2006) and intense outbreaks of white plague-II and yellow band disease mainly affecting Montastraea, Diploria, and Colpophyllia species in Puerto Rico (Turgeon 2008).

Atlantic-Caribbean coral reef ecosystems are in the midst of an unprecedented outbreak of a newly described coral disease, SCTLD. This particular disease is characterized by rapid spread, rapid tissue loss, and high mortality rates. While SCTLD was first reported on Florida's Coral Reef in 2014, reports of its spread to the wider Caribbean region began to occur in early 2018. As of August 2020, SCTLD has affected corals along the entirety of Florida's 360 mile-long reef system, with the exception of the Dry Tortugas region, and has been reported in thirteen Caribbean countries/territories, including the U.S. Virgin Islands and Puerto Rico. Nearly half of the 45 known scleractinian (stony coral) species that comprise Florida's Coral Reef and at least one third of species documented throughout the Caribbean are susceptible to SCTLD, including some of the slowest-growing and longest-lived primary reef-building species of brain, star, and starlet coral, and threatened species such as pillar coral and cactus coral (Neely 2018). Five of the susceptible species are listed under the ESA. While the causative agent remains unknown, the success of antibiotics in halting or slowing the spread in field and laboratory studies for some coral species suggests that a bacterium may be involved; however, it remains unclear whether or not the bacterium (or bacterial assemblage) is the primary agent, a secondary infection, or an opportunistic component of the coral microbiome (Skrivanek 2020).

There is still much to learn about SCTLD. Since 2016, NOAA and response partner organizations, have been working to document the outbreak, identify potential pathogen(s), understand how environmental factors may be contributing to the outbreak/spread of the disease, develop innovative treatments to slow or halt the spread of the disease, and implement best practices to restore damaged habitats. While we know that SCTLD can be transmitted via direct contact and seawater, we do not know how it is traveling across the broader Caribbean region as its appearance has not followed known oceanographic circulation patterns (Skrivanek 2020).

Natural Disturbance. The status and health of coral reef ecosystems are heavily influenced by their oceanographic setting and natural disasters. This includes ocean temperatures, ocean pH, relative sea level, coastal erosion, tropical storms, earthquakes, tsunamis, and volcanic activity. Energy from ocean currents and waves affect coral reefs and coastlines through erosion and breaking corals and other hard structures. Nearshore ocean currents transport and deposit eroded and other loose sediments. Sediment deposition on coral reefs can suffocate corals or reduce light available for photosynthesis of zooxanthellae, thus affecting coral health. In contrast to the chronic effects of routine ocean currents and wave energy, U.S. coral reefs are also subject to acute and extreme impacts of storms. Due to their branching morphologies, corals are especially susceptible to breakage from extreme wave action and storm surges. In the Atlantic, storms form off the West African coast and move toward the Caribbean. Most recently, Hurricanes Irma (2017) and Maria (2017) damaged coral reefs in the USVI, Puerto Rico, and Florida. More storms hit Florida than any other U.S. state, and since 1851 only 18 hurricane seasons have passed without a known storm impacting the state, with a cumulative impact from the storms totaling over $191 billion in damage (2017 USD) (Pasch et al. 2023). Historically, large storms potentially resulted in asexual reproductive events, if the fragments encountered suitable substrate, attached, and grew into new colonies. However, recently, the amount of suitable substrate has been significantly reduced; therefore, many fragments created by storms die.

Furthermore, severe storms can also transport marine debris that could physically damage corals. Storms may also increase suspension of sediments, erosion, and transport of inland sediments and debris via rivers swollen from large precipitation events, all of which negatively affect coral directly, via smothering, and indirectly, through reduced water clarity and salinity.

Sedimentation. Sediment is a significant stressor for corals. Puerto Rico waters have been burdened by sediment due to a legacy of deforestation in the 1950’s to support sugarcane agriculture which endured into the 1980s (Marinez and Lugo 2008). Increasing urban expansion and associated construction activities, in some cases construction converting agricultural land to a built environment, contribute to these sediment loads. Sediment favors competition by macro algae and reduces the availability of suitable colonizing substrate, smothers new recruits, attenuates light penetration and therefore symbiont photosynthesis, and reduces fertilization (Humphrey et al. 2008; Jokiel et al. 2014; Jones et al. 2015). There can be substantial natural variability in turbidity/suspended sediment among coral reef environments due to tides, storms, and river input and sediment tolerance varies among coral species (Anthony et al. 2004; Erftemeijer et al. 2012; Harmelin-Vivien 1994; Jouon et al. 2008; Orpin et al. 2004; Storlazzi et al. 2004) Certain morphologies are prone to collect more sediment from the water column than the coral species is able to clear (Hubbard, 1972; Bak, 1976; Dodge, 1977; Rogers, 1990; Stafford-Smith, 1993).

Non-indigenous Species. The lionfish, originally from the indo-pacific is a particularly harmful non-indigenous species in Florida's waters and in the Caribbean. Lionfish are a major predator on commercial and sport fish species and the herbivorous fish species that are important to controlling algal growth on coral reefs (Albins and Hixon 2013; Cote et al. 2013; Lesser and Slattery 2011). Their presence in reef systems has been associated with severe declines in fish abundance (Albins and Hixon 2008). Initial observations in the mid-1980's are attributed to aquarium releases. They are established in coastal waters from North Carolina to South America. Lionfish have invaded the Loxahatchee estuary (i.e., Jupiter Inlet on the Atlantic coast of Florida). Over 200 young-of-year individuals ranging from 23 to 185 mm were collected over a 1-year survey period. They were primarily associated with man-made structures and associated debris along the shoreline as far as 5.5 km inland (Jud et al. 2011).

9.12.2 Nassau Grouper

Fishing Effects. Two different aspects of fishing effect Nassau grouper stocks, fishing effort throughout the non-spawning months and fishing effort directed at spawning aggregations or migratory access to spawning aggregations. Nassau grouper are fished commercially and recreationally throughout the year by handline, longline, fish traps, spear guns, and gillnets (NMFS General Canvas Landing System). Aggregations are mainly exploited by handlines or by fish traps, although gillnets were being used in Mexico in the early to mid-1990s (Aguilar-Perera 2004). Prior to regulations prohibiting the harvest and possession, the USVI and Puerto Rico's reef fisheries commonly took Nassau groupers at aggregation sites (SAFMC 1990, CFMC 1993).

Habitat Loss. During its various life history stages, the Nassau grouper uses many different communities or habitat types within the coral reef ecosystem. The increase in urban, industrial, and tourist developments throughout the species' range impacts coastal mangroves, seagrass beds, estuaries, and live coral (Mahon 1990). Suitable habitat for the Nassau grouper is particularly susceptible to impacts from human activity because of the relatively shallow water depth range where these features occur as well as their proximity to the coast. As a result, habitat features may be impacted by activities such as coastal and in-water construction, dredging and disposal activities, beach nourishment, stormwater run-off, wastewater and sewage outflow discharges, point and non-point source pollutant discharges, and fishing activities. Loss of juvenile habitat, such as macroalgae, seagrass beds, and mangrove channels is likely to negatively affect recruitment rates. As shown in the Bahamas (Dahlgren and Eggleston 2001), habitat preferences or selection may be key to early survival and subsequent population size and loss of those preferred coral-algal settlement habitats may pose a threat to grouper populations (Kaufman and Romero 2011).

Poor water quality is a threat to both corals and macroalgae in nearshore areas. Increased sedimentation resulting from poor land development practices adds turbidity and pollutants into nearshore habitats and can change water flow patterns in creeks, where newly settled juveniles may be found. Dredging operations are also capable of destroying macroalgal beds that may be used as grouper nursery areas. Stormwater run-off, wastewater and sewage outflow discharges, and point and non-point source pollutant discharges can adversely impact settlement, development, refuge, and foraging essential features by allowing nutrients and sediments from point and non-point sources to alter the natural levels of nutrients or sediments in the water column, which could negatively impact the substrate characteristics or health (e.g., seagrass and corals). Pollution leading to significant declines in water quality may render spawning locations unusable or reduce adult or egg survival. Acoustic disturbances may also inhibit spawning activity due to the acoustic cues used by the animal during courtship and spawning behaviors. Further, because the spawning aggregation sites are so discrete and rare, and the species' reproduction depends on their use of these sites, the species is highly vulnerable at these locations and loss of an aggregation site could lead to significant population impacts.

Climate Change. Nassau grouper have been found across a range of temperatures with the only implication being that spawning occurs when sea surface temperatures are approximately 25°C. If sea surface temperatures rise, the geographic range of the species may shift in response to any changes. One of the other potential effects of climate change could relate to the loss of structural habitat in the coral reef ecosystems (Munday et al. 2008).

9.13 Gulf of Mexico Region

Unique coastal ecosystems in the Gulf Coast include hypersaline lagoons, coral reefs, and mangrove forests. More than half of the coastal wetlands in the conterminous United States occur along the Gulf Coast. The coastal areas of the Gulf of Mexico includes over 750 estuaries over 10 thousand square miles.

Florida's karst geology makes it particularly vulnerable to subsidence, the gradual caving in or sinking of an area of land, and sea level rise. Expansion of inland tidal marshes replacing lowland coastal forests over the last 120 years was demonstrated along the Big Bend of Florida (Raabe and Stumpf 2016). In the past, many of Florida's wetlands were drained for agriculture, logging, and urban development, and numerous rivers were channelized for navigation. The modifications were most intense in south Florida, where, beginning in the 1920s, canals and levees were built to control flooding and to drain wetlands. These modifications resulted in the loss of much of the original Everglades wetlands from Lake Okeechobee south.

9.13.1 Water Quality

As described in the General Factors, Section 9.2.6 above, impaired baseline water temperature, DO, Nutrients, BOD, COD, toxics and other 303(d) impairments are significant detriments to the health, diversity, and distribution of aquatic life affecting the survival of native fish in the Gulf of Mexico Region and elsewhere. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. Gulf sturgeon adults and large juveniles swim upriver from the Gulf of Mexico in the spring when the water temperature is 15-20°C (Chapman and Carr 1995; Fox et al. 2000) and return to the Gulf in the fall when water temperatures range from 18 to 23°C. Temperature exceedances and other water quality stressors would disrupt this pattern. Sawfish have similar temperature tolerances, and a number of factors, in addition to water temperature, such as water depth, shoreline vegetation, and salinity, affect how and when a sawfish uses a habitat. Generally, smalltooth sawfish live in waters warmer than 17.5°C. Smalltooth sawfish tend to live in shallow water and move to deeper waters as they grow.

Figure 98 shows a summary of findings from the EPA’s 2010 NCCA Report for the Gulf Region (USEPA 2015). A total of 240 sites were sampled to assess approximately 10,700 square miles of Gulf coastal waters. Biological quality was rated as good in 61% of the Gulf coast region based on the benthic index.

According to the more recent 2015 NCCA Report, the biological condition was good in 68% of the estuaries in the Gulf of Mexico, according to the M-AMBI marine benthic index. This is similar to the overall national estimate of estuaries in good biological condition of 71% and is not significantly different from the proportion of Gulf of Mexico area rated good in 2010. About 75% of Gulf of Mexico estuarine area had good sediment quality based on measures of chemical contaminants found in sediments and laboratory tests of toxicity. This is similar to the rest of the estuarine area in the continental US in 2015 and shows statistically significant improvement over the NCCA results for sediment quality in the Gulf of Mexico in 2010. Ecological fish tissue contamination was degraded in estuaries of the Gulf of Mexico in 2015 with 74% of waters in poor condition and 15% in fair. Only 9% of the area is rated good. The proportion of area rated poor for ecological fish tissue contaminants in the Gulf of Mexico is higher than the national estimate of 55%. However, a closer look at the data reveals that the area not assessed (area for which fish suitable for analysis could not be caught) in the Gulf is only 1%, while the national estimate of not assessed area is about 10%, and is as high as 28% in estuaries of the West Coast. Care must be taken when comparing populations when there is a wide range of area for which assessments are not available for an indicator. The eutrophication index, which examines the potential for estuarine area to undergo social eutrophication based upon measurements of nutrients, chlorophyll a, dissolved oxygen and water clarity, found that 18% of Gulf of Mexico estuarine area was in good condition, 55% of area was in fair condition and 28% in poor condition. This indicates that the estuarine area in the Gulf is more likely to experience eutrophication than the country as a whole, but doesn't represent a statistically significant change since 2010. For more information, see EPA's website (https://www.epa.gov/national-aquatic-resource-surveys/gulf-coast-estuaries-national-coastal-condition-assessment-2015).

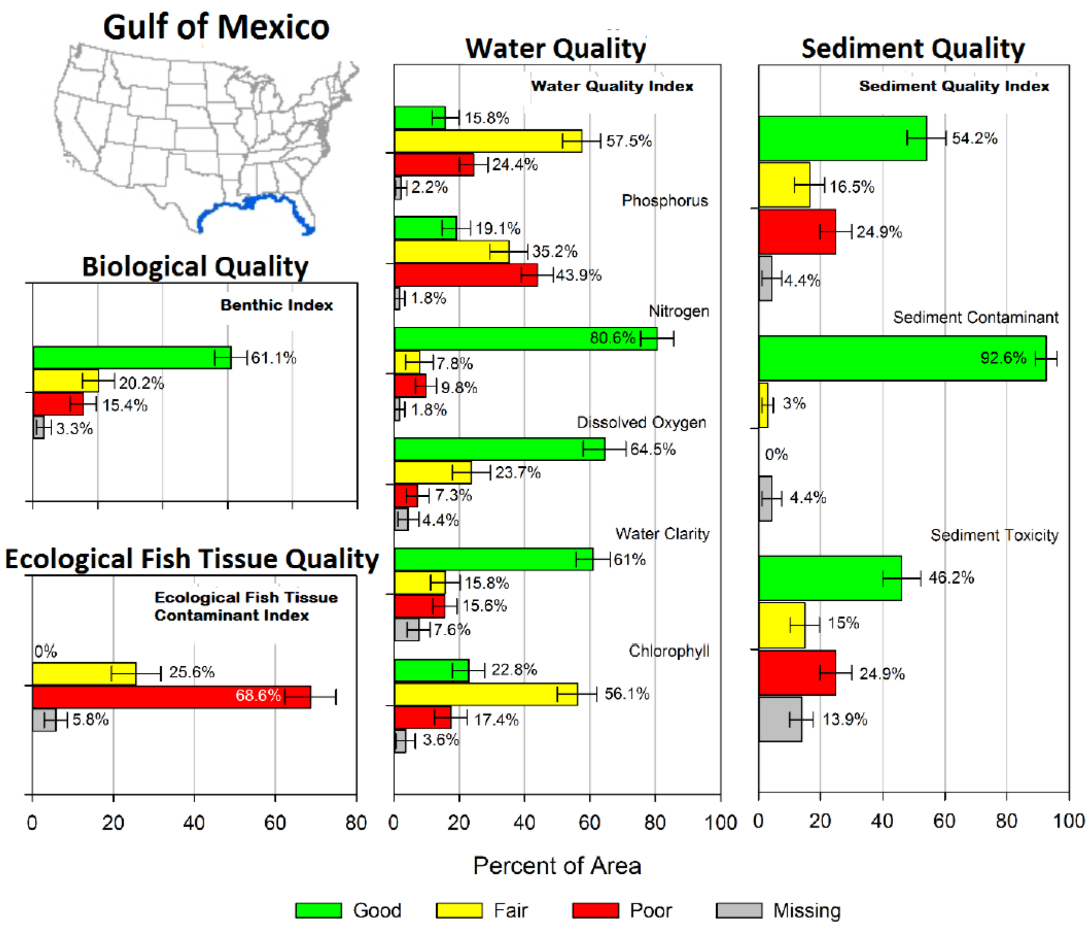


Figure 98. National Coastal Condition Assessment 2010 Report findings for the Gulf Coast. Bars show the percentage of coastal area within a condition class for a given indicator (n = 240 sites sampled). Error bars represent 95% confidence levels (USEPA 2015)

9.13.2 Contaminants

Arsenic has recently arisen as the pollutant of concern in this subregion. The Tampa Bay Tributaries, Withlacoochee, Sarasota–Peace–Myakka, and Ocklawaha Basins have had the highest number of water systems reporting samples with elevated arsenic. The basins with the highest number of wells with exceedances for the 2-year period associated with the Tampa Bay Tributaries, Suwannee, Withlacoochee, and Springs Coast Basins. Arsenic in ground water may be naturally occurring, of anthropogenic origin due to human-induced geochemical changes, or a true contaminant released as a result of human activities. The prevalence of elevated arsenic detections in the southwest Florida basins and the Suwannee Basin may be due to the chemical makeup of the aquifer in these areas. In addition to this natural source, potential anthropogenic sources include arsenic-based pesticides applied to cotton fields; citrus groves; road, railroad, and power line rights-of way; golf courses; and cattle-dipping vats, which were in use in Florida until 1961 (McKinnon et al. 2011). In recent years, the use of arsenical pesticides has significantly decreased, and as of 2013 its use is restricted only to monosodium methanearsonate on cotton fields, golf courses, sod farms, and highway rights-of-way (78 CFR 59). However, residues from past use, when bound to soil particles, do not readily dissipate. Higher numbers of reported exceedances may be considered an artifact of the change in the EPA arsenic standard for ground water, which was reduced from 50 to 10 μg/L in 2001, and was fully implemented in 2006.

9.13.3 Harmful Algal Blooms

The Gulf of Mexico, in particular waters along Florida's coast are susceptible to harmful algal blooms (HABs). Florida monitors for HABs in fresh, estuarine, and marine waters. Blooms can occur any time of year in Florida, due to its subtropical climate. The HABs are caused by a suite of unique taxa that can bloom under particular physical, chemical, and biological conditions. The drivers of some HABs are well understood, while the drivers of other HABs, such as the red tide organism, *Karenia brevis,* are still unclear. While HABs can occur naturally, they are frequently associated with elevated nutrient concentrations. HABs may produce toxins that contaminate shellfish or finfish, making them unsuitable for human consumption. They can also affect plant and animal communities. The Gulf of Mexico Alliance, a partnership between Alabama, Florida, Louisiana, Mississippi, and Texas, is working to increase regional collaboration to enhance the Gulf's ecological and economic health. Reducing the effects of HABs is 1 of its water quality priorities (Anderson et al. 2019)

Freshwater cyanobacteria (or blue-green algae) blooms have received increased attention in recent years because of their potential to produce toxins that can harm humans, livestock, domestic animals, fish, and wildlife. While blooms of cyanobacteria can occur naturally, they are frequently associated with elevated nutrient concentrations, slow-moving water, and warm temperatures. Cyanotoxins are bioactive compounds naturally produced by some species of cyanobacteria that can damage the liver (hepatotoxins), nervous system (neurotoxins), and skin (dermatotoxins) of humans and other animals. Potentially toxigenic cyanobacteria have been found statewide in Florida's rivers, streams, lakes, and estuaries. There are also concerns that freshwater cyanotoxins can be transported into coastal systems. The results of the Cyanobacteria Survey Project (1999–2001), managed by the Harmful Algal Bloom Task Force and Wildlife Research Institute, indicated that the taxa *Microcystis aeruginosa*, *Anabaena* spp., and *Cylindrospermopsis raciborskii* were dominant, while species with the genera *Aphanizomenon*, *Planktothrix*, *Oscillatoria*, and *Lyngbya* were also observed statewide but not as frequently. Cyanotoxins (microcystins, saxitoxin [STX], cylindrospermopsins, and anatoxin) were also found statewide (Williams et al. 2007). Other cyanobacteria of concern in Florida are reported in (Abbott et al. 2009).

More than 50 marine and estuarine HAB species occur in Florida and have the potential to affect public health, water quality, living resources, ecosystems, and the economy. Any bloom can degrade water quality because decomposing and respiring cells reduce or deplete oxygen, produce nitrogenous byproducts, and form toxic sulfides. Declining water quality can lead to animal mortality or chronic diseases, species avoidance of an area, and reduced feeding. Such sublethal, chronic effects on habitats can have far-reaching impacts on animal and plant communities. *Karenia brevis*, sometimes mixed with related *Karenia* species, causes red tides that are an ongoing threat to human and environmental health in the U.S. Gulf of Mexico. Blooms occur annually on the west coast of Florida and less frequently in the Panhandle and east coast. *Karenia brevis* produces brevetoxins that can kill fish and other marine vertebrates, including manatees, sea turtles, and seabirds. Blooms of the STX-producing dinoflagellate, *Pyrodinium bahamense*, have been linked to the bioaccumulation of the neurotoxin STX in puffer fish and more than 20 cases of saxitoxin puffer fish poisoning in Florida (Landsberg et al. 2006). While these blooms raise serious concerns about the ecology of affected ecosystems, there have not been any wide-scale animal mortality events attributed to STXs in Florida. As a tropical species, *P. bahamense*, has seldom bloomed north of Tampa Bay on the west coast or north of the Indian River Lagoon on the east coast. Blooms are generally limited to May through October (Phlips et al. 2006). In Florida, *Pyrodinium* is most prevalent in flow-restricted lagoons and bays with long water residence times and salinities between 10 and 30 practical salinity units. The latter conditions competitively favor *Pyrodinium* because of its slow growth rates and euryhaline character (Phlips et al. 2006). Blooms also appear to be accentuated during periods of elevated rainfall and nutrient loads to lagoons (Phlips et al. 2010), suggesting a link between coastal eutrophication and the intensity and frequency of blooms. However, discharges of naturally tannic waters from wetlands during high-rainfall events can also produce favorable conditions for this organism. These observations also point to the potential role of future climate trends in defining the dynamics of HAB species in Florida (Phlips et al. 2010).

Other bloom-forming marine species can be divided into 2 categories: toxin-producing species and taxa that form blooms associated with other problems, such as low oxygen concentrations, physical damage to organisms, and general loss of habitat. Potential toxin-producing planktonic marine HAB species include the diatom group *Pseudo-nitzschia* spp.; the dinoflagellates *Alexandrium monilatum, Takayama pulchella, K. mikimotoi, K. selliformis, Karlodinium veneficum, Prorocentrum minimum, P. rhathymum*, and *Cochlodinium polykrikoides*; the prymnesiophytes *Prymnesium* spp. and *Chrysochromulina* spp.; and the raphidophyte *Chattonella* sp. (Abbott et al. 2009). Many of these species are associated with fish or shellfish kills in various ecosystems around the world (Landsberg 2002). Additionally, benthic cyanobacteria and macroalgae blooms have been observed on Florida's coral reefs and have been associated with mortality and disease events involving various organisms (Lapointe et al. 2004; Paul et al. 2005; Richardson et al. 2007).

Although many HAB species have been observed at bloom levels in Florida (Phlips et al. 2011), uncertainty remains over the relative toxicity of the specific strains. In addition to ichthyotoxic HAB species that directly cause fish kills, the list of HAB species linked to hypoxia or other density-related issues (e.g., allelopathy, physical damage to gills of fish) is extensive and includes almost any species that reaches exceptionally high biomass. Examples include the widespread bloom-forming planktonic dinoflagellate, *Akashiwo sanguinea*, in the Indian River Lagoon and the St. Lucie Estuary, and the cyanobacterium, *Synechococcus*, in Florida Bay (Phlips et al. 2011; Phlips et al. 1999). Many fish kills, particularly those occurring in the early morning hours, are due to low DO levels in the water associated with the algal blooms and are not necessarily the result of toxins.

Another important issue associated with HABs is the loss or alteration of overall habitat quality. Prolonged and intense coastal eutrophication can result in domination by a select few species, resulting in a loss of diversity and alteration of food web structure and function. For example, during major *Pyrodinium* blooms, 80% to 90% of total phytoplankton biomass is attributable solely to this species (Phlips et al. 2006). Similar domination by a single species occurs in benthic ecosystems, where massive blooms of green and red macroalgae have periodically over-run some shallow habitats of the Florida coast (Lapointe and Bedford 2007).

10 EFFECTS OF THE ACTION, INTRODUCTION

Section Contents

10.1 Introduction

Our analysis of the effects of the action to threatened and endangered species includes 3 primary components, which are integrated into the risk analysis: exposure analysis, response analysis, and species life-history considerations. This effects analysis section is organized following the stressor, exposure, response, risk assessment framework.

Section 7 regulations define "effects of the action" as "all consequences to ESA-listed species or critical habitat that are caused by the action, including the consequences of other activities that are caused by the action. A consequence is caused by the action if it would not occur but for the action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action" (50 C.F.R. § 402.02). Later, in the Integration and Synthesis (Chapters 16, 17, 18, and 19), we add the effects of the action to the status of the species, environmental baseline, and the cumulative effects to formulate the agency's conference and biological opinion as to whether EPA is able to insure that their action is not likely to jeopardize ESA-listed species, and is not likely to destroy or adversely modify designated critical habitat.

10.1.1 Stressors Associated with the Action

For this consultation, EPA's action encompasses all approved product labels containing the a.i.s carbaryl and methomyl. The potential stressors we expect to result from the action include carbaryl and methomyl; other ingredients of these product formulations (including "inert" ingredients and other a.i.s); label recommended tank mixtures (including other pesticide formulations and adjuvants); and toxic metabolites and degradates of product formulation ingredients. An abiotic stressor (e.g., temperature) that is present in the habitat of ESA-listed species may influence response of the species to stressors associated with the action.

10.1.2 Mitigation to Minimize or Avoid Exposure

For carbaryl and methomyl, EPA has committed to label restrictions beyond those originally described and analyzed in the March 9, 2023 draft opinion. These label changes are incorporated as conservation measures into the final description of the action (described in Chapter 5).

10.2 Exposure Analysis

In this section, we describe the methods used to characterize pesticide exposure to ESA-listed species. The procedures rely on models that identify potential interactions of pesticides with ESA-listed species and quantify the magnitude of exposure based on how the pesticides and the ESA-listed species behave in the environment. We begin with a description of the development of aquatic habitat bins, linking physical characteristics that define aquatic habitats used by ESA-listed species with modeling parameters used to predict exposure. Next, we identify information used to derive exposure estimates for different routes of exposure including contact with contaminated surface or pore water. Finally, we identify information on the co-occurrence of pesticide use and ESA-listed species to assess the likelihood of exposure to pesticides.

10.2.1 Aquatic Habitat Bins

The National Research Council Committee of the National Academy of Sciences recommended that fate and transport models be used to estimate time-varying and space-varying pesticide concentrations in generic habitats relevant to ESA-listed species (NAS 2013b). Physical characteristics of aquatic habitats, including depth, width, and flow rate affect the environmental concentrations and dissipation patterns of pesticides. A generic habitat defines these physical parameters and uses them to derive EECs. The 2-meter deep, static "Farm Pond" that is routinely used by EPA in screening level assessments is an example of a generic habitat. Defining generic habitats to represent all ESA-listed species is a challenge given the diversity in the habitats they occupy. Ultimately, the Services identified 10 habitat "bins," a number EPA felt could feasibly be evaluated given the scope of the analysis (Table 129). The generic habitats included: 1 aquatic-associated terrestrial habitat; 3 static freshwater habitats of varying volume; 3 flowing water habitats of variable volume and flow rates; and 3 marine/estuarine habitats representative of nearshore tidal, nearshore subtidal, and offshore habitats. EPA and the Services previously agreed to utilize these bins to develop EEC for ESA-listed species that rely on aquatic habitats (EPA 2015).

Table 129. Generic aquatic habitats parameters for exposure modeling

Generic Habitat Bins	Depth (meters)	Width (meters)	Length (meters)	Flow (m^3/second)
1 – Aquatic-associated terrestrial habitats	NA	NA	NA	NA
2- Flowing water (low-flow)	0.1	2	length of field[1]	0.001
3- Flowing water (moderate-flow)	1	8	length of field	1
4- Flowing water (high-flow)	2	40	length of field	100
5 – Static freshwater (low-volume)	0.1	1	1	0
6- Static freshwater (moderate-volume)	1	10	10	0
7- Static freshwater (high-volume)	2	100	100	0
8- Intertidal nearshore	0.5	50	Length of field	NA
9- Subtidal nearshore	5	200	Length of field	NA
10- Offshore marine	200	300	Length of field	NA

[1]length of field – The habitat being evaluated is the reach or segment that abuts or is immediately adjacent to the treated field. The habitat is assumed to run the entire length of the treated area.

The Services identified the bin(s) representative of habitats utilized by each ESA-listed species. A single species may occur in a range of habitats represented by multiple bins. EPA's carbamate BE's identify each of the species bin assignments originally recommended by the Services.

Bin 1 represents habitats in the terrestrial-aquatic transition zone, such as riparian habitats, dunes, beaches, and rocky shorelines. Examples of species that utilize these habitats include sea turtles for nesting, and pinnipeds for nesting and lounging. These species may be exposed to pesticides in either the terrestrial or aquatic environment. For example, black abalone occupy intertidal habitats and remain in the intertidal zone during periods of low tide when their habitats are dewatered. The intertidal also produces important invertebrate prey for juvenile Chum and

Chinook salmon. These species can be exposed to pesticides present in the surface water during periods of inundation, or be exposed directly to spray drift during periods of low tide.

Flowing water habitats represented by bins 2, 3, and 4 vary considerably in depth, width, and velocity, which influence both initial concentration and rates of dissipation. These bins are defined by differing flow rates that are products of velocity (influenced by the gradient and other factors) and habitat volume (width and depth). Flow rates vary temporally and spatially in these habitats and are influenced by several factors. For example, bends in the shoreline, shoreline roughness, and organic debris can create back currents or eddies that can concentrate allochthonous inputs. Dams and other water control structures would also significantly influence flow. Some small streams and channels are intermittent and can become static and temporally cut off from connections with surface water flows during dry seasons. Low flow habitats may also occur on the margins of higher flow systems (e.g., floodplain habitats associated with higher flowing rivers).

Bin 2 is intended to represent habitats with flow rates occurring at 0.001-1 m^3/second including springs, seeps, brooks, small streams, and a variety of floodplain habitats (oxbows, side channels, alcoves, etc.). Examples of ESA-listed species that utilize habitats fitting the characteristics of bin 2 include the Pacific eulachon and several salmonid species. During spawning migration, Pacific eulachon migrate upstream through shoreline habitats during low hydrograph periods at depths of 0.1-1 m. While there are different habitat preferences among the species, listed Pacific salmonids utilize lower flow habitats in some phase of their lifecycle for activities such as spawning, rearing, or migration. Bin 3 flow rates are representative of small to large streams (1-100 m^3/second) and bin 4 definitions (larger volumes and flow rates exceeding 100 m^3/second) correspond with larger riverine habitats. These habitats are used by listed anadromous species during spawning migrations (e.g., salmonids, sturgeons, and eulachon). Smalltooth sawfish ascend inland river systems and juveniles have been found in streams and canals consistent with bin 3 and bin 4 parameters. Additionally, juvenile green turtles shelter and forage in coastal streams and rivers.

Bins 5, 6, and 7 represent freshwater habitats that are relatively static, where flow is less likely to substantially influence the rate of pesticide dissipation. Examples of bin 5 habitats (volumes <100 m^3) include vernal pools, small ponds, floodplain habitats that are cut off from main channel flows, and seasonal wetlands. Salmonid juveniles use a variety of small volume floodplain habitats to forage, over-winter, and shelter from larger predators such as backwater areas and off-channel ponds that are relatively static and may temporarily loose connection to the main stream channel. Bin 6 volumes (100 – 20,000 m^3) correspond with many ponds, vernal pools, wetlands, and small shallow lakes and Bin 7 represents larger volume habitats (>20,000 m^3) such as lakes, impoundments, and reservoirs. Impoundments are frequently encountered by anadromous fish during spawning migrations of adults and out-migrations of juveniles. Ponds and lakes are also utilized by salmonids for rearing, particularly juvenile sockeye salmon which rear in lakes for 1 to 3 years.

Bins 8, 9, and 10 were designed to characterize marine habitats. Marine habitats are generally defined by water depth and distance from shoreline. The nearshore, or neritic zone is the relatively shallow area that extends from the coastlines to the edge of the continental shelf at depths of approximately 200m. Nearshore habitats are subdivided into the intertidal zone (Bin 8, the area between shoreline and mean low tide mark), and the subtidal zone (Bin 9, nearshore habitats that extend from the mean low tide mark to the continental shelf and are generally submerged). Bin 10 is intended to represent the deep offshore habitats (>200m in depth) that extend beyond the continental shelf. Depths within the intertidal zone are variable between locations but generally range from 0 to <10 m. Depth within the intertidal habitat depends on the tidal cycle and tidal range. Surface waters can persist during low tides and may be used by ESA-listed species. For example black abalone may be found in tide pools and salmonids may use distributary channels exposed during low tide periods. Depth in the subtidal habitats range from 0 – approximately 200m. Listed corals occur primarily in the subtidal zone. Southern resident killer whale and beluga whale also utilize nearshore habitats, as do all listed sea turtles, pinnipeds, and anadromous fish, and several listed marine fish (e.g., rockfish, grouper, and sawfish). Offshore habitats are used by all listed cetaceans and sea turtles and are also used by anadromous fish (e.g., salmonids, sturgeon), marine fish (e.g., hammerhead sharks), and pinnipeds (e.g., Hawaiian monk seal).

10.2.2 Exposure Estimates

Exposure modeling to characterize risk

EPA and NMFS rely on chemical fate and transport models to estimate pesticide concentrations that are likely to occur in listed species' habitats, analyze the effects of transport to habitats, and to determine whether EPA has insured that a pesticide is not likely to jeopardize ESA-listed species, and is not likely to destroy or adversely modify designated critical habitat. These models incorporate empirical data on chemical-specific transport, degradation, and partitioning in the environment. Additionally, these models are field-validated and incorporate regionally-specific meteorological data and soil types typical of pesticide use sites. These models provide a standard method to estimate time-varying and space-varying pesticide concentrations expected to occur following defined pesticide applications in varying aquatic habitats with physical parameters consistent with those used by listed species. As such, they provide the best available data, closely linking specific applications (e.g., a label rate and method) to estimated exposure concentrations in listed species habitats (e.g., a nearby small, shallow stream). This approach for characterizing risk to listed species was recommended through an independent scientific review by the National Academy of Sciences (NAS 2013b).

EPA revisions to exposure modeling for listed species habitats

While the carbaryl and methomyl BEs refer to the aquatic habitat "bins" described in Table 129, the physical parameters (e.g., width and depth of habitat) no longer align with the Services' original recommendations for these bins (EPA 2015). In 2020, EPA revised their approach for BEs, reducing the number of habitat bins evaluated in an effort to improve the efficiency of the consultation process (EPA 2020). In doing so, EPA's model estimates exclude several key aquatic habitats (e.g. differentiate between a medium flowing water body, bin 3, versus a larger flowing water body, bin 4) and instead provide estimates for 3 generic scenarios intended to represent potential exposures over a broad range of specific aquatic habitats.

EPA now utilizes only 2 standard water bodies (the farm pond and index reservoir) to estimate pesticide concentrations in larger static (habitat bins 6 and 7) and flowing water bodies (habitat bins 3 and 4). EPA derived estimates for these 2 generic habitats to estimate pesticides in surface waters and benthic sediment pore water using the PWC fate and transport model. More information and details (e.g., PWC inputs and outputs) are available in the BEs (EPA 2021a; EPA 2021c). While some of the assumptions for physical parameters of the model of aquatic habitats have been revised, they still represent some of the habitats utilized by listed species.

For small volume habitats that offer little in terms of dilution potential (e.g., characteristic of bins 2 and 5), the EECs are expected to be comparable to pesticide concentration in runoff because these habitats can easily be inundated and dominated by surface water runoff. Therefore, EPA uses PRZM daily runoff data (e.g., ZTS files) to estimate pesticide concentration for these habitats. The impact of the revisions on risk estimates is discussed below and in the section entitled Weighing the Uncertainties (10.6). The methodology used in the BEs to estimate runoff and drift utilized inputs consistent with application requirements specified on product labels. Additionally, inputs representing application site characteristics (e.g., meteorological conditions) were selected at the HUC2 regional scale to generate geographically-specific EECs (EPA 2021a; EPA 2021c).

EPA Farm pond: EECs provided for medium-large volume static habitats (e.g., ponds, lakes, and surrogate for marine off-shore)
The PWC is a field-scale, fate and transport model designed to predict pesticide concentrations in aquatic habitats on the edge of a treated field. These estimates factor in pesticide deposition from drift and runoff. We expect this model provides reasonable estimates of exposure in habitats located in close proximity to treated areas, particularly when the model assumptions closely match site conditions (e.g., the drainage area and physical parameters of the habitat are comparable to those being modeled). However, it is also important to consider that the output will underestimate potential exposure for some habitats (e.g., small ponds and other relatively static habitats with volumes less than 20,000 m^3) and overestimate potential exposure in others (e.g., offshore habitats and other water body with volumes greater than 20,000 m^3).

EPA Index reservoir: EECs provided for medium - large volume flowing habitats (e.g., streams, rivers, and surrogate for marine subtidal and offshore)
EPA also used the PWC to estimate concentrations in medium to large flowing water habitats assuming the index reservoir generic habitat parameter inputs. There is a large amount of uncertainty with regard to the accuracy of these estimates as neither the assumptions regarding watershed size, nor the physical parameters that define the receiving water body in index reservoir appear characteristic of streams, rivers, or the marine habitats they are intended to represent. The PWC does not account for the physical properties of marine habitats. Additionally, models to estimate pesticides in marine habitats do not currently exists and it is not feasible to utilize watershed-scale models to estimate pesticide concentrations in streams and rivers across the action area given the inputs required. Rather than relying on watershed models which require making highly uncertain assumptions regarding the presence/absence and timing of multiple pesticide applications, we relied on the PWC model estimates which are essentially field scale models which generate realistic exposure estimates for treatment to a single use site. The PWC EECs for both farm pond and index reservoir represent concentrations that are expected to occur in larger volume aquatic habitats that occur near the edge of the treated field when the pesticide is applied according to product labeling. While they are quantitative in nature,

we apply them qualitatively recognizing that they represent only the modeled situations and are more representative of some habitats than others. For example, when considering exposure potential for adult sturgeon, we looked at the general range of exposure values, rather than focusing on any particular estimate.

EPA Runoff calculations: EECs provided for low volume habitats (e.g., pools, small ponds, headwater streams, off-channel/floodplain habitats, tide pools)
EPA derived pesticide concentrations in daily runoff to inform our estimation of exposures that could occur in shallow habitats with physical characteristics comparable to bin 2 and 5 (e.g., off-channel/floodplain habitats, tide pools, headwater streams, etc.). While runoff itself is not an aquatic habitat, low volume habitats can be dominated by storm water during runoff events, with little to no dilution during the event. NMFS relied on these calculations to define runoff transport into these habitats. As with all modeled estimates, these values are applied qualitatively. While these estimates are useful for characterizing realistic contributions from runoff in the most vulnerable, low volume habitats, they do not consider additional contributions that may occur from other important exposure pathways (e.g., aerial drift discussed below).

EPA PFAM – EECs provided to estimate pesticide discharge from cranberry
EPA also derived EECs associated with flooded (rice and cranberry) agriculture using the PFAM model (Appendix 3-2, EPA 2021a). NMFS relied on the "manual_release" EECs from the PWC output to define transport into listed species habitats that could occur when water is intentionally released from cranberry bogs following harvest operations. Estimates for rice were not utilized as this use was removed from the action.

Additional surface water estimates
NMFS derived additional exposure estimates to characterize likely exposure from drift transport into low volume habitats that are important for many listed species (e.g., pools, small ponds, headwater streams, off-channel/floodplain habitats, tide pools). Low volume habitats are susceptible to drift transport when located in proximity to pesticide use sites. While EPA did not provide EECs associated with this route of exposure in their BEs, NMFS worked with EPA to derive estimates based on AgDRIFT deposition curves and assuming the bin 2 generic habitat parameters to predict deposition and initial concentrations that could result from applications considering buffers and application methods from the labels. Aquatic degradation (i.e., chemical half-life) and sediment sorption (i.e., EPA Tier 1 Rice Model) were then used to factor in dissipation of the compound and develop EECs for comparison to toxicity estimates.

Updates to EECs to reflect recent label changes
Agreements to modify carbaryl and methomyl labeling occurred after the release of EPA's carbaryl and methomyl BEs. Applicants and EPA worked together to generate new EECs that reflect recent agreements to modify carbaryl labels. NMFS also coordinated with EPA to generate new EECs to reflect recent changes to methomyl labeling discussed in the description of the action (Chapter **Error! Reference source not found.**).

Attachment 2 of this document contains: 1) the PWC batch files used to generate EECs for the farm pond, index reservoir, and runoff; 2) the spreadsheets used to generate the drift EECs; and

3) all of the EECs gathered into a single file (Carbamate_eecs_111622.csv). These EECs form the basis for the Risk-plots presented in the Risk Characterization.

NMFS did consider the degree to which label restrictions on applications within 48-hours of rain ('rainfast') would reduce runoff but did not quantitatively adjust any of the EECs prior to the Risk Characterization (e.g., Risk-plots). Rather, NMFS applied a 10% reduction in EECs in the effects analysis based on the chemical properties (i.e., half-lives) and analysis of the PWC data (e.g., comparing runoff versus time between application date and precipitation).

NMFS also considered the impacts of the addition to the carbaryl labels of authorization to use the pesticide to treat red scale in California citrus (Chapter 5) subsequent to the issuance of the draft conference and biological opinion for public comment. The EECs described above and presented in the Risk-plots do not include this single application to citrus at 12 lbs a.i./A (fewer applications but 2.4x the rate). The increased application rate is expected to produce higher EECs. For drift, this would be a proportional increase of 2.4x in EECs. For the other estimates (farm pond, index reservoir, and runoff), NMFS generated EECs by modifying the PWC inputs for citrus in California to reflect a single carbaryl application at 12 lbs a.i./A. The resulting EECs ranged from 1.0 - 2.9x higher with the increased application rate. NMFS determined that the label restrictions for this specific carbaryl use (e.g., prohibiting application within proximity to listed species, see Chapter 5) are sufficient to address the increased risk (12 lbs a.i./A) posed by the higher EECs. The maximum single application rate of 5 lbs a.i./A that can be applied in proximity to species was assessed to evaluate this use.

For the risk characterization, NMFS considered the EECs discussed above (Farm Pond, Index Reservoir, Runoff, and Drift), along with other relevant information, to qualitatively represent estimates for the range of potentially relevant exposure concentrations based on species habitat use. When considering estimates for larger volume habitats, we factored in the expectation that contributions from other sites within the watershed that did not see applications will serve to reduce the EECs via dilution. Given the lack of adequate exposure models, NMFS relied on exposure estimates for the freshwater bodies to represent marine aquatic habitats (e.g., nearshore tidal, subtidal, and offshore habitats). For example, drift EECs were used to characterize the likely range of acute exposure that could occur in tide-pool habitats.

Similarly, NMFS qualitatively used 4 different time-weighted averaging periods (TWAs) to assess the effects of different exposure durations. NMFS's risk characterization compared acute exposure durations (i.e., 1-day and 4-day TWAs) to acute toxicity tests (e.g., 4-day LC50 test) to characterize the likelihood of acute effects. Chronic exposure estimates (i.e., 21-day and 60-day TWAs) were compared to toxicity tests of chronic duration to evaluate risk hypotheses. Uncertainties associated with using the available EECs to represent exposure concentrations present in aquatic habitats and compare to toxicity data are considered in the risk characterization as part of the confidence and discussed in the uncertainty section (10.6).

Water Quality Monitoring Data

In the NAS's independent review of the NMFS and EPA procedures for characterizing risk to listed species, the committee noted the importance of distinguishing between monitoring data that can and cannot be used to estimate pesticide exposure or evaluate exposure model predictions (NAS 2013b). The exposure models used by EPA and NMFS are field-scale models designed to estimate concentrations in aquatic habitats that occur in close proximity to the application site. The National Academy of Sciences advised that only field-scale monitoring data should be used to estimate exposure or evaluate the accuracy of the model predictions. Further, the field-scale modeling must be associated with specific applications of the pesticide "under well-described conditions such as application rate, field characteristics, water characteristics, and meteorological conditions (NAS 2013b)." This is due to the unlikelihood of observing comparable results between monitoring and modeling data when the scenarios are so different. It is vital to consider the spatial and temporal relationships between the pesticide application and the pesticide monitoring given the influence of these variables on the magnitude of pesticide exposure.

We grouped water quality monitoring information into 2 categories: (1) field studies, and (2) general monitoring studies (NAS 2013b). Field studies are defined as the monitoring of specific applications of the pesticide at the field-scale under well-described conditions. These studies can provide the information needed to make direct comparison to exposure model estimates depending on how well conditions of the field study being monitored match up with the model inputs assumptions [including, but not limited to application rate, method, buffers, meteorological conditions (e.g., wind direction, wind speed, temperature, relative humidity), characteristics of the treated site (e.g., soil type), and characteristics of the habitat sampled (e.g., width, depth, and flow)]. Model inputs used by EPA to generate exposure values are selected to provide a range of exposure estimates based on ranges of application and site conditions. Field studies monitoring other situations are not directly comparable and cannot be used to evaluate model estimates. Studies lacking this well-described information are defined as general monitoring studies. General monitoring studies provide information on pesticide concentrations in surface water or ground water at specific locations and times. They may also be coordinated with the use of pesticides (e.g., at the watershed level). However, they cannot be used to evaluate exposure model predictions.

The available monitoring data are applicable to assessing exposure in ESA-listed species habitats to varying degrees. Common aspects that limit the utility of the available monitoring data as accurate depictions of exposure within ESA-listed species habitats include: 1) protocols were not designed to capture peak concentrations or durations of exposure in habitats occupied by ESA-listed species; 2) limited utility as a surrogate for other non-sampled surface waters; 3) lack of representativeness of current and future pesticide uses and conditions; and 4) lack of information on actual pesticide usage to correlate with observed surface water concentrations. These aspects are discussed in more detail below.

1. Protocols not designed to capture peak exposure: The NAWQA monitoring studies contain the largest data set evaluated. However, these studies were designed to evaluate trends in water quality and were not designed to characterize exposure of pesticides to ESA-listed species. Sampling from the NAWQA studies and other studies (see Monitoring Data, Section 9.5.3 of the Environmental Baseline) reviewed was not conducted in coordination with specific applications of carbaryl and methomyl and do not meet the criteria (discussed above) that are necessary to estimate pesticide exposure or evaluate exposure model predictions. Similarly, sampling was not designed to target the

habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of these pesticides in flowing water habitats, it is not surprising that pesticide concentrations from these datasets were generally much lower than predicted by modeling efforts. While these studies provide useful information on pesticide trends, and have proved useful in addressing documented exceedance of water quality criteria through adaptive management, they have limited utility for our purposes because concentrations monitored at specific locations cannot be placed into context without considerations of the temporal and spatial relationships to pesticide applications.

2. Limited applicability to other locations: Pesticide runoff and drift are influenced by a variety of site-specific variables such as meteorological conditions, soil type, slope, and physical barriers to runoff and drift. Additionally, surface water variables such as volume, flow, and pH influence both initial concentrations and persistence of pesticides in aquatic habitats. Finally, cropping patterns and pesticide use have high spatial variability. Given these and other site-specific factors, caution should be used when extrapolating monitoring data to other sites.
3. Representativeness of current and future uses: Pesticide usage varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future usage of products containing the 2 a.i.s.
4. Lack of information on actual usage to correlate with observed concentrations: A common constraint in the monitoring data was lack of information on actual usage of pesticides containing the 2 a.i.s. For example, the ability to relate surface water monitoring data to the action was severely hampered because information on application rates, setbacks/buffers, and applications methods associated with the monitoring were frequently not reported. In most cases, the temporal and spatial aspect of pesticide use relative to sampling was not reported, further limiting the utility of the information.

We characterized the available studies as general monitoring. As suggested by the National Academy of Sciences (NAS 2013b), we did not use these studies to estimate pesticide concentrations after a pesticide application (i.e., to predict maximum concentrations that are likely to occur from approved uses) or to evaluate the performance of EPA's fate and transport models.

10.2.3 Estimating Co-Occurrence Associated with Pesticide Uses

NMFS evaluated co-occurrence of ESA-listed species with pesticide use to assess the likelihood of exposure by comparing the spatial distribution of species with the labeled uses of the a.i.s. We relied, in part, on previous analyses performed by EPA and developed as part of their BEs. Details of the procedure and rationale are available in sections of the EPA BEs. In brief, use sites described on the pesticide labels (e.g., carrots) were assigned to land use categories (Use Data Layers; UDLs). Some use sites were grouped into an aggregate category (e.g., carrots as part of the Vegetables and Ground Fruit UDL), while some crops (e.g., corn) were kept as an individual land use category (i.e., the Corn UDL). Geospatial information associated with the use sites and the land use categories were primarily based on data from the National Land Cover Database (2011) and the NASS Cropland Data Layer (CDL; 2013-2017). The CDL has over 100 different cultivated classes which were grouped by EPA in order to reduce the likelihood of errors of omission and commission between similar crop categories; these groupings were designed to minimize uncertainties. Over the 15-year period of the action, cropping patterns for many crops may change due to market demand or crop rotations. Additionally, there is the potential for mis-classification of crops. Relying on broader aggregate land use categories for specific use sites was considered conservative and not likely to undergo substantial changes during the 15-year duration of the action.

EPA's BEs provided overlap data for each UDL on a species basis (range and habitat). Rather than rely on this data, NMFS choose to use the approach taken in NMFS's previous organophosphate opinions (available at https://doi.org/10.25923/mqyt-xh03). NMFS relied on EPA's GIS data for each UDL and calculated the overlap with United States Geological Survey hydrologic units at the 12-digit hydrologic unit code scale (HUC12) using the 2021 Watershed Boundary Dataset. The resulting overlap data are provided as part of Attachment 2 of this document. For each use category, overlap data for a species range, habitat, or other critical area (e.g., nursey estuaries) was calculated by combining the data from all the HUC12 regions making up the area of interest and displayed as part of the Risk Characterization (e.g., on a Risk Plot). The specific list of HUC12s used for a given species, range, or area are provided in Attachment 2 of this document. This process allowed the overlap data representing the area of interest to be readily modified. The overlap data represents an estimate of the area where authorized use could occur. Use site groupings (e.g., vegetables and ground fruit) were designed to minimize uncertainties, however they also introduce the possibility that overlap percentages include uses for which the active ingredient has not been registered. These uncertainties were considered in our assessment of confidence. NMFS does not assume that use will occur on all the acres associated with a use category. The uncertainty in the actual extent of use is discussed below and handled qualitatively in the assessment (e.g., in the Risk Characterization).

NMFS did not rely on usage data quantitatively in the Risk Characterization (e.g., in the Risk-plots). Usage data provides information on the extent of pesticide use that has occurred in a region. NMFS assessed various sources of information on past use (e.g., water quality monitoring, proprietary surveys, and required reporting) for their robustness and quality and identified numerous uncertainties and limitations. Details are presented in Appendix D. NMFS concluded that usage data – alone – are not sufficient to represent the extent of pesticide use that will occur over the 15-year period of the action, particularly given NMFS's need to insure the action doesn't jeopardize the species or adversely modify the habitat. This is very distinct from NMFS assessment of the uncertainties associated with the use categories (e.g., the CDL and UDL layers) discussed above.

Specific examples of NMFS's analysis of usage data are shown in Figure 99 and Figure 100. The data are from California Department of Pesticide Regulation's (CalDPR) Pesticide Use Reporting database (PUR). The PUR data is publicly available (https://www.cdpr.ca.gov/docs/pur/purmain.htm) and based on required reporting of most pesticide use in California (both agricultural and non-agricultural). For each pesticide, the figure shows the total acres treated (Y-axis) in CA with the pesticide each year from 2003-2017 for every site name (X-Axis) for which use was reported. Sites with no use will not appear. Due to the log scale, years with no use of the pesticide on a site are omitted. For the 2 pesticides and across all sites, use is highly variable year-to-year. The difference between the minimum year of use and the maximum is often 10x and can be as much as 10,000 acres. The large variability in past use means that any quantitative estimate of future use over the 15-year action period is highly uncertain. Unlike the land use categories, NMFS determined that the usage (the extent of pesticide use) can substantially change during the 15-year period of the action.

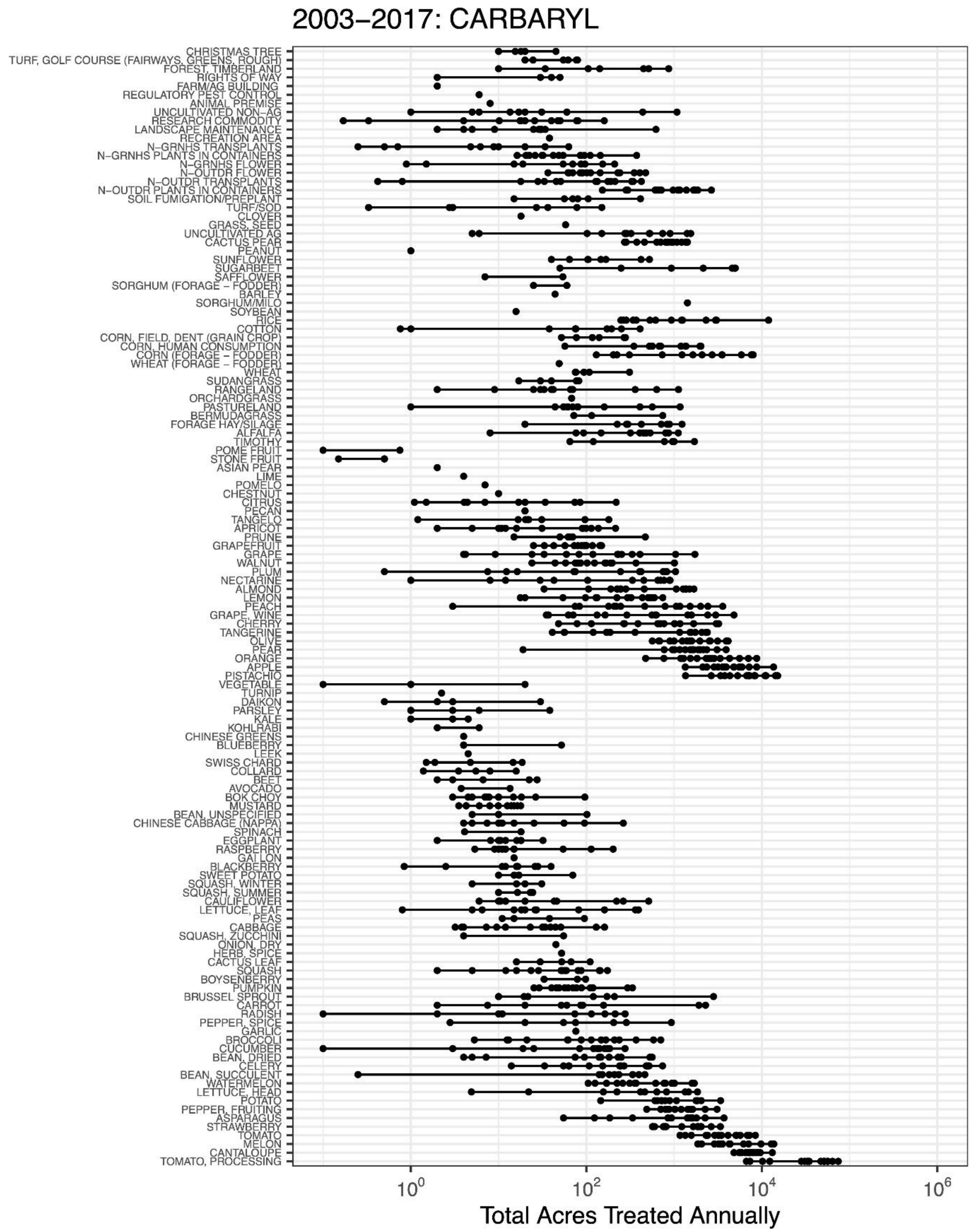


Figure 99. Total Acres Treated with carbaryl in CA each year from 2003-2017. Sites are grouped by Use Categories used in the overlap analysis (e.g., Vegetables and Ground Fruit sites, Orchard sites, etc.). Years with no use reported are not shown. Note the log scale.

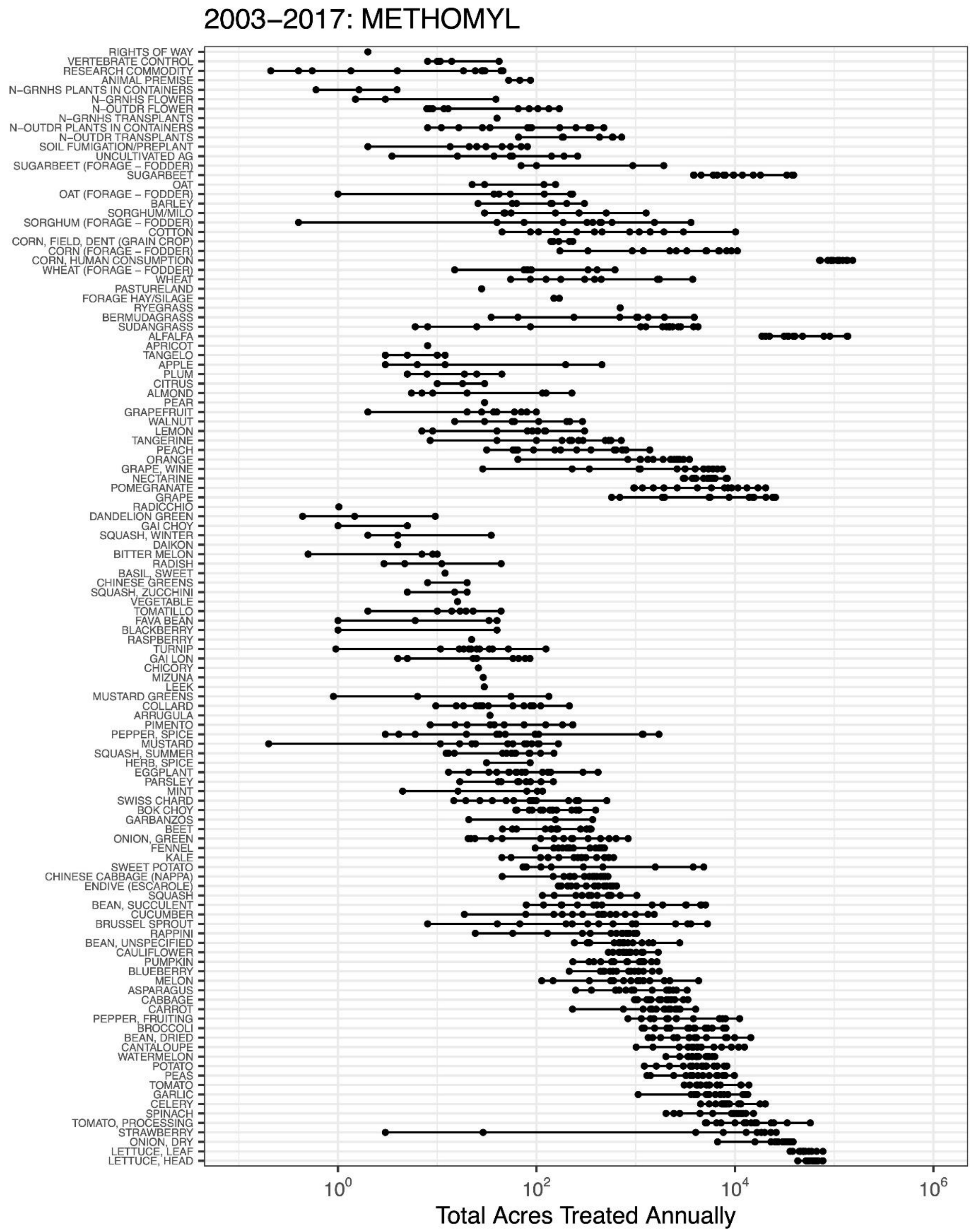


Figure 100. Total Acres Treated with methomyl in CA each year from 2003-2017. Sites are grouped by Use Categories used in the overlap analysis (e.g., Vegetables and Ground Fruit sites, Orchard sites, etc.). Years with no use reported are not shown. Note the log scale.

Response Analysis

10.3.1 Introduction

NMFS is charged under the ESA to evaluate all effects of an action on ESA-listed species and critical habitat in the action area that may be affected by the action. We evaluate all aspects of an action that may reduce fitness of individuals or reduce the conservation value of PBFs of designated critical habitat. We relied on the available response information (i.e., toxicity data) for the stressors of the action reported by EPA in the BEs, and located additional information through our own search of available toxicity information (EPA 2021a; EPA 2021b) to perform our effects analysis. In addition to the information provided in the BEs, NMFS performed its own systematic literature review. Searches were made for any new information starting from the search dates in both BEs through July 2022. In addition, NMFS performed an open literature search without date limitations in order to see if any information was missing from the BEs. Databases searched were ECOTOX, Web of Science, and Google Scholar. A number of references in this section take either of the forms: a) MRID#### or b) E#### (which comes from EPA's ECOTOX knowledgebase - https://cfpub.epa.gov/ecotox/index.cfm). The full citations associated with these references can be found by accessing EPA's BEs for carbaryl and methomyl at https://www.epa.gov/endangered-species, or in the case of references with the E#### identifier, by querying ECOTOX. Summaries of the totality of toxicological response information considered follows.

The majority of available toxicity data was for mortality (i.e., lethality). Sublethal toxicity data included assessment endpoints for growth, reproduction, behavior, sensory function, and AChE inhibition. These sublethal endpoints are reasonably linked to fitness level consequences in individuals and populations. For example, sublethal effects to essential behaviors such as reductions in a fish's ability to swim can clearly translate to fitness level consequences by impairing an individual's ability to feed, escape predation, migrate, etc. The importance of sublethal effects is supported by numerous empirical studies. Notably, an adverse outcome pathway has been developed for carbamate pesticides establishing causal links between molecular effects (i.e., AChE inhibition) and individual level impacts (i.e., mortality), thereby establishing the relevance of the AChE endpoint to this analysis.

Relevant toxicological data for each endpoint are displayed on Risk-plots to aid in comparing response levels to EECs. In the following sections, the data selected for display on the Risk-plots is described. In some cases, each endpoint (e.g., LOEC) was displayed separately (e.g., growth) whereas, in other cases, endpoints were combined to form dose-response curves or ranges. In generating dose-response curves we sought to use the most appropriate combinations of EC50, slope and equation. In cases where a representative slope was not available, we assumed a default slope of 4.5. A slope of 4.5 is considered a 'typical' slope based on a cross-section of dose-response data available in 1975 – when the 'typical' slope was estimated for the Special Review of Pesticides; Criteria and Procedures; Final Rule [40 CFR Part 154: 49005; 49007; 49016 § 154.7(a)(3), (4), (5), and (6)].

Several of the references/citations in this section take the form of ECOTOX or MRID identification numbers. These references can be found in the BEs for carbaryl and methomyl, which are available at https://www.epa.gov/endangered-species.

10.3.1.1 Mode and Mechanism of Action

Carbaryl and methomyl share a similar mode and mechanism of toxic action and are a part of a group known as N-methyl carbamates. Both have similar chemical structures and act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates. They inhibit the enzyme AChE, which is present in cholinergic synapses. The normal function of AChE is to break down (hydrolyze) the neurotransmitter, acetylcholine, thereby serving as an “off-switch” for the electrochemical signal transmissions along nerve cells and neuromuscular junctions. AChE is prevalent in a variety of cell and organ types throughout the body of vertebrates and invertebrates (Walker 1991). Interference of normal nerve transmission by N-methyl carbamates may affect a wide array of physiological systems in animals. Organophosphate pesticides share this mode of action and physiological responses are similar.

The mechanism of action of N-methyl carbamates, inhibition of AChE, involves a series of enzyme-mediated reactions. Briefly, in a reversible reaction carbamates bind to AChE, thereby inhibiting AChE’s normal activity to hydrolyze the neurotransmitter acetylcholine at nerve synapses. This reaction is similar to organophosphorus insecticides with the main exception being a carbamylation of AChE instead of a phosphorylation. Carbamate inhibition of AChE is “reversible” in cases of sublethal exposure and recovery of N-methyl carbamate-inhibited AChE is typically rapid compared to OP-inhibited AChE. The key result of AChE inhibition by carbamate and OP insecticides is accumulation of acetylcholine in a cholinergic synapse. The buildup of acetylcholine causes continuous nerve firing and eventual failure of nerve impulse propagation. A variety of adverse effects to organisms can result ranging from sublethal behavioral effects to death (Mineau 1991b).

Incidences of acute poisoning from AChE inhibitors are prevalent for wildlife, particularly for birds and fish (Mineau 1991b). The following passage describes the classic signs of AChE-inhibiting insecticide poisonings of fish:

> *“Fish initially change normal swimming behavior to rapid darting about with loss of balance. This hyper excitability is accompanied by sharp tremors which shake the entire fish. The pectoral fins are extended stiffly at right angles from the body instead of showing the usual slow back and forth motion normally used to maintain balance. The gill covers open wide, and opercular movements become more rapid. With death the mouth is open and the gill covers are extended. Hemorrhaging appears around the pectoral girdle and base of the fins (Weiss 1957).”*

Numerous reports, peer-reviewed journal articles (Antwi 1985; Coppage 1974; Haines 1981; Holland 1967; Rabeni 1975; Williams 1966) as well as multiple reviews, text books ((Geisy 1999; Mineau 1991b; Smith 1993), and wildlife poisoning cases document inhibition of AChE activity in exposed invertebrates (Detra 1986; Detra 1991) and vertebrates including salmonids following exposures to carbamates and OPs (Beyers and Sikoski 1994; Grange 2002; Hoy 1991;

Laetz et al. 2009; Li et al. 2008; Li 1996; Sandahl 2004; Sandahl 2005; Scholz 2000; Scholz et al. 2006; St. Aubin 2004; Tierney 2007; Zinkl 1987).

One study relevant to our effects analysis measured inhibition of brain AChE, duration of recovery, survival at 24 h, and tissue concentrations in juvenile rainbow trout (*O. mykiss)* following exposure to carbaryl at 0, 250, 500, 1,000, 2,000, and 4,000 µg/L (Zinkl 1987). Rainbow trout showed dose-dependent AChE inhibition from 61 to 91% when exposed to 250 – 4,000 µg/L for 24 h. Most trout that died had 85% or greater inhibition. Trout recovered AChE activity following 24 h in uncontaminated water, indicating that fish recover if given the opportunity following carbamate exposures (Zinkl 1987). This study showed that carbaryl is acutely toxic to rainbow trout in a matter of hours (incidences of death at 1.5 – 4 h) at concentrations at or above 1,000 µg/L.

10.3.1.2 Degradates

For carbaryl, 3 major degradates (1-naphthol, 1, 4 napthoquinone, and carbon dioxide) were detected in various environmental fate studies. According to EPA's BE, these degradates are not considered to be of toxicological concern because they do not contain a carbamate functional group. Additionally, 1-naphthol can also be generated by a variety of natural and anthropogenic processes, including the breakdown of the polycyclic aromatic hydrocarbon (PAH) naphthalene.

Acute fish and invertebrate toxicity data is available for 1-napthol, the principal hydrolysis degradate of carbaryl. LC50s presented for aquatic invertebrates include 700-730 ug/L for *D. magna* (freshwater organism), 200-210 ug/L for *M. bahia* (estuarine organism) and 2,100 ug/L for *C. virginica* (estuarine organism). Another study compared acute lethalities between carbaryl and 1-naphthol in 2 species of fish (Shea and Berry 1983). In goldfish (*Carassius auratus*) and killifish (*Fundulus heteroclitus*), 1-naphtol was significantly more toxic than carbaryl based on 10-day acute lethality tests (Shea and Berry 1983). The degradate 1-naphthol was approximately 5 times more toxic than carbaryl in goldfish, and in killifish twice as toxic as carbaryl (Shea and Berry 1983). Additionally, fish exposed to 1-naphthol showed neurological trauma including pronounced erratic swimming behaviors and increased opercula beats following exposure to 5 and 10 mg/L. None of these symptoms were observed in the carbaryl treatments (Shea and Berry 1983).

For methomyl, 4 major degradates (i.e., methomyl oxime, acetonitrile, acetamide and CO2) were detected in various environmental fate studies. According to EPA's BE, these degradates are not considered to be of toxicological concern because they do not contain a N-methylcarbamate functional group. EPA's 1998 Reregistration Eligibility Decision (RED) (EPA 1998) describes a study evaluating the degradate thiolacetohyroxamic acid 5-methyl ester, which was tested and found to be practically nontoxic to bluegill. The bluegill LC50 for this degradate is 462,000 µg/L. No other degradate toxicity data were presented.

10.3.1.3 Adjuvant Toxicity

Although no data were provided in the BEs related to adjuvant toxicity, an abundance of toxicity information is available on the effects of the alkylphenol polyethoxylates, a family of non-ionic surfactants used extensively in combination with pesticides as dispersing agents, detergents, emulsifiers, adjuvants, and solubilizers (Xie et al. 2005). Two types of alkylphenol polyethoxylates, NP ethoxylates and octylphenol ethoxylates, degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates, octylphenol and NP, respectively. We note that the technical registrant of methomyl stated that no nonylphenol ethoxylates are used within methomyl formulations. We did not receive information on the presence or absence of alkylphenolpolyethoxylates in carbaryl- or carbofuran-containing formulations. Adjuvants are frequently mixed with formulations prior to applications so, although they may not be present in the formulations, they could still be co-applied. Below we discuss NP's toxicity as an example of potential adjuvant toxicity, as we received no information on adjuvant use or toxicity within the BEs.

We queried EPA's ECOTOX online database and retrieved 707 records of NP's acute toxicity to freshwater and saltwater species. The lowest reported LC50 for a salmonid was 130 ug/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of NP, lowest reported LC50 = 1 ug/L for *H. azteca*. These data indicate that a wide array of aquatic species are killed by NP at ug/L concentrations. We also queried EPA's ECOTOX database for sublethal toxicity and retrieved 689 records of freshwater and saltwater species tested in chronic experiments. The lowest fish LOEC reported was 0.15 ug/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 ug/L. Additionally, salmonid prey species are sensitive to sublethal effects of NP. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation (Brown et al. 1999). Multiple studies with fish indicated that NP disrupts fish endocrine systems by mimicking the female hormone 17-B-estradiol (Arsenault et al. 2004; Brown and Fairchild 2003; Hutchinson et al. 2006; Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005; Segner 2005). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 ug/L (Arukwe and Roe 2008; Hemmer et al. 2002; Ishibashi et al. 2006; Schoenfuss et al. 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol; however, immature male fish have the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Brown and Fairchild 2003; Fairchild et al. 1999). Additionally, processes involved in seawater adaptation of salmonid smolts are impaired by NP (Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005).

These results demonstrate NP is of concern to aquatic life, particularly salmonid endocrine systems involved in reproduction and smoltification. This summary is for one of the more than 4,000 inerts/other ingredients and adjuvants currently registered for use in pesticide formulations and there are likely others with equally deleterious effects. Unfortunately, the effects that these other ingredients may have on listed species and designated critical habitat remain uncertain.

10.3.1.4 Types of Information Presented in the BEs

Toxicity data available for carbaryl and methomyl were reviewed and divided into major taxonomic groups, including: fish and aquatic amphibians, aquatic invertebrates, aquatic plants, birds, reptiles, terrestrial-phase amphibians, mammals, terrestrial invertebrates, and terrestrial plants. For each of these groups, endpoints are determined for each taxon for mortality (animals only) and sublethal effects (i.e., growth or reproduction). These endpoints are used to establish thresholds, which are then used in conjunction with exposure data to make effects determinations based on the taxon with which they are associated. Each BE primarily summarized acute and chronic toxicity data from “standardized toxicity tests” submitted by pesticide registrants during the registration process, tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally included aspects of survival (death), reproduction, and growth measured in laboratory dose-response experiments (EPA 2004). Survival is measured in both acute and chronic tests. Reproduction and growth are generally measured and reported in the chronic tests. For this opinion, NMFS translates effects to individuals of species into potential population level consequences as described in the Assessment Framework chapter.

Survival of individual fish is typically measured by incidences of death following 96-h exposures (acute test) and incidences of death following 21-day, 30-day, 32-day, and “full life cycle” exposures (chronic tests) to a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, etc.) (EPA 2004). Lethality of the pesticide is usually reported as the median lethal concentration (LC50), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as an EC50, because death of these organisms may be difficult to detect and immobilization is considered a terminal endpoint. An LC50 is derived from the number of surviving individuals at each concentration tested following a 96-h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC50s are typically calculated by probit analysis. If the data are not normally distributed for a probit analysis, than either a moving average or binomial is used, resulting in no slope being reported. Ideally, to maximize the utility of a given LC50 study, a slope, variability around the LC50, and a description of the experimental design such as experimental concentrations tested, number of treatments and replicates used, and solvent controls are needed. The slope of the observed dose-response relationship is particularly useful in interpolating incidences of death at concentrations below or above an estimated LC50. The variability of an LC50 is usually depicted by a confidence interval (95% CI) or standard deviation/error and is illustrative of the degree of confidence associated with a given LC50 estimate, i.e., the smaller the range of uncertainty the higher the confidence in the estimate. Without an estimate of variability, it is difficult to infer the precision of the estimate. Furthermore, survival experiments are of most utility when conducted with the most sensitive life stage of the listed species or a representative surrogate.

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of impacted growth derived from a guideline study to fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. What is generally assessed is size or weight of fish measured at several times during an experiment. The test fish are usually fed twice daily, *ad libitum*, i.e., an overabundance of food is available to the fish. Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected.

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the scale of a population by measuring the number of offspring per females in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic freshwater fish experiments where hatchability and juvenile and larval survival are measured. NMFS considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels. These endpoints are not generally measured in standardized toxicity assays used in pesticide registration.

10.3.1.5 Impacts to Prey

One uncertainty in the effects analysis is the degree to which secondary poisoning of listed-species may occur from feeding on contaminated prey (e.g., dead and dying drifting insects). Secondary poisoning is a frequent occurrence with OPs and carbamates in bird deaths (Mineau 1991a), yet is much less studied in fish. Uptake, metabolism, and accumulation of carbaryl by a salmonid prey item, *Chironomus riparius* (midge), exposed for 24 h indicated significant uptake over the first 8 h, significant metabolism (more than 85-99%) of parent carbaryl to metabolites, and low bioconcentration factors (5-10) (Lohner and Fisher 1990). These results suggest that contaminated prey items, such as aquatic invertebrates, do not accumulate significant carbaryl, and what they do accumulate is likely rapidly metabolized. That said, listed fish species could still get a dose of carbaryl from feeding on drifting, contaminated insects that have not had time to metabolize carbaryl. Juvenile brook trout gorged on drifting insects following applications of carbaryl, and AChE activity was reduced (15-34%) in the trout (Haines 1981). However, it is not possible to differentiate the contribution to AChE inhibition from the aqueous and dietary routes because concentrations were not measured in the water, prey, or fish. In another study, resident brook trout feeding on dying and dead drifting invertebrates (from the pyrethroid cypermethrin) exhibited a range of physiological symptoms: loss of self-righting ability and startle response; lethargy; hardening and haemolysis of muscular tissue similar to muscle tetany; and anemic appearance of blood and gills (Davies and Cook 1993). The possibility that the adverse effects in the trout manifested from exposure to the water column instead of from feeding on contaminated prey was ruled out by the authors as measured field concentrations of pesticides did not produce known toxic responses. In a laboratory feeding study with the OP fenitrothion, brook trout (*S. fontinalis*) were fed contaminated pellets (1 or 10 mg/g fenitrothion for 4 wks) (Wildish and Lister 1973). Growth was reduced in both treatments. AChE inhibition was measured at 2, 12, and 27 d following termination of contaminated diet treatments. Trout had lower AChE activity than unexposed fish at both treatments, and by 27 d following termination, contaminated diet-induced AChE levels regained some of their activity. The treatment concentrations used in this study are very high and indicate that brook trout are not sensitive to diet-induced toxicity of fenitrothion. The experiment did show that AChE inhibition from the diet is possible, yet it is difficult to determine the relative toxicity of carbaryl and methomyl found in contaminated prey consumed by listed-species.

Anticholinesterase insecticides also reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities (Chang et al. 2005; Fleeger et al. 2003; Liess and Schulz 1999; Relyea 2005; Schulz 2004; Schulz and Liess 1999). Spray drift and runoff from agricultural and urban areas can expose aquatic invertebrates to relatively low concentrations of insecticides for as little as minutes or hours, but populations of many taxa can take months or even years to recover to pre-exposure or reference densities (Anderson et al. 2003; Liess and Schulz 1999; Stark et al. 2004; Wallace et al. 1991). For example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an acetylcholinesterase inhibitor) and fenvalerate (another commonly used insecticide), 8 of 11 abundant species disappeared and the remaining 3 were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in invertebrate prey availability. Therefore, in addition to the direct impacts that acetylcholinesterase inhibitors have on listed species, there may also be, independently, significant indirect effects to individuals via their prey (Peterson et al. 2001). For example, wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991; Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. Davies and Cook (1993) found that, several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches consumed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with acetylcholinesterase inhibitors (Liess and Schulz 1999), suggesting indirect effects to salmon via prey availability may be similar.

One likely biological consequence of reduced feeding, foraging, and prey availability is a reduction in food uptake and, subsequently, a reduction in somatic growth of exposed fish. For example, juvenile growth is a critical determinant of freshwater and marine survival for Chinook salmon (Higgs et al. 1995). Reductions in the somatic growth rate of salmon fry and smolts are believed to result in increased size-dependent mortality (Healey 1982; West and Larkin 1987; Zabel and Achord 2004). Zabel and Achord (2004) observed size-dependent survival for juvenile salmon during the freshwater phase of their outmigration. Mortality is also higher among smaller and slower growing salmon because they are more susceptible to predation during their first winter (Beamish and Mahnken 2001; Healey 1982; Holtby et al. 1990). These studies suggest that factors affecting the organism and reducing somatic growth, such as anticholinesterase insecticide exposure, could result in decreased first-year survival. Finally, a population model has linked reductions in juvenile survival due to reductions in prey abundance following short-term exposures to three anticholinesterase pesticides, one being carbaryl, to reductions in population productivity (Macneale et al. 2014).

10.3.2 Carbaryl Response: Fish

The information provided by EPA in their BE for carbaryl addressed aspects of survival, growth and reproduction of aquatic species (freshwater and saltwater), and provided some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. Because NMFS is charged under the ESA to evaluate all effects of an action, we therefore evaluate all aspects that may reduce fitness of individuals or reduce the conservation value of PBFs of designated critical habitat. Our evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broad range of endpoints including behaviors, endocrine disruption, and other physiological alterations. The information we assessed is derived from published, scientific journals and information from government agency reports, theses, books, information and data provided by the registrants identified as applicants, and independent reports.

10.3.2.1 AChE

Carbaryl is a *N*-methyl carbamate with a mode and mechanism of AChE inhibition that has been shown in many types of organisms including fish. NMFS searched the BE, ECOTOX, and open literature for studies on AChE inhibition in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. In total, 29 studies were collected, representing 36 AChE endpoints (LOECs, LC50s, and IC50s) and 29 different fish species. Response values ranged from 0.0006 mg/L (Mdegela et al. 2010) to 12.25 mg/L (Pereira et al. 2019). The majority of these endpoints were acute, in vitro exposures of brain and/or muscle homogenate.

Data displayed on Risk-plots

A single dose-response bar was plotted on the Risk-plot. The IC50 (0.423 mg/L) for this dose-response bar was constructed by taking the geometric mean of 8 in vivo IC50 values from 6 different studies (Ferrari et al. 2004a; Ferrari et al. 2004b; Ferrari et al. 2007; Labenia et al. 2007; Laetz et al. 2009; Troiano and Grue 2016). IC50 values ranged from 0.019 to 2.6 mg/L for 5 different fish species, both freshwater and estuarine/marine. The majority of the studies had exposure durations of 96-hrs and were on juvenile fish. The lowest IC50 value of 0.019 mg/L was for AChE inhibition of the brain in juvenile rainbow trout (Ferrari et al. 2004b; Ferrari et al. 2007). The lowest IC50 value representing AChE inhibition of the brain for an estuarine/marine fish was in juvenile coho salmon, with a value of 0.1458 mg/L (Laetz et al. 2009). One study exposed 2-year old white sturgeon to a formulated carbaryl product with 80% purity for a duration of 6-hrs and reported a IC50 value of 1.751 mg/L for AChE brain inhibition (Troiano and Grue 2016). The dose-response bar displayed on the plot was calculated using a logit equation with an IC50 of 0.423 mg/L and a slope of 0.81 (Laetz et al. 2009).

10.3.2.2 Fish Mortality

According to EPA's carbaryl BE, the acute mortality studies conducted with technical grade carbaryl were used to derive a SSD, including toxicity data for all fish exposed to carbaryl. Five distributions were tested, and a variety of methods were used. The triangular distribution and maximum likelihood method were ultimately chosen to represent HC05 through HC95 values for freshwater and estuarine/marine fish. Acute toxicity estimates (96-hour LC50s) for carbaryl range from 0.14 - 1188 mg/L and span 4 orders of magnitude, indicating a wide range of sensitivity to carbaryl among fish. The lowest LC50 for carbaryl is for TGAI (Technical Grade Active Ingredient) tested on *Ictalurus punctatus* (LC50 = 0.14 mg/L; E5722). Toxicity data for carbaryl when tested as a formulated product are also available. The most sensitive endpoint for the formulated product was an LC50 value of 0.44 mg/L (rainbow trout; E112236). Additional mortality endpoints reported as “survival” and “hatch” in ECOTOX are available for fish. Reported toxicity endpoint concentrations associated with these studies ranged from 0.36 mg/L (threshold concentration as estimated by linear-plateau regression; E13270) to 10 mg/L (NOAEL/No LOAEL; E112238).

NMFS found 20 additional LC50 values, not included in EPA’s BE, during the search for new data. NMFS assessed whether the new data would influence the analyses by adding the new data to the existing data. New geometric means were calculated for each species and HC05s were estimated by fitting the data with a log-triangle distribution. The resulting new HC05 was within the confidence limits of the respective HC05 provided in EPA’s BE. NMFS chose to continue to use the HC05 provided by EPA. However, we did take into consideration these additional data when determining the level of confidence we had in the SSD. In this case, our confidence increased due to the fact that the new LC50 observations fell reasonably within the existing SSD.

Data displayed on Risk-plots

A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 1.0554 mg/L (HC05 value from the “Fish Only” SSD) and a slope of 4.5 (assumed).

10.3.2.3 Invertebrate Prey Availability

According to EPA’s carbaryl BE, the aquatic invertebrate mortality endpoint used to derive a threshold for direct and indirect effects is based on the HC05 value from the SSD for the taxon. SSDs were generated for mollusk and non-mollusk aquatic invertebrates separately, with freshwater and estuarine/marine species pooled together in both groups. SSDs were based on acute 48 and 96-hr LC50 values from studies using TGAI only (LC50 values from formulation/mixture testing were not included), usually using juvenile invertebrates. There were 25 orders and 53 species of non-mollusk invertebrates and 5 orders and 15 species of mollusk invertebrates used in the SSD. The HC05 value is 1.6 µg/L for non-mollusk aquatic invertebrates and 6600 µg/L for mollusks. For freshwater aquatic invertebrates (non-mollusks), the most sensitive mortality endpoint reported is 0.0001 mg/L based on 100% mortality of the common prawn (pink shrimp; *Palaemon serratus*; E151760). However, there was not enough information available in the study to use the endpoint quantitatively. The highest mortality-based endpoint for non-mollusks is 3000 mg/L (LOAEC for hatching) for trematodes (E87871). This value exceeds the solubility of carbaryl in water of 32 mg/L at 32 ºC (Suntio et al., 1988). When toxicity endpoints exceed the water solubility, the endpoint may overestimate the exposure concentration in the study and additional analysis is needed to see if that study is reliable. For mollusks, the most sensitive mortality-based endpoint is an LD50 value of 0.35 mg/L for the pond snail (*Lymnaea acuminate*; E65606). The highest mortality-based endpoint for mollusks is 100 mg/L (which exceeds the water solubility) and is based on 100% mortality for a snail species in the Neogastropoda order (E74591). For mollusks, acute LC50 values range from 3.08 to 67.01 mg/L (exceeding solubility). For non-mollusks, the reported mortality data for carbaryl encompass a wide range of toxicity values from acute LC50 values of 0.00066 to 100 mg/L, which exceeds the water solubility of carbaryl. Immobility in aquatic invertebrates is often used as a surrogate for mortality, and effects ranged from 0.00066 to >18 mg/L, with most endpoints representing water flea, midge, caddisfly and Ostracod shrimp toxicity. Additional mortality endpoints reported as "survival", "lifespan", and "hatch" in ECOTOX are available for aquatic invertebrates for a variety of species such as shrimp, daphnia, lobsters, and crayfish. Concentrations associated with the reported endpoints ranged from 0.00072/0.00072 mg/L (NOAEL/LOAEL for survival; E114283) reported for *Daphnia magna* to 4.7 mg/L for the purple spined sea urchin (*Arbacia punctulata*; EC50 for survival; E115739).

NMFS found 2 additional LC50 values during the search for new data. While NMFS did not perform a thorough review of the new toxicity data, NMFS did assess whether the new data would influence the analyses. The new toxicity data was added to the existing data, new geometric means were calculated for each species and HC05s were estimated by fitting the data with a log-logistic distribution. The resulting new HC05 was within the confidence limits of the respective HC05 provided in EPA's BE. NMFS, therefore, chose to continue to use the HC05 provided by EPA.

Data displayed on Risk-plots

A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.005 mg/L (HC10 value from the "All non-mollusk aquatic invertebrates" SSD) and a slope of 4.5 (assumed). In our conference and biological opinion, we present effects to invertebrates that serve as prey for our listed species by plotting a single dose-response bar on our risk plots. The dose-response bar was calculated using a probit equation with an assumed slope of 4.5 and an LC50 representing the HC10 from the SSDs presented in EPA's BEs, specifically the "non-mollusk aquatic invertebrates" SSDs. EPA's invertebrate SSDs are, by nature, considerate of various sensitivities across invertebrate species. For example, the carbaryl non-mollusk SSD considered 25 orders and 53 species of invertebrates. NMFS selected the HC10 from the invertebrate SSD to use in our effects analysis using best professional judgement. NMFS's listed fish (e.g., salmonids, sturgeon, and sawfish) species consume many types of food, so if one species of invertebrate is more affected by a carbamate pesticide than another, our listed species will have other food options available to them despite the loss of certain prey items. The ecological importance of any one invertebrate species will depend on its value as food to the listed fish species and their abundance in the environment. While they can switch food, both an overall decline in prey biomass (Cuffney et al. 1984; Macneale et al. 2010) and long recovery times for invertebrate communities (Liess and Schulz 1999; Macneale et al. 2014) will still be ecologically relevant to their food availability. Because it is not possible within the scope of this conference and biological opinion to assess effects to all invertebrate species, and, because the presence and relative abundances of the SSD invertebrate species are not known, we selected the HC10 as a metric to be protective of 90% of the invertebrate prey species that our listed species could potentially consume in both fresh and marine water.

10.3.2.4 Growth

EPA's carbaryl BE did not report any effects on growth for estuarine/marine fish species, so the growth data for freshwater fish were used as a surrogate. The carbaryl BE reported endpoint values for growth in fish (and aquatic-phase amphibians) ranging from 0.25 to 9.99 mg/L, spanning 15-fold differences in carbaryl-mediated effects on the growth of fish. There were 2 studies that reported low growth-related endpoint values for carbaryl. In the first study, a NOAEC value of 0.25 mg/L was reported based on dry weight (LOAEC of 0.50 mg/L) reductions in 4-day old fathead minnow larvae (*Pimephales promelas*) exposed to 99.8% pure carbaryl for 7 days (E16510). In the other study, an IC25 of 0.25 mg/L from a 7-day study was based on the reduction of biomass (using an inhibition concentration methodology) for bonytail freshwater fish following exposure to 99.7% TGAI (E93091). However, both studies contained limitations (including limited information on methodology, nonstandard endpoint analysis, and lack of raw data), and therefore were not considered as the growth endpoint threshold for freshwater fish. The least sensitive growth-related endpoint (NOAEL of 9.99 mg/L) was for general developmental changes in freshwater zebrafish (*Danio rerio*) exposed for 4 days to 99.9% TGAI (E109343). In these studies, the tested species belonged to the same fish order (i.e., Cypriniformes), suggesting that fish species within the same order could potentially display different sensitivities to carbaryl TGAI with regards to growth-related effects. There was another study that evaluated growth effects of carbaryl to freshwater fish (Carlson 1972, MRID 40644801, E5073), which was classified as acceptable for quantitative use. This was a study in which chronic exposure of fathead minnows to carbaryl for 9 months resulted in no effects on growth, but rather reduced survival (27.5% reduction) and fecundity (98.5% reduction in eggs/mature female; 92.4% reduction in eggs/spawning) at 0.68 mg/L.

NMFS searched the BE, ECOTOX, and open literature for studies on growth endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS evaluation of the action. In total, 9 different studies were considered which reported a variety of growth-related endpoints for 7 different fish species. The most sensitive endpoint was a LOEC representing reduced dry weight in fathead minnow larvae at a concentration of 0.2 mg/L (Pickering et al. 1996). The highest endpoint concentration was 4.05 mg/L representing a LOEL for reduced body weight of the walking catfish after a 25.6-day exposure to 99% carbaryl (Lata et al. 2001).

Data displayed on Risk-plots

NMFS reviewed the available growth studies and determined that 6 contained endpoints that were appropriate for displaying on the risk plot (Arunachalam and Palanichamy 1982; Dwyer et al. 2005b; Flynn et al. 2022; Lata et al. 2001; Norberg-King 1989; Pickering et al. 1996). Endpoints included: IC25s, IC50s, LOELs, NOECs and LOECs, and ranged in concentration from 0.2 mg/L to 4.05 mg/L. Exposure durations included: 7, 12, 25.6, 26, 28, and 32-day exposures.

10.3.2.5 Reproduction

EPA's carbaryl BE reported 3 available studies that evaluated reproductive effects to freshwater fish for carbaryl. No reproduction studies were available for estuarine/marine fish, so the reproduction data for freshwater fish were used as a surrogate. The study with the lowest reproductive endpoint was the same study as discussed above for growth effects. In this study, chronic exposure of fathead minnows (*Pimephales promelas*) to carbaryl resulted in reduced reproductive effects (NOEC = 0.21 mg/L; LOAEC 0.68 mg/L; and the calculated MATC was 0.378 mg a.i./L.) including reduced number of eggs per female and reduced number of eggs spawned (Carlson 1972). In the other 2 studies, decreases in hatching were reported at 1.7 mg/L (E162695), whereas, in the other study, no effects in fecundity or fertility were observed up to 0.82 mg/L (MRID 48669601).

NMFS searched the BE, ECOTOX, and open literature for studies on reproductive endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. In total, 5 different studies were considered with endpoint concentrations ranging from 0.68 mg/L to 7.915 mg/L.

Data displayed on Risk-plots
NMFS reviewed the available reproduction studies and determined that 3 contained endpoints that were appropriate for displaying on the risk plot. For the purpose of this assessment, reproductive endpoints were further classified based on the life-stage of the chemical exposure. For example, the endpoint hatch-success would be considered a reproductive endpoint if it were measured in the offspring of an exposed parent. In contrast, hatch-success would be considered a developmental endpoint if the unhatched eggs themselves were exposed. In total, 3 reproductive endpoints were displayed on the Risk-plot. The first endpoint is a LOEC of 0.68 mg/L which comes from a study in the BE described above (Carlson 1972). The second endpoint was a LOEC of 1.0 mg/L representing a 2-week delay in spawning time and a 39% reduction in the number of eggs produced by fathead minnow after a 28-day exposure to 98% pure carbaryl (Flynn et al. 2022). The third study reported effects on ovarian function, the E2 hormone, and the Gonadosomantic Index in adult freshwater perch at a concentration of 7.915 µmg/L after 15 days and beyond of a 90-day total exposure to a product containing 50% carbaryl (Choudhury et al. 1993).

10.3.2.6 Behavior

NMFS considered impacts of carbaryl to behavioral endpoints in fish. NMFS searched the BE, ECOTOX, and open literature for studies on behavioral endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. In total, 13 different studies were considered, representing 9 different species of both freshwater and estuarine/marine fish. Endpoints included: L-serine-evoked EOG, predator vulnerability, altered swimming/impaired orientation, and chemical avoidance, hyperactivity, burrowing behavior, surfacing behavior, and decreased feeding rate. The most sensitive endpoint was a LOAEL at a concentration of 0.1 mg/L representing a 49.7% decrease in L-serine-evoked EOG in sockeye salmon, rainbow trout, and coho salmon after exposure to 99.8% pure carbaryl for 30 minutes (Tierney 2007).

Data displayed on Risk-plots

NMFS reviewed the available behavioral studies and determined that 8 were appropriate for displaying on the risk plot. In total, 11 behavioral endpoints are plotted on the Risk-plot (Arunachalam and Palanichamy 1982; Beauvais et al. 2001; Hansen et al. 1972; Hussain et al. 2020; James and Sampath 1994; Labenia et al. 2007; Pozarycki 1999; Tierney et al. 2007). Endpoint concentrations ranged from 0.1 mg/L to 10.0 mg/L.

10.3.2.7 Development

NMFS searched the BE, ECOTOX, and open literature for studies on developmental endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. For the purpose of this assessment, endpoints were considered to be "developmental" when exposures to test species were made pre-hatch. In total, 12 different studies were considered. These studies reported developmental effects to 7 different species of both freshwater and estuarine/marine fish. Endpoints included: gastrulation, spontaneous movement, histopathological observations, heart morphology, spinal cord deformations, etc. Endpoint concentrations ranged from 0.003 mg/L to 11.69 mg/L. The most sensitive endpoint for freshwater fish was an EC50 of 3 µg/L representing impacts to gastrulation, spontaneous movement, and extension of the tail in zebrafish (Schulte and Nagel 1994). The most sensitive endpoint for estuarine/marine fish was a LOEL of 1000 µg/L representing heart deformation in mummichog (*Fundulus heteroclitus*) after exposure to 100% pure carbaryl for 5-days (Clark and Di Giulio 2012).

Data displayed on Risk-plots

Developmental endpoints were not displayed on the Risk-plots. Instead, these types of endpoints were evaluated qualitatively when considering the available evidence in support of particular risk hypotheses. For example, endpoints such as heart deformities, low weight, and decreased locomotion may translate into fitness-level impacts in later life stages (e.g., impacts to growth). However significant uncertainties exist. NMFS determined a qualitative approach to these endpoints was appropriate as a number of the endpoints captured in our search have uncertain connections to fitness (e.g., spontaneous tail movement in embryos). Additionally, the degree to which individuals may recover from or compensate for the early developmental impacts is uncertain.

10.3.3 Carbaryl Response: Coral

Corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion. In addition, corals have associated microbiota comprised of a diverse consortium of organisms, including bacteria, which reside in the surficial mucus layer and are important for coral health, but remain poorly understood. Corals obtain nutrition through 2 primary pathways: sunlight (via the symbiotic dinoflagellate); and zooplankton and other suspended particulate matter (via the polyps). In evaluating the potential responses of coral to carbaryl, NMFS considered direct mortality to both the polyp and algal symbiont; sublethal impacts (e.g., impacts to metamorphosis or settlement); as well as prey availability (e.g., zooplankton).

Few studies have evaluated the effects of carbamate insecticides on corals (Acevedo 1991; Markey et al. 2007). Greater attention has been given to the impact of herbicides, which have been demonstrated to negatively affect coral algal symbionts (Jones 2005). For example, diuron exposures at concentration as low as 10 ug/L have been shown to cause photoinhibition in the coral species, *Pocillopora damicornis*, resulting in bleaching followed by full colony mortality (Cantin et al. 2007). The impact of insecticides on corals has received somewhat less attention, however studies that do exist suggest that insecticides, including AChE inhibitors (e.g., organophosphates, carbamates) have negative impacts on larval settlement, metamorphosis, and survival (Flores et al. 2020; Markey et al. 2007; Ross et al. 2015), among other adverse effects. This is not surprising, as post-fertilized coral contain acetylcholine, the target of action for AChE inhibitors such as carbaryl and methomyl (Talesa et al. 1992).

Carbaryl and methomyl both exhibit phytotoxicity, in addition to being toxic to animals via the inhibition of AChE. It can be assumed therefore that both compounds will induce some adverse impacts to coral algal symbionts. The concentration at which adverse impacts are anticipated is uncertain. In regards to the algal symbiont, coral-based studies have shown significant impacts at environmentally relevant concentrations of AChE inhibitor profenofos, causing bleaching (i.e., reductions in dinoflagellate density) at 10 ug/L (Markey et al. 2007).

Adverse effects to the larval stage of polyps have also been observed at environmentally relevant concentrations. Markey et al. (2007) observed 60 to 100% reductions in larval settlement at 0.3 ug/L chlorpyrifos and profenofos, and 3 ug/L carbaryl after an 18-hour exposure. Fertilization was not impacted in this study at the highest concentration tested 30 ug/L. In another study, Ross et al. (2015) observed significant decreases in larval survival at 2.96 ug/L naled following an 18-20-hour exposure. In this study settlement and zooxanthellae density were not impacted at the highest concentration tested 10 ug/L. One study (Acevedo 1991) observed impacts occurring at much higher concentrations in carbaryl. In this study, larval coral exhibited no noticeable adverse effects at concentrations up to 10,000 ug/L carbaryl. This same study reported impacts to larval swimming at the lowest concentration tested of chlorpyrifos: 10 ug/L, and 100% mortality at 1000 ug/L. NMFS did not find any studies evaluating the effects of insecticides on developed coral colonies; however colony-level impacts have been observed as a result of herbicide exposure via adverse impacts to the algal symbiont (Flores et al. 2020; Nalley et al. 2021). Studies described above demonstrate the ability of insecticides to also impact algal symbionts; colony level impacts from insecticides is therefore plausible.

10.3.3.1 Polyp Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for carbaryl, we determined that aquatic invertebrates were the closest available surrogate for coral polyps. Note that surrogate uncertainty is considered when making effect of exposure determinations. For coral polyp mortality we considered the information described above under the section "Invertebrate Prey" (Section 10.3.2.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.0016 mg/L (HC05 value from the "All non-mollusk aquatic invertebrates" SSD) and a slope of 4.5 (assumed).

10.3.3.2 Algal Symbiont Mortality (surrogate: algae)

In evaluating the available data for carbaryl, we determined that algae were the closest available surrogate for coral algal symbionts. EPA's carbaryl BE reported 11 available studies that evaluated effects to algae for carbaryl. Endpoints included: biomass, photosynthesis, growth rate, density, etc. Effect concentrations ranged from 0.005 mg/L to 37.9 mg/L. Species tested included green algae, blue-green algae, and diatom.

Data displayed on Risk-plots
NMFS reviewed the available studies and determined that 6 contained endpoints that were appropriate for displaying on the risk plot. These studies reported IC50 or EC50 values. A single dose-response bar was plotted on the Risk-plot. The IC50 (1.55 mg/L) for this dose-response bar was constructed by taking the geometric mean of 22 values from these 6 studies. IC50 values ranged from 0.34 mg/L (Brooke 1993) to 6.1 mg/L (Ma et al. 2006). The dose-response bar displayed on the plot was calculated using a probit equation with an IC50 of 1.55 mg/L and a slope of 4.5 (assumed).

10.3.3.3 Carbaryl-specific Studies on Coral

NMFS found 2 studies that evaluating the effects of carbaryl on coral (Acevedo 1991; Markey et al. 2007). The first study observed a 96-hour larval mortality NOEC and LOEC of 10 mg/L and 100 mg/L respectively. The other study observed 60 to 100% reductions in larval settlement at 0.003 mg/L carbaryl after an 18-hour exposure.

10.3.3.4 Zooplankton Prey (surrogate: aquatic invertebrates)

In evaluating the available data for carbaryl, we determined that aquatic invertebrates were the closest available surrogate for marine zooplankton. For coral prey availability we considered the information described above under the section "Invertebrate Prey" (section 10.3.2.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.005 mg/L (HC10 value from the "All non-mollusk aquatic invertebrates" SSD) and a slope of 4.5 (assumed).

10.3.4 Carbaryl Response: Abalone and Conch

Abalone

Black abalone is a large marine gastropod mollusk belonging to the taxonomic genus of Haliotidae, a group of sea snails with convex spiral structured shells. Black abalone have separate sexes and are broadcast spawners. As spawning occurs, gametes are dispersed from the gonads of both parents into the sea and fertilization is entirely external. The embryos and larvae that result from this process are small and unprotected, obtain no parental care or safeguard of any kind, and are exposed to a wide range of physical and biological sources of mortality. The average life expectancy for an abalone that reaches adulthood is 30 years. Adults attain a maximum shell length of approximately 200 millimeters (indexed by linear measure of the maximum diameter of the elliptical shell). Female black abalone become sexually mature at a length of about 50 millimeters, and males at about 40 millimeters (Ault 1985). Ault (1985) projected that sexually mature female black abalone may discharge over 2 million unfertilized eggs per spawning episode and are capable of undergoing multiple episodes each spawning season. Black abalone spawning season is between April and September with peak times occurring during the late summer and early autumn (Leighton 2005 as cited in Butler et al. 2009). Black abalone are most commonly observed in the mid to low intertidal zone, in complex habitats with deep crevices that provide shelter for juvenile recruitment and adult survival. Adults eat different types of algae. They can catch kelp drifting along the seabed or attached to rocks. Black abalone feed on giant kelp and feather boa kelp in southern California (south of Point Conception) habitats, and bull kelp in central and northern California habitats.

The white abalone historically was found in coastal waters between 5-60 meters deep from Point Conception, California to Punta Abreojos, Baja California, Mexico (Cox 1960; Stierhoff et al. 2012). Prior to the fishery collapse, major concentrations of white abalone occurred between 25-30 meters deep (Stierhoff et al. 2012). Since the fishery collapse, the depth distribution of white abalone has shifted toward deeper depths, as most living individuals are those that were too deep to be fished during the 1960s and 1970s (Lafferty et al. 2004). In surveys conducted at an offshore bank from 2002 – 2010 between depths of 30 to 60 m, white abalone were most abundant and dense at depths of 40-50 meters (Stierhoff et al. 2012). The duration of the larval stage is roughly 1 to 2 weeks where they drift in the water current.

Limited information exists on the effects of environmental pollutants and toxins on abalone. Three specific cases of mortality events have been documented after exposure to: 1) sewage discharge in the 1960s; 2) power plant effluent containing copper; and 3) ballast release after the grounding of a vessel (NMFS 2020). The black abalone 2020 recovery plan also notes the potential effects of oil exposure. Effects include injury or mortality and the destruction of other intertidal organisms that black abalone rely upon for settlement cues (e.g., crustose coralline algae), food (e.g., diatoms, macroalgae), and shelter. Martello et al. (2000) observed impacts to the immune system of adult black abalone after a 6.5-hour exposure to 1.2 mg/L pentachlorophenol (a fungicidal wood preservative). In another study (Viant et al. 2001) adult red abalone were exposed to 3 mg/L of the piscicide 3-trifluoromethyl-4-nitrophenol for 5-hours. Exposed abalone exhibited metabolic responses indicative of cellular stress.

10.3.5 Carbaryl Response: Sunflower Sea Star

Sunflower Sea Star

The sunflower sea star (*Pycnopodia helianthoides*) spans the Northeastern Pacific Ocean from the Aleutian Islands to Baja California. They dwell in the low intertidal and subtidal zones to a depth of 435 meters (1,427 feet) but are most common at depths less than 25 meters (82 feet) and rare in waters deeper than 120 meters (394 feet) (Fisher 1928; Gravem et al. 2021; Lambert 2000). Sunflower sea stars are broadcast spawners that require close proximity to mates for successful fertilization (Hodin et al. 2021; Lambert 2000; Lundquist and Botsford 2004; Morris et al. 1980). While individuals are generally considered solitary, documentation of seasonal, patchy distribution suggests that individuals aggregate to spawn (Gravem et al. 2021; Mauzey et al. 1968). Spawning has been reported to occur between November and July, and larval duration may be as short as 7 weeks or as long as 21 (Lowry et al. 2022). The diet of adult *P. helianthoides* generally consists of benthic and mobile epibenthic invertebrates, including sea urchins, snails, crab, sea cucumbers, and other sea stars (Mauzey et al. 1968; Shivji et al. 1983), and appears to be driven largely by prey availability. Larval and pre-metamorphic life stages are planktonic feeders and no data exist to suggest a prey preference.

Few studies have evaluated the effects of carbaryl on echinoderms. Thursby (1991) observed an LC50 of 4,700 ug/L carbaryl after a 2-day exposure. Three studies reported developmental effects, although the variability in effect concentrations is notable. Pesando et al. (2003) observed morphological effects on fertilization and first cleavages of *Paracentrotus lividus* after acute exposure to carbaryl concentrations of 20,122 ug/L. Similar developmental effects were observed at significantly lower concentrations by Hernández et al. (1990). In that study, EC50s associated with developmental cleavage and cellular mobilization of *Pseudecheneis magellanicuswere* observed at concentrations ranging from 6.3 to 157.5 ug/L. Another study (Falk-Petersen et al. 1985) reported an EC50 of 1,500 ug/L after early life stage exposures of *Strongylocentrotus droebachiensis*. No studies evaluating the effects of methomyl on echinoderms were found.

Despite the lack of studies investigating direct effects of carbaryl and the absence of studies investigating direct effects of methomyl on echinoderms, several studies have investigated the impacts of other known AChE inhibitors, the shared mechanism of action of carbaryl and methomyl, on echinoderms. These effects include effects on the reproduction capability of adults, effects on the development of embryos, effects on growth, behavioral effects, effects on enzymes and metabolism at the molecular-level, and mortality.

In a study on black sea urchin (*Tetrapygus niger*) Iannacone et al. (2007) observed IC50s of reduced fertilization success after a 1 hour in vitro exposure of sperm to methaminidophos in 2 formulations, Monofos and Tamaron, at concentrations of 1,423,000 ug/L and 608,000 ug/L respectively. In another study, exposure of *P. lividus* sperm and embryos to dimethoate resulted in alteration in segmentation planes and altered cell division in egg cells fertilized by treated sperm starting at concentrations of 400 ug/L, while the fertilizing ability of the sperm cells were unaffected (Scalisi et al. 2020).

After acute exposure to chlorpyrifos, 90% of cells during development were abnormal in both *Strongylocentrotus droebachiensis* and *Strongylocentrotus purpurarus* sea urchin embryos at concentrations between 876 and 1,752 ug/L (Buznikov et al. 2001). After acute exposure to 14,023 ug/L chlorpyrifos at or just before the mid-blastula 2 stage of development, there were observable malformations in *Lytechinus variegatus* embryos, with the greatest sensitivity observed at the late blastula stage (Buznikov et al. 2007). In another study, the development of *P. lividus* larvae was investigated after the eggs were exposed for 48-hours, and the reduction in percent normal larvae was denoted by a NOEC of 50 ug/L, a LOEC of 100 ug/L, and an EC50 of 350 ug/L (Bellas et al. 2005). After chronic exposure to chlorpyrifos for 15-days, *P. lividus* displayed an increase in metamorphosis at 35 ug/L (Aluigi et al. 2010). In Buono et al. (2012), the impacts of chlorpyrifos and azinphose-methyl, both organophosphate insecticides, on the development of *P. lividus* larvae after an acute exposure were investigated. Exposure to both organophosphates resulted in abnormal sea urchin development; chlorpyrifos resulted in a NOEC of 0.8 ug/L, a LOEC of 1 ug/L, an EC1 of 0.29 ± 0.11 ug/L, and an EC50 of 194.6 ± 16.11 ug/L while azinphose-methyl resulted in a NOEC of 0.7 ug/L, a LOEC of 1.6 ug/L, an EC1 of 0.24 ± 0.11 ug/L, and an EC50 of 141.23 ± 28.18 ug/L. After exposure to eserine, *P. lividus* exhibited a delay in growth and an increase in abnormal larvae at concentrations of 35,059 ug/L (Marchi et al. 1996). Exposure of *P. lividus* at the gastrula stage of development to Basudin, a diazinon formulation, resulted in effects on the speed of development and the length of plutei between 0.0002 to 20 ug/L diazinon (Morale et al. 1998). Exposure to azametiphos resulted in developmental effects on 2 species of urchin; after 48-hours, *P. lividus* showed 66% malformation at a concentration of 1000 ug/L, while after 96-hours, *Sphaerechinus granularis* had 15% of larvae with delayed development at 1 ug/L and 25% of larvae with delayed development at 1000 ug/L (Sanhueza-Guevara et al. 2018).

The percent reduction in *P. lividus* larval growth was denoted by an EC10 of 60 ug/L and an EC50 of 279 ug/L (Bellas et al. 2022), indicating that growth is a more sensitive endpoint than developmental defects in *P. lividus* (Bellas et al. 2005; Bellas et al. 2022). Yao et al. (2010) is the only study, to our knowledge, that investigates the impacts of organophosphate exposure on echinoderm behavior. The authors calculated an acute LC50 of 43,000 ug/L after exposure of *Hemicentrotus pulcherrimus* to monocrotophos, with exposure to a concentration of 5,000 ug/L resulting in 70% of larvae swimming actively, and exposure to a concentration of 30,000 ug/L resulting in 30% of larvae swimming actively. Acute exposure of *H. pulcherrimus* to monocrotophos resulted in a variety of molecular effects; at concentrations as low as 4 ug/L assuming a 40% product, AChE activity was inhibited and its distribution was effected (Zhang et al. 2017a), while at a concentration of 400 ug/L, the expression of HpNetrin and its receptor neogenin were inhibited, which has been linked to developmental defects (Zhang et al. 2017b). Xu et al. (2012) found a 48-hour LC50 of 4,472 ug/L after exposure of *H. pulcherrimus* to monocrotophos, and found metabolic impacts at concentrations as low as 4 ug/L.

Adverse effects resulting from exposure to several other classes of pesticides have also been documented. For example: fungicides (Albutra and Adamat 2015; Hosoya and Mikami 2008; Hosoya et al. 2019; Moschino and Marin 2002; Pagano et al. 2001), herbicides (Bellas et al. 2005; Larrain et al. 1999; Manzo et al. 2006), antifoulants (Bellas 2006; Bellas et al. 2005; Manzo et al. 2006; Moschino and Marin 2002), organochlorines (Beiras and Tato 2018; Bellas et al. 2005; Dinnel et al. 1989; Green et al. 1997; Larrain et al. 1999; Pesando et al. 2004; Stabili and Pagliara 2015), pyrethroids (Erkmen 2015; Gharred et al. 2015), and surfactants (Bellas et al. 2005).

In evaluating the potential responses of the sunflower sea star to carbaryl, NMFS considered direct mortality, developmental effects, and prey availability.

10.3.5.1 Larval Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for carbaryl, we determined that aquatic invertebrates were the closest available surrogate for the larval life-stage of the sunflower sea star. Note that surrogate uncertainty is considered when making effect of exposure determinations. For sunflower sea star larval mortality, we considered the information described above under the section "Invertebrate Prey" (Section 10.3.2.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots

One dose-response bar was plotted on the Risk-plot. The "mortality larvae. HC05" dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.0016 mg/L (HC05 value from the "All non-mollusk aquatic invertebrates" SSD) and a slope of 4.5 (assumed). EPA did not generate a combined SSD (mollusks and non-mollusks). Our decision to use the non-mollusk SSD here was in order to capture the increase in sensitivity to chemical stressors that would be anticipated in the embryo and larval phases of the sunflower sea star, as compared to the adult phase.

The "mortality larvae.ssd" bar represents single point values (the HC05, HC50, HC95) that were extracted from the "All non-mollusk aquatic invertebrates" SSD described above. In addition, we considered the echinoderm EC50 from Thursby (1991) which occurred at 4,700 ppb after a 2-day exposure.

10.3.5.2 Developmental Effects (to echinoderms)

As described in section 10.3.5, the effects of carbaryl on echinoderm development was reported in several studies. Three endpoints were extracted from these studies and displayed on the Risk-plot. First, the most sensitive EC50 of 0.0063 mg/L reported in Hernández et al. (1990) associated with developmental cleavage and cellular mobilization of *Pseudecheneis magellanicuswere* was plotted. An EC50 of 1.5 mg/L after early life stage exposures of *Strongylocentrotus droebachiensis* (Falk-Petersen et al. 1985) was also plotted. Finally, 20.122 mg/L was plotted, which denotes Pesando et al. (2003) observed morphological effects on fertilization and first cleavages of *Paracentrotus lividus* after acute exposure to carbaryl.

10.3.5.3 Prey Availability

In evaluating the available data for carbaryl, we determined that aquatic invertebrates and mollusks were the best representation of the sunflower sea star's prey base. As described in section 10.3.5, sunflower sea stars are generalists. Since EPA did not generate a combined SSD (mollusks and non-mollusks), NMFS decided to plot the non-mollusk SSD as well as the mollusk SSD.

Data displayed on Risk-plots

A single dose-response bar, “prey.invert”, was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.005 mg/L (HC10 value from the “All non-mollusk aquatic invertebrates” SSD) and a slope of 4.5 (assumed). The “prey.mollusk” bar represents single point values (HC05, HC50, and HC95) that were extracted EPA’s mollusk SSD.

10.3.6 Queen conch

The queen conch is a large gastropod mollusk belonging to the same taxonomic group as abalone (Mollusca). Queen conch are slow growing and late to mature, reaching up to 12 inches in length and living up to 30 years. Adult queen conch prefer sandy algal flats, but are also found on gravel, coral rubble, smooth hard coral, and beach rock bottom, while juveniles are primarily associated with seagrass beds. Queen conch have a protracted spawning season of 4 to 9 months, with peak spawning during warmer months. They reproduce through internal fertilization, meaning individuals must be in contact to mate. Females can store fertilized eggs for several weeks, and eggs may be fertilized by multiple males. Egg laying takes 24 to 36 hours, with each egg mass containing about 750,000 eggs. After an incubation period of about 5 days the eggs hatch, and the veligers (larvae) drift in the water column from 21 to 30 days before settling to the bottom and metamorphosing into the adult form. Larval conch feed on phytoplankton, juvenile conch feed primarily on seagrass, detritus, macroalgae, and organic material in the sediment, and adults feed primarily on different types of filamentous algae.

Naled and permethrin (brand names Dibrom and Biomist 30-30, respectively) are pesticides commonly used to control mosquitos in proximity to conch habitat. Both are sprayed as an ultra-low volume mist with naled applied from aircraft and permethrin applied from a truck-mounted mister. Queen conch generally breed during the spring and summer months; consequently, their larvae are most abundant when pesticide usage is at its peak. Aerial drift and runoff can carry pesticides into non-targeted areas (Hennessey et al. 1992; Pierce et al. 2005). In addition, queen conch larvae are associated with surface layers (Barile et al. 1994; Stoner and Davis 1997) where many contaminants, including pesticides, accumulate (Rumbold and Snedaker 1997). Several studies have indicated that pesticides have both direct and indirect impacts on queen conch early life stages. For example, McIntyre et al. (2006) recorded that permethrin and naled have significant toxicological effects on the development and survival of queen conch embryos in laboratory experiments. In this study, larvae were exposed to pesticide concentrations representative of those following pesticide application. After 24 hours larvae exposed to Biomist 30-30 experienced 50-95% mortality; at 48-hours the mortality was 100%. Abnormalities were also observed during embryogenesis, with slow development seen in all pesticide treatments. Defects increased with increased pesticide concentrations, and in some cases, if the larvae hatched were so deformed as to be unviable. Similarly, Delgado et al. (2013) exposed queen conch larvae to naled and permethrin for 12-hours. No mortality was observed, however exposed larvae demonstrated an increase in the proportion undergoing metamorphosis. The authors hypothesize that the pesticides are able to stimulate the regulatory pathway which induces metamorphosis. Exposure to pesticides may cause larval conch to metamorphose in suboptimum habitat. The LOEC was observed at 26.28 ng/ml which corresponds to 6.3 ng/ml. Concentrations tested were environmentally relevant, i.e., within the range of those detected in nearshore environments in Florida.

Abalone and conch are taxonomically related (marine gastropod mollusks) and exist within similar trophic levels. From a toxicological perspective it is a reasonable assumption that responses to chemical stressors will be comparable between the 2 species. We do, however, acknowledge that large variations in toxic response to pesticides are sometimes observed in closely related species. In evaluating the potential responses of abalone and conch to carbaryl, NMFS considered direct mortality to both the adult and larval stages, sublethal impacts, and prey availability.

10.3.6.1 Juvenile and Adult Mortality (surrogate: mollusks)

In evaluating the available data for abalone and conch, we determined that mollusks were the closest available surrogate for the juvenile and adult life-stages. Note that surrogate uncertainty is considered when making effect of exposure determinations. For mortality, we considered the information provided in EPA's BE describing the mollusk SSD.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 6.6 mg/L (HC05 value from the "Mollusks" SSD) and a slope of 4.5 (assumed).

10.3.6.2 Larval Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for abalone, we determined that aquatic invertebrates were the closest available surrogate for the larval life-stage. Note that surrogate uncertainty is considered when making effect of exposure determinations. For abalone larval mortality we considered the information described above under the section "Invertebrate Prey" (Section 10.3.2.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.0016 mg/L (HC05 value from the "All non-mollusk aquatic invertebrates" SSD) and a slope of 4.5 (assumed). EPA did not generate a combined SSD (mollusks and non-mollusks). The non-mollusk SSD was used to capture the embryo and larval phases of abalone.

10.3.6.3 Sublethal Effects (surrogate: mollusks)

EPA's BE included sublethal data for mollusks. These data were organized into 4 endpoints: growth, reproduction, behavior, and AChE inhibition.

For growth, the BE included 5 studies with concentrations ranging from 1 mg/L to 19.2 mg/L. Endpoints observed in these studies included impacts to shell deposition, development, and growth rate. NMFS did not find any additional studies related to the impact of carbaryl on growth to mollusks. In total, 13 values were displayed on the Risk-plots (Armstrong and Millemann 1974; Butler et al. 1960; Davis and Hidu 1969; Mayer 1987; Stewart et al. 1967).

For reproduction, the BE included 2 studies (Tripathi and Singh 2003; Tripathi and Singh 2004). Both studies evaluated the effect of carbaryl on pond snails. Effects observed included a decrease in eggs laid, decrease in eggs hatched, and decrease in the survivability of hatchlings. NMFS did not find any additional reproductive studies. The Risk-plot displays LOACs from these studies at 1 mg/L and 2 mg/L.

For behavior, the BE included 2 studies (Armstrong and Millemann 1974; Donkin et al. 1997). These 2 studies contain 3 EC50 values representing decreases in food consumption and filtration rate. NMFS did not find any additional behavior studies. The Risk-plot displays the 3 EC50s at 13.6 mg/L, 22 mg/L, and 41.6 mg/L.

For AChE inhibition, the BE included 4 studies (Binelli et al. 2005; Kopecka-Pilarczyk 2010; Mora et al. 1999; Tripathi and Singh 2003). Mora (1999) reported an IC50 for Mediterranean Mussel at 0.089 mg/L. The other studies reported LOECs ranging from 0.0001 mg/L to 1.0 mg/L. NMFS did not find any additional behavior studies. The Risk-plot displays 5 AChE endpoints with this range.

10.3.6.4 Algal Prey

In evaluating the available data for carbaryl, we determined that algae were the closest available surrogate for marine macroalgal species such as giant kelp. EPA’s carbaryl BE reported 11 available studies that evaluated effects to algae for carbaryl. Endpoints included: biomass, photosynthesis, growth rate, density, etc. Effect concentrations ranged from 0.005 mg/L to 37,900 ug/L. Species tested included green algae, blue-green algae, and diatom.

Data displayed on Risk-plots
NMFS reviewed the available studies and determined that 6 contained endpoints that were appropriate for displaying on the risk plot. These studies reported IC50 or EC50 values. A single dose-response bar was plotted on the Risk-plot. The IC50 (1.55 mg/L) for this dose-response bar was constructed by taking the geometric mean of 22 values from these 6 studies. IC50 values ranged from 0.34 mg/L (Brooke 1993) to 6.1 mg/L (Ma et al. 2006). The dose-response bar displayed on the plot was calculated using a probit equation with an IC50 of 1.55 mg/L and a slope of 4.5 (assumed).

10.3.7 Carbaryl Response: Additional Habitat Endpoints

10.3.7.1 Aquatic Plants

Most of the available toxicity studies with aquatic plants have focused on growth, mortality, physiological effects, and population effects. All but 3 of the available toxicity endpoints for aquatic plants involve non-vascular species. Because of the variability in study designs and endpoints, it was not possible to derive an SSD with the available plant data. For aquatic plants, the Risk-plot displays the minimum (0.22 mg/L), maximum (336.23 mg/L) and geometric mean (2.75 mg/L) of the available 33 endpoints.

10.3.7.2 Terrestrial Plants

NMFS considered the data provided in EPA's BE, Chapter 2. This information was evaluated qualitatively when assessing the impact of carbaryl on terrestrial vegetation such as riparian habitat.

10.3.8 Methomyl Response: Fish

The information provided by EPA in their BE for methomyl addressed aspects of survival, growth and reproduction of aquatic species (freshwater and saltwater), as well as providing some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. Because NMFS is charged under the ESA to evaluate all effects that are caused by the action and reasonably certain to occur, we evaluate all of the effects' consequences that may reduce the fitness of individuals or reduce the conservation value of PBFs of designated critical habitat. Our evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broad range of endpoints including behaviors, endocrine disruption, and other physiological alterations. The information we assessed is derived from published, scientific journals and information from government agency reports, theses, books, information and data provided by the registrants identified as applicants, and independent reports.

10.3.8.1 AChE

Methomyl is a N-methyl carbamate with a mode and mechanism of AChE inhibition that has been shown in many types of organisms including fish. NMFS searched the BE, ECOTOX, and open literature for studies on AChE inhibition in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. In total, 5 studies were collected representing 5 different endpoints and fish species.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The IC50 (25.97 ug/L) for this dose-response bar was constructed by taking the geometric mean of 2 in vivo endpoint values from 2 different studies (Carter 1971; Li et al. 2008). The first endpoint was an IC50 of 16 ug/L measured in channel catfish (*Ictalurus punctatus)*. The other endpoint was a LOEC of 42.14 ug/L measured in stone moroko (*Pseudorasbora parva*). The dose-response bar displayed on the plot was calculated using a logit equation with an IC50 of 25.97 ug/L and a combined slope of 0.95 (Laetz et al. 2009).

10.3.8.2 Fish Mortality

According to EPA's methomyl BE, acute mortality studies conducted with technical grade methomyl were used to derive SSDs, which were fit to test results for fish exposed to methomyl. Five distributions were tested and a variety of methods were used to determine whether different subsets of data should be modeled independently. Acute toxicity estimates (96-hour LC50) for methomyl range from 300 (MRID 40098001) to 32,000 µg/L (MRID 40098001) and span 2 orders of magnitude, indicating a wide range of sensitivity to methomyl among fish. The lowest definitive LC50 for methomyl is for a formulation (24% a.i.) tested on the channel catfish (LC50 = 300 µg/L; Meyer and Ellersieck, 1986). Despite the lowest 3 LC50 values (300 – 370 µg/L) being derived from studies that used formulated methomyl products, the lowest TGAI derived LC50 values (417 – 425 µg/L) are close in magnitude. The most sensitive species, channel catfish, is represented by a TGAI study used in the all aquatic vertebrate SSD used to derive the HC05 value, so it should be noted that all species represented by 96-hour typical end-use product TEP studies are also represented by at least 1 96-hour TGAI study that has been incorporated in the all vertebrate SSD. For the fish families for which methomyl toxicity data are available, Atheriniformes (Ictaluridae and Centrarchidae families), in general, appear to be the most sensitive to methomyl with LC50 values ranging from 320-2800 µg/L. The Cyprinodontidae family (the only saltwater representative, with the exception that 1 salmonid inhabits both freshwater and saltwater habitats) only had 1 data point for comparison but all represented families (also including Cyprinidae, Salmonidae and Cichlidae) had somewhat similar toxicity ranges, with Salmonidae having the widest range of sensitivities (from 560 to 32,000 µg/L).

NMFS found only a single additional LC50 value. While NMFS did not perform a thorough review of this study, NMFS did assess whether the new data would influence the analyses. The new toxicity data was added to the existing data, new geometric means were calculated for each species and HC05s were estimated by fitting the data with a log-triangle distribution. The resulting new HC05 was within the confidence limits of the HC05 provided in EPA's BE. NMFS, therefore, chose to continue to use the HC05 provided by EPA.

Data displayed on Risk-plots

A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 472 ug/L (HC05 value from the "All Fish" SSD) and a slope of 4.5 (assumed).

10.3.8.3 Invertebrate Prey

According to EPA's methomyl BE, the aquatic invertebrates mortality threshold is based on the HC05 value from the pooled freshwater and estuarine/marine SSD for the taxon (SSDs were generated for non-mollusks aquatic invertebrates only, as no acute mortality/LC50 data for mollusks was available). By comparing results from pooled invertebrates with freshwater invertebrates alone; the SSD results support pooling the data into a single SSD for all invertebrates. SSDs were based on acute 48 and 96-hr LC50 values from studies using TGAI only (LC50 values from formulation/mixture testing were not included). To generate SSDs, 5 potential distributions were considered (log-normal, log-logistic, log-triangular, log-gumbel and Burr) and the log-gumbel distribution was found to provide the best fit. Model-averaged SSDs and model-averaged quantiles, including the HC05 were estimated. The mortality thresholds for aquatic invertebrates are based on the HC05 of 3.94 µg a.i./L as determined from the pooled freshwater and estuarine/marine invertebrate SSD.

For aquatic invertebrates overall, when considering the acute mortality data from either a 48 or 96-hour exposure duration, there is a large range in sensitivity with a 3 orders of magnitude difference in the values from 2.11 µg/L [Water flea (*Ceriodaphnia reticulata*); Mano et al., 2010; E154905] to 8930 µg/L [Northern house mosquito (*Culex pipiens*); E167166]. From the data available from the 48 and 96-hour exposure durations, the values from tests conducted with the active ingredient, methomyl, form the basis of the SSD. For aquatic invertebrates, immobility is also considered as a surrogate measure for mortality (EC50) because the size of the organisms makes determination of mortality difficult. Aside from the acute mortality and immobility endpoints, other mortality-related endpoints are for reduced survival/survivorship, hatch and lifespan.

NMFS did not find any additional LC50 data for methomyl and aquatic invertebrates. Thus, there was no need to update EPA's mortality SSD.

Data displayed on Risk-plots

A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 6.7 ug/L (HC10 value from the "Aquatic Invertebrate" SSD) and a slope of 4.5 (assumed). In our conference and biological opinion, we present effects to invertebrates that serve as prey for our listed species by plotting a single dose-response bar on our risk plots; the dose-response bar was calculated using a probit equation with an assumed slope of 4.5 and an LC50 representing the HC10 from the SSDs presented in EPAs BEs, specifically the "non-mollusk aquatic invertebrates" SSDs. EPA's invertebrate SSDs are, by nature, considerate of various sensitivities across invertebrate species. For example, the carbaryl non-mollusk SSD considered 25 orders and 53 species of invertebrate. NMFS selected the HC10 from the invertebrate SSD to use in our effects analysis using best professional judgement. NMFS's listed species consume many types of food, so if 1 species of invertebrate is more affected by a carbamate pesticide than another, our listed species will have other food options available to them despite the loss of certain prey items. The ecological importance of any one invertebrate species will depend on its value as food to the listed fish species and their abundance in the environment. While they can switch food, both an overall decline in prey biomass (Cuffney et al. 1984; Macneale et al. 2010) and long recovery times for invertebrate communities (Liess and Schulz 1999; Macneale et al. 2014) will still be ecologically relevant to their food availability. Since it is not possible within the scope of this conference and biological opinion to assess effects to all invertebrate species, and since the presence and relative abundances of the SSD invertebrate species are not known, we selected the HC10 as a metric to be protective of 90% of the invertebrate prey species that our listed species could potentially consume in both fresh and marine water.

10.3.8.4 Growth

EPA's methomyl BE reports available acceptable endpoint values for growth in fish (and aquatic-phase amphibians) that range from 73 to 1030 µg/L, spanning 2 orders of magnitude. An additional lower growth-related endpoint value is available for the fathead minnow, with a length and dry weight LOAEC (without a NOAEC) of 46 µg/L (Call et al., 1989; E14097); however, it is not suitable for quantitative use. Therefore, the lowest values for growth-related endpoints are for the fathead minnow, with a length NOAEC/LOAEC of 73/145 µg/L and an MATC of 103 (Howard et al. 1991). The data range extends to a NOAEC (without an accompanying LOAEC) of 1030 µg/L (Hicks 2012). The former was based on significant inhibitions ($p < 0.05$) of 9% reduction in length and 19% reduction in wet weight in the 145 µg/L treatment. Although, in that study, neither egg hatchability nor fry survival were significantly reduced at the highest concentration tested (261 µg/L), another 2 studies are available to support effects in this treatment range—with significant reduction in length of the F1 generation at 142 µg/L (NOAEC/LOAEC for fathead minnow of 76/142 µg/L; Strawn et al. 1993), and survival at 117 µg/L (NOAEC/LOAEC for fathead minnow of 57/142 µg/L; Driscoll 1982). With this additional support for effects in the range of NOAEC/LOAEC = 73/145 µg/L (MRID 46015305), this study provides the lowest sublethal effects endpoint for growth and freshwater vertebrate species. The only other species represented is also an estuarine/marine species, the sheepshead minnow (*Cyprinodon variegatus*) with a NOAEC/LOAEC of 260/490 µg/L and an MATC of 357 for significantly reduced growth (12.9% reduction in length) (Boeri 1998).

NMFS searched the BE, ECOTOX, and open literature for studies on growth endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS evaluation of the action. No additional growth related studies were found. In total, 3 different studies were considered which reported growth-related endpoints for the fathead minnow. Endpoint concentrations range from 142 ug/L (Strawn 1993) to 731 ug/L (Call DJ and DL 1992).

Data displayed on Risk-plots
NMFS reviewed the available growth studies and determined that 3 contained endpoints that were appropriate for displaying on the risk plot (Call DJ and DL 1992; Driscoll 1982; Strawn 1993). These studies contain 5 endpoints ranging from 142 ug/L (Strawn 1993) to 731 ug/L (Call DJ and DL 1992). Exposure durations range from 28-days to 56-days.

10.3.8.5 Reproduction

EPA's methomyl BE reports reproductive effects of methomyl on freshwater fish identified from registrant-submitted studies and open-literature studies range from 94.7-312 µg/L. There were 2 studies, representing 1 species (fathead minnow). There were no data for estuarine/marine fish, so the freshwater fish reproduction data were used as a surrogate. The lowest value summaries in the BE are from a 21-day registrant submitted short term reproduction assay with the fathead minnow (*P. promelas*) (Hicks 2012). In this assay, fecundity (eggs per surviving female per reproductive day) and fertilization success were significantly reduced (23.3 and 1.6%, respectively) at 312 µg/L (NOAEC/LOAEC of 94.7/312 µg/L). The slight reduction of 1.6% would not provide convincing evidence of an effect alone; however, the dose/response pattern followed at the next higher concentration with a 33% reduction and the GSI (gonad-somatic index) was also significantly affected at this concentration. The only other available study had a NOAEC/LOAEC for F1 hatching success in the same concentration range (142/280 µg/L); hatching success was significantly ($p \leq 0.05$) reduced (8.5%) in the 280 µg/L treatment (MRID 43072101). Significant effects were also seen in length and wet weight at this treatment level (and in length at 142 µg/L), as mentioned in the growth and development section above. However, no significant effects were seen in time to first spawn, F0 hatching success, mean eggs per spawn, mean spawning days or spawns per pair at the highest test concentration (280 µg/L).

NMFS searched the BE, ECOTOX, and open literature for studies on reproductive endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. In total, 3 different studies were considered with endpoint concentrations ranging from 20 ug/L to 312 ug/L.

Data displayed on Risk-plots
NMFS reviewed the available reproduction studies and determined that 3 contained endpoints that were appropriate for displaying on the risk plot. For the purpose of this assessment, reproductive endpoints were further classified based on the life-stage of the chemical exposure. For example, the endpoint hatch-success would be considered a reproductive endpoint if it were measured in the offspring of an exposed parent. In contrast, hatch-success would be considered a developmental endpoint if the unhatched eggs themselves were exposed. In total, 2 reproductive endpoints were displayed on the Risk-plot (Hicks 2012; Meng et al. 2021). The first study observed significant decrease/inhibition in sex hormones cholesterol (CHO), pregnenolone (PREG), and progesterone (PROG) in testes and serum of Nile tilapia (*Oreochromis niloticus*) following a 48-day exposure. The other study observed a decrease in the number of spawns and fecundity in fathead minnow following a 21-day exposure.

10.3.8.6 Development

NMFS searched the BE, ECOTOX, and open literature for studies on developmental endpoints in fish in pursuit of finding the best scientific and commercial data available to support NMFS's evaluation of the action. For the purpose of this assessment, endpoints were considered to be "developmental" when exposures to test species were made pre-hatch. In total 8 different studies were considered. These studies reported developmental effects to 3 different fish species: fathead minnow, zebrafish, and sheepshead minnow. Endpoints included: body length, hatch success, mobility, and spinal curvature. Endpoint concentrations ranged from 46 ug/L (Call et al. 1989) to 6,702 ug/L (Padilla et al. 2012).

Data displayed on Risk-plots
Developmental endpoints were not displayed on the Risk-plots. Instead, these types of endpoints were evaluated qualitatively when considering the available evidence in support of particular risk hypotheses. For example, endpoints such as heart deformities, low weight, and decreased locomotion may translate into fitness-level impacts in later life stages (e.g., impacts to growth). However significant uncertainties exist. NMFS determined a qualitative approach to these endpoints was appropriate as a number of the endpoints captured in our search have uncertain connections to fitness (e.g., spontaneous tail movement in embryos). Additionally, the degree to which individuals may recover or compensate for the early developmental impacts is uncertain.

10.3.9 Methomyl Response: Coral

See Section 10.3.3 for an overview of the impacts of methomyl (and AChE inhibiting pesticides, generally) on coral species.

10.3.9.1 Polyp Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for methomyl, we determined that aquatic invertebrates were the closest available surrogate for coral polyps. Note that surrogate uncertainty is considered when making effect of exposure determinations. For coral polyp mortality we considered the information described above under the section "Invertebrate Prey" (Section 10.3.8.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 3.94 ug/L (HC05 value from the "Aquatic Invertebrate" SSD) and a slope of 4.5 (assumed).

10.3.9.2 Algal Symbiont Mortality (surrogate: algae)

In evaluating the available data for methomyl, we determined that algae were the closest available surrogate for coral algal symbionts. EPA's methomyl BE reported 6 available studies that evaluated effects to algae for methomyl. Endpoints included: biomass, phosphorus content, growth rate, abundance, etc. Effect concentrations ranged from 3,690 ug/L to 900,000 ug/L. Species tested included green algae and blue-green algae.

Data displayed on Risk-plots
NMFS reviewed the available studies and determined that 3 contained endpoints that were appropriate for displaying on the risk plot. These studies reported EC50 values. A single dose-response bar was plotted on the Risk-plot. The EC50 (84,112 ug/L) for this dose-response bar was constructed by taking the geometric mean of 6 values from these 3 studies. IC50 values ranged from 43,140 ug/L (MRID 48983401) to 184,000 ug/L (Pereira et al. 2009). The dose-response bar displayed on the plot was calculated using a probit equation with an EC50 of 84,112 ug/L and a slope of 4.5 (assumed).

10.3.9.3 Methomyl-specific Studies on Coral

NMFS did not find any methomyl-specific studies on coral. See the discussion above for information on impacts from similar insecticides.

10.3.9.4 Zooplankton Prey (surrogate: aquatic invertebrates)

In evaluating the available data for methomyl, we determined that aquatic invertebrates were the closest available surrogate for marine zooplankton. For coral prey abundance we considered the information described above under the section "Invertebrate Prey" (Section 10.3.8.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 3.94 ug/L (HC05 value from the “Aquatic Invertebrate” SSD) and a slope of 4.5 (assumed). The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 6.7 ug/L (HC10 value from the “Aquatic Invertebrate” SSD) and a slope of 4.5 (assumed).

10.3.10 Methomyl Response: Abalone and Conch

See Section 10.3.4 for an overview of the impacts of methomyl (and AChE inhibiting pesticides, generally) on abalone and conch.

10.3.10.1 Abalone and Conch Adult Mortality (surrogate: mollusks)

EPA’s BE contained only a single study evaluating the impacts of methomyl on mollusks (MRID 42074601). This study did not evaluate mortality (see Section 10.3.10.3 ”Sublethal Effects” below).

10.3.10.2 Abalone and Conch Larval Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for methomyl, we determined that aquatic invertebrates were the closest available surrogate for larval abalone. Note that surrogate uncertainty is considered when making effect of exposure determinations. For larval mortality we considered the information described above under the section “Invertebrate Prey” (Section 10.3.8.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
A single dose-response bar was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 3.94 ug/L (HC05 value from the “Aquatic Invertebrate” SSD) and a slope of 4.5 (assumed).

10.3.10.3 Sublethal Effects (surrogate: mollusks)

EPA’s BE contained only a single study evaluating the impacts of methomyl on mollusks (MRID 42074601). This study reported a shell deposition EC50 of 140 mg/L and a NOAEC of 0.12 mg/L in Eastern oyster (*Crassostrea virginica*). NMFS did not find any additional mollusk studies for methomyl.

10.3.10.4 Algal Prey

In evaluating the available data for methomyl, we determined that algae were the closest available surrogate for marine macro algae such as giant kelp. EPA’s methomyl BE reported 6 available studies that evaluated effects to algae for methomyl. Endpoints included: biomass, phosphorus content, growth rate, abundance, etc. Effect concentrations ranged from 3,690 ug/L to 900,000 ug/L. Species tested included green algae and blue-green algae.

Data displayed on Risk-plots
NMFS reviewed the available studies and determined that 3 contained endpoints that were appropriate for displaying on the risk plot. These studies reported EC50 values. A single dose-response bar was plotted on the Risk-plot. The EC50 (84,112 ug/L) for this dose-response bar was constructed by taking the geometric mean of 6 values from these 3 studies. IC50 values ranged from 43,140 ug/L (MRID 48983401) to 184,000 ug/L (Pereira et al. 2009). The dose-response bar displayed on the plot was calculated using a probit equation with an EC50 of 84,112 ug/L and a slope of 4.5 (assumed).

10.3.11 Methomyl Response: Sunflower Sea Star

See Section 10.3.5 for an overview of the impacts of methomyl (and AChE inhibiting pesticides, generally) on the sunflower sea star. In evaluating the potential responses of the sunflower sea star to methomyl, NMFS considered direct mortality to the larval stages and prey availability. Sublethal endpoints, including development, were not included in the methomyl analysis due to lack of available data.

10.3.11.1 Larval Mortality (surrogate: aquatic invertebrates)

In evaluating the available data for methomyl, we determined that aquatic invertebrates were the closest available surrogate for the larval life-stage of the sunflower sea star. Note that surrogate uncertainty is considered when making effect of exposure determinations. For sunflower sea star larval mortality, we considered the information described above under the section “Invertebrate Prey” (Section 10.3.2.3). As described above, we did not alter the SSD created by EPA in the BE.

Data displayed on Risk-plots
One dose-response bar was plotted on the Risk-plot. The “mortality larvae.HC05” dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.00394 mg/L (HC05 value from the “pooled aquatic invertebrates” SSD) and a slope of 4.5 (assumed).

The “mortality larvae.ssd” bar represents single point values (the HC05, HC10, HC50, HC90, and HC95) that were extracted from the “pooled aquatic invertebrates” SSD described above.

10.3.11.2 Prey Availability

In evaluating the available data for methomyl, we determined that aquatic invertebrates were the best representation of the sunflower sea star’s prey base. As described in section 10.3.5, sunflower sea stars are generalists. However, EPA did not report mollusk data for methomyl, so there is no mollusk data included in the analysis for methomyl.

Data displayed on Risk-plots
A single dose-response bar, “prey.invert”, was plotted on the Risk-plot. The dose-response bar displayed on the plot was calculated using a probit equation with an LC50 of 0.0067 mg/L (HC10 value from the “All pooled aquatic invertebrates” SSD) and a slope of 4.5 (assumed).

10.3.12 Methomyl Response: Additional Habitat Endpoints

10.3.12.1 Aquatic Plants

Available toxicity studies with aquatic plants have focused on growth/biomass, population abundance, and biochemical effects such as phosphorous and nitrogen content. Most of the available toxicity endpoints for aquatic plants involves non-vascular plant species, although vascular plant data are available for 1 species, *Lemna gibba*. Because of the variability in study designs and endpoints, it was not possible to derive an SSD with the available plant data. For aquatic plants, the Risk-plot displays the minimum (7,240 ug/L), maximum (700,000 ug/L) and geometric mean (148,400 ug/L) of the available 47 endpoints.

10.3.12.2 Terrestrial Plants

NMFS considered the data provided in EPA's BE, Chapter 2. This information was evaluated qualitatively when assessing the impact of methomyl on terrestrial vegetation such as riparian habitat.

10.4 Risk Hypotheses

The following sections outline risk hypotheses that we developed for several groups of species.

10.4.1 Anadromous Fish

10.4.1.1 Primary Route of Exposure

The primary route of pesticide exposure in anadromous fish is contact with runoff and drift transport pathways of pesticides deposited in surface waters. Exposure from contact with contaminated surface water will be evaluated. Quantitative estimates of exposure are evaluated using surface water concentration estimates derived by EPA.

10.4.1.2 Risk Hypotheses

We constructed the following risk hypotheses for the species effects analysis, considering the available exposure, response, and life history information referenced above. Modifications to these risk hypotheses were made on a species-by-species basis, as appropriate.

- Exposure to the pesticide is sufficient to reduce abundance via acute lethality.
- Exposure to the pesticide is sufficient to reduce abundance via reduction in prey availability.
- Exposure to the pesticide is sufficient to reduce abundance via impacts to growth (direct toxicity).
- Exposure to the pesticide is sufficient to reduce productivity via impairments to reproduction.

- Exposure to the pesticide is sufficient to reduce abundance and productivity via impairments to ecologically significant behaviors.
- Exposure to the pesticide is sufficient to reduce ChE activity; the identified mechanism of toxicity

For the assessment of impacts to designated critical habitat, we constructed risk hypotheses based on the PBFs identified. Water quality and prey availability are key attributes that are either designated as PBFs of the critical habitat, or are relevant to the PBFs of most species' habitats.

Additional species-specific risk hypothesis considerations
Atlantic salmon. NOAA Fisheries and the FWS share jurisdiction for the recovery of the Maine DPS Atlantic salmon. USFWS evaluates the effects of this action on the species during its freshwater residency. In this opinion, we evaluate the effects of the action in marine and estuarine habitats. Consequently, our evaluation of risk hypotheses for Atlantic salmon do not consider freshwater reproductive and rearing activities.

Gulf Sturgeon. NOAA Fisheries and USFWS share jurisdiction for the recovery of Gulf sturgeon. USFWS evaluates the effects of this action on the species during its freshwater residency. In this opinion, we evaluate effects of the action in marine and estuarine habitats. Consequently, we do not include risk hypotheses associated with freshwater exposures of Gulf sturgeon.

10.4.2 Marine Fish

10.4.2.1 Primary Route of Exposure

The primary route of pesticide exposure in marine fish is contact with runoff and drift transport pathways of pesticides deposited in surface waters. Exposure from contact with contaminated surface water will be evaluated. Quantitative estimates of exposure are evaluated using surface water concentration estimates derived by EPA.

10.4.2.2 Risk Hypotheses

We constructed the following risk hypotheses for the species' effects analysis, considering the available exposure, response, and life history information referenced above. Modifications to these risk hypotheses were made on a species-by-species basis, as appropriate.

- Exposure to the pesticide is sufficient to reduce abundance via acute lethality.
- Exposure to the pesticide is sufficient to reduce abundance via reduction in prey availability.
- Exposure to the pesticide is sufficient to reduce abundance via impacts to growth and development (direct toxicity).
- Exposure to the pesticide is sufficient to reduce productivity via impairments to reproduction.
- Exposure to the pesticide is sufficient to reduce abundance and productivity via impairments to ecologically significant behaviors.

- Exposure to the pesticide is sufficient to reduce ChE activity; the identified mechanism of toxicity

For the assessment of impacts to designated critical habitat, we constructed risk hypotheses based on the PBFs identified. Water quality and prey availability are key attributes that are either designated as PBFs of the critical habitat, or are relevant to the PBFs of most species habitats.

10.4.3 Marine Invertebrates

10.4.3.1 Primary Route of Exposure

The primary route of exposure in marine invertebrates is contact with contaminated surface waters from the runoff and drift pathways. Black abalone occur in the intertidal zone and may, along with the other marine invertebrates, occur in the nearshore subtidal zone.

10.4.3.2 Risk Hypotheses

We constructed the following risk hypotheses for the species effects analysis, considering the available exposure, response, and life history information referenced above.

Abalone and Conch

- Exposure to the pesticide is sufficient to reduce abundance via acute lethality to juvenile and adults.
- Exposure to the pesticide is sufficient to reduce abundance via acute lethality to larval stages.
- Exposure to the pesticide is sufficient to reduce abundance via reduction in forage (algae).
- Exposure to the pesticide is sufficient to reduce abundance and/or productivity via sublethal impacts to growth, reproduction or behavior.
- Exposure to the pesticide is sufficient to reduce ChE activity; the identified mechanism of toxicity.

Sunflower Sea Star

- Exposure to the pesticide is sufficient to reduce abundance via acute lethality.
- Exposure to the pesticide is sufficient to reduce abundance via reduction in prey availability.

Corals

- Exposure to the pesticide is sufficient to reduce abundance via acute lethality to coral polyps.
- Exposure to the pesticide is sufficient to reduce abundance via acute lethality to algal symbionts.
- Exposure to the pesticide is sufficient to reduce abundance via reduction in prey availability.
- Exposure to the pesticide is sufficient to reduce abundance and/or productivity via sublethal impacts to growth, reproduction or behavior.

For the assessment of impacts to designated critical habitat, we constructed risk hypotheses based on the primary biological features identified. Water quality and prey availability are key attributes that are either designated as PBFs of the critical habitat, or are relevant to the PBFs of most species habitats.

10.4.4 Cetaceans

10.4.4.1 Primary Route of Exposure

SRKW is the only listed cetacean likely to be adversely affected by the action. EPA established that direct toxicity to listed killer whales is not expected given the deeper water habitats they typically occupy. EPA also concluded that these pesticides are likely to adversely affect SRKW due to reductions in their prey including Chinook salmon, the species' preferred prey (EPA BEs 2017).

10.4.4.2 Risk Hypotheses

SRKW primarily feed on salmon and prefer Chinook salmon. Pacific salmonids may be exposed to carbaryl and methomyl during residency in freshwater and nearshore habitats. Localized depletions in the prey base may result in increased energy demands of SRKW due to abandonment of foraging areas in search of more abundant prey or expending substantial effort to find depleted prey resources within their range. Reductions in prey can lead to nutritional stress, reduced body size and condition, and can lower reproductive and survival rates. Food scarcity can also cause whales to draw on fat stores, mobilizing persistent contaminants that can affect reproduction and immune function. We constructed the following risk hypothesis for carbaryl and methomyl considering the available exposure, response, and life history information referenced above for SRKW:

- Exposure to the pesticide is sufficient to reduce SRKW abundance via reduction in prey availability (primarily salmonids and other fish).

10.5 Risk Analysis

In this section, we integrate the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action at the population and species level. We use the Risk-plot tool for integrating exposure and response. Where applicable, we may also use population models to estimate responses. A weight-of-evidence approach, which considers the limitations and uncertainties inherent in the available information, is then applied to characterize risk.

10.5.1 Risk-plots

Risk-plots are a tool developed by NMFS that utilize the R programming language to collectively display EECs, the extent of pesticide use sites within a species range, and effects data so that the user can visually assess the risk at the population scale. The response data and exposure estimates summarized by the Risk-plots are the same as those presented in EPA's BEs (e.g., the ranges of EECs and the spatial overlaps with species range associated with each use site). Both mortality and sublethal effect data are summarized. Effects on mortality are displayed as a range of percent mortalities based on a selected LC50 and slope. Sublethal effects are displayed as the ranges of LOECs, EC50s, etc. associated with available sublethal endpoints (e.g., growth). Effect data and EECs are displayed quantitatively using the same axis to visually estimate response magnitudes associated with each labeled use site. For a given species and pesticide, a Risk-plot provides a graphic summary of the sources of information (i.e., exposure, response, and use) needed to qualitatively assess the risk to the population posed by all labeled uses across the range of the species and across their different habitat uses (e.g., habitat bins).

10.5.2 Population Models

Sufficient data and empirical relationships were available to construct population models for 4 Pacific salmon life history strategies. We ran life-history matrix models for ocean-type and stream-type Chinook salmon (*O. tshawytscha*), coho salmon (*O. kisutch*), and sockeye salmon (*O. nerka*). The basic salmonid life history we modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death.

Somatic growth reductions - We integrated 2 avenues of effect to subyearling salmonids' growth from exposure to the 2 carbamates into the somatic growth model (Appendix A). The first avenue is a result of AChE inhibition on the feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors (Sandahl et al. 2005). Salmon are often food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to the AChE inhibitor, chlorpyrifos (Brazner and Kline 1990). Furthermore, based on our review of the sensitivities of aquatic invertebrates to carbaryl and methomyl, we expect reductions in densities and altered composition of the salmonid prey communities. Therefore, the second avenue the model addresses is the potential for reductions in juvenile growth due to reductions in available prey.

Reductions in aquatic prey are included in the model because of the high relative toxicity of pesticides to salmonid prey and the extended duration of effects on prey communities (e.g., slow recovery). Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities. Carbaryl and methomyl are highly toxic to aquatic macroinvertebrates; concentrations that are not expected to kill salmonids are often lethal for their invertebrate prey. In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, caddisflies, and stoneflies) are among the most sensitive aquatic macroinvertebrates. In addition, effects on the prey community can persist for extended periods of time (weeks, months, years), resulting in effects on fish feeding and growth long after an exposure has ended (Colville et al. 2008; Liess and Schulz 1999; Van den Brink et al. 1996; Ward et al. 1995).

These results show that all 4 species can be severely affected by changes in juvenile growth driven primarily by reduced prey availability. The concentrations that elicit reductions in population growth rate are expected to occur in salmonid habitats. The degree to which an actual threatened or endangered population is affected will depend on a host of factors including the number of individuals exposed, the duration of exposure, when they are exposed, and if they are exposed more than once. It is also important to realize that these are idealized populations and we did not incorporate other factors that can affect the sensitivity of exposed salmonids such as elevated temperatures, presence of mixtures of carbamates and organophosphate pesticides (also AChE inhibitors), and the condition of the fish. We also did not incorporate incidences of death due to acute toxicity in the growth model, and none of the concentrations examined up to the AChE EC50 (Figure 101) caused any acute mortality. We show however, that even without these other stressors taken into account there is strong evidence the expected concentrations in salmonid habitats will adversely affect salmonid populations due to loss of prey base.

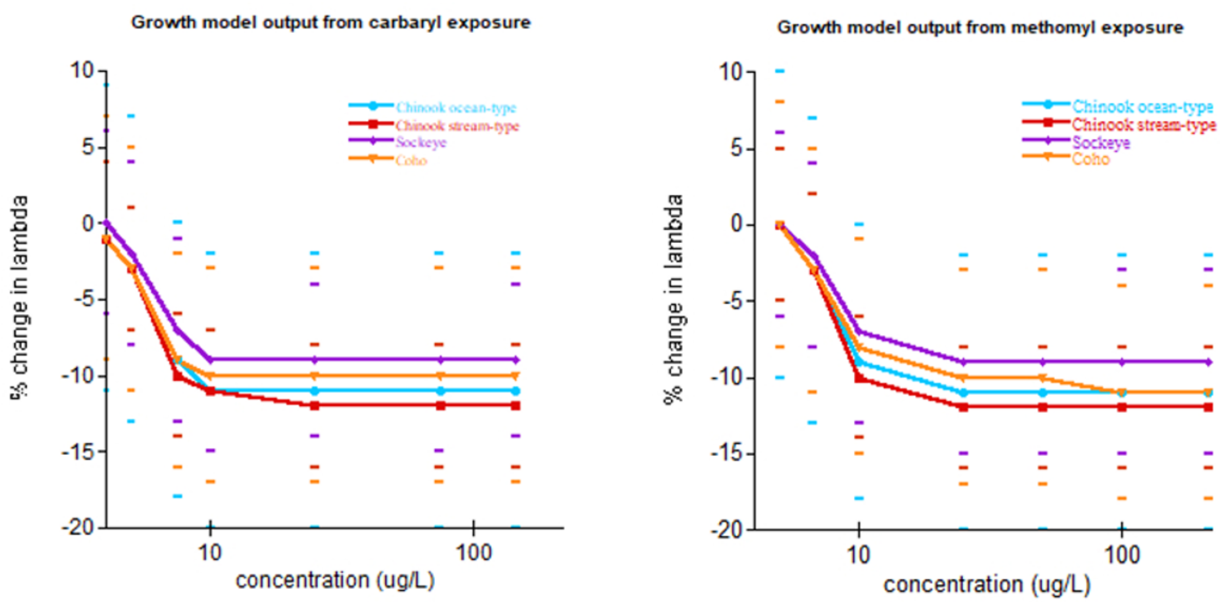


Figure 101. Somatic growth model output for carbaryl and methomyl. Scenario used to generate output was a single, 4-day exposure beginning on day 1 of the somatic growth period with 100% of the population exposed. Lines indicate mean percent change in lambda and caps show 1 standard deviation.

Acute Mortality: An acute toxicity model was constructed that estimated the population-level impacts of sub-yearling juvenile mortality resulting from exposure. For specific information on the construction and parameterization of the models see Appendix A. Potential population-level impacts resulting from mortality following freshwater exposure to pesticides were integrated into the models as alterations in the first year survival rate. We also evaluated population level responses resulting from varying the proportion of the population exposed. Population level impacts were assessed as changes in the intrinsic population growth rate and quantified as the percent change in population growth rate. The model output is shown in Table 130, Table 131, Table 132, and Table 133. Changes that exceeded the variability in the baseline (i.e., a standard deviation) were considered to be different. Importantly, the acute toxicity models excluded sublethal and indirect effects of the pesticide exposures. For example, the potential population-level impacts of reduced prey abundance are not captured by these models.

Table 130. Acute mortality model output for ocean-type Chinook. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold (and shaded) indicates a percent change in population growth rate of greater than 1 standard deviation from control values. The baseline values for ocean-type Chinook are: lambda=1.09, standard deviation of 0.1, standard deviation as a percent of lambda is 9, and first year survival S1=5.64E-03.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (12.9)	0 (12.9)	-1 (12.8)	-1 (12.8)	-1 (12.7)
10	0 (130)	-1 (12.9)	-1 (12.8)	-3 (12.6)	-3 (12.4)
15	0 (12.9)	-1 (12.9)	-2 (12.8)	-4 (12.5)	-5 (12.2)
20	-1 (13.0)	-2 (13.0)	-3 (12.9)	-5 (12.5)	-6 (12.1)
25	-1 (13.1)	-2 (13.0)	-4 (13.3)	-6 (12.7)	-8 (11.8)
30	-1 (13.0)	-2 (13.3)	-5 (13.4)	-8 (12.7)	**-10 (11.5)**
35	-1 (13.3)	-3 (13.8)	-6 (13.9)	**-9 (13.0)**	**-12 (11.4)**
40	-1 (13.4)	-3 (14.0)	-7 (14.3)	**-11 (13.5)**	**-14 (11.1)**
45	-1 (133.6)	-4 (14.3)	-8 (15.4)	**-13 (14.1)**	**-16 (10.7)**
50	-2 (13.6)	-5 (14.9)	**-9 (16.0)**	**-15 (15.3)**	**-18 (10.5)**
55	-2 (14.0)	-5 (15.5)	**-11 (17.5)**	**-17 (16.5)**	**-21 (10.2)**
60	-2 (14.2)	-6 (16.9)	**-12 (18.6)**	**-20 (17.9)**	**-23 (9.7)**
65	-2 (14.3)	-7 (16.9)	**-14 (19.8)**	**-22 (19.1)**	**-26 (9.5)**
70	-3 (14.6)	-7 (17.8)	**-16 (21)**	**-24 (20.3)**	**-29 (8.9)**
75	-3 (15.2)	-8 (18.4)	**-17 (22.1)**	**-27 (21.6)**	**-33 (8.5)**
80	-3 (15.3)	**-9 (19.7)**	**-18 (23.2)**	**-30 (22.3)**	**-37 (8.1)**
85	-4 (15.8)	**-10 (20.4)**	**-20 (24)**	**-32 (23.1)**	**-42 (7.3)**
90	-4 (16.1)	**-10 (21.5)**	**-21 (24.9)**	**-34 (23.4)**	**-48 (6.6)**
95	-4 (16.5)	**-11 (22.7)**	**-22 (25.3)**	**-36 (23.2)**	**-56 (5.5)**
100	-4 (17.1)	**-12 (23.0)**	**-23 (25.9)**	**-38 (23.6)**	**-100 (NA)**

Table 131. Acute mortality model output for stream-type Chinook. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold (and shaded) indicates a percent change in population growth rate of greater than 1 standard deviation from control values. The baseline values for stream-type Chinook are: lambda=1.00, standard deviation of 0.03, standard deviation as a percent of lambda is 3, and first year survival S1=6.43E-03.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (4.4)	0 (4.4)	-1 (4.4)	-1 (4.4)	-1 (4.3)
10	0 (4.5)	-1 (4.5)	-1 (4.5)	-2 (4.4)	**-3 (4.3)**
15	0 (4.6)	-1 (4.7)	-2 (4.7)	**-3 (4.6)**	**-4 (4.2)**
20	-1 (4.7)	-1 (4.9)	**-3 (5.1)**	**-4 (4.8)**	**-5 (4.1)**
25	-1 (4.8)	-2 (5.1)	**-3 (5.5)**	**-6 (5.1)**	**-7 (4.1)**
30	-1 (4.9)	-2 (5.6)	**-4 (6.0)**	**-7 (5.6)**	**-8 (4.0)**
35	-1 (5.1)	-2 (6.0)	**-5 (6.8)**	**-8 (6.1)**	**-10 (4.0)**
40	-1 (5.4)	**-3 (6.5)**	**-6 (7.5)**	**-10 (6.9)**	**-12 (3.9)**
45	-1 (5.6)	**-3 (7.0)**	**-7 (8.5)**	**-11 (7.8)**	**-14 (3.7)**
50	-2 (5.8)	**-4 (7.5)**	**-8 (9.8)**	**-13 (9.3)**	**-16 (3.7)**
55	-2 (6.2)	**-4 (8.3)**	**-9 (11.1)**	**-15 (10.9)**	**-18 (3.6)**
60	-2 (6.5)	**-5 (9.3)**	**-11 (13.0)**	**-17 (13.1)**	**-20 (3.5)**
65	-2 (6.9)	**-6 (10.1)**	**-12 (14.7)**	**-19 (14.7)**	**-23 (3.4)**
70	-2 (7.2)	**-6 (11.1)**	**-13 (15.7)**	**-22 (16.7)**	**-26 (3.2)**
75	**-3 (7.7)**	**-7 (12.4)**	**-15 (17.5)**	**-24 (17.9)**	**-29 (3.1)**
80	**-3 (8.1)**	**-8 (13.5)**	**-15 (18.3)**	**-27 (18.8)**	**-33 (2.9)**
85	**-3 (8.6)**	**-8 (14.6)**	**-17 (19.3)**	**-29 (19.7)**	**-37 (2.7)**
90	**-3 (9.1)**	**-9 (15.4)**	**-18 (20.2)**	**-30 (20.0)**	**-43 (2.4)**
95	**-4 (9.5)**	**-10 (16.4)**	**-20 (21.1)**	**-32 (20.2)**	**-52 (2.0)**
100	**-4 (10.3)**	**-11 (17.6)**	**-21 (21.4)**	**-33 (20.0)**	**-100 (NA)**

Table 132. Acute mortality model output for sockeye. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold (and shaded) indicates a percent change in population growth rate of greater than 1 standard deviation from control values. The baseline values for sockeye are: lambda=1.01, standard deviation of 0.06, standard deviation as a percent of lambda is 6, and first year survival S1=2.57E-02.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (8.0)	0 (7.9)	-1 (7.9)	-1 (7.8)	-1 (7.8)
10	0 (8.0)	-1 (8.0)	-1 (8.0)	-2 (7.9)	-3 (7.7)
15	0 (8.0)	-1 (8.0)	-2 (8.1)	-3 (7.9)	-4 (7.7)
20	-1 (8.0)	-1 (8.2)	-3 (8.2)	-4 (8.1)	-5 (7.5)
25	-1 (8.1)	-2 (8.4)	-3 (8.5)	-5 (8.2)	**-7 (7.4)**
30	-1 (8.2)	-2 (8.8)	-4 (9.0)	**-7 (8.4)**	**-8 (7.3)**
35	-1 (8.4)	-2 (8.9)	-5 (9.6)	**-8 (8.8)**	**-10 (7.1)**
40	-1 (8.6)	-3 (9.2)	**-6 (10.1)**	**-9 (9.6)**	**-11 (7.0)**
45	-1 (8.7)	-3 (9.7)	**-7 (10.9)**	**-11 (10.4)**	**-13 (6.9)**
50	-1 (9.0)	-4 (10.4)	**-8 (12.0)**	**-13 (11.2)**	**-15 (6.7)**
55	-2 (9.2)	-4 (10.9)	**-9 (13.4)**	**-15 (12.9)**	**-17 (6.5)**
60	-2 (9.4)	-5 (11.9)	**-10 (14.4)**	**-17 (14.4)**	**-19 (6.4)**
65	-2 (9.7)	-5 (12.3)	**-12 (16.1)**	**-19 (15.7)**	**-22 (6.2)**
70	-2 (10.0)	**-6 (13.4)**	**-13 (16.9)**	**-21 (17.3)**	**-25 (5.9)**
75	-3 (10.4)	**-7 (14.3)**	**-14 (18.2)**	**-23 (18.1)**	**-28 (5.6)**
80	-3 (10.9)	**-8 (15.6)**	**-16 (19.0)**	**-26 (19.1)**	**-32 (5.4)**
85	-3 (11.3)	**-8 (16.3)**	**-17 (19.9)**	**-28 (19.7)**	**-39 (5.0)**
90	-3 (11.6)	**-9 (17.0)**	**-18 (20.8)**	**-29 (19.8)**	**-42 (4.5)**
95	-3 (12.3)	**-10 (17.7)**	**-19 (20.9)**	**-30 (19.9)**	**-51 (3.8)**
100	-4 (12.7)	**-10 (18.3)**	**-20 (21.5)**	**-32 (19.8)**	**-100 (NA)**

Table 133. Acute mortality model output for coho. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold (and shaded) indicates a percent change in population growth rate of greater than 1 standard deviation from control values. The baseline values for coho are: lambda=1.03, standard deviation of 0.05, standard deviation as a percent of lambda is 5, and first year survival S1=2.97E-02.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (7.4)	0 (7.5)	-1 (7.5)	-1 (7.4)	-2 (7.4)
10	0 (7.5)	-1 (7.6)	-2 (7.6)	-3 (7.4)	-3 (7.2)
15	0 (7.6)	-1 (7.7)	-3 (7.8)	-4 (7.5)	**-5 (7.1)**
20	-1 (7.7)	-2 (8.0)	-4 (8.1)	**-6 (7.7)**	**-7 (7.0)**
25	-1 (7.9)	-2 (8.4)	**-5 (8.5)**	**-7 (8.0)**	**-9 (6.9)**
30	-1 (7.9)	-3 (8.5)	**-6 (9.1)**	**-9 (8.4)**	**-11 (6.6)**
35	-1 (8.2)	-3 (9.2)	**-7 (9.9)**	**-11 (8.9)**	**-13 (6.5)**
40	-1 (8.5)	-4 (9.7)	**-8 (10.7)**	**-13 (9.8)**	**-16 (6.4)**
45	-2 (8.8)	-4 (10.3)	**-9 (11.8)**	**-14 (11.0)**	**-18 (6.1)**
50	-2 (9.1)	**-5 (11.1)**	**-10 (13.4)**	**-17 (12.2)**	**-21 (5.9)**
55	-2 (9.5)	**-6 (11.7)**	**-12 (14.9)**	**-20 (14.2)**	**-23 (5.8)**
60	-3 (9.9)	**-6 (12.6)**	**-14 (17.0)**	**-23 (16.5)**	**-26 (5.5)**
65	-3 (10.3)	**-7 (14.1)**	**-15 (18.5)**	**-25 (18.7)**	**-30 (5.3)**
70	-3 (10.7)	**-8 (15.1)**	**-17 (20.6)**	**-28 (20.6)**	**-33 (5.0)**
75	-3 (11.2)	**-9 (16.4)**	**-19 (22.3)**	**-31 (22.4)**	**-37 (4.7)**
80	-4 (11.6)	**-9 (17.7)**	**-20 (23.6)**	**-34 (23.7)**	**-42 (4.4)**
85	-4 (12.3)	**-11 (19.3)**	**-22 (25.0)**	**-37 (24.5)**	**-47 (4.0)**
90	-4 (12.9)	**-12 (20.4)**	**-24 (26.0)**	**-39 (25.2)**	**-54 (3.4)**
95	-4 (13.4)	**-13 (21.6)**	**-25 (27.3)**	**-42 (25.2)**	**-63 (2.8)**
100	**-5 (14.1)**	**-14 (22.9)**	**-27 (27.6)**	**-43 (25.7)**	**-100 (NA)**

In analyzing risk, we integrate the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action at the population and species level. We use 2 tools to integrate exposure and response. Risk-plots and where applicable, population models. A weight-of-evidence approach which considers the limitations and uncertainties inherent in the available information is then applied to characterize risk. Whenever possible, most sensitive toxicological endpoints used in the Risk-plots are from those studies that were conducted on species with best fit as surrogates to Pacific salmonids (e.g., rainbow trout).

The following effects scenarios of carbaryl and methomyl on Pacific salmonids (chum, chinook, coho, sockeye, and steelhead) are based on the life history, exposure, and response considerations described in the previous sections of this chapter.

- *Substantial exposure throughout ESU & High Prey Mortality*
  Exposure to habitats and individuals is likely to result in indirect impairments through prey loss, sublethal impacts, and/or direct mortality. The somatic growth population model and acute mortality models indicate that this exposure scenario will appreciably reduce the population growth rate of the ESU.
- *Exposure limited to population(s) & High Prey Mortality*
  We anticipate exposure, but exposure is geographically limited to [one or several populations] of the ESU. Exposure to habitats and individuals is likely to result in indirect impairments through prey loss, sublethal impacts, and/or direct mortality. The somatic growth population model and acute mortality models indicate that this exposure scenario will appreciably reduce the growth rate of these populations within the ESU.
- *Minimal/no exposure*
  Exposure to individuals is improbable. However, if exposure does occur, it is likely to result in indirect impairments through prey loss, sublethal impacts, and/or direct mortality. The acute and somatic growth population models both indicate that this exposure scenario will result in a dismissible reduction in population growth rate within the ESU.

10.6 Weighing the Uncertainties

All estimates of exposure and response must rely on assumptions with associated uncertainties that may contribute to the possibility of overestimating or underestimating risk, or in some circumstances may do either, depending on the specific context. Uncertainties may be due to natural variability, lack of knowledge, measurement error, or model error. One way to account for uncertainties associated with variability is to integrate measures of variability into models to calculate the probability of risk; the underlying assumption is that risk can be accurately predicted by mathematically accounting for variability. Accounting for uncertainty is critical when weighing model outputs and when applying outputs in risk conclusions. This section describes how we utilized a variety of tools with different assumptions to increase our confidence in risk estimates, and how we weighed key assumptions and associated uncertainties of our risk assessment to reach conclusions consistent with the purpose of section 7(a)(2) of the ESA (to insure the actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat).

In Table 134, we identify key assumptions associated with estimates utilized in our assessment of the effects of the action. X indicates if the assumption contributes to the possibility that risk will be underestimated or overestimated. In some cases, the assumption may contribute to the possibility of either underestimating or overestimating risk, depending on the specific circumstances being evaluated. We then discuss how these assumptions and associated uncertainties are factored into our weight-of-evidence approach presented in the risk characterization section below.

Table 134. Assessment assumptions and influence on risk estimates.

Assumption (estimate)	Underestimate Risk	Overestimate Risk
1. Pesticide application rates- Pesticides will be applied at the highest labeled rate for the use site or crop grouping (EECs)		**x**
2. Treatment of authorized use sites- Pesticides will be applied on authorized use sites (Risk-plot)		**x**
3. Annual maximal exposures– the risk calculation only considers the likelihood of exposure to maximum annual values (e.g., 24-hr EEC). It does not account for effects over the full effective range of predicted exposures (Risk-plot)	**x**	
4. GIS data layers accurately represent the presence and absence of use sites (pesticide/species overlap analysis)	**x**	**x**
5. Exposure to multiple stressors does not increase risk – The risk estimates or information do not account for other real world stressors known to exacerbate response (e.g., temperature, other pesticides, etc.) (Risk-plot)	**x**	
6. Species surrogacy – The sensitivity of endangered species and their prey to pesticide exposure is comparable to that of available surrogate species (Risk-plot)	**x**	**x**
7. Exposure estimates accurately predict pesticide concentrations in habitats relevant to ESA-listed species (EECs,	**x**	**x**

Assumption (estimate)	Underestimate Risk	Overestimate Risk
Risk-plot)		
8. Responses to pesticides that degrade over time in the environment can be accurately predicted using toxicity data generated under test conditions that maintain concentrations at relatively constant concentrations (EECs, Risk-plot, Mixtures).	**x**	**x**
9. Effects to essential behaviors are assumed to have fitness consequences regardless of the presence/absence of a quantitative link to an apical endpoint (mortality, reproduction, or growth).	**x**	**x**

1. Pesticide application rate assumptions tend to **overestimate** risk: Exposure estimates generated by EPA using fate and transport models assume the pesticides are applied at the highest labeled rate for a particular crop, crop grouping, or other use site. This assumption contributes to the possibility that exposure and risk will be overestimated because applications may occur at lower than maximum rates. However, EPA's action encompasses all uses authorized by approved product labels, so this assumption is needed to determine whether label requirements are sufficient to insure jeopardy is not likely for ESA-listed species and destruction or adverse modification is not likely for designated critical habitat (NAS 2013a). Note, however, the RPA mitigation (described below) does account for the typical or actual application rate in determining the appropriate mitigation level.

2. Treatment of authorized use sites assumptions tend to **overestimate** risk**:** Similar to the pesticide application rate assumptions, this assumption allows us to evaluate the full extent of EPA's authorized approval of pesticide use based on labels. If EPA has authorized pesticide application for a particular site, that site may receive one or more pesticide applications during the course of the 15-year action. An important consideration in estimating the likelihood of exposure to a species is the extent to which authorized use sites occur in direct proximity to species habitats and/or known occupied areas. However, we do not assume that usage will occur to every authorized use site nor do we assume that all usage occurs at the same day and time. Instead, we assume that: 1) the pesticide may be applied to any authorized site; and 2) the greater the extent of authorized use sites in the species range equates to a greater chance that application may occur in close proximity to species habitat. The distinction between "will be applied to every" and "may be applied to any," is important in understanding the assumptions underlying our analysis. The assumption that every use site will receive application is not realistic nor appropriate. While we do not expect every site to be treated, it is imperative to consider the potential responses to treatments that may occur in close proximity to ESA-listed

species locations to insure existing controls (i.e., product labeling) are adequate to avoid jeopardy and adverse modification. Usage data (data on past use) are not available at a useful scale to predict exposure to threatened and endangered species. The proximity of pesticide use relative to the ESA-listed species is a much more important driver of risk than the percent of treated crop over a large area (e.g., a state). Additionally, the existing usage information has significant data gaps for agricultural crops and non-agricultural uses and is based on limited geographical sampling. Also, NMFS does not have adequate information on sources of usage data (particularly the proprietary sources) to assess their accuracy. Finally, pesticide usage is highly variable over time and we cannot reliably predict the changes in usage that will occur during the 15-year duration of the action. More details on these uncertainties and limitations can be found in Appendix D.

3. Annual maximum exposures assumptions tend to **underestimate** risk: Risk-plots display annual time-weighted average concentrations. However, exposure to lesser concentrations (submaximal) can also contribute to risk. While the maximum daily peak occurs one day a year, toxic residues may persist for days, weeks, or months due to repeated applications of pesticides and their persistence. The assumption of annual maximum exposures omits the entire range of exposures expected to cause mortality and other effects, and thus contributes to the likelihood that risk will be underestimated. Therefore, to mitigate the impact of this assumption, chemical persistence and the number of applications allowed were adopted as factors in our analysis to weigh the likelihood of exposure.

4. GIS data layer assumptions may **overestimate or underestimate** risk: Our analysis relies on GIS data layers representing land use classifications, which we use as surrogates for locations where pesticides can be applied (pesticide use sites). Four issues arise that may contribute to an over- or under-estimate of risk.
    a. Accuracy of data layers. The GIS data layers contain many inaccuracies and local knowledge suggests that land use type is frequently misclassified. The extent of the inaccuracies is uncertain because information quantifying the level of inaccuracy was not available.
    b. Overlapping data layers. In some cases, data layers for use sites overlap. The overlap may be due to a valid overlap in uses on a single site. For example, it is relatively common to plant more than one crop at a single location during the course of a year (double cropping).
    c. Data layer availability. The Cropland Data Layer used to identify locations of crop use sites does not include coverage for Alaska, Hawaii, and other lands under U.S. jurisdiction. We used the National Land Cover Database to identify cropland in Alaska and Hawaii. Additionally, we used regional data as surrogates to approximate the magnitude of EECs for pesticide use on U.S. territorial islands (e.g., Southeastern US-HUC3 for the Caribbean; Hawaii-HUC20 for Pacific islands).
    d. Changes in time. All of the GIS data layers (e.g., NLCD and CDL) are a snapshot in time based on the year(s) they cover. For agricultural locations, aggregating multiple years of CDLs and multiple CDLs to create some UDLs (e.g., vegetables and ground fruit) was used to partially address this uncertainty (e.g., crop

rotation). Nonetheless, potential changes in the data layers over the 15-year duration of the action mean that assumptions based on specific year(s) of data may over- or under-estimate risk in the future.

Overall, these different kinds of inaccuracy in GIS data would not tend to systematically over- or under-estimate risk, and we assumed these 4 sources of uncertainty could contribute equally to the likelihood of underestimating or overestimating exposure. When data layers were not available to evaluate the presence/absence of use sites, we expressed low confidence in risk estimates.

5. Assumption that exposure to multiple stressors will not increase risk may **underestimate** risk: The information summarized in the Risk-plots does not account for other real world stressors known to exacerbate responses to carbamate insecticides (e.g., exposure to other pesticides). This assumption contributes to the likelihood that risk will be underestimated. To account for potential increases in risk associated with multiple stressors, we evaluated the available information supporting risk hypotheses that (a) elevated temperatures could enhance the toxicity of pesticides in listed coldwater fishes, and (b) pesticide mixtures applied as multi-a.i. formulations or tank mixtures could increase risk from direct and indirect effects for the ESA-listed species. Exposure to temperature stress was evaluated based on the occurrence of impaired water quality due to exceedance of temperature thresholds (Clean Water Act section 303(d) listings) in the habitat of the ESA-listed species. The mixtures' risk hypotheses were evaluated qualitatively by generating exposure and response estimates for examples of multi-a.i. pesticide formulations and tank mixtures as described in the Effects of the Action below.

6. Species surrogacy assumptions may **underestimate or overestimate** risk: In most instances, the sensitivity of endangered species and their prey to the stressors of the action have not been tested; their sensitivities are assumed to be comparable to surrogate species that have been tested. These assumptions may underestimate or overestimate risk, depending on the relative sensitivity of the species. Species surrogacy represents a large source of uncertainty because sensitivities among even closely related species can span several orders of magnitude and, frequently, extrapolations across taxa groups are necessary due to the absence of information with closely related species (e.g., the BEs include extrapolations of toxicity response data from mallard duck to estimate responses in sea turtles because they sometimes represented the nearest taxonomic relation with available information). EPA's BEs summarized the range of available toxicity data for different taxa as data arrays (e.g., LC50s for mortality and LOECs for sublethal endpoints). When enough data were available, SSDs were used to describe the variability in sensitivity among species to pesticides by utilizing empirical toxicity data and fitting them to a distribution curve. For example, Figure 102 shows an SSD curve derived for malathion based on variability in toxicity of malathion among saltwater invertebrate species (y-axis, LC50s; EPA 2017c). Species in the figure with corresponding quantile values of >0.5 (x-axis) are less sensitive than the median, or 50th percentile of the distribution. When EPA had sufficient data to develop SSDs (e.g., fish mortality endpoints), the 5th percentile in sensitivity was used to generate output for the Risk-plots (i.e., we selected a values that suggested a 95% probability that toxicity to species would

not be underestimated). For endpoints with too little data available to generate an SSD (e.g., sublethal responses), the range of available data was considered (e.g., behavior LOECs) with an emphasis on the greatest sensitivity (e.g., lowest behavior LOEC). In either case, we considered the best scientific and commercial data available and designed the analysis to ensure we captured any potential for effects to our listed species.

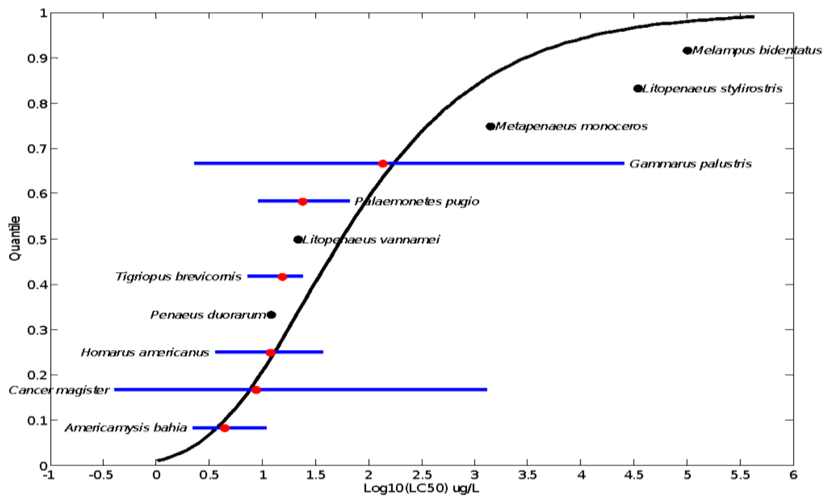


Figure 102. Species Sensitivity Distribution (SSD) example from the malathion BE (EPA 2017c). Log-Gumbel distribution fit to malathion saltwater invertebrate data. Black points indicate single toxicity values. Red points indicate average of multiple toxicity values for a single species. Blue line indicates full range of toxicity values for a given species.

7. Exposure estimate assumptions may **underestimate or overestimate** risk: EPA developed estimates for the aquatic habitat with the PWC model (an integration of PRZM5 and the VVWM), as described in their BE (EPA 2021a; EPA 2021c). The output generated using Risk-plot relies on the EEC estimates generated with the PWC model and AgDRIFT. The accuracy of the exposure estimates depends on how well model inputs represent site-specific conditions. Below are several factors that NMFS considered in assessing how well modeled EECs represent actual exposures and, therefore, how much confidence to place on them in the Risk Characterization.
    a. EPA generated geographically-specific EECs for a variety of aquatic habitats for all HUC2 regions in the U.S. A substantial amount of variability in environmental conditions occurs at the HUC2 scale that influences exposure. While a more refined HUC scale may be possible, we expect the range of values to be comparable to those generated at the HUC2 scale. For this reason, and others, it was not deemed appropriate, necessary, or feasible for the analysis given the nation-wide scale of the assessment that includes all federally-listed endangered and threatened species. Input variables were selected to represent sites vulnerable to runoff within the region as described in EPAs BEs (EPA 2017 a, b, c). While alternative inputs (e.g., selection of a different meteorological station) could result in higher or lower EECs, overall the Service agreed with EPA that the approach was appropriate to evaluate the likelihood of fitness level impacts at the individual scale for the 15-year action (Step 2 described within the EPA's BEs). The models

are designed to predict pesticide concentrations in aquatic habitats on the edge of a treated field considering buffers required by labeling. We expect the models to provide reasonable estimates of exposure in habitats located in close proximity to treated areas, particularly when the size of the assumed drainage area is comparable with the size of single spray applications (e.g., smaller drainages areas such as by the generic "farm pond"). While inputs are weighted to generate estimates at the higher end of the exposure range within the region, it's possible that exposure is underestimated for some sites (e.g., those that receive greater rainfall than assumed, or site with soil characteristics more conducive to runoff). These modeled estimates represent the best scientific and commercial data available for anticipated concentrations resulting from applications, and, overall we expect the EECs to provide reasonably accurate estimates with a tendency to overestimate exposure. However, we do not equate any one modeled scenario directly to any one habitat. Instead, we consider all of the information available (including monitoring data, habitat type, etc.) in order to define the most appropriate range of exposure values for comparison against toxicological data.

b. There is much greater uncertainty with regard to extrapolating estimates generated using "index reservoir" because the physical parameters modelled are not characteristic of many streams, rivers, and marine habitats; Rivers have much larger watersheds that would include multiple land uses, use sites, and areas where use may not be permitted. The assumption that all of the use sites within these large watersheds are treated with pesticides tends to overestimate risk (assumption #2 above). Conversely, using a watershed model that averages concentrations over such large areas (e.g., those derived for catchments) does not account for potential variation within the watershed and likely underestimates risk when individuals are distributed in close proximity to use sites (see #3 above).
c. Even greater uncertainty exists for marine habitats where model estimates that account for complex currents and tidal exchange are not available. While the modelled generic freshwater habitats can serve as surrogates to estimate exposure in marine and estuarine habitats, we feel the values derived for these bins are most likely overestimates of exposure for most habitats given the potential for dissipation due to tidal action and the dilution potential of deeper water habitats. Therefore, Risk-plot exposure estimates were given little weight in drawing conclusions in our evaluation of population level effects associated with species in marine habitats.
d. Uncertainty also exists regarding estimates of exposures following applications to developed locations (i.e., the Developed UDL). The available models such as the PWC are designed for modelling transport to habitats resulting from applications to agricultural sites. In contrast to applications to agricultural sites (e.g., the entire field of a crop), applications to developed land typically involves a small piece of a larger area (e.g., the foundation of a building). EPA's modeling for Developed uses acknowledged this limitation in the modeling and adjusted the estimates as an attempt to account for the situation. These EECs may still be overestimates of exposure, but more appropriate models for this use site do not exist. NMFS recognized the greater uncertainty in EECs associated with Developed uses and

reduced the confidence we had in exposures resulting from applications to those locations.

e. Uncertainties around the inputs to any model will also affect the confidence in modelled outputs. For EPA's models this includes uncertainties associated with a chemical's fate parameters (e.g., Kow). For EPA's exposure estimates, NMFS relied on EPA's evaluation of the range of possible input parameters associated with a chemical's physical properties and selection of appropriately conservative values (as presented in their BEs). NMFS also relied on EPA's evaluation of the available studies to determine which are of appropriate quality to consider for inclusion as an input value to model pesticide transport. NMFS acknowledges there are added uncertainties introduced in the EECs in cases where input values are uncertain (e.g., different values available for input parameters). These choices have the potential to contribute to overestimates, or underestimates in exposure.

8. The assumption that field and laboratory exposure result in comparable responses may **underestimate or overestimate** risk: Standardized laboratory toxicity tests typically require that pesticide concentrations be maintained at a relatively stable concentration for the duration of the exposure period. In the natural environment, pesticides continue to degrade and dissipate at varying rates depending on site-specific conditions and the pesticide's physical-chemical properties. The conventional approach for handling the uncertainty associated with the differing exposure patterns was assumed; exposure estimates using time-weighted average (TWA) concentrations that factor in degradation and dissipation were assumed to produce similar responses to toxicity test conducted under relatively constant exposure concentrations conducted with comparable exposure durations. TWA exposure estimated for acute durations (1d and 4d) were used to estimate responses based on acute toxicity studies and TWA estimates for chronic durations (21-d) were used to estimate responses using chronic studies. Utilizing average concentrations estimated under natural conditions can either underestimate or overestimate risk because response is a function of both exposure duration and concentration. Actual response may vary depending on site-specific dissipation pattern and toxicokinetic factors. For example, cholinesterase inhibition occurs rapidly, particularly with carbamate insecticides, and adverse responses can occur with exposure at much shorter durations than the standard 96-hr duration acute study used to estimate lethality in fish. Consequently, a 4-day TWA may underestimate lethality if exposure for shorter durations is sufficient to elicit mortality. Given the rapid onset of AChE-inhibition, the primary mechanism of action of the two pesticides, we used both 1-day and 4-day TWAs to evaluate responses to acute exposures.

9. Assumptions on lack of information empirically linking effect endpoints with fitness level consequences may **underestimate or overestimate** risk: An adverse outcome pathway establishing causal links from the molecular level to individual and population level effects exists for these and other AChE-inhibiting compounds. Carbaryl and methomyl inhibit AChE, which interferes with normal nervous system transmission, and has been linked to behavioral, reproductive, and lethal effects. Yet, these links frequently do not provide the information needed to predict the degree to which the "apical endpoints" of growth, reproduction, and survival may be impaired. Sublethal effects to

essential behaviors, such as impacts to a fish's ability to swim or a bird's ability to fly, can clearly translate to fitness level consequences by impairing an individual's ability to feed, escape predation, migrate, etc. If information is lacking to establish the degree to which impacts to a fish's ability to swim impacts its ability to survive and reproduce, we can either assume the apical endpoints will not be impacted and likely underestimate the risk, or we can assume they will impact individual fitness, which may overestimate risk. To ensure protection of the species, we logically inferred that impacts to a species' essential behaviors (e.g., effects on the ability of salmon to feed, escape predation, migrate, home, osmoregulate, etc.) and impacts to the availability of food were capable of producing fitness level consequences regardless of the presence of empirical studies quantitatively linking these assessment measures to an apical endpoint. However, the uncertainty in sublethal effects translating into impacts to the species fitness is factored into final conclusions by downgrading the confidence in the effect.

10.7 Pesticide Mixtures

Consideration of the toxicity resulting from exposure to pesticide mixtures is an important part of the Effects Analysis of this opinion. This is due in part to the identified need to consider all effects of the action when making jeopardy determinations and establishing RPAs/RPMs. Pesticide mixtures are explicitly permitted on EPA-authorized product labels and are, therefore, part of the action under consultation here. Additionally, monitoring data showing that pesticide mixtures are common in aquatic habitats throughout the United States (Bradley et al. 2017; Covert et al. 2020; Gilliom et al. 2007) supports the expectation that ESA-listed species are likely to be exposed to complex pesticide mixtures. Failing to consider mixtures may underestimate pesticide risk to such an extent as to lead to erroneous conclusions and ineffective protections for listed species.

Pesticide mixtures can be organized into 3 categories. The first category is formulated products, which are produced and sold as 1 product containing multiple a.i.s. Since the exact types and application rates of the a.i.s are shown on the product labels, it is possible to predict the resulting aquatic concentrations following their use. Several formulated products containing carbaryl or methomyl have been identified as part of this action and are shown in Table 135. However, due to the uncertainty of the resulting environmental concentrations produced from labeled use, quantitative examples of mixture toxicity were not developed for these formulated products. The second category, environmental mixtures, result from unrelated pesticide use over the landscape and are typically detected in ambient water quality monitoring efforts (Bradley et al. 2017; Covert et al. 2020; Gilliom et al. 2007).

Quantitative examples of toxicity from environmental mixtures containing carbaryl or methomyl were not developed here. The final category, tank mixes, refer to a situation where the pesticide user applies multiple pesticides simultaneously at the use site. Tank mixes are explicitly allowed on product labels and their use is often encouraged to increase pesticide efficacy. Tank mixes containing carbaryl or methomyl and other a.i.s on various crops occurred frequently, as reported in the California Pesticide Use Reporting Database (PUR 2018). The 3 quantitative examples of coapplication that are presented in the examples below were reported in the 2018 PUR data.

Table 135. Formulated products containing carbaryl or methomyl.

REGISTRATION #	NAME	PERCENT ACTIVE INGREDIENT	ACTIVE INGREDIENT
4-122	BONIDE A COMPLETE FRUIT TREE SPRAY	11.76	Captan
		6.00	Malathion
		0.30	Carbaryl
4-458	COPPER DRAGON TOMATO & VEGETABLE DUST	7	Basic copper sulfate
		2	Carbaryl
4-474	BONIDE VEGETABLE-FLORAL DUST	13.72	Basic copper sulfate
		1.25	Carbaryl
8119-5	CORRY'S SLUG, SNAIL & INSECT KILLER	5	Carbaryl
		2	Metaldehyde
9198-234	THE ANDERSONS BICARB LAWN INSECT KILLER GRANULES	0.058	Bifenthrin
		2.3	Carbaryl
9198-235	THE ANDERSONS BICARB INSECTICIDE + FERTILIZER	0.058	Bifenthrin
		2.3	Carbaryl
71096-18	GET-A-BUG SNAIL, SLUG & INSECT KILLER	5	Carbaryl
		2	Metaldehyde
2724-274	GOLDEN MALRIN RF-128 FLY KILLER	1	methomyl
		0.049	cis-9-Tricosene
7319-6	LURECTRON SCATTERBAIT	1	methomyl
		0.026	cis-9-Tricosene
53871-3	STIMUKIL FLY BAIT	1	methomyl
		0.04	cis-9-Tricosene

Quantitative examples of risk from 3 tank mix scenarios were generated using usage information on current product labels, fish toxicity values (i.e., LC_{50}s), and exposure concentrations (EECs) presented earlier in this opinion and summarized below in Table 136. The mixture toxicity predictions presented here use mortality as the endpoint. For all chemicals, a standard probit slope of 4.5 was used to describe the concentration-response relationship. Using calculated

EECs, standard slope, and reported LC_{50} values, the percent mortality resulting from each ingredient of the tank mix was calculated using the following equation (in Microsoft Excel):

$$\textit{percent mortality single chemical} = \textit{NORMDIST}((\textit{slope} * (\log(\textit{EEC})-\log(LC_{50}))))$$

The first 2 examples contain pesticides of different classes (i.e., carbamates and pyrethroids) with dissimilar modes of action, therefore calculations of combined toxicity utilized the following response-addition equation:

$$\textit{mixture toxicity} = ((\textit{mortality A} + \textit{mortality B}) - (\textit{mortality A} * \textit{mortality B}))$$

The third example contains a tank mixture of the pesticides carbaryl and methomyl, which are both carbamates. Therefore, calculations of combined toxicity utilized the following concentration-addition equation:

$$\textit{mixture toxicity} = \textit{NORMDIST}((\textit{slope}*(\log(\textit{combined relative } LC_{50}))$$

As shown in Table 136, tank mixes of carbaryl and esfenvalerate applied to carrots produced elevated toxicity compared to applications of carbaryl alone. The resulting combined toxicity is driven by esfenvalerate, which alone may produce up to 100% mortality in fish. Similarly, tank mixes of methomyl and lamda-cyhalothrin applied to lettuce are predicted to produce 100% mortality in fish, again with the resulting toxicity being driven by the pyrethroid constituent of the mixture. The final tank mixture of carbaryl and methomyl represents airblast application to citrus crops, with the required 25-foot buffer factored into the EEC calculations. Here each of the chemicals on its own is predicted to produce only up to 2.5% mortality, while the mixture is predicted to produce just over 10% mortality in exposed fish. These quantitative estimates of risk show greater toxicity than what is expected from use of either carbaryl or methomyl alone. Therefore, the co-application of multiple pesticides increases the potential for adverse impacts and increased risk to threatened and endangered species. The magnitude of this toxicity, using mortality as the endpoint, is in some cases expected to adversely impact the health of listed fish.

Table 136. Predicted tank mixture toxicity to fish.

Pesticide	EEC (ppb)	LC_{50} (ug/l)	Single Chemical Toxicity (percent mortality)	Mixture Toxicity (percent mortality)
Carbaryl	48.6	1055.4[1]	0	100
Esfenvalerate	1.3	0.142[2]	100	
Methomyl	29	335[1]	0	100
Lambda-cyhalothrin	1	0.029[2]	100	

Carbaryl	382.83[3]	1055.4	2.37	10.2
Methomyl	75.26[3]	335	0.02	

[1]HC05 values from EPA's BE for carbaryl and methomyl
[2]LC50 values from EPA 2016 Pyrethroid Registration Review
[3]Applied to citrus crops by airblast

We did not perform a robust, quantitative mixtures analysis of all the formulated products, tank mixtures, and environmental mixtures containing carbaryl or methomyl across the geographic and temporal scope of this opinion. This is due in part to inconsistent reporting of the frequency and magnitude of mixture concentrations, the large nationwide scale of this action, and the long temporal duration of the action (i.e., 15-year registration). Therefore, following NRC guidance (NRC, 2013), this opinion utilizes a qualitative mixtures analysis that contains examples of quantitatively-estimated mixture toxicity. Despite long-standing uncertainties regarding mixtures (e.g., their temporal and geographic extent, understanding combined biological effects), it remains reasonable to conclude that exposure to pesticide mixtures containing carbaryl or methomyl, especially when applied with other neurotoxic pesticides, poses a threat to listed aquatic species. Our overall qualitative analysis of mixtures supports the stated mixtures risk hypothesis.

11 Species Effects Analysis: Carbaryl

In this section, we summarize the results of the carbaryl effects analysis for ESA-listed species. The species-specific analyses themselves can be found in Attachment 3 of this document.

The effects analysis integrates the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action at the population and species level. The information is organized by species. See Chapters 3 (Assessment Framework) and 10 (Effects Analysis Introduction) for descriptions of the methods and information used in the effects analysis.

Table 137. Carbaryl species effects analysis conclusions

Common Name	Scientific Name	Risk	Confidence
Atlantic salmon, Gulf of Maine ESU	*Salmo salar*	Low	Medium
Chum salmon , Columbia River ESU	*Oncorhynchus keta*	Medium	Medium
Chum salmon, Hood Canal summer-run ESU	*Oncorhynchus keta*	Medium	Medium
Chinook salmon, California coastal ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Central Valley spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Lower Columbia River ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Puget Sound ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Sacramento River winter-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River fall-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River spring/summer run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Columbia River spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Willamette River ESU	*Oncorhynchus tshawytscha*	Medium	Medium

Coho salmon, Central California coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Lower Columbia River ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Oregon coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, S. Oregon and N. Calif coasts ESU	*Oncorhynchus kisutch*	Medium	Medium
Sockeye, Ozette Lake ESU	*Oncorhynchus nerka*	Low	Medium
Sockeye, Snake River ESU	*Oncorhynchus nerka*	Medium	Medium
Steelhead, California Central Valley DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Lower Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Middle Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Northern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Puget Sound DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Snake River Basin DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, South-Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Southern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Willamette River DPS	*Oncorhynchus mykiss*	Medium	Medium
Eulachon, Pacific smelt, Southern DPS	*Thaleichthys pacificus*	Medium	Medium
Green sturgeon, Southern DPS	*Acipenser medirostris*	Medium	Medium
Shortnose sturgeon	*Acipenser brevirostrum*	Medium	Medium

Atlantic sturgeon, Carolina DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Chesapeake Bay DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Gulf of Maine DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, New York Bight DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, South Atlantic DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Gulf sturgeon	*Acipenser oxyrinchus desotoi*	Low	Medium
Yelloweye rockfish	*Sebastes ruberrimus*	Low	Medium
Boccacio, Puget Sound/Georgia Basin	*Sebastes paucispinis*	Medium	Low
Nassau grouper	*Epinephelus striatus*	Low	High
Smalltooth sawfish, U.S. DPS	*Pristis pectinata*	Medium	Medium
Giant manta ray	*Manta birostris*	Low	High
Black abalone	*Haliotis cracherodii*	Medium	Low
White abalone	*Haliotis sorenseni*	Low	High
Sunflower sea star	*Pycnopodia helianthoides*	Medium	Low
Queen conch – Proposed	*Alger gigas*	Low	Medium
Staghorn coral – Proposed	*Acropora cervicornis*	Medium	Low
Elkhorn coral	*Acropora palmata*	Medium	Low
Coral, no common name	*Acropora globiceps*	Low	Medium
Coral, no common name	*Acropora retusa*	Low	Medium
Coral, no common name	*Acropora speciosa*	Low	Medium
Coral, no common name	*Euphyllia pardivisa*	Low	Medium
Coral, no common name	*Isopora crateriformis*	Low	Medium

Boulder star coral	*Orbicella franksi*	Medium	Low
Lobed star coral	*Orbicella annularis*	Medium	Low
Mountainous star coral	*Orbicella faveolata*	Medium	Low
Pillar coral	*Dendrogyra cylindrus*	Medium	Low
Rough cactus coral	*Mycetophyllia ferox*	Medium	Low
Killer whale, Southern Resident DPS	*Orcinus orca*	Medium	Medium

12 Species Effects Analysis: Methomyl

In this section we summarize the results of the methomyl effects analysis for ESA-listed species. The species-specific analyses can be found in Attachment 3 of this document.

The effects analysis integrates the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action at the population and species level. The information is organized by species. See Chapters 3 (Assessment Framework) and 10 (Effects Analysis Introduction) for descriptions of the methods and information used in the effects analysis.

Table 138. Methomyl species effects analysis conclusions

Common Name	Scientific Name	Risk	Confidence
Atlantic salmon, Gulf of Maine ESU	*Salmo salar*	Low	Medium
Chum salmon , Columbia River ESU	*Oncorhynchus keta*	Low	Medium
Chum salmon, Hood Canal summer-run ESU	*Oncorhynchus keta*	Low	Medium
Chinook salmon, California coastal ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Central Valley spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Lower Columbia River ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Puget Sound ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Sacramento River winter-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River fall-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River spring/summer run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Columbia River spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Willamette River ESU	*Oncorhynchus tshawytscha*	Medium	Medium

Coho salmon, Central California coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Lower Columbia River ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Oregon coast ESU	*Oncorhynchus kisutch*	Low	Medium
Coho salmon, S. Oregon and N. Calif coasts ESU	*Oncorhynchus kisutch*	Medium	Medium
Sockeye, Snake River ESU	*Oncorhynchus nerka*	Medium	Medium
Steelhead, California Central Valley DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Lower Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Middle Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Northern California DPS	*Oncorhynchus mykiss*	Low	Medium
Steelhead, Puget Sound DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Snake River Basin DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, South-Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Southern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Willamette River DPS	*Oncorhynchus mykiss*	Medium	Medium
Eulachon, Pacific smelt, Southern DPS	*Thaleichthys pacificus*	Low	Medium
Green sturgeon, Southern DPS	*Acipenser medirostris*	Medium	Medium
Shortnose sturgeon	*Acipenser brevirostrum*	Medium	Medium
Atlantic sturgeon, Carolina DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Chesapeake Bay DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Gulf of Maine DPS	*Acipenser oxyrinchus oxyrinchus*	Low	Medium

Atlantic sturgeon, New York Bight DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, South Atlantic DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Gulf sturgeon	*Acipenser oxyrinchus desotoi*	Low	Medium
Yelloweye rockfish	*Sebastes ruberrimus*	Low	Medium
Boccacio, Puget Sound/Georgia Basin	*Sebastes paucispinis*	Medium	Low
Nassau grouper	*Epinephelus striatus*	Low	High
Smalltooth sawfish, U.S. DPS	*Pristis pectinata*	Medium	Medium
Giant manta ray	*Manta birostris*	Low	High
Black abalone	*Haliotis cracherodii*	Low	Medium
White abalone	*Haliotis sorenseni*	Low	High
Sunflower sea star – Proposed	*Pycnopodia helianthoides*	Medium	Low
Queen conch – Proposed	*Alger gigas*	Low	High
Staghorn coral	*Acropora cervicornis*	Medium	Low
Elkhorn coral	*Acropora palmata*	Medium	Low
Coral, no common name	*Acropora globiceps*	Low	Medium
Coral, no common name	*Acropora retusa*	Low	Medium
Coral, no common name	*Acropora speciosa*	Low	Medium
Coral, no common name	*Euphyllia pardivisa*	Low	Medium
Coral, no common name	*Isopora crateriformis*	Low	Medium
Boulder star coral	*Orbicella franksi*	Medium	Low
Lobed star coral	*Orbicella annularis*	Medium	Low
Mountainous star coral	*Orbicella faveolata*	Medium	Low
Pillar coral	*Dendrogyra cylindrus*	Medium	Low
Rough cactus coral	*Mycetophyllia ferox*	Medium	Low
Killer whale, Southern Resident DPS	*Orcinus orca*	Medium	Medium

13 Critical Habitat Effects Analysis: Carbaryl

In this section, we summarize the results of the carbaryl effects analysis for designated critical habitats. The habitat-specific analyses for each species can be found in Attachment 3 of this document.

The effects analysis integrates the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action. The information is organized by species habitat. See Chapters 3 (Assessment Framework) and 10 (Effects Analysis Introduction) for descriptions of the methods and information used in the effects analysis.

Table 139. Carbaryl critical habitat effects analysis conclusions

Critical Habitat Unit	Scientific Name	Risk	Confidence
Atlantic salmon, Gulf of Maine ESU	*Salmo salar*	Low	Medium
Chum salmon , Columbia River ESU	*Oncorhynchus keta*	Medium	Medium
Chum salmon, Hood Canal summer-run ESU	*Oncorhynchus keta*	Medium	Medium
Chinook salmon, California coastal ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Central Valley spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Lower Columbia River ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Puget Sound ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Sacramento River winter-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River fall-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River spring/summer run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Columbia River spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Willamette River ESU	*Oncorhynchus tshawytscha*	Medium	Medium

Coho salmon, Central California coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Lower Columbia River ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Oregon coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, S. Oregon and N. Calif coasts ESU	*Oncorhynchus kisutch*	Medium	Medium
Sockeye, Ozette Lake ESU	*Oncorhynchus nerka*	Low	Medium
Sockeye, Snake River ESU	*Oncorhynchus nerka*	Medium	Medium
Steelhead, California Central Valley DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Lower Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Middle Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Northern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Puget Sound DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Snake River Basin DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, South-Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Southern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Willamette River DPS	*Oncorhynchus mykiss*	Medium	Medium
Eulachon, Pacific smelt, Southern DPS	*Thaleichthys pacificus*	Medium	Medium
Green sturgeon, Southern DPS	*Acipenser medirostris*	Medium	Medium
Atlantic sturgeon, Carolina DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium

Atlantic sturgeon, Chesapeake Bay DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Gulf of Maine DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, New York Bight DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, South Atlantic DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Yelloweye rockfish	*Sebastes ruberrimus*	Low	Medium
Boccacio, Puget Sound/Georgia Basin	*Sebastes paucispinis*	Medium	Low
Smalltooth sawfish, U.S. DPS	*Pristis pectinata*	Medium	Medium
Black abalone	*Haliotis cracherodii*	Medium	Low
Nassau grouper	*Epinephelus striatus*	Low	Medium
Caribbean Corals	*Orbicella annularis, Orbicella faveolata, Orbicella franksi, Dendrogyra cylindrus, Mycetophyllia ferox*	Medium	Low
Indo-Pacific Corals*	*Acropora globiceps, Acropora retusa, Acropora speciosa, Euphyllia paradivisa, Isopora crateriformis*	Medium	Low
Killer whale, Southern Resident DPS	*Orcinus orca*	Medium	Medium

**Proposed critical habitat*

14 Critical Habitat Effects Analysis: Methomyl

In this section, we summarize the results of the methomyl effects analysis for designated critical habitats. The habitat-specific analyses for each species can be found in Attachment 3 of this document.

The effects analysis integrates the exposure and response information to evaluate the likelihood of adverse effects from stressors of the action. The information is organized by species. See Chapters 3 (Assessment Framework) and 10 (Effects Analysis Introduction) for descriptions of the methods and information used in the effects analysis.

Table 140. Methomyl critical habitat effects analysis conclusions

Critical Habitat Unit	Scientific Name	Risk	Confidence
Atlantic salmon, Gulf of Maine ESU	*Salmo salar*	Low	Medium
Chum salmon , Columbia River ESU	*Oncorhynchus keta*	Low	Medium
Chum salmon, Hood Canal summer-run ESU	*Oncorhynchus keta*	Low	Medium
Chinook salmon, California coastal ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Central Valley spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Lower Columbia River ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Puget Sound ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Sacramento River winter-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River fall-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Snake River spring/summer run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Columbia River spring-run ESU	*Oncorhynchus tshawytscha*	Medium	Medium
Chinook salmon, Upper Willamette River ESU	*Oncorhynchus tshawytscha*	Medium	Medium

Coho salmon, Central California coast ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Lower Columbia River ESU	*Oncorhynchus kisutch*	Medium	Medium
Coho salmon, Oregon coast ESU	*Oncorhynchus kisutch*	Low	Medium
Coho salmon, S. Oregon and N. Calif coasts ESU	*Oncorhynchus kisutch*	Medium	Medium
Sockeye, Snake River ESU	*Oncorhynchus nerka*	Medium	Medium
Steelhead, California Central Valley DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Lower Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Middle Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Northern California DPS	*Oncorhynchus mykiss*	Low	Medium
Steelhead, Puget Sound DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Snake River Basin DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, South-Central California coast DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Southern California DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Columbia River DPS	*Oncorhynchus mykiss*	Medium	Medium
Steelhead, Upper Willamette River DPS	*Oncorhynchus mykiss*	Medium	Medium
Eulachon, Pacific smelt, Southern DPS	*Thaleichthys pacificus*	Low	Medium
Green sturgeon, Southern DPS	*Acipenser medirostris*	Medium	Medium
Atlantic sturgeon, Carolina DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium

Atlantic sturgeon, Chesapeake Bay DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, Gulf of Maine DPS	*Acipenser oxyrinchus oxyrinchus*	Low	Medium
Atlantic sturgeon, New York Bight DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Atlantic sturgeon, South Atlantic DPS	*Acipenser oxyrinchus oxyrinchus*	Medium	Medium
Yelloweye rockfish	*Sebastes ruberrimus*	Low	Medium
Boccacio, Puget Sound/Georgia Basin	*Sebastes paucispinis*	Medium	Low
Nassau grouper	*Epinephelus striatus*	Low	Medium
Smalltooth sawfish, U.S. DPS	*Pristis pectinata*	Low	Medium
Black abalone	*Haliotis cracherodii*	Low	Medium
Caribbean Corals	*Orbicella annularis, Orbicella faveolata, Orbicella franksi, Dendrogyra cylindrus, Mycetophyllia ferox*	Medium	Low
Indo-Pacific Corals*	*Acropora globiceps, Acropora retusa, Acropora speciosa, Euphyllia paradivisa, Isopora crateriformis*	Medium	Low
Killer whale, Southern Resident DPS	*Orcinus orca*	Medium	Medium

**Proposed critical habitat*

15 CUMULATIVE EFFECTS

Section Contents

15.1 Introduction

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02). Future Federal actions that are unrelated to the action under consultation are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

This section attempts to identify the likely future changes and their impact on ESA-listed species and their critical habitats in the action area. This section is not meant to be a comprehensive socio-economic evaluation, but a brief outlook on future changes in the environment. Projections are based upon recognized organizations producing best-available information and reasonable rough-trend estimates of change stemming from these data. However, all changes are based upon projections that are subject to error and alteration by complex economic and social interactions.

During this consultation, we searched for information on future state, tribal, local, or private (non-Federal) actions reasonably certain to occur in the action area. The majority of information found has already been described in the Environmental Baseline (Chapter 9), most of which we expect will continue in the future. An increase in these activities could similarly increase their effect on ESA-protected resources and for some, an increase in the future is considered reasonably certain to occur. Given current trends in global population growth, demand for pest control, threats associated with climate change, pollution, fisheries, etc., these trends are likely to continue to increase in the future, although any increase in effect may be somewhat countered by an increase in conservation and management activities.

15.2 Population Growth and Development

From 1970 to 2010, the population in coastal counties increased by 34.8 million people or by approximately 40%, with projections indicating another 8% increase of 10 million people from 2010 to 2020. Population growth will require greater and greater demand on resources, greater demand for food and water, and greater demand for energy. The increase in demand for these essential items are likely to extend pressures on many threatened and endangered species populations and their designated critical habitats. As many cities border coastal or riverine systems, with over 128 million people, or almost 40% of the U.S. population living on the coast, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and runoff from expanding urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from expanding impervious surfaces and existing and additional roadways is typically warmer than natural surface waters and may also contain oil, heavy metals, polycyclic aromatic hydrocarbons, and other chemical pollutants. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for ESA-listed species. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with impaired thermal, dissolved oxygen, and nutrient regimes and impairments by high pollutant concentrations. Continued growth into forested and other natural areas alter landscapes to the detriment of species habitat. Altered landscapes, such as the loss of riparian vegetation along rivers and increases in impervious surfaces, adversely affect the delivery of sediment and gravel and significantly alter stream hydrology and water quality.

A nationwide rise in the population necessitates a rise in agricultural output, and the potential conversion of forested and other natural lands to agriculture. As most of the coastal states have large tracts of irrigated agriculture, this rise in agricultural output is anticipated to affect coastal areas and aquatic species. Impacts from heightened agricultural production will likely result in 2 negative impacts on listed species. The first impact may come from a needed reliance and greater use and application of insecticides, herbicides, and fertilizers resulting in their increased concentrations and entry into freshwater systems. Toxics and other pollutants from agricultural runoff may further degrade habitats supporting listed species. Second, increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce flow rates and alter habitat throughout freshwater systems. Reductions in flows could mean higher water temperatures, and as water is drawn off, contaminants will become more concentrated in these systems, exacerbating toxicity issues in habitats for protected species.

A rise in population will also require pesticide use to protect public health from disease vectors, control invasive species, and maintain public areas such as recreational waters. This can require the application of pesticides at, near, or over waters where the ESA-listed species occur. The residue left by non-agricultural pesticide applications affecting waters of the U.S. are regulated as discharges under state-issued NPDES permits. Discharges of pesticides are also expected to occur in waters not designated as waters of the U.S. such that ESA-listed species will be exposed to pesticide residues from unregulated discharges.

The above issues are likely to pose continuous, unquantifiable, negative effects on listed species addressed in this opinion, particularly freshwater and anadromous species, and those species adapted to and requiring nearshore and estuarine habitats. Each activity has negative effects on water quality. They include increases in sedimentation, increased point and non-point pollution discharges, and decreased infiltration of rainwater resulting in increased runoff into surface waters. Decreased rainwater infiltration leads to decreases in shallow groundwater recharge, decreases in hyporheic flow (e.g., water that spreads laterally beneath river gravels outside the channel where surface flows occur), decreases in summer base flows and elevated temperatures. For example, EPA's National Rivers and Streams Assessment 2013-2014 – Collaborative Survey (USEPA 2020) reported that only 51% of the 186,538 miles of western rivers and streams represented in the survey were in good biological condition based on macroinvertebrate data. These observations did not differ significantly from the 2008-2009 survey. The biological condition of fish communities was significantly lower in the 2013-2014 survey relative to the 2008-2009 survey: Only 38% of fish communities assessed in 126,846 miles of western rivers and streams were found to be in good biological condition. Biological condition is the most comprehensive indicator of water body health. When the biology of a stream is healthy, the chemical and physical components of the stream are also typically in good condition. Nationally, the amount of stream length in good quality for fish condition dropped from 34.8% in 2009 to 26.4% in 2014. Stream lengths in good condition for macroinvertebrate communities was essentially unchanged: with the proportion of assessed stream lengths in good condition at 29.6% in 2009 and 30.2% in 2014.

15.3 Climate Change

The current and past impacts of climate change on species and habitat were discussed in the Environmental Baseline. A set of 5 scenarios were developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections of future emissions or concentrations of GHGs are employed consistently across the various branches of climate science while taking into consideration assumptions of how socio-economic systems could evolve over the course of the 21st century. Scenario uncertainty is explored by assessing alternative socio-economic futures; the 5 scenarios described in IPCC (2021) are based on Shared Socio-economic Pathways (SSPs). These 5 new SSP scenarios cover a broader range of GHG and air pollutant futures than did the previous IPCC scenarios, or representative concentration pathways (RCPs) described in IPCC (2014). Although not directly comparable, both sets of scenarios are labelled by the level of radiative forcing that will be reached in 2100, and modelling studies relying on RCPs complement the assessment based on SSP scenarios (IPCC 2021).

The 5 SSP scenarios start in 2015 and project to either 2100 or 2300, and they include both high-CO2 emissions pathways without climate change mitigation as well as new low-CO2 emissions pathways. Differences in the level of climate change mitigation and air pollution control strongly affect anthropogenic emissions trajectories of short-lived climate forcers (SLCFs) (IPCC 2021). Scenario SSP1-1.9 represents the low end of future emissions pathways, leading to warming below 1.5°C in 2100 and limited temperature overshoot of 1.5°C over the 21st century; it is characterized by very low GHG emissions. Scenario SSP1-2.6 is characterized by low GHG emissions and, along with SSP1-1.9, CO2 emissions declining to net zero around or after 2050 followed by varying levels of net negative CO_2 emissions. Scenario SSP2-4.5 is characterized by intermediate GHG emissions and CO_2 emissions remaining around current levels until the middle of the century. Scenario SSP3-7.0 is characterized by high GHG and CO_2 emissions that will approximately double from the current levels by 2100. Scenario SSP5-8.5 represents the very high warming end of the range of future emissions pathways that have been presented in the literature, and is characterized by very high GHG and CO_2 emissions that will approximately double from the current levels by 2050 (IPCC 2021). SSP2-4.5 and SSP1-2.6 represent scenarios with stronger climate change mitigation and thus lower GHG emissions, but only SSP1-2.6 was designed to limit warming to below 2°C. The Paris Agreement aims to limit the future rise in global average temperature to 2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios (Hayhoe et al. 2018).

Global mean surface temperature, calculated by merging sea surface temperature over the ocean and air temperature 2 meters over land and sea ice areas, is used in most paleo, historical, and present-day observational estimates of global warming (IPCC 2021). Warming greater than the global average has already been experienced in many regions and seasons, with most land regions experiencing greater warming than over the ocean and with the Arctic region warming most rapidly (Allen et al. 2018; IPCC 2021). Global warming has led to more frequent heatwaves in most land regions and an increase in the frequency and duration of marine heatwaves (Allen et al. 2018). Global mean surface temperature has increased by 1.09 (0.95 to 1.20)°C from pre-industrial times to 2011-2020; global temperatures have risen at an unprecedented rate since 2012, with the period from 2016-2020 being the hottest 5-year period between 1850 and 2020 (IPCC 2021). Average global warming up to 1.5°C as compared to pre-industrial levels is expected to lead to regional changes in extreme temperatures, and increases in the frequency and intensity of precipitation and drought (Allen et al. 2018). Projections show that the average global surface temperature during the period from 2081-2100 is very likely to be higher by 1.0°C to 1.8°C under the low CO_2 emissions scenario SSP1-1.9 and by 3.3°C to 5.7°C under the high CO_2 emissions scenario SSP5-8.5 as compared to 1850-1900 (IPCC 2021).

15.4 Regions, Species and Topics of Focus

15.4.1 West Coast

As described in NOAA Fisheries' Western Regional Action Plan (NOAA 2016), marine and estuarine environments along the West Coast will likely experience increased stressors in the future due to climate change and oceanographic regime shifts. Some of the oceanic changes observed to be occurring or expected to continue into the future due to anthropogenic climate change include: a) timing of the onset, duration, and strength of coastal upwelling; b) changes in atmospheric wind patterns that drive ocean circulation, including changes in transport in the California Current that affect the lower trophic levels of the food web; c) increased water column stratification as observed during marine heat waves (including those associated with El Niño events); d) more frequent occurrences of hypoxia; e) pH-related declines in aragonite saturation, which are likely to impact lower, middle, and upper trophic levels of the food web; and f) rising coastal sea level.

Freshwater environments will also experience increased stresses due to changes in physical forcing. The major stressors occurring now that are expected to affect watersheds into the future are: a) increased average air and stream temperature; b) an increased fraction of annual precipitation falling as rain rather than snow; c) a contraction in the snow accumulation season that also comes with reduced springtime mountain snowpack; and d) more natural runoff and stream flow in winter and less snowmelt runoff in spring. Rising temperature alone increases annual water deficits that drive drought stress and moisture content in vegetation. Increasing water deficits on the landscape leads to substantial negative impacts on forests by making them more vulnerable to pests, pathogens, and wild fire. Because snowpack serves as a critical natural reservoir for fresh water in many West Coast watersheds, reduced snowpack typically increases human conflict over already fully or over-allocated freshwater resources. Without management actions that mitigate or resist climate change impacts in freshwater habitats, these changes will very likely diminish the productive capacity of many West Coast watersheds for Pacific salmon and steelhead.

Estuaries experience climate change forcings from the atmosphere, the ocean and the tributary freshwater environments. Changes in estuarine systems due to rising coastal sea level, warming temperatures and altered stream temperature, stream flow timing and volume will cause multiple stresses on anadromous species through habitat modification, changes in primary and secondary production, altered species composition and food-web structure, and changes in fish metabolism.

15.4.2 Northeast

As described in NOAA Fisheries' Northeast Regional Action Plan (Hare et al. 2016), as a result of climate change and natural variability, there have been changes in a number of physical parameters in the Northeast U.S. Shelf over the past 30-40 years (EcoAp 2015) and climate models project that these changes will continue. Air and ocean temperatures are increasing in the Northeast U.S., which can impact organisms, their habitats, and ultimately the human communities that depend on these organisms and habitats. Over the last 2 decades, ocean temperatures in the Northeast have warmed faster than the global ocean. In particular, the Gulf of Maine has warmed faster than 99% of the global ocean. The Northeast U.S. is also a "hotspot" for sea-level rise, with rates in the past 5 decades approximately 3–4 times higher than the global average (Sallenger Jr et al. 2012). Annual precipitation and river flows have increased and the timing of snowmelt is earlier, while the magnitude of extreme precipitation events has also increased (Karl and Knight 1998; McCabe and Wolock 2002; Walsh et al. 2014). Climate projections from global climate models suggest that both temperature and precipitation will increase over time in the Northeast US.

15.4.3 Southeast

As described in NOAA Fisheries' Southeast Regional Action Plan (Gore et al. 2020), there is limited information on large-scale patterns of environmental change that can be attributed to climate change in the Southeast region, due in part to incomplete region-wide ocean observing systems and limited knowledge on the influence of natural long-term variability (Hoegh-Guldberg et al. 2014). The Gulf Stream appears to be weakening along with the broader, related Atlantic Meridional Overturning Circulation (AMOC) (Ezer et al. 2013; Rahmstorf et al. 2015), which may have implications for regional primary and secondary productivity patterns if it results in declines in the magnitude, duration or frequency of Gulf Stream-related upwelling events. The coastline of the Southeast region is dominated by flat coastal marshes in the Carolinas and the limestone landscapes of south Florida. These habitats are vulnerable to flooding and displacement due to inundation. Changes in spatial extent and water quality of estuarine habitats will likely be ecologically significant because of important estuarine dependent species. Increased ocean temperature is expected to have a range of impacts to ocean ecosystems affecting biodiversity redistribution, water quality, physiology, and eutrophication (García Molinos et al. 2016; Holmyard 2014; IPCC 2021). In the Southeast region, sea surface temperature is predicted to increase by as much as 3º C by 2100 (Ingram et al. 2013).

15.4.4 Gulf of Mexico

As described in NOAA Fisheries' Gulf of Mexico Regional Action Plan (Lovett et al. 2016), warming ocean temperatures may have the most wide-ranging effects on aquatic species in the Gulf of Mexico, through both indirect and direct effects. Temperature can affect preference-driven shifts in population distributions (Pinsky et al. 2013; Sydeman et al. 2015). In addition, changes in water temperature and circulation can result in the loss of suitable pelagic habitat. Future responses of marine organisms are challenging to predict because of gaps in knowledge on the physiology and plasticity of many organisms. The recent, relatively rapid increase in sea level rise is predicted to result in seawater inundation of estuaries, coastal flooding, and erosion causing loss of estuaries and freshwater wetlands, with potential negative effects to estuarine species less tolerant of salinity changes and changes in estuarine productivity (Ezer and Atkinson 2014; Martínez Arroyo et al. 2011; Ogden et al. 2005; Zhang et al. 2004). The Gulf of Mexico has been experiencing accelerated losses of saltwater wetlands (95,000 acres from 2004-2009, more than double the loss between 1998-2004 according to Dahl and Stedman (2013)) due primarily to storms, but land subsidence, changes in freshwater inflows (due to diversions or changes in storm patterns), and sea level rise also have been playing a role.

15.4.5 Pacific Salmonids

Continued climate change will increasingly affect Pacific salmon and steelhead in various ways at different points in their life cycle. Because their lifecycle stretches from freshwater rivers into the ocean and back, Pacific salmon face climate-related changes and other challenges in each of those environments.

Salmonids may respond to climate change in a number of different ways including changes in their behavior, morphology (body shape), growth rates, performance, survival, and population growth rate or productivity. They may also adapt by shifting the timing of their runs. Salmon populations migrate through all major river systems on the West Coast in an ordered sequence. Each population has a characteristic run timing that has evolved over thousands of years to ensure that adults can reach their spawning grounds in time to produce the next generation. For example, each Snake River salmon population has its own adaptation to the thermal regime it generally encounters. Summer steelhead have plenty of time before spawning and often use cool-water tributaries to avoid high temperatures in the mainstem river. In contrast, Snake River spring/summer Chinook and sockeye travel directly to their spawning grounds, and delays to migration because of high temperature tend to increase mortality. Among endangered populations in the Pacific Northwest, Snake River sockeye is the most sensitive to temperature, especially in the mainstem rivers. Adults from this population migrate at the warmest time of year through some of the hottest rivers in the region. In addition, declining summer flows exacerbate the risk to Snake River sockeye, especially in the free-flowing Salmon River. In contrast to the Snake River sockeye population, Columbia River sockeye has adapted to rising water temperatures by migrating earlier in summer. These fish shifted their migration period 11 days earlier from the 1950s to the 2010s (Crozier et al. 2011). NMFS believes this change in behavior was caused in part by plastic responses to flow management, but also by natural selection due to temperature increases over this period. Because later migrants became more likely to encounter lethal temperatures, earlier migrants were more common in subsequent generations. For populations to evolve like this, they need to be relatively large and heterogeneous. This is 1 reason why preserving large, diverse, wild populations is vital for natural adaptation to climate change.

Because climate effects in 1 life stage often carry over into subsequent stages, accounting for cumulative effects on salmon is especially challenging and important given their complex life history and migration behavior. For example, population dynamic models, or life-cycle models, are used to calculate the cumulative climate effects that influence salmon extinction risk by integrating multiple climate-related impacts that affect salmon throughout all stages of life. More details about the population dynamic model for Chinook salmon can be found at: https://www.fisheries.noaa.gov/west-coast/climate/extinction-risk-chinook-salmon-due-climate-change. When climate is assumed to be stable, with historical levels of variability, individual simulations showed variability in spawner numbers but, on average, stayed constant over time. However, when climate change was assumed under various emission scenarios, all populations rapidly declined below the quasi extinction threshold. Using the life-cycle model to project changes in the Snake River spring/summer Chinook salmon population for instance, NMFS found that the population will decline dramatically in the coming decades due primarily to rising sea surface temperatures and changes in freshwater temperature and flow. Consistent with these projections, there have been record-low returns of salmon species across the West Coast in response to the marine heat wave of 2014-2016.

We assessed Pacific salmon’s vulnerability to changing climate and ocean conditions to better understand these changes and their impact on different population groups. More details about the report can be found here: https://www.fisheries.noaa.gov/feature-story/west-coast-salmon-vulnerable-climate-change-some-show-resilience-shifting-environment. Salmon have long thrived in the west coast region, proving themselves resilient to past shifts in climate. However, climate is now changing at an unprecedented rate, and most populations now lack access to habitat that once provided refuge from climate extremes. Salmon and steelhead stocks will be affected differently by the environmental shifts expected with climate change, and the assessment examined the stocks' sensitivities to a variety of threats and constraints. The assessment also examined salmon and steelhead population groups for their ability to adapt to climate change, a quality known as “adaptive capacity.” Overall, the West Coast salmon stocks most vulnerable to climate change and its associated environmental shifts (including more extreme high and low flows and hotter oceans and rivers) and least able to adjust to differences are: Chinook salmon in California’s Central Valley; Coho salmon in California and southern

Oregon; Snake River sockeye salmon; and Spring-run Chinook salmon in the interior Columbia and Willamette River basins. Steelhead, pink and chum salmon face less risk, either because they are more adaptable to varying conditions (steelhead) or spend less time in freshwater (pink and chum).

In California, low river flows have posed a challenge for juvenile salmon migrating towards the ocean in spring, with juvenile mortality increasing as river flows decrease (Michel et al. 2021). Water is heavily regulated in California's Central Valley watershed, which exacerbates the problem of already low river flows due to dam construction, water diversions, and diking on the Sacramento River which has reduced the flows from their historic state (Michel et al. 2021). The authors identified 3 key thresholds for stream flow and salmon survival that can serve as targets for the management of water and its associated resources in the Sacramento River (Michel et al. 2021), and discovered that the flow-survival relationship was nonlinear. The "minimum" threshold river flow was 4,259 cubic feet per second (cfs), the "historic mean" threshold river flow was 10,712 cfs, and the "maximum" threshold river flow was 22,872 cfs (Michel et al. 2021). Only 3% of tagged salmon survived the outmigration below the minimum threshold, 18.9% survived between the minimum and historic mean thresholds, 50.8% survived between historic mean and high thresholds, and only 35.3% survived above the high threshold (Michel et al. 2021). The authors concluded that the historic mean threshold was an important target for water resource managers to try to attain.

15.4.6 East Coast Sturgeon

Numerous studies indicate that East Coast sturgeon are and will continue to be impacted by climate change. Gunderson (1998) found that juvenile metabolism and survival were impacted by increasing hypoxia in combination with increasing temperature. Niklitschek and Secor (2005) used a multivariable bioenergetics and survival model to generate spatially explicit maps of potential production in the Chesapeake Bay; a 1^{o}C temperature increase reduced productivity by 65% (Niklitschek and Secor 2005).

A population viability analysis for shortnose sturgeon at the southern end of their range found that salt-water intrusion and decreases in summer dissolved oxygen could reduce population productivity (Jager et al. 2013). In the Hudson River, Woodland and Secor (2007) found that flow volume and water temperature in the fall months preceding spawning were significantly correlated with subsequent year-class strength. Numerous aspects of shortnose sturgeon life history and ecology are linked to temperature, river flow, dissolved oxygen, salinity, but the effect of change in these environmental variables on shortnose sturgeon is unclear (Cech and Doroshov 2005; Ziegeweid et al. 2008a; Ziegeweid et al. 2008b). Habitat models coupled with global climate models for the cogener, European Atlantic Sturgeon (*Acipenser sturio*) indicate strong climate effects throughout the range, especially in the southern portions (Lassalle et al. 2010).

15.4.7 Atlantic Salmon

The effects of climate change on the Gulf of Maine DPS of Atlantic salmon are very likely to be negative. Warming will change freshwater and marine habitats and potentially affect the phenology of Atlantic salmon migration. Ocean acidification could also affect olfaction, which Atlantic salmon use for natal homing. In a review, Jonsson and Jonsson (2009) concluded that the thermal niche of Atlantic salmon will likely shift northward causing decreased production and possibly extinction at the southern end of the range. The Northeast U.S. Shelf Ecosystem represents the southern extent of the range of Atlantic salmon in the Northwest Atlantic Ocean. In a more recent review, Friedland et al. (2014) found that declines in post-smolt survival were associated with ocean warming. Friedland et al. (2014) hypothesized that in the Northwest Atlantic, the decline in survival was a result of early ocean migration by post-smolts. Similarly, Mills et al. (2013) suggested that poor trophic conditions, likely due to climate-driven environmental factors, and warmer ocean temperatures are constraining the productivity and recovery of Atlantic salmon in the Northwest Atlantic. Thus, there is ample evidence that climate change and long-term climate variability will reduce the productivity of the Gulf of Maine DPS of Atlantic salmon.

15.4.8 Corals

Climate change will affect coral reef ecosystems through sea level rise, changes to the frequency and intensity of tropical storms, and altered ocean circulation patterns. As temperatures rise, mass coral bleaching events and infectious disease outbreaks are becoming more frequent. Additionally, carbon dioxide absorbed into the ocean from the atmosphere has already begun to reduce calcification rates in reef-building and reef-associated organisms by altering seawater chemistry through decreases in pH. When combined, all of these impacts dramatically alter ecosystem function, as well as the goods and services coral reef ecosystems provide to people around the globe. Coral communities will start to change when bleaching events occur more than twice in a decade-corals that are susceptible to bleaching will be less common on reefs and the structural complexity of many coral reefs will decline. These changes will occur more rapidly if/when bleaching events begin to occur annually. In the U.S., coral reefs occur in the Indo-Pacific, in the Atlantic, in the Caribbean, and off the Gulf of Mexico.

16 Species Integration and Synthesis: Carbaryl

In this section we summarize the results of the carbaryl integration and synthesis for ESA-listed species. The integration and synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the action. In this section, we add the effects of the action to the environmental baseline and the cumulative effects, in light of the status of the species, to formulate NMFS's conference and biological opinion as to whether EPA was able to insure their action is not likely to jeopardize the continued existence of the species.

The species-specific scorecards with results for each species can be found in the Integration and Synthesis portions of Attachment 3. Consideration of the risk reduction achieved through additional conservation measures described in Chapter 5 is incorporated into the final conclusions as described in the "Impact of Conservation Measures" subsections of the Effects Analysis and Integration and Synthesis chapters in Attachment 3. These final conclusions are presented in the table below in the "With Conservation Measures (Final BiOp Conclusions)" column.

Table 141. Carbaryl species conclusions with conservation measures incorporated

Species Name	With Conservation Measures (Final BiOp Conclusions)
Atlantic salmon, Gulf of Maine ESU	No Jeopardy
Chum salmon , Columbia River ESU	No Jeopardy
Chum salmon, Hood Canal summer-run ESU	No Jeopardy
Chinook salmon, California coastal ESU	No Jeopardy
Chinook salmon, Central Valley spring-run ESU	No Jeopardy
Chinook salmon, Lower Columbia River ESU	No Jeopardy
Chinook salmon, Puget Sound ESU	No Jeopardy
Chinook salmon, Sacramento River winter-run ESU	No Jeopardy
Chinook salmon, Snake River fall-run ESU	No Jeopardy
Chinook salmon, Snake River spring/summer run ESU	No Jeopardy
Chinook salmon, Upper Columbia River spring-run ESU	No Jeopardy
Chinook salmon, Upper Willamette River ESU	No Jeopardy
Coho salmon, Central California coast ESU	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Coho salmon, Lower Columbia River ESU	No Jeopardy
Coho salmon, Oregon coast ESU	No Jeopardy
Coho salmon, S. Oregon and N. Calif coasts ESU	No Jeopardy
Sockeye, Ozette Lake ESU	No Jeopardy
Sockeye, Snake River ESU	No Jeopardy
Steelhead, California Central Valley ESU	No Jeopardy
Steelhead, Central California coast ESU	No Jeopardy
Steelhead, Lower Columbia River ESU	No Jeopardy
Steelhead, Middle Columbia River ESU	No Jeopardy
Steelhead, Northern California ESU	No Jeopardy
Steelhead, Puget Sound ESU	No Jeopardy
Steelhead, Snake River Basin ESU	No Jeopardy
Steelhead, South-Central California coast ESU	No Jeopardy
Steelhead, Southern California ESU	No Jeopardy
Steelhead, Upper Columbia River ESU	No Jeopardy
Steelhead, Upper Willamette River ESU	No Jeopardy
Eulachon, Pacific smelt, Southern DPS	No Jeopardy
Green sturgeon, Southern DPS	No Jeopardy
Shortnose sturgeon	No Jeopardy
Atlantic sturgeon, Carolina DPS	No Jeopardy
Atlantic sturgeon, Chesapeake Bay DPS	No Jeopardy
Atlantic sturgeon, Gulf of Maine DPS	No Jeopardy
Atlantic sturgeon, New York Bight DPS	No Jeopardy
Atlantic sturgeon, South Atlantic DPS	No Jeopardy
Gulf sturgeon	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Yelloweye rockfish	No Jeopardy
Boccacio, Puget Sound/Georgia Basin	No Jeopardy
Gulf grouper	No Jeopardy
Nassau grouper	No Jeopardy
Smalltooth sawfish, U.S. DPS	No Jeopardy
Giant Manta Ray	No Jeopardy
Chambered Nautilus	No Jeopardy
Black abalone	No Jeopardy
White abalone	No Jeopardy
Sunflower sea star – Proposed	No Jeopardy
Queen Conch – Proposed	No Jeopardy
Staghorn coral	No Jeopardy
Elkhorn coral	No Jeopardy
Coral, Acropora globiceps	No Jeopardy
Coral, Acropora jacquelineae	No Jeopardy
Coral, Acropora retusa	No Jeopardy
Coral, Acropora speciosa	No Jeopardy
Coral, Euphyllia pardivisa	No Jeopardy
Coral, Isopora crateriformis	No Jeopardy
Coral, Seriatopora aculeata	No Jeopardy
Boulder star coral	No Jeopardy
Lobed star coral	No Jeopardy
Mountainous star coral	No Jeopardy
Pillar coral	No Jeopardy
Rough cactus coral	No Jeopardy
Green sea turtle, Central North Pacific DPS	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Green sea turtle, Central South Pacific DPS	No Jeopardy
Green sea turtle, Central West Pacific DPS	No Jeopardy
Green sea turtle, East Pacific DPS	No Jeopardy
Green sea turtle, North Atlantic DPS	No Jeopardy
Green sea turtle, South Atlantic DPS	No Jeopardy
Hawksbill sea turtle	No Jeopardy
Kemp's ridley sea turtle	No Jeopardy
Leatherback sea turtle	No Jeopardy
Loggerhead sea turtle, North Pacific Ocean DPS	No Jeopardy
Loggerhead sea turtle, Northwest Atlantic Ocean DPS	No Jeopardy
Olive ridley sea turtle, Mexico's Pacific Coast breeding colonies	No Jeopardy
Olive ridley sea turtle, all other areas	No Jeopardy
Killer whale, Southern Resident DPS	No Jeopardy
Steller sea lion, Western	No Jeopardy
Guadalupe fur seal	No Jeopardy
Hawaiian monk seal	No Jeopardy

17 SPECIES INTEGRATION AND SYNTHESIS: METHOMYL

In this section we summarize the results of the methomyl integration and synthesis for ESA-listed species. The integration and synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the action. In this section, we add the effects of the action to the environmental baseline and the cumulative effects, in light of the status of the species, to formulate NMFS's conference and biological opinion as to whether EPA was able to insure their action is not likely to jeopardize the continued existence of the species.

The species-specific scorecards with results for each species can be found in the Integration and Synthesis portions of Attachment 3. Consideration of the risk reduction achieved through additional conservation measures described in Chapter 5 is incorporated into the final conclusions as described in the "Impact of Conservation Measures" subsections of the Effects Analysis and Integration and Synthesis chapters in Attachment 3. These final conclusions are presented in the table below in the "With Conservation Measures (Final BiOp Conclusions)" column.

Table 142. Methomyl species conclusions with conservations measures incorporated

Species Name	With Conservation Measures (Final BiOp Conclusions)
Atlantic salmon, Gulf of Maine ESU	No Jeopardy
Chum salmon , Columbia River ESU	No Jeopardy
Chum salmon, Hood Canal summer-run ESU	No Jeopardy
Chinook salmon, California coastal ESU	No Jeopardy
Chinook salmon, Central Valley spring-run ESU	No Jeopardy
Chinook salmon, Lower Columbia River ESU	No Jeopardy
Chinook salmon, Puget Sound ESU	No Jeopardy
Chinook salmon, Sacramento River winter-run ESU	No Jeopardy
Chinook salmon, Snake River fall-run ESU	No Jeopardy
Chinook salmon, Snake River spring/summer run ESU	No Jeopardy
Chinook salmon, Upper Columbia River spring-run ESU	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Chinook salmon, Upper Willamette River ESU	No Jeopardy
Coho salmon, Central California coast ESU	No Jeopardy
Coho salmon, Lower Columbia River ESU	No Jeopardy
Coho salmon, Oregon coast ESU	No Jeopardy
Coho salmon, S. Oregon and N. California coasts ESU	No Jeopardy
Sockeye, Snake River ESU	No Jeopardy
Steelhead, California Central Valley ESU	No Jeopardy
Steelhead, Central California coast ESU	No Jeopardy
Steelhead, Lower Columbia River ESU	No Jeopardy
Steelhead, Middle Columbia River ESU	No Jeopardy
Steelhead, Northern California ESU	No Jeopardy
Steelhead, Puget Sound ESU	No Jeopardy
Steelhead, Snake River Basin ESU	No Jeopardy
Steelhead, South-Central California coast ESU	No Jeopardy
Steelhead, Southern California ESU	No Jeopardy
Steelhead, Upper Columbia River ESU	No Jeopardy
Steelhead, Upper Willamette River ESU	No Jeopardy
Eulachon, Pacific smelt, Southern DPS	No Jeopardy
Green sturgeon, Southern DPS	No Jeopardy
Shortnose sturgeon	No Jeopardy
Atlantic sturgeon, Carolina DPS	No Jeopardy
Atlantic sturgeon, Chesapeake Bay DPS	No Jeopardy
Atlantic sturgeon, Gulf of Maine DPS	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Atlantic sturgeon, New York Bight DPS	No Jeopardy
Atlantic sturgeon, South Atlantic DPS	No Jeopardy
Gulf sturgeon	No Jeopardy
Yelloweye rockfish	No Jeopardy
Boccacio, Puget Sound/Georgia Basin	No Jeopardy
Nassau grouper	No Jeopardy
Smalltooth sawfish, U.S. DPS	No Jeopardy
Giant Manta Ray	No Jeopardy
Black abalone	No Jeopardy
White abalone	No Jeopardy
Sunflower sea star – Proposed	No Jeopardy
Queen Conch – Proposed	No Jeopardy
Staghorn coral	No Jeopardy
Elkhorn coral	No Jeopardy
Boulder star coral	No Jeopardy
Lobed star coral	No Jeopardy
Mountainous star coral	No Jeopardy
Pillar coral	No Jeopardy
Rough cactus coral	No Jeopardy
Coral, *Acropora globiceps*	No Jeopardy
Coral, *Acropora jacquelineae*	No Jeopardy
Coral, *Acropora retusa*	No Jeopardy
Coral, *Acropora speciose*	No Jeopardy
Coral, *Euphyllia pardivisa*	No Jeopardy
Coral, *Isopora crateriformis*	No Jeopardy
Coral, *Seriatopora aculeata*	No Jeopardy

Species Name	With Conservation Measures (Final BiOp Conclusions)
Killer whale, Southern Resident DPS	No Jeopardy

18 Critical Habitat Integration and Syntheses: Carbaryl

In this section we summarize the results of the carbaryl integration and synthesis for ESA-listed species' designated and proposed critical habitats. The integration and synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the action. In this section, we add the effects of the action to the environmental baseline and the cumulative effects, in light of the status of the species, to formulate NMFS's conference and biological opinion as to whether EPA was able to insure their action is not likely to jeopardize the continued existence of the species.

The species-specific scorecards with results for each species can be found in the Integration and Synthesis portions of Attachment 3. Consideration of the risk reduction achieved through additional conservation measures described in Chapter 5 is incorporated into the final conclusions as described in the "Impact of Conservation Measures" subsections of the Effects Analysis and Integration and Synthesis chapters in Attachment 3. These final conclusions are presented in the table below in the "With Conservation Measures (Final BiOp Conclusions)" column.

Table 143. Carbaryl species critical habitat conclusions

Species Name	**With Conservation Measures (Final BiOp Conclusions)**
Atlantic salmon, Gulf of Maine ESU	No Destruction or Adverse Modification
Chum salmon , Columbia River ESU	No Destruction or Adverse Modification
Chum salmon, Hood Canal summer-run ESU	No Destruction or Adverse Modification
Chinook salmon, California coastal ESU	No Destruction or Adverse Modification
Chinook salmon, Central Valley spring-run ESU	No Destruction or Adverse Modification
Chinook salmon, Lower Columbia River ESU	No Destruction or Adverse Modification
Chinook salmon, Puget Sound ESU	No Destruction or Adverse Modification
Chinook salmon, Sacramento River winter-run ESU	No Destruction or Adverse Modification

Species Name	With Conservation Measures (Final BiOp Conclusions)
Chinook salmon, Snake River fall-run ESU	No Destruction or Adverse Modification
Chinook salmon, Snake River spring/summer run ESU	No Destruction or Adverse Modification
Chinook salmon, Upper Columbia River spring-run ESU	No Destruction or Adverse Modification
Chinook salmon, Upper Willamette River ESU	No Destruction or Adverse Modification
Coho salmon, Central California coast ESU	No Destruction or Adverse Modification
Coho salmon, Lower Columbia River ESU	No Destruction or Adverse Modification
Coho salmon, Oregon coast ESU	No Destruction or Adverse Modification
Coho salmon, S. Oregon and N. California coasts ESU	No Destruction or Adverse Modification
Sockeye, Ozette Lake ESU	No Destruction or Adverse Modification
Sockeye, Snake River ESU	No Destruction or Adverse Modification
Steelhead, California Central Valley DPS	No Destruction or Adverse Modification
Steelhead, Central California coast DPS	No Destruction or Adverse Modification
Steelhead, Lower Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Middle Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Northern California DPS	No Destruction or Adverse Modification

Species Name	With Conservation Measures (Final BiOp Conclusions)
Steelhead, Puget Sound DPS	No Destruction or Adverse Modification
Steelhead, Snake River Basin DPS	No Destruction or Adverse Modification
Steelhead, South-Central California coast DPS	No Destruction or Adverse Modification
Steelhead, Southern California DPS	No Destruction or Adverse Modification
Steelhead, Upper Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Upper Willamette River DPS	No Destruction or Adverse Modification
Eulachon, Pacific smelt, Southern DPS	No Destruction or Adverse Modification
Green sturgeon, Southern DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Carolina DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Chesapeake Bay DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Gulf of Maine DPS	No Destruction or Adverse Modification
Atlantic sturgeon, New York Bight DPS	No Destruction or Adverse Modification
Atlantic sturgeon, South Atlantic DPS	No Destruction or Adverse Modification
Yelloweye rockfish	No Destruction or Adverse Modification
Boccacio, Puget Sound/Georgia Basin	No Destruction or Adverse Modification

Species Name	**With Conservation Measures (Final BiOp Conclusions)**
Nassau grouper – Proposed	No Destruction or Adverse Modification
Smalltooth sawfish, U.S. DPS	No Destruction or Adverse Modification
Black abalone	No Destruction or Adverse Modification
White abalone	No Destruction or Adverse Modification
Boulder star coral	No Destruction or Adverse Modification
Lobed star coral	No Destruction or Adverse Modification
Mountainous star coral	No Destruction or Adverse Modification
Pillar coral	No Destruction or Adverse Modification
Rough cactus coral	No Destruction or Adverse Modification
Coral, *Acropora globiceps* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora jacquelineae* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora retusa* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora speciose* – Proposed	No Destruction or Adverse Modification
Coral, *Euphyllia pardivisa* – Proposed	No Destruction or Adverse Modification
Coral, *Isopora crateriformis* – Proposed	No Destruction or Adverse Modification

Species Name	**With Conservation Measures (Final BiOp Conclusions)**
Coral, *Seriatopora aculeata* – Proposed	No Destruction or Adverse Modification
Killer whale, Southern Resident DPS	No Destruction or Adverse Modification

19 Critical Habitat Integration and Synthesis: Methomyl

In this section we summarize the results of the methomyl integration and synthesis for ESA-listed species' designated and proposed critical habitats. The integration and synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the action. In this section, we add the effects of the action to the environmental baseline and the cumulative effects, in light of the status of the species, to formulate NMFS's conference and biological opinion as to whether EPA was able to insure their action is not likely to jeopardize the continued existence of the species.

The species-specific scorecards with results for each species can be found in the Integration and Synthesis portions of Attachment 3. Consideration of the risk reduction achieved through additional conservation measures described in Chapter 5 is incorporated into the final conclusions as described in the "Impact of Conservation Measures" subsections of the Effects Analysis and Integration and Synthesis chapters in Attachment 3. These final conclusions are presented in the table below in the "With Conservation Measures (Final BiOp Conclusions)" column.

Table 144. Methomyl species critical habitat conclusions

Species Name	With Conservation Measures (Final BiOp Conclusions)
Atlantic salmon, Gulf of Maine ESU	No Destruction or Adverse Modification
Chum salmon , Columbia River ESU	No Destruction or Adverse Modification
Chum salmon, Hood Canal summer-run ESU	No Destruction or Adverse Modification
Chinook salmon, California coastal ESU	No Destruction or Adverse Modification
Chinook salmon, Central Valley spring-run ESU	No Destruction or Adverse Modification
Chinook salmon, Lower Columbia River ESU	No Destruction or Adverse Modification
Chinook salmon, Puget Sound ESU	No Destruction or Adverse Modification
Chinook salmon, Sacramento River winter-run ESU	No Destruction or Adverse Modification

Species Name	With Conservation Measures (Final BiOp Conclusions)
Chinook salmon, Snake River fall-run ESU	No Destruction or Adverse Modification
Chinook salmon, Snake River spring/summer run ESU	No Destruction or Adverse Modification
Chinook salmon, Upper Columbia River spring-run ESU	No Destruction or Adverse Modification
Chinook salmon, Upper Willamette River ESU	No Destruction or Adverse Modification
Coho salmon, Central California coast ESU	No Destruction or Adverse Modification
Coho salmon, Lower Columbia River ESU	No Destruction or Adverse Modification
Coho salmon, Oregon coast ESU	No Destruction or Adverse Modification
Coho salmon, S. Oregon and N. California coasts ESU	No Destruction or Adverse Modification
Sockeye, Snake River ESU	No Destruction or Adverse Modification
Steelhead, California Central Valley DPS	No Destruction or Adverse Modification
Steelhead, Central California coast DPS	No Destruction or Adverse Modification
Steelhead, Lower Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Middle Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Northern California DPS	No Destruction or Adverse Modification
Steelhead, Puget Sound DPS	No Destruction or Adverse Modification

Species Name	With Conservation Measures (Final BiOp Conclusions)
Steelhead, Snake River Basin DPS	No Destruction or Adverse Modification
Steelhead, South-Central California coast DPS	No Destruction or Adverse Modification
Steelhead, Southern California DPS	No Destruction or Adverse Modification
Steelhead, Upper Columbia River DPS	No Destruction or Adverse Modification
Steelhead, Upper Willamette River DPS	No Destruction or Adverse Modification
Eulachon, Pacific smelt, Southern DPS	No Destruction or Adverse Modification
Green sturgeon, Southern DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Carolina DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Chesapeake Bay DPS	No Destruction or Adverse Modification
Atlantic sturgeon, Gulf of Maine DPS	No Destruction or Adverse Modification
Atlantic sturgeon, New York Bight DPS	No Destruction or Adverse Modification
Atlantic sturgeon, South Atlantic DPS	No Destruction or Adverse Modification
Yelloweye rockfish	No Destruction or Adverse Modification
Bocaccio, Puget Sound/Georgia Basin	No Destruction or Adverse Modification
Nassau grouper – Proposed	No Destruction or Adverse Modification

Species Name	**With Conservation Measures (Final BiOp Conclusions)**
Smalltooth sawfish, U.S. DPS	No Destruction or Adverse Modification
Black abalone	No Destruction or Adverse Modification
Boulder star coral	No Destruction or Adverse Modification
Lobed star coral	No Destruction or Adverse Modification
Mountainous star coral	No Destruction or Adverse Modification
Pillar coral	No Destruction or Adverse Modification
Rough cactus coral	No Destruction or Adverse Modification
Coral, *Acropora globiceps* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora jacquelineae* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora retusa* – Proposed	No Destruction or Adverse Modification
Coral, *Acropora speciose* – Proposed	No Destruction or Adverse Modification
Coral, *Euphyllia pardivisa* – Proposed	No Destruction or Adverse Modification
Coral, *Isopora crateriformis* – Proposed	No Destruction or Adverse Modification
Coral, *Seriatopora aculeata* – Proposed	No Destruction or Adverse Modification
Killer whale, Southern Resident DPS	No Destruction or Adverse Modification

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20 CONCLUSION

EPA made NLAA and LAA determinations in their 2021 BEs for carbaryl and methomyl. We reviewed those conclusions and made several modifications (see Chapter 7). Together, the agencies made LAA determinations for 61 species, including 2 proposed species. The agencies also made LAA determinations for 56 critical habitats, including 6 proposed critical habitats. NLAA determinations were made for 38 species, 16 designated critical habitats, and 8 proposed critical habitats. The NLAA and LAA determinations for species and habitats were the same for both carbaryl and methomyl.

20.1 Carbaryl

It is NMFS's conference and biological opinion that EPA is able to insure that their registration of the uses of all pesticide products containing carbaryl (as described by product labels, or on EPA Endangered Species Protection Program Bulletins), is not likely to jeopardize the continued existence of listed or proposed species under NMFS's jurisdiction. Similarly, it is NMFS's biological and conference opinion that EPA is able to insure this action will not destroy or adversely modify the designated or proposed critical habitat of those species. We reached this conclusion upon reviewing the current status of the ESA-listed and proposed species and designated and proposed critical habitat, the environmental baseline within the action area, the effects of the action that now includes the RPA from the draft opinion and additional mitigation measures, and cumulative effects.

20.2 Methomyl

It is NMFS's conference and biological opinion that EPA is able to insure that their registration of the uses of all pesticide products containing methomyl (as described by product labels, or on EPA Endangered Species Protection Program Bulletins), is not likely to jeopardize the continued existence of listed or proposed species under NMFS's jurisdiction. Similarly, it is NMFS's biological and conference opinion that EPA is able to insure this action will not destroy or adversely modify the designated or proposed critical habitat of those species. We reached this conclusion upon reviewing the current status of the ESA-listed and proposed species and designated and proposed critical habitat, the environmental baseline within the action area, the effects of the action that now includes the RPA from the draft opinion and additional mitigation measures, and cumulative effects.

21 INCIDENTAL TAKE STATEMENT

Section Contents

21.1 Introduction

Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA, either as proposed by the action agency or modified by a RPA, and the action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species, or an ITS. To minimize such impacts, NMFS provides RPMs and terms and conditions that must be complied with by the Federal agency or any applicant in order to be exempt from the prohibitions against "take" of ESA-listed species. RPMs are measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take." (50 CFR §402.02). This conference and biological opinion has assessed the probable effects of the action and determined incidental take of proposed species is reasonably certain to occur. However, there are no take prohibitions under section 9 or 4(d) until proposed species are listed under the ESA as endangered (or threatened with a 4d rule). The incidental take statement provided with this conference opinion does not become effective after the listing is final unless EPA requests to adopt the conference opinion as a biological opinion and NMFS, after ensuring the effects of the action are no different than identified here, responds in the affirmative (50 CFR 402.10(d)).

Section 9(a)(1) of the ESA prohibits the taking of endangered species without a specific permit or exemption. Protective regulations adopted pursuant to section 4(d) of the ESA extend the prohibition to threatened species. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct (50 CFR §222.102). We interpret "harass" as meaning to create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns with include, but are not limited to, breeding, feeding, or sheltering (Wieting 2016). Harm is further defined by NMFS as an act which actually kills or injures fish or wildlife, and may to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering (50 CFR §222.102). Incidental take is defined as takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR §402.02). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action, whether implemented as proposed or as modified by RPAs, is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this ITS. NMFS cannot issue an ITS to cover any take of marine mammals that would also be prohibited under the Marine Mammal Protection Act, unless such take has been authorized pursuant to section

101(a)(5) of that Act. Consequently, any exemption of incidental take of marine mammals under this ITS is conditional upon the issuance of an authorization for such take under the MMPA should effects from EPA's action be determined to result in Level A or Level B harassment.

21.2 Amount or Extent & Effects of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 C.F.R. § 402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by the action, which in this case would occur only upon implementation of the RPA. When it is not possible or practicable to specify the amount or extent of take, a surrogate may be used if we: describe the causal link between the surrogate and take of the listed species, explain why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and set a clear standard for determining when the level of anticipated take has been exceeded. 50 C.F.R. §402.14(h)(i)(1)(i).

For this opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products to 62 ESA-listed species, as well as 2 proposed species under NMFS's jurisdiction during the 15-year duration of the action. Pesticide runoff and drift of carbaryl and methomyl are most likely to reach streams and other aquatic sites when they are applied to crops and other land use settings located adjacent to wetlands, riparian areas, ditches, floodplain habitats, intermittent streams, nearshore estuarine and marine habitats. The likelihood for these inputs into aquatic habitats are especially high when rainfall immediately follows applications, or if wind conditions exacerbate inputs from drift. The effects of pesticides and other contaminants found in urban runoff, especially from areas with a high degree of impervious surfaces, may also exacerbate degraded water quality conditions of receiving waters. Urban runoff is also generally warmer in temperature, and elevated water temperature poses negative effects to many ESA-listed species. The range of effects of carbaryl and methomyl on ESA-listed species includes killing or injuring individuals directly, and reductions in prey leading to starvation and impaired growth. For example, impaired growth increases the susceptibility of juveniles becoming prey to predators, and starvation may make species more susceptible to disease. In addition, exposed individuals may change normal behaviors (e.g., feeding, sheltering, breeding). These results are not the purpose of the action. Therefore, incidental take of ESA-listed species is reasonably certain to occur over the 15-year duration of the action.

Given the variability of real-life conditions, the broad nature and scope of the action, and the wide-ranging distributions of individuals of ESA-listed species, the best scientific and commercial data available are not sufficient to enable NMFS to directly estimate a specific amount of incidental take associated with the action for each listed species that may be exposed and respond to this exposure. As explained in the Description of the Action and the Effects of the Action sections, NMFS identified multiple uncertainties associated with the action. Areas of uncertainty include:

- Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;

- Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
- Minimal information on tank mixtures and associated exposure estimates;
- Limited data on toxicity of environmental mixtures;
- Variability in annual land use, crop cover, and pest pressure;
- Temporal and spatial variability of individuals;
- Uncertainty about pesticide concentrations that may occur in nearshore estuarine and marine habitats; and
- Uncertainty about pesticide concentrations resulting from non-agricultural uses.

Additionally, NMFS recognizes there are multiple impediments that reduce the likelihood of detecting take to ESA-listed species from the use of pesticides. It is important to place the significance of mortality incidents in the proper context. Vyas (1999) concluded that most wildlife mortality is unaccounted for as only a small fraction are likely observed, reported, and confirmed. Data show that most effects on wildlife are not observed, the majority of incidents observed are not reported, only a portion of those that are reported are investigated and, of those investigated, confirmation of pesticides is challenging given a general lack of resources for such investigations and the need to immediately secure samples for analysis prior to chemical dissipation. The likelihood of detecting impacts becomes even more difficult in species that are not abundant. Sublethal effects such as reduced reproduction are nearly impossible to detect without rigorous environmental monitoring. Additionally, there are generally no mandates requiring investigation or reporting of pesticide incidents. The exception is that pesticide registrants are required to report ecological incidents to EPA under FIFRA 6(a)(2). EPA maintains an incident database (the EIIS) to document reported incidents. The EIIS uses criteria to categorize incidents as “major” or “minor” depending on the scale of effect observed. Additionally, the EIIS also characterizes the likelihood that the incident was caused by a particular pesticide using defined criteria (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#eiis). For these reasons, NMFS uses surrogates for the allowable extent of take of listed species, as described below within each of the species groupings.

Anadromous and Marine Fish

NMFS identifies, as a surrogate for the anticipated extent of take of anadromous and marine fish, the ability of this action to proceed without any fish mortality within the action area attributed to the legal use of carbaryl or methomyl, or any compounds, degradates, or mixtures of these a.i.s affecting aquatic habitats containing ESA-listed species. Note that “any fish mortality” is a higher threshold than might be assumed on first reading. This threshold is appropriate as a surrogate for take given the qualifiers described below (e.g. attributable to carbaryl and methomyl), as well as the discussion on observing ecological incidents outlined in the previous section. Because of the difficulty of detecting mortality or other adverse effects on ESA-listed species of fish, individuals killed do not have to be ESA-listed species in order for their death to be considered a relevant surrogate for take. In addition, mortalities of other species of fish provide an acceptable surrogate because they are causally linked to effects on the ESA-listed fish species in the same habitat. For example, salmonids are relatively sensitive to pesticides compared to other species of fish so, if there are kills of other anadromous or freshwater fishes attributed to use of these pesticides, it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes of listed fishes in downstream areas increases. Because fish mortalities can easily go unobserved or unaccounted for, we consider an exceedance of take to have occurred when any fish mortality is reported to EPA, as described below, and attributed by EPA to the lawful use of these a.i.s according to EPA's guidelines for evaluating ecological incident data (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#eiis). Incidental take consists of incidents (as defined in 40 CFR 159.184) involving fish mortalities that are considered attributable to 1 of these a.i.s, its metabolites, or degradates, if the available information suggests a certainty index of "probable" or "highly probable" as defined in EPA's guidance for using incident data (EPA October 13, 2011; https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#guidance).

Marine Invertebrates

NMFS identifies, as a surrogate for the allowable extent of take of marine invertebrates, the ability of this action to proceed without any mortality or adverse reproductive effects to corals or mollusks within the action area attributed to the legal use of carbaryl or methomyl, or any compounds, degradates, or mixtures affecting aquatic habitats containing ESA-listed species. Note that "any mortality or adverse reproductive effects" is a higher threshold than might be assumed on first reading. This threshold is appropriate as a surrogate for take given the qualifiers described below (e.g. attributable to carbaryl and methomyl), as well as the discussion on observing ecological incidents outlined in the previous section. Similar to the situation with fish, because of the difficulty of detecting adverse effects on ESA-listed or proposed species of marine invertebrates, an exceedance of take occurs when any coral or mollusks mortality or adverse reproductive effect is reported to EPA and attributed by EPA to the lawful use of these a.i.s according to EPA's guidelines for evaluating ecological incident data (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#eiis). In addition, mortality or adverse reproductive effects on other species of coral or mollusks provide an acceptable surrogate because they are causally linked to similar effects on the ESA-listed species. For example, species that are taxonomically similar tend to have similar toxicological sensitivities to pesticides. Therefore, if there are mortalities or adverse reproductive effects on other species of coral or mollusks, it is likely that listed marine invertebrates have also been adversely affected, even if these effects were not observed and reported. Reproductive effects, and both "minor" and "major" incidents involving marine invertebrate corals or mollusks are considered attributable to 1 of these a.i.s, its metabolites, or degradates, if the available information suggests a certainty index of "probable" or "highly probable" as defined in EPA's guidance for using incident data (EPA October 13, 2011; https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#guidance).

Cetaceans - Southern Resident Killer Whale (SRKW)

NMFS identifies, as a surrogate for the allowable take of SRKW, the ability of this action to proceed without any mortality to Pacific salmonids attributed to the legal use of carbaryl or methomyl. Mortality of salmon is causally linked to take on SKRW. Salmon, in particular Chinook salmon, are the prey for SRKW. Note that "any mortality to Pacific salmonids" is a higher threshold than might be assumed on first reading. This threshold is appropriate as a surrogate for take given the qualifiers described below (e.g. attributable to carbaryl and methomyl), as well as the discussion on observing ecological incidents outlined in the previous section. The reduction in production of Pacific salmon throughout their range that would occur under the action would therefore result in harm to SRKW by further reducing prey availability, which may cause animals to forage for longer periods, travel to alternate locations, or abandon foraging efforts. These effects are difficult to directly observe. The extent of take from the action is not anticipated to cause direct take by serious injury or mortality to SRKWs. However, the action is expected to result in take in the form of harm to SRKWs through a reduction in the availability of their prey, which can impact individual survival and reproductive success. An exceedance of take occurs when any Pacific salmonid mortality is reported to EPA and attributed by EPA to the lawful use of these a.i.s according to EPA's guidelines for evaluating ecological incident data (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#eiis). In relation to EPA's guidelines, both "minor" and "major" incidents involving Pacific salmonids are considered attributable to 1 of these a.i.s, its metabolites, or degradates, if the available information suggests a certainty index of "probable" or "highly probable" as defined in EPA's guidance for using incident data (EPA October 13, 2011; https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-using-incident-data-evaluating-listed-and#guidance).

21.3 Reasonable and Prudent Measures

"Reasonable and prudent measures" are actions that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR §402.02). Only incidental take resulting from the implementation of the RPA as the agency action and any specified RPMs, and terms and conditions identified in the ITS are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of ESA.

NMFS believes the RPMs described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

- RPM 1. Revise and approve all carbaryl and methomyl product labels and develop relevant EPA Endangered Species Protection Plan Bulletins with measures to conserve ESA-listed species (that are in addition to those included in the action).
- RPM 2. Improve ecological incident reporting, develop ESA educational materials, and report label compliance.

21.4 Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agency must comply (or must ensure that any applicant complies) with the following terms and conditions. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). The terms and conditions described below must be undertaken by the U.S. Environmental Protection Agency and applicants for the exemption in section 7(o)(2) to apply.

RPM 1: Revise and approve all carbaryl and methomyl product labels and develop relevant EPA Endangered Species Protection Plan Bulletins to conserve ESA-listed species.

A. Terms and Conditions for EPA to Coordinate with Applicants

To address RPM number 1, applicants with registrations for products containing carbaryl and methomyl shall submit to EPA the following label amendments. Label amendments shall be submitted to EPA within 60 days of the issuance date of this biological opinion.

a. **Amendments beyond the conservation measures to minimize the effects of incidental take**
   i. **For carbaryl, when applying this product within 300 meters of ESA-listed species habitat:**
      - When applying carbaryl by airblast at rates ≥1 lb carbaryl/A, do not apply within 55 ft of ESA-listed species habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams, marshes or natural ponds, estuaries and commercial fish farm ponds) when wind is blowing toward the aquatic habitat or when wind is blowing ≤ 2 mph.[9]
      - Implement 1 or more runoff reduction measures from **Error! Reference source not found.**
      - Do not apply in coral Pesticide Use Limitation Areas during peak coral spawning and larval developmental periods (July – September).
      - Do not tank mix with other neurotoxic pesticides (i.e., carbamate, organophosphate, pyrethroid, and neonicotinoid pesticides) at application rates that exceed 50% the maximum labeled rate of any pesticide active ingredient used in the tank mixture.
   ii. **For methomyl, when applying this product within 300 meters of ESA-listed species habitat:**
      - When applying methomyl by airblast at rates ≥0.5 lb methomyl/A, do not apply within 55 ft of ESA-listed species habitats (such as, but not limited to, lakes, reservoirs, rivers, permanent streams, marshes or natural ponds, estuaries and commercial fish farm ponds) when wind is blowing toward the aquatic habitat or when wind is blowing ≤ 2 mph.
      - Implement 1 or more runoff reduction measure from **Error! Reference source not found.**
      - Do not apply in coral Pesticide Use Limitation Areas during peak coral spawning and larval developmental periods (July – September).
      - Do not tank mix with other neurotoxic pesticides (i.e., carbamate, organophosphate, pyrethroid, and neonicotinoid pesticides) at

[9] This requirement can be waived in cases where grower maintains a functional riparian system > 10m wide alongside waterways adjacent to treatment area.

application rates that exceed 50% the maximum labeled rate of any pesticide active ingredient used in the tank mixture.

b. **Amendments to reference EPA's Endangered Species Bulletins**
Applicants shall submit to EPA the following label amendments for all technical and manufacturing use products containing carbaryl and methomyl:

The following statements shall be placed at the beginning of the Directions for Use section:

"This product may only be formulated into end-use products that contain the following language on their labeling (to be placed at the beginning of the Directions for Use section of all end-use product labels) when they are released for shipment:

"ENDANGERED SPECIES PROTECTION REQUIREMENTS": Before using this product, you must obtain any applicable Endangered Species Protection Bulletins ('Bulletins') within 6 months prior to or on the day of application. To obtain Bulletins, go to Bulletins Live! Two (BLT) at https://www.epa.gov/pesticides/bulletins. When using this product, you must follow all directions and restrictions contained in any applicable Bulletin(s) for the area where you are applying the product, including any restrictions on application timing if applicable. It is a violation of Federal law to use this product in a manner inconsistent with its labeling, including this labeling instruction to follow all directions and restrictions contained in any applicable Bulletin(s). For general questions or technical help, call 1-844-447-3813, or email ESPP@epa.gov""

c. **Amendments to improve ecological incident reporting**
Applicants shall submit to EPA the following label amendments for all technical and manufacturing use products containing carbaryl and methomyl:

The following statements shall be placed at the beginning of the Directions for Use section:

"This product may only be formulated into end-use products that contain the following language on their labeling (to be placed at the beginning of the Directions for Use section of all end-use product labels) when they are released for shipment:

"Reporting Ecological Incidents: For guidance on reporting ecological incidents, including death, injury, or harm to plants and animals, including bees and other non-target insects, see EPA's Pesticide Incident Reporting website: https://www.epa.gov/pesticide-incidents or call (registrant phone number)".

B. Terms and Conditions for EPA

a. **Within 60 days of the issuance date of this conference and biological opinion**
EPA shall notify all end-use product registrants of carbaryl and methomyl to submit, within 60-days of EPA's notification, the necessary amendments to their end-use product labels, to be consistent with the technical/manufacturing use

product label amendments described in RPM 1(A, B, C, D) Terms and Conditions for applicants.

b. **Within 18 months of the issuance date of this conference and biological opinion**
    i. EPA shall review and act on all of the registrants' requests to amend labels.
    ii. EPA shall develop Endangered Species Protection Bulletins to incorporate the registrants label amendments described above.

RPM 2. Improve ecological incident reporting, develop ESA educational materials, and report label compliance

A. **Terms and Conditions for EPA**

a. **Label modifications for Ecological Incidents**
Within 60 days of the issuance date of this conference and biological opinion EPA shall notify all end-use product registrants of carbaryl and methomyl to submit within 60-days of EPA's notification, the necessary amendments to their end-use product labels, to be consistent with the technical/manufacturing use product label amendments described above (RPM 1[a,b,c,d]). Terms and Conditions for applicants. EPA shall review and act on the registrants' requests to amend labels as described above within 18 months of the issuance date of this conference and biological opinion.

b. **Annual Reporting of Ecological Incidents**
Within 2 years of the conference and biological opinion, EPA shall commence annual reporting to NMFS the occurrence of all minor and major ecological incidents involving aquatic species attributable to the use of products containing carbaryl and methomyl.

c. **ESA Conservation Educational Materials**
EPA shall amend the Endangered Species Protection Bulletins to include a link to generic ESA conservation educational materials. This material is to be jointly developed by NMFS and EPA and maintained on either a NMFS or EPA website. In addition to providing a link, the Endangered Species Protection Bulletins should include an advisory encouraging applicators to review the information.

d. **Label Compliance Monitoring**
EPA shall work with NMFS to determine a feasible means by which EPA will report to NMFS a summary of relevant compliance data on an annual basis. The goal of this term and condition is to establish a process by which NMFS can better access information regarding label compliance for pesticides subject to ESA Section 7 consultations. EPA shall work with NMFS to develop a process of effectiveness monitoring which utilizes existing FIFRA compliance monitoring strategies.

21.5 Minimizing Adverse Effects to Proposed Species

NMFS determined that the carbaryl and methomyl actions are likely to adversely affect 2 proposed ESA-listed species, the sunflower sea star and the queen conch. Given the status of these species, NMFS provides advisory recommendations to minimize or avoid adverse effects. In this case, NMFS recommends the same mitigation be applied within the sunflower sea star and queen conch PULAs as described in the Terms and Conditions for other ESA-listed species (Section 21.4). If the proposed listing is finalized, and no significant new information is developed, these recommendations will become mandatory once this conference opinion is adopted as the biological opinion.

22 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of an action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect ESA-listed species:

1. Develop models that more accurately quantify pesticide exposure in estuarine and near-shore ocean environments.

2. Work with other appropriate federal, state, and local partners to determine efficacy of riparian area management methods in reducing pesticide loading from authorized uses especially the types of vegetation and width of riparian areas needed.

3. Identify and implement other methods that eliminate or significantly reduce pesticide loading into species' habitats.

4. Develop and implement educational outreach on pesticide risks to threatened and endangered species.

5. Develop improved methods for characterizing exposure from non-agricultural uses.

In order for NMFS's Office of Protected Resources Endangered Species Act Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, the EPA should notify the Endangered Species Act Interagency Cooperation Division of any conservation recommendations they implement in their final action.

23 Reinitiation Notice

This concludes formal consultation for the EPA's proposed registration of pesticide products containing carbaryl and methomyl to ESA-listed species and their critical habitats under the jurisdiction of the NMFS. As 50 C.F.R. §402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if:

1. The amount or extent of taking specified in the ITS is exceeded.
2. New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered.
3. The identified action is subsequently modified in a manner that causes an effect to ESA-listed species or designated critical habitat that was not considered in this opinion.
4. A new species is listed or critical habitat designated under the ESA that may be affected by the action.

NMFS's analysis and conclusions are based on EPA's action and the implementation of the RPA. If changes to product labeling result in modifications to the action that were not considered in this opinion, including but not limited to label modifications authorizing pesticide application in new locations, additional application methods, or increased application rates or numbers of applications, EPA must contact NMFS to discuss reinitiation. If reinitiation of consultation appears warranted due to 1 or more of the above circumstances, EPA must contact NMFS Office of Protected Resources, ESA Interagency Cooperation Division. In the event reinitiation condition (1), (2), or (3) is met, reinitiation will be only for the a.i.(s) that meet that condition. It is recommended that EPA request reinitiation with sufficient time to consult and to prevent lapse of coverage for the a.i.s in this opinion.

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APPENDIX A: PACIFIC SALMON POPULATION MODELING

Introduction

To assess the potential for adverse impacts of the anticholinesterase insecticides on Pacific salmon populations, a model was developed that explicitly links impairments in the biochemistry, behavior, prey availability and somatic growth of individual salmon to the productivity of salmon populations. More specifically, the model connects known effects of the pesticides on salmon physiology and behavior with community-level effects on salmon prey to estimate population-level effects on salmon. The model used here is an extension of one developed for investigating the direct effects of pesticides on the biochemistry, behavior and growth of ocean-type Chinook salmon (Baldwin et al., 2009) and includes indirect impacts on prey base (Macneale et al, 2014).

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). Acetylcholinesterase is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al. 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), antipredator behaviors (Scholz et al. 2000) and reproductive physiology (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000).

Changes to the size of juvenile salmon from exposure to anticholinesterase pesticides were linked to salmon population demographics (Baldwin et al., 2009). We used size-dependent survival of juveniles during a period of their first year of life. We did this by constructing and analyzing general life-history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). A steelhead (*O. mykiss*) life-history model was not constructed due to the lack of demographic information relating to the proportions of resident and anadromous individuals, the freshwater residence time of steelhead, and rates of repeated spawning. The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure 103). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook females can mature at age 3, 4 or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as subyearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life-history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the lack of sufficient demographic and reproductive data for a single population.

To assess the potential for adverse impacts of the pesticides on Pacific salmon populations, another model was developed that explicitly links mortality due to exposure of young-of-the-year to the productivity of salmon populations. The acute toxicity model estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of contaminants. This model excluded sublethal and indirect effects of the exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life-history strategies.

The model endpoint used to assess population-level impacts for both the somatic growth and acute mortality models was the percent change in the intrinsic population growth rate (lambda, λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. NMFS uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing Endangered Species Recovery Plans, composing biological opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda<1.0$ indicate a declining population, negative changes in lambda greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline at minimum reduces the species recovery and could push a population's growth rate below 1.0, and into decline.

Assessing the results from different pesticide exposure scenarios relative to a control (i.e., unexposed) scenario can indicate the potential for pesticide exposures to lead to changes in the first year survival. Consequently, subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists in forecasting the potential population-level impacts to listed populations. The model conveys the potential influence of life-history strategies that might explain differential results within the species modeled.

Methods

In order to understand the relative impacts of a short-term pesticide exposure on exposed vs. unexposed fish, we used parameters for an idealized baseline population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. There were no definitive data available on the populations to support specific density dependent relationships, so rather than assign an unsupported relationship, the National Research Council recommendation was followed to utilize density independent parameters (NAS, 2013). The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability reported in the literature as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent so that they remain in the range they occupied during the period when demographic data were collected.

In the models an individual fish experiences an exposure scenario once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur to the population annually. All individuals in 1 cohort within a given population are assumed to be exposed to the pesticide during their subyearling spring-summer growth period. No other age classes experience the exposure.

Somatic Growth Model

We integrated 2 avenues of effects to subyearling salmonids' growth from exposure to the 2 carbamates. The first avenue is a result of AChE inhibition on the feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors (Sandahl et al. 2005). Salmon are often food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to the AChE inhibitor, chlorpyrifos (Brazner and Kline 1990). Furthermore, based on our review of the sensitivities of aquatic invertebrates to the 2 insecticides, we expect reductions in densities and altered composition of the salmonid prey communities. Therefore, the second avenue the model addresses is the potential for reductions in juvenile growth due to reductions in available prey.

Reductions in aquatic prey are included in the model because of the high relative toxicity of pesticides to salmonid prey and the extended duration of effects on prey communities. Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities. The 2 insecticides are highly toxic to aquatic macroinvertebrates; concentrations that are not expected to kill salmonids are often lethal for their invertebrate prey. In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, caddisflies, and stoneflies) are among the most sensitive aquatic macroinvertebrates. In addition, effects on the prey community can persist for extended periods of time (weeks, months, years), resulting in effects on fish feeding and growth long after an exposure has ended (Colville et al. 2008; Liess and Schulz 1999; Van den Brink et al. 1996; Ward et al. 1995).

The somatic growth model consists of 2 parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance due to insecticide exposure to potential reductions in the growth of individual fish. The population portion of the model links the sizes of individual subyearling salmon to their survival and the subsequent growth of the population. Models were constructed using MATLAB 7.7.0 (R2008b) (The MathWorks, Inc. Natick, MA).

Organismal Model

The organismal model tracks individual somatic growth of salmonid fingerlings using a series of relationships between pesticide exposure, AChE activity, feeding behavior, food uptake, and somatic growth rate (Figure 104, Figure 105, Figure 106). The model incorporates empirical data when available (Baldwin et al., 2009). Since growth and toxicity data are limited, extrapolation from 1 salmon species to the others was done with the assumption that the salmon stocks would exhibit similar physiological and toxicological responses. Sigmoidal dose-response relationships based upon the AChE inhibition EC50 values and their slopes are used to determine the level of

AChE activity (Figure 104A, B, C) from the exposure concentration of each pesticide exposure or pulse.

A linear relationship based on empirical data related AChE activity to feeding behavior (Sandahl et al. 2005, Figure 104D). Feeding behavior was then assumed to be directly proportional to food uptake, defined as potential ration (Figure 104E, Brett 1969). The potential ration expresses the amount of food the organism can consume when prey abundance is not limiting. Potential ration over time (Figure 104F) depicts how the food intake of individual fish changes in response to the behavioral effects of the pesticide exposure over the modeled growth period. Potential ration is equal to final ration if no effects on prey abundance are incorporated (Figure 106). When effects of pesticide exposure on prey abundance are incorporated, final ration is the product of potential ration (relating to the fish's ability to capture prey, Figure 104) and the relative abundance of prey available following exposure (Figure 105). Next, additional empirical data (e.g., Weatherley and Gill 1995) defined the relationship between final ration and somatic growth rate (Figure 106C). While the empirical relationship is more complex (e.g., somatic growth rate plateaus at rations above maximum feeding), a linear model was considered sufficient for the overall purpose of this model. Finally, the model combines these linear models relating AChE activity to feeding behavior, feeding behavior to potential ration, and final ration to somatic growth rate to produce a linear relationship between AChE activity and somatic growth rate (Figure 106D). One important assumption of the model is that the relationships are stable, i.e., do not change with time. The relationships would need to be modified to incorporate time as a variable if, for example, fish are shown to compensate over time for reduced AChE activity to improve their feeding behavior and increase food uptake.

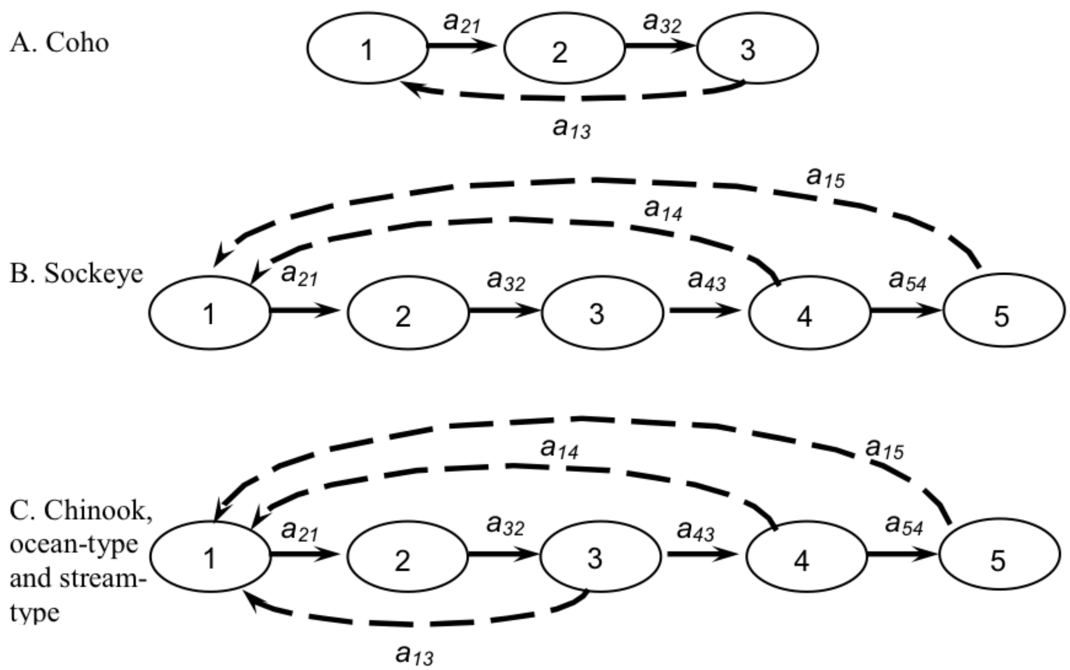
The models allow exposures that can include multiple AChE-inhibiting pesticides over various time pulses. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific EC50s and EC50s and slopes (Figure 104B, Figure 105B). The timecourse for each exposure was built into the model as a pulse with a defined start and end during which the exposure remained constant (Figure 104A, Figure 105A). The timecourse for AChE activity, on the other hand, was modeled using 2 single-order exponential functions, 1 for the time required for the exposure to reach full effect and the other for time required for complete recovery following the end of the exposure (time-to-effect AChE activity and time-to-recovery AChE activity, respectively; Figure 104C). The apparent activity level was back-calculated to result in a relative concentration (concentration/ AChE inhibition EC50) for each day of the growth period for each pulse. The relative concentration for each day was summed across all the pulses to result in a total apparent concentration for each day. The sigmoid slope used in the calculation of AChE activity using the apparent concentration was the arithmetic mean of the sigmoid slopes for each pesticide present on each day. The timecourse for relative prey abundance was modeled incorporating a 1 day spike in prey drift relative to the toxicity and available prey base followed by a drop in abundance due to the toxic impacts (Figure 105C). Recovery is assumed to be due to a constant influx of invertebrates from connected habitats (aquatic and terrestrial) that are not exposed to the pesticide. Incoming organisms are subject to toxicity if pesticides are still present and this alters the rate of recovery during exposures. Incorporating dynamic effects and recovery variables allows the model to simulate differences in the pharmacokinetics (e.g., the rates of uptake from the environment and of detoxification) of various pesticides and simulate differences in invertebrate community response and recovery rates (see below).

The relationship between final ration and somatic growth rate (Figure 106C) produces a relationship representing somatic growth rate over time (Figure 106D), which is then used to model individual growth rate and size over time. The growth models were run for 1000 individual fish, with initial weight selected from a normal distribution with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent subyearling size in the spring prior to the onset of pesticide application. For each iteration of the model (one day for the organismal model), the somatic growth rate is calculated for each fish by selecting the parameter values from normal distributions with specified means and standard deviations (Table 145). The weight for each fish is then adjusted based on the calculated growth rate to generate a new weight for the next iteration. The length (days) to run the growth portion of the model was selected to represent the time from when the fish enter the linear portion of their growth trajectory in the mid to late spring until they change their growth pattern in the fall due to reductions in temperature and resources or until they migrate out of the system. The outputs of the organismal model that are handed to the population models consist of mean weights (with standard deviations) after the species-appropriate growth period (Table 146). A sensitivity analysis was run to determine the influence of the parameter values on the output of the growth model.

The option of exposing only a specified percent of the population to the pesticide(s) during the somatic growth period is provided. The exposed percent of the population is applied to the number of individuals run in the individual growth model. After running all 1000 individual growth trajectories (with X percent exposed and 100-X percent control) the mean weight and standard deviation of the whole is determined and handed to the population model to run as the size distribution of the impacted population.

The parameter values defining control conditions that are constant for all the modeled species are listed in Table 145. Model parameters such as the length of the growth period and control daily growth rate that are species specific are listed in Table 146. Each exposure scenario was defined by a concentration and exposure time for each pesticide. The duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al. 2004), with a half-life of 0.5 days. Toxicity values describing 50% inhibition of AChE activity (IC_{50}) and the slope for each active ingredient are shown in Table 147, as reported in the carbaryl and methomyl biological evaluations (USEPA 2021).

The effects of exposures on the prey base are incorporated in the somatic growth model as the available ration (Macneale et al., 2014). For prey, it is assumed there is a constant, independent influx of prey from upstream habitats that will eventually (depending on the rate selected) return prey abundance to 1. As mentioned above, however, these invertebrates are subject to exposure once added to the system, and therefore prey recovery rate is a product of the influx rate as well as the exposure scenario. While recovery rates reported in the literature vary, it is assumed a 1% recovery rate is ecologically realistic (Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a specific floor (Figure 105B). This assumption depends on a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals that would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate. No studies specify floors per se, but studies quantifying invertebrate densities following highly toxic exposures indicate a floor of 0.2 is ecologically realistic (i.e., regardless of the exposure, 20% of a fish's ration will be available daily; e.g., Cuffney et al. 1984). Finally, because prey availability has been found to increase dramatically albeit briefly following pesticide exposures (due to immediate mortality and/or emigration of benthic prey into the water column; Davies and Cook 1993, Schulz 2004), a 1-day prey spike is included for the day following an exposure. The relative magnitude of the spike is calculated as the product of the standing prey availability the day prior to exposure (minus the floor), the toxicity of the exposure, and a constant of 20. This calculation therefore accounts for the potential prey that are available and the severity of the exposure. The spike will be greater when more prey are available and/or the toxicity of the exposure is greater; alternatively, the spike will be small when few prey are available and/or the exposure toxicity is low. The toxicity values for prey abundance (EC_{50} and sigmoid slope) were calculated as the lower 10^{th} percentile of the invertebrate species sensitivity distribution from the USEPA BE (Table 147).



D. Transition Matrix

$$A=\begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$

Figure 103. Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook (C) salmon. The life-history graph for a population labeled by age, with each transition element labeled according to the matrix position, aij, i row and j column. Dashed lines represent reproductive contribution and solid lines represent survival transitions. D) The transition matrix for the life-history graph depicted in C.

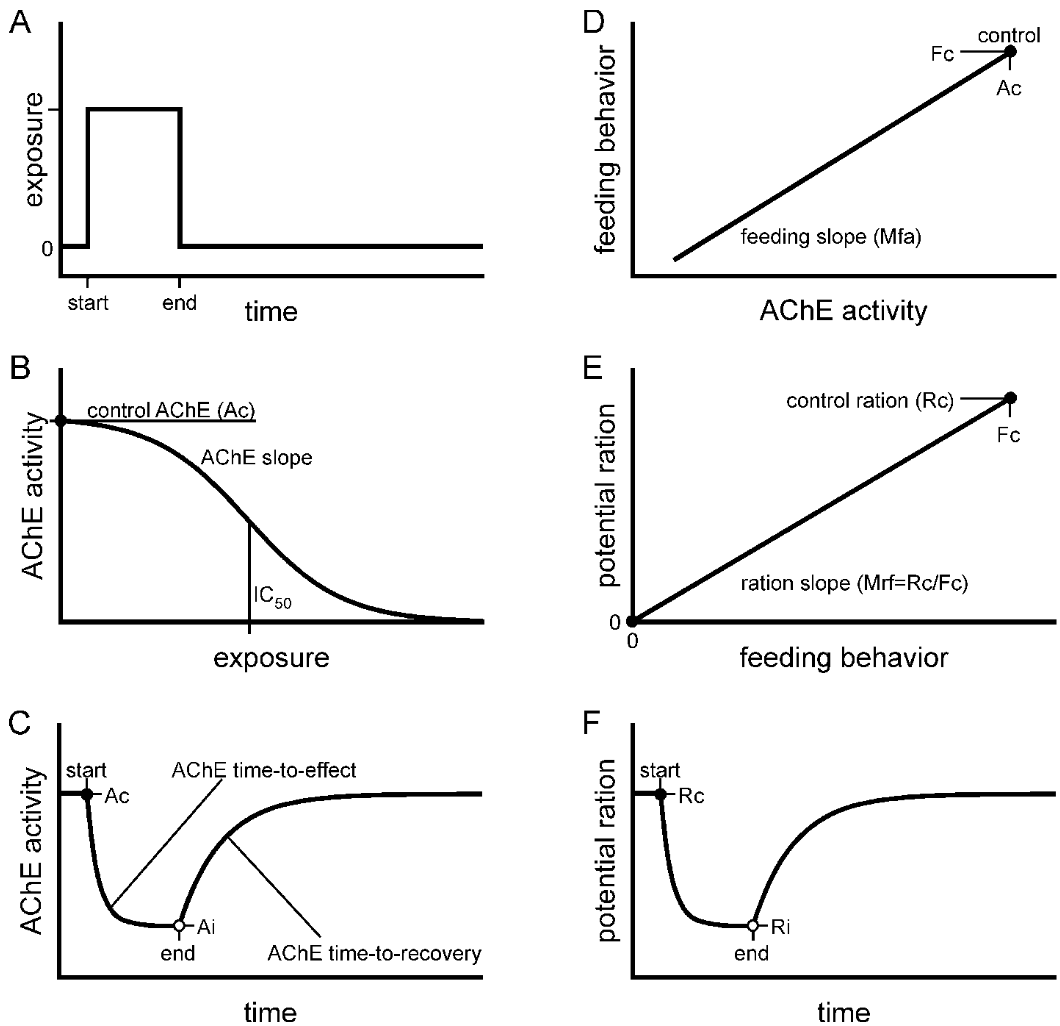


Figure 104. Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state acetylcholinesterase (AChE) activity showing a dose-dependent reduction defined by control activity (horizontal line, Ac), sigmoidal (i.e., hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, EC50). C) Timecourse of acetylcholinesterase inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (Ai) based on Panel B. D) Linear model relating acetylcholinesterase activity to feeding behavior using a line that passes through the feeding (Fc) and activity (Ac) control conditions with a slope of Mfa. E) The relationship between feeding behavior and the potential ratio an organism could acquire (if not food limited) used a line passing through the control conditions (Fc as in Panel D and the control ration, Rc) and through the origin producing a slope (Mrf) equal to Rc/Fc. F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C & E.

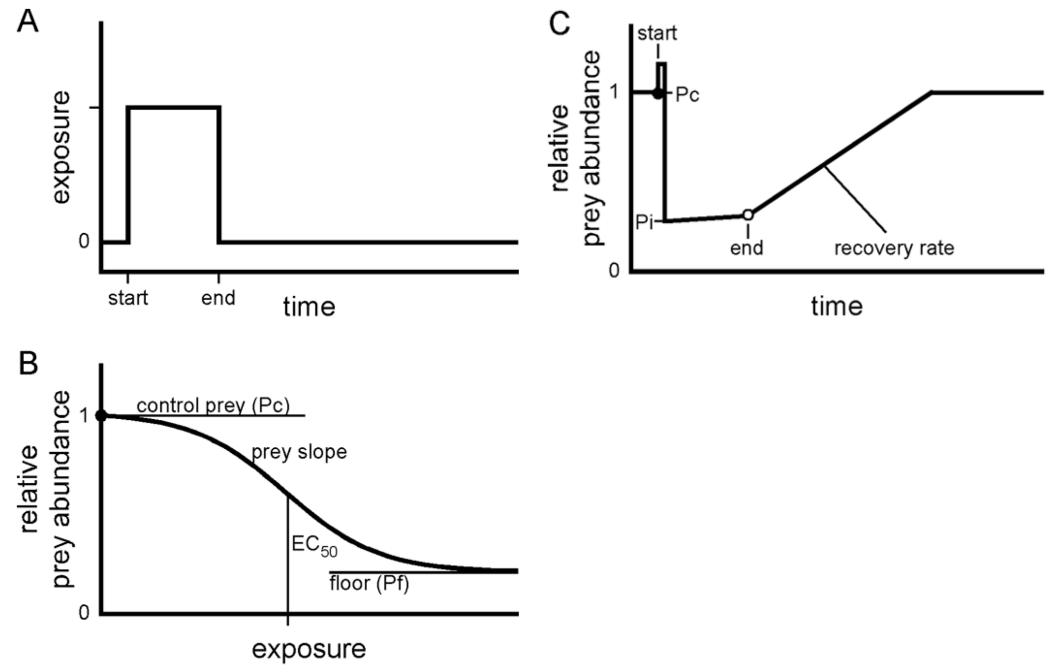


Figure 105. Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either single compound or mixtures). B) Sigmoidal relationship between exposure concentration and relative prey abundance showing a dose-dependent reduction defined by control abundance (horizontal line at 1, Pc), sigmoid (i.e., hille) slope (prey slope), the concentration producing a 50% reduction in prey (vertical line, EC50), and a minimum abundance always present (horizontal line denoted as floor, Pf). C) Timecourse of prey abundance including a 1-day spike in prey drift relative to the available prey and the level of toxicity followed by a drop to the level of impact or the floor whichever is greater. During extended exposures at low toxicity recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure recovery to control prey drift is at the constant rate of influx from upstream habitats.

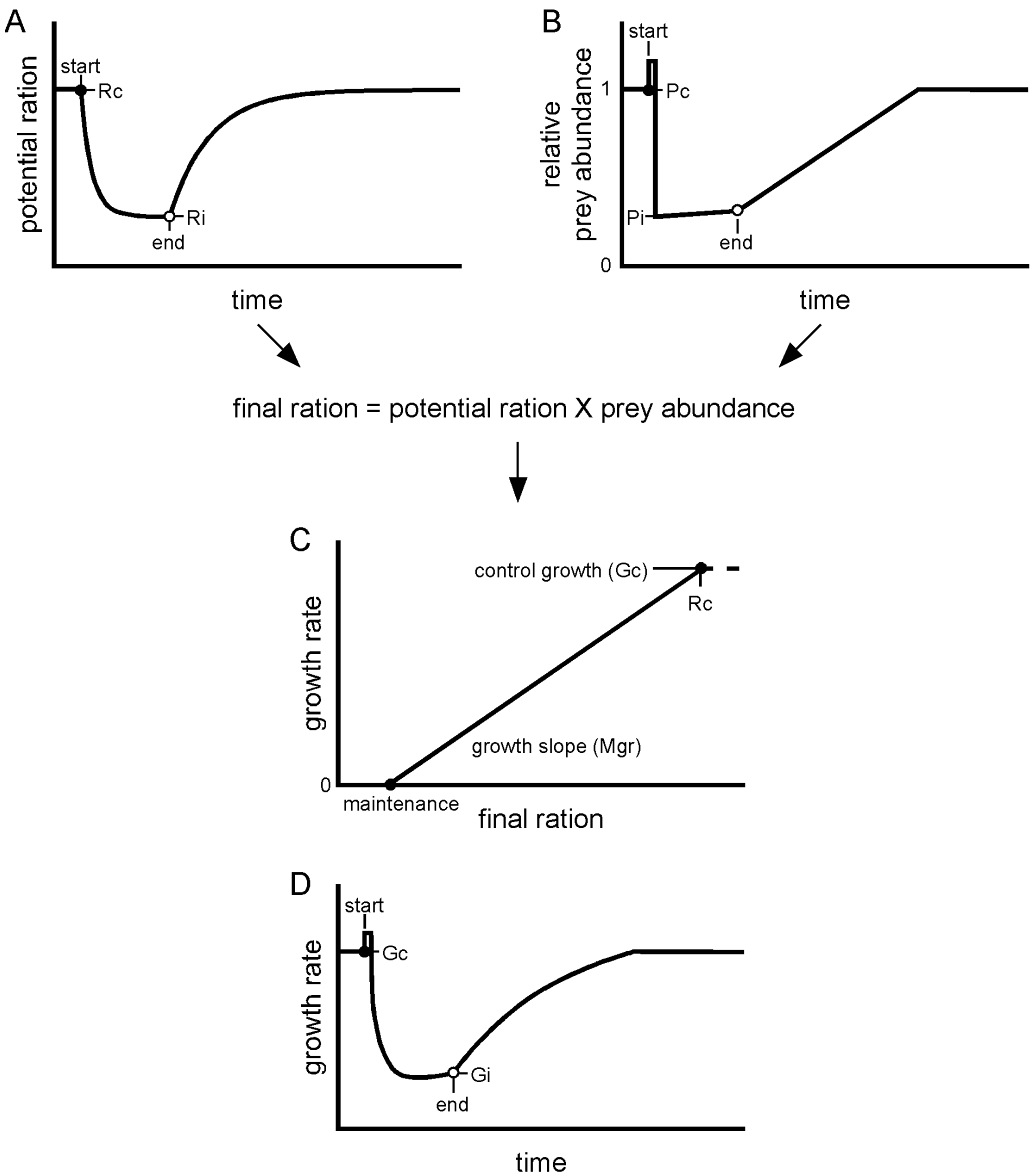


Figure 106. Relationships used to link anticholinesterase exposure to growth rate relating to long-term weight gain of each fish. See text for details. Relationships in A, B, and C utilize empirical data. Closed circles represent control conditions. Open circles (e.g., Ai) represent the exposed (inhibited) condition. A&B) Relationships describing the Timecourse of the effects of anticholinesterase exposure on the organisms ability to capture food (Panel A, potential ration) and the availability of food to capture (Panel B, relative prey abundance). The figures are the same as those in Figures A1-2F and 3C, respectively. For a given exposure concentration and time, multiplying potential ration by relative prey abundance yields the final ration acquired by the organism. C) A linear model was used to relate final ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. D) Timecourse for effect of exposure to anticholinesterase on growth rate produced by combining A, B, & C.

Below are the mathematical equations used to derive Figure 104, Figure 105, and Figure 106.

Figures Figure 104A and Figure 105A use a step function:

time < start; exposure = 0
start ≤ time ≤ end; exposure = exposure concentration(s)
time > end; exposure = 0.

Figures Figure 104B and Figure 105B use a sigmoid function:

y = bottom + (top – bottom)/(1 + (exposure concentration/EC50)^slope).
For 2B, y = AChE activity, top = Ac, bottom = 0.
For Figure 3B, y = prey abundance, top = Pc (in this case 1), bottom = Pf.

Figures Figure 104D,E, and Figure 106C use a linear function (the point-slope form of a line):

y = m*(x – x1) + y1.
For 2D, m= Mfa, x1 = Ac, and y1 = Fc.
For 2E, m= Mrf (computed as Rc/Fc), x1 = Fc, and y1 = Rc.
For 4C, m= Mgr, x1 = Rc, and y1 = Gc.

Figure Figure 104C uses a series of exponential functions:

time < start; y = c
start ≤ time ≤ end; y = c – (c – i)*(1 – exp(-ke*(time – start)))
time > end; ye = c – (c – i)*(1 – exp(-ke*(end – start)))
y = ye + (c – ye)*(1 – exp(-kr*(time – end))).

For Figure Figure 104C, c = Ac, i = Ai, ke = ln(2)/AChE effect half-life, kr = ln(2)/AChE recovery half-life. For Figure Figure 104C the value of ye is calculated to determine the amount of inhibition that is reached during the exposure time, which may not be long enough to reach the maximum level of inhibition.

For Figure Figure 105C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. During exposures resulting in low prey toxicity, toxicity-limited recovery can occur. After exposure ends, a constant rate of recovery proceeds until control drift is reached or another exposure occurs

preyavail=preydrift(day-1)-floor;
preytox=1/(1+(concentration)^preyslope);
preyrecrate=0.01;
preydriftrec = preyrecrate*preytox.
time=start; spike=(-1+10^(1.654*preyavail))*(1-preytox)
preydrift =preydrift+spike
start ≤ time ≤ end; preydrift=(preyavail*preytox)+preyrdriftrec+floor;
time>end; preydrift = preydrift(day-1)+preydriftrec

Figure Figure 104F is generated by using the output of Figure Figure 104C for a given time as the input for Figure 104D and using the resulting output of Figure 104D as the input for Figure 104E. The resulting output of Figure 104E produces a single time point in the relationship in Figure 104F. Performing this series of computations across multiple days produces the entire relationship in Figure 104F. Figure 106D is generated by taking the outputs of Figure 106A and Figure 106B for the same day. Note the relationship of Figure 106A is equivalent to Figure 104F. The resulting outputs of Figure 106A and Figure 106B are multiplied to produce a final ration for a given day. The prey abundance (Figure 106B) available for consumption during a prey spike is capped at a maximum of 1.5*control drift to provide a limited benefit to the individual fish. The final ration is used as input for Figure 106C to generate Figure 106D.

Salmonid Population Model

The weight distributions from the organismal growth portion of the model are used to calculate size-dependent first-year survival for a life-history matrix population model for each species and life-history type. This incorporates the impact that reductions in size could have on population growth rate and abundance. The first-year survival element of the transition matrix incorporates a size-dependent survival rate for a 3- or 4-month interval (depending upon the species) which takes the subyearlings up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type Chinook, coho, and sockeye models. For ocean-type Chinook, it is the 3-month period the subyearling smolt spend in the estuary and nearshore marine habitats (i.e., estuary survival). The weight distributions from the organismal model are converted to length distributions by applying condition factors from data for each modeled species (cf; 0.0095 for sockeye and 0.0115 for all others) as shown in Equation L.

Equation L: length(mm) = ((fish weight(g)/cf)^(1/3))*10

The relationship between length and early winter or estuary survival rate was adapted from Zabel and Achord (2004) to match the survival rate for each control model population (Howell et al. 1985, Kostow 1995, Myers et al. 2006). The relationship is based on the length of a subyearling salmon relative to the mean length of other competing subyearling salmon of the same species in the system, Equation D, and relates that relative difference to size-dependent survival based upon Equation S. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 146. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

Equation D: Δlength = fish length(mm) – mean length(mm)

Equation S: Survival $\phi = (e^{(\alpha+(0.0329*\Delta length))}) / (1 + e^{(\alpha+(0.0329*\Delta length))})$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations D and S generates a size-dependent survival probability for each fish. This process was replicated 1000 times for each exposure scenario and simultaneously 1000 times for the paired control scenario and results in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates are inserted in the calculation of first-year survival in the respective control and pesticide-exposed transition matrices.

The investigation of population-level responses to pesticide exposures uses life-history projection matrix models. Individuals within a population exhibit various growth, reproduction, and survivorship rates depending on their developmental or life-history stage or age. These age specific characteristics are depicted in the life-history graph (Figure 103A-D) in which transitions are depicted as arrows. The nonzero matrix elements represent transitions corresponding to reproductive contribution or survival, located in the top row and the subdiagonal of the matrix, respectively (Figure 103E). The survival transitions in the life-history graph are incorporated into the n x n square matrix (A) by assigning each age a number (1 through n) and each transition from age i to age j becomes the element a_{ij} of matrix A (i = row, j = column) and represent the proportion of the individuals in each age passing to the next age as a result of survival. The reproductive element (a_{1j}) gives the number of offspring that hatch per individual in the contributing age, j. The reproductive element value incorporates the proportion of females in each age, the proportion of females in the age that are sexually mature, fecundity, fertilization success, and hatch success.

In order to understand the relative impacts of a short-term pesticide exposure on exposed vs. unexposed fish, we used parameters for an idealized baseline population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. There were no definitive data available on the populations to support specific density dependent relationships, so rather than assign an unsupported relationship, the NAS recommendation was followed to utilize density independent parameters (NAS 2013a). The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent so that they remain in the range they occupied during the period when demographic data were collected.

In the model an individual fish experiences an exposure scenario once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All individuals in 1 cohort within a given population are assumed to be exposed to the pesticide during their subyearling spring-summer growth period. No other age classes experience the exposure. Regardless of the level of AChE inhibition due to the direct exposure, only the sublethal effects related to somatic growth are incorporated in the somatic growth model.

The model recalculates first-year survival for each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation S. Population model output consists of the percent change in lambda from the unexposed control populations derived from the mean of 2 thousand calculations of both the unexposed control population and the pesticide exposed population. Change in lambda, representing alterations to the population productivity, was selected as the primary model output for reasons outlined previously.

A prospective analysis of the transition matrix, A, (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ, equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ. Variability was integrated by repeating the calculation of λ 2000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij}, on λ was assessed by calculating the sensitivity values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij}, defined by $\delta\lambda / \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ. The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij} and equals (a_{ij}/λ) times the sensitivity of a_{ij}. One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life-history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, 4 life-history models were constructed. The differences in life history may result in different freshwater pesticide exposure profiles which can translate into potentially different population-level responses. Separate models were constructed for coho salmon, sockeye salmon, ocean-type and stream-type Chinook salmon. In all cases, transition values were determined from literature data on survival and reproductive characteristics of each species for populations that exhibit the life history strategy and were listed as endangered, threatened, or a species of concern under the ESA. All transition values are listed in Table 148.

A life-history transition matrix was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 148. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

The life-history matrix for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend 1 winter in freshwater, then migrate to the ocean to spend 3 to 4 winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only 1 winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al.1997). All age 3 returning adults are males. Hatch rate and first year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989) and fecundity (McGurk 2000). Fecundity values for age 4 females were 3374 (473) and for age 5 females were 4058 (557) (McGurk 2000). First year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table A1-4. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life-history matrix was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4 or 5. Ocean-type Chinook migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Greene and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4511(65), 5184(89), and 5812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the Chinook model are listed in Table 148. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first 3 months of estuary/ocean survival are the size-dependent stage. Size data for determining subyearling Chinook condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life-history matrix for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life-history matrices described above (transition matrix values in Table 148). The acute responses are modeled as direct reduction in the first year survival rate (S1). Two options are available to run, direct mortality estimates and exposure scenarios. Direct mortality can be input as percent mortality and is multiplied by the first-year survival rate in the transition matrix. Calculated EEC values can be assessed in the Risk-Plots to identify the appropriate level of mortality. In contrast, modelling exposure scenarios results in a cumulative reduction in survival as defined by the concentration and the dose-response curve (the LC_{50} and slope for each pesticide). A sigmoid dose-response relationship is used to accurately handle responses well away from LC_{50} and to be consistent with other does-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC_{50}, as well as the sigmoid slope for the LC_{50}. For a given concentration, a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life-history matrix. The model allows for a specified percentage of the population (0-100%) to experience the exposure.

Demographic variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 10,000 calculations of both the unexposed control population and the pesticide exposed population. For the purposes of this assessment, the percent change in lambda is defined as different from control when the difference between the mean percent change is greater than the percent of 1 standard deviation from the control lambda.

For examining acute mortality, only direct mortality was used as inputs for the models. Exposure scenarios using specific EECs were not modeled. Mortality rates from 5% to 100% were run in 5% increments. The mortality values were assessed across a combination of percent overlap values (10%, 25%, 50%, 80%, and 100%) to estimate population productivity across differences in pesticide use area overlap with the species distribution.

Results

Sensitivity Analysis

A sensitivity analysis conducted on the organismal model revealed that changes in the control somatic growth rate had the greatest influence on the final weights (Table 145). While this parameter value was experimentally derived for another species (sockeye salmon; Brett et al. 1969), this value was adapted for each model species and is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Other parameters related to the daily growth rate calculation, including the growth to ration slope (Mgr) and the control ration produced strong sensitivity values. Initial weight, the prey recovery rate and the prey floor also strongly influenced the final weight values (Table 145). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all 4 of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 145. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook, 0.25 for stream-type Chinook, and 0.24 for sockeye.

Acute Model Output

While trends in effects were seen for acute toxicity across all 4 life-history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all 4 salmon, while the differences are consequences of distinctions between the life-history matrices. The stream-type Chinook and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life-history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook than for stream-type Chinook and sockeye. While some life-history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Shifts in population growth rate occurred across mortality levels and increased with the percentage of the population exposed (Tables Table 149, Table 150, Table 151, Table 152 and Figures Figure 107, Figure 108, Figure 109, Figure 110). Percent changes in lambda were considered significant if they were outside of 1 standard deviation from the unexposed population. The tables can be used to estimate losses in productivity due to mortality resulting from expected environmental concentrations in habitat utilized by juvenile salmonids. The likelihood of population effects from death of juveniles increases for those populations that spend longer periods in freshwaters such as stream-type Chinook, sockeye, and coho salmon.

For those populations with lambdas greater than 1, reductions in lambda from death of subyearlings can lead to consequences to abundance and productivity. Attainment of recovery and time-associated goals would be delayed for populations with reduced lambdas. Many of the populations that are categorized as core populations or are important to individual strata have lambdas just above 1 and are essential to survival and recovery goals. Slight changes in lambda, even as small as 3-4%, would result in reduced abundances and increased time to meet population recovery goals. For those natural populations with current lambdas of less than 1, risk of extinction would increase, especially if several successive generations were exposed.

Somatic Growth Model Output

These results show that all 4 species can be severely affected by changes in juvenile growth resulting from AChE inhibition and reduced prey availability (Tables Table 153&Table 154 and Figure 111). This is driven by the loss of prey availability for these compounds. The concentrations that elicit reductions in population growth rate are expected to occur in salmonid habitats. The degree to which an actual threatened or endangered population is affected will depend on a host of factors including the number of individuals exposed, the duration of exposure, when they are exposed, and if they are exposed more than once. It is also important to realize that these are idealized populations and we did not incorporate other factors that can affect the sensitivity of exposed salmonids such as elevated temperatures, presence of mixtures of carbamates and organophosphate pesticides (also AChE inhibitors), and the condition of the fish. We also did not incorporate incidences of death due to acute toxicity in the growth model. We show however, that even without these other stressors taken into account there is strong evidence that given the expected concentrations in salmonid habitats that populations will be adversely affected if juvenile life stages are exposed.

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Table 145. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value[1]	Error[2]	Sensitivity[3]
acetylcholinesterase activity (Ac)	1.0[4,5]	0.06[5]	-0.167
feeding (Fc)	1.0[4,5]	0.05[5]	0.088
ration (Rc)	5% weight/day[6]	0.05[7]	-0.547
feeding vs. activity slope (Mfa)	1.0[5]	0.1[5]	-0.047
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.547
growth vs. activity slope (Mga)	1.75 (Mfa*Mrf*Mgr)	-	-
initial weight	1 gram[8]	0.1[8]	1.00
control prey drift	1.0[4]	0.05[11]	0.116
AChE impact time-to-effect ($t_{1/2}$)	0.5 day[9]	n/a	0.005
AChE time-to-recovery ($t_{1/2}$)	30 days[10]	n/a	-0.0001
prey floor	0.20[11]	n/a	0.178
prey recovery rate	0.01[12]	n/a	0.323
somatic growth rate (Gc)	1.3[13]	0.06[6]	2.531

[1] mean value of a normal distribution used in the model or constant value when no corresponding error is listed
[2] standard deviation of the normal distribution used in the model
[3] mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold
[4] other values relative to control
[5] derived from Sandahl et al. 2005
[6] derived from Brett et al. 1969
[7] data from Brett et al. 1969 has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability
[8] consistent with field-collected data for juvenile Chinook (Nelson et al. 2004)
[9] estimated from Ferrari et al. 2004
[10] consistent with Eder et al., 2007; Ferrari et al., 2004; Chambers et al., 2002
[11] estimated from Van den Brink et al. 1996
[12] derived from Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008
[13] derived from Brett et al. 1969 and adapted for ocean-type Chinook, used for sensitivity analysis

Table 146. Species specific control parameters to model organismal growth and survival rates. Growth period and survival rate are determined from the literature data listed for each species. Gc and α were calculated to make the basic model produce the appropriate size and survival values from the literature.

	Chinook Stream-type[1]	Chinook Ocean-type[2]	Coho[3]	Sockeye[4]
days to run organismal growth model	184	140	184	168
growth rate percent body wt/day (Gc)	1.28	1.30	0.90	1.183
α from equation S	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

[1] Values from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006
[2] Values from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007
[3] Values from data in Pess et al. 2002, Knudsen et al. 2002
[4] Values from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table 147. Effects values (ug/L) and slopes for AChE activity, and prey abundance dose-response curves.

compound	AChE Activity EC_{50}[1] ug/L	AChE Activity slope	Prey Abundance EC_{50}[3] ug/L	Prey Abundance Slope
Carbaryl	145.8	0.81	5.0	5.5
Methomyl	213	0.95	6.7	5.5

[1] Values are geometric means of those reported in EPA BEs.
[2] Values from EPA BEs and are the 5th percentile of the LC50 SSD.
[3] Values from analysis of global search of reported LC50 and EC50s reported in EPA's Ecotox database. See text.

Table 148. Matrix transition element (standard deviation) and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life-history graphs as depicted in Figure A1-1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element aij equals the rate of change in λ with respect to the transition element, defined by δλ/ δa. The elasticity of transition element aij is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1] (std)	S	E	Value[2] (std)	S	E	Value[3] (std)	S	E	Value[4]	S	E
S1	0.0643 (0.003)	3.844	0.247	0.0056 (0.001)	57.13	0.292	0.0296 (0.002)	11.59	0.333	0.0257 (0.003)	9.441	0.239
S2	0.1160 (0.002)	2.132	0.247	0.48 (0.097)	0.670	0.292	0.0505 (0.005)	6.809	0.333	0.183 (0.003)	1.326	0.239
S3	0.17006 (0.004)	1.448	0.246	0.246 (0.050)	0.476	0.106				0.499 (0.003)	0.486	0.239
S4	0.04 (0.002)	0.319	0.0127	0.136 (0.023)	0.136	0.0168				0.1377 (0.003)	0.322	0.0437
R3	0.5807 (0.089)	0.00184	0.0011	313.8 (38.1)	0.0006	0.186	732.8 (75.0)	0.000469	0.333			

R4	746.73 (86.62)	0.000313	0.233	677.1 (80.7)	0.000146	0.0896				379.57 (53.2)	0.000537	0.195
R5	1020.36 (101.33)	1.25E-05	0.0127	1028 (117.5)	1.80E-05	0.0168				608.7 (83.0)	7.28E-05	0.0437

[1] Value calculated from data in Healey and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006
[2] Value calculated from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007
[3] Value calculated from data in Pess et al. 2002, Knudsen et al. 2002
[4] Value calculated from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table 149. Acute mortality model output for ocean-type Chinook. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold indicates a percent change in population growth rage of greater than 1 standard deviation from control values. The baseline values for ocean-type Chinook are: lambda=1.09, standard deviation of 0.1, standard deviation as a percent of lambda is 9, and first year survival S1=5.64E-03.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (12.9)	0 (12.9)	-1 (12.8)	-1 (12.8)	-1 (12.7)
10	0 (130)	-1 (12.9)	-1 (12.8)	-3 (12.6)	-3 (12.4)
15	0 (12.9)	-1 (12.9)	-2 (12.8)	-4 (12.5)	-5 (12.2)
20	-1 (13.0)	-2 (13.0)	-3 (12.9)	-5 (12.5)	-6 (12.1)
25	-1 (13.1)	-2 (13.0)	-4 (13.3)	-6 (12.7)	-8 (11.8)
30	-1 (13.0)	-2 (13.3)	-5 (13.4)	-8 (12.7)	**-10 (11.5)**
35	-1 (13.3)	-3 (13.8)	-6 (13.9)	**-9 (13.0)**	**-12 (11.4)**
40	-1 (13.4)	-3 (14.0)	-7 (14.3)	**-11 (13.5)**	**-14 (11.1)**
45	-1 (133.6)	-4 (14.3)	-8 (15.4)	**-13 (14.1)**	**-16 (10.7)**
50	-2 (13.6)	-5 (14.9)	**-9 (16.0)**	**-15 (15.3)**	**-18 (10.5)**
55	-2 (14.0)	-5 (15.5)	**-11 (17.5)**	**-17 (16.5)**	**-21 (10.2)**
60	-2 (14.2)	-6 (16.9)	**-12 (18.6)**	**-20 (17.9)**	**-23 (9.7)**
65	-2 (14.3)	-7 (16.9)	**-14 (19.8)**	**-22 (19.1)**	**-26 (9.5)**
70	-3 (14.6)	-7 (17.8)	**-16 (21)**	**-24 (20.3)**	**-29 (8.9)**
75	-3 (15.2)	-8 (18.4)	**-17 (22.1)**	**-27 (21.6)**	**-33 (8.5)**
80	-3 (15.3)	**-9 (19.7)**	**-18 (23.2)**	**-30 (22.3)**	**-37 (8.1)**
85	-4 (15.8)	**-10 (20.4)**	**-20 (24)**	**-32 (23.1)**	**-42 (7.3)**
90	-4 (16.1)	**-10 (21.5)**	**-21 (24.9)**	**-34 (23.4)**	**-48 (6.6)**
95	-4 (16.5)	**-11 (22.7)**	**-22 (25.3)**	**-36 (23.2)**	**-56 (5.5)**
100	-4 (17.1)	**-12 (23.0)**	**-23 (25.9)**	**-38 (23.6)**	**-100 (NA)**

Table 150. Acute mortality model output for stream-type Chinook. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold indicates a percent change in population growth rage of greater than 1 standard deviation from control values. The baseline values for stream-type Chinook are: lambda=1.00, standard deviation of 0.03, standard deviation as a percent of lambda is 3, and first year survival S1=6.43E-03.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (4.4)	0 (4.4)	-1 (4.4)	-1 (4.4)	-1 (4.3)
10	0 (4.5)	-1 (4.5)	-1 (4.5)	-2 (4.4)	**-3 (4.3)**
15	0 (4.6)	-1 (4.7)	-2 (4.7)	**-3 (4.6)**	**-4 (4.2)**
20	-1 (4.7)	-1 (4.9)	**-3 (5.1)**	**-4 (4.8)**	**-5 (4.1)**
25	-1 (4.8)	-2 (5.1)	**-3 (5.5)**	**-6 (5.1)**	**-7 (4.1)**
30	-1 (4.9)	-2 (5.6)	**-4 (6.0)**	**-7 (5.6)**	**-8 (4.0)**
35	-1 (5.1)	-2 (6.0)	**-5 (6.8)**	**-8 (6.1)**	**-10 (4.0)**
40	-1 (5.4)	**-3 (6.5)**	**-6 (7.5)**	**-10 (6.9)**	**-12 (3.9)**
45	-1 (5.6)	**-3 (7.0)**	**-7 (8.5)**	**-11 (7.8)**	**-14 (3.7)**
50	-2 (5.8)	**-4 (7.5)**	**-8 (9.8)**	**-13 (9.3)**	**-16 (3.7)**
55	-2 (6.2)	**-4 (8.3)**	**-9 (11.1)**	**-15 (10.9)**	**-18 (3.6)**
60	-2 (6.5)	**-5 (9.3)**	**-11 (13.0)**	**-17 (13.1)**	**-20 (3.5)**
65	-2 (6.9)	**-6 (10.1)**	**-12 (14.7)**	**-19 (14.7)**	**-23 (3.4)**
70	-2 (7.2)	**-6 (11.1)**	**-13 (15.7)**	**-22 (16.7)**	**-26 (3.2)**
75	**-3 (7.7)**	**-7 (12.4)**	**-15 (17.5)**	**-24 (17.9)**	**-29 (3.1)**
80	**-3 (8.1)**	**-8 (13.5)**	**-15 (18.3)**	**-27 (18.8)**	**-33 (2.9)**
85	**-3 (8.6)**	**-8 (14.6)**	**-17 (19.3)**	**-29 (19.7)**	**-37 (2.7)**
90	**-3 (9.1)**	**-9 (15.4)**	**-18 (20.2)**	**-30 (20.0)**	**-43 (2.4)**
95	**-4 (9.5)**	**-10 (16.4)**	**-20 (21.1)**	**-32 (20.2)**	**-52 (2.0)**
100	**-4 (10.3)**	**-11 (17.6)**	**-21 (21.4)**	**-33 (20.0)**	**-100 (NA)**

Table 151. Acute mortality model output for sockeye. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold indicates a percent change in population growth rage of greater than 1 standard deviation from control values. The baseline values for sockeye are: lambda=1.01, standard deviation of 0.06, standard deviation as a percent of lambda is 6, and first year survival S1=2.57E-02. Bold indicates values greater than or equal to 1 standard deviation away from baseline.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (8.0)	0 (7.9)	-1 (7.9)	-1 (7.8)	-1 (7.8)
10	0 (8.0)	-1 (8.0)	-1 (8.0)	-2 (7.9)	-3 (7.7)
15	0 (8.0)	-1 (8.0)	-2 (8.1)	-3 (7.9)	-4 (7.7)
20	-1 (8.0)	-1 (8.2)	-3 (8.2)	-4 (8.1)	-5 (7.5)
25	-1 (8.1)	-2 (8.4)	-3 (8.5)	-5 (8.2)	**-7 (7.4)**
30	-1 (8.2)	-2 (8.8)	-4 (9.0)	**-7 (8.4)**	**-8 (7.3)**
35	-1 (8.4)	-2 (8.9)	-5 (9.6)	**-8 (8.8)**	**-10 (7.1)**
40	-1 (8.6)	-3 (9.2)	**-6 (10.1)**	**-9 (9.6)**	**-11 (7.0)**
45	-1 (8.7)	-3 (9.7)	**-7 (10.9)**	**-11 (10.4)**	**-13 (6.9)**
50	-1 (9.0)	-4 (10.4)	**-8 (12.0)**	**-13 (11.2)**	**-15 (6.7)**
55	-2 (9.2)	-4 (10.9)	**-9 (13.4)**	**-15 (12.9)**	**-17 (6.5)**
60	-2 (9.4)	-5 (11.9)	**-10 (14.4)**	**-17 (14.4)**	**-19 (6.4)**
65	-2 (9.7)	-5 (12.3)	**-12 (16.1)**	**-19 (15.7)**	**-22 (6.2)**
70	-2 (10.0)	**-6 (13.4)**	**-13 (16.9)**	**-21 (17.3)**	**-25 (5.9)**
75	-3 (10.4)	**-7 (14.3)**	**-14 (18.2)**	**-23 (18.1)**	**-28 (5.6)**
80	-3 (10.9)	**-8 (15.6)**	**-16 (19.0)**	**-26 (19.1)**	**-32 (5.4)**
85	-3 (11.3)	**-8 (16.3)**	**-17 (19.9)**	**-28 (19.7)**	**-39 (5.0)**
90	-3 (11.6)	**-9 (17.0)**	**-18 (20.8)**	**-29 (19.8)**	**-42 (4.5)**
95	-3 (12.3)	**-10 (17.7)**	**-19 (20.9)**	**-30 (19.9)**	**-51 (3.8)**
100	-4 (12.7)	**-10 (18.3)**	**-20 (21.5)**	**-32 (19.8)**	**-100 (NA)**

Table 152. Acute mortality model output for coho. Shown are the percent changes in population growth rate (lambda, λ) with the standard deviations in parentheses. The toxicity values were applied as direct mortality on first year survival (left column). The percent of the population exposed was also varied (top row). Bold indicates a percent change in population growth rage of greater than 1 standard deviation from control values. The baseline values for coho are: lambda=1.03, standard deviation of 0.05, standard deviation as a percent of lambda is 5, and first year survival S1=2.97E-02.

	percent population experiencing mortality				
percent mortality	**10**	**25**	**50**	**80**	**100**
5	0 (7.4)	0 (7.5)	-1 (7.5)	-1 (7.4)	-2 (7.4)
10	0 (7.5)	-1 (7.6)	-2 (7.6)	-3 (7.4)	-3 (7.2)
15	0 (7.6)	-1 (7.7)	-3 (7.8)	-4 (7.5)	**-5 (7.1)**
20	-1 (7.7)	-2 (8.0)	-4 (8.1)	**-6 (7.7)**	**-7 (7.0)**
25	-1 (7.9)	-2 (8.4)	**-5 (8.5)**	**-7 (8.0)**	**-9 (6.9)**
30	-1 (7.9)	-3 (8.5)	**-6 (9.1)**	**-9 (8.4)**	**-11 (6.6)**
35	-1 (8.2)	-3 (9.2)	**-7 (9.9)**	**-11 (8.9)**	**-13 (6.5)**
40	-1 (8.5)	-4 (9.7)	**-8 (10.7)**	**-13 (9.8)**	**-16 (6.4)**
45	-2 (8.8)	-4 (10.3)	**-9 (11.8)**	**-14 (11.0)**	**-18 (6.1)**
50	-2 (9.1)	**-5 (11.1)**	**-10 (13.4)**	**-17 (12.2)**	**-21 (5.9)**
55	-2 (9.5)	**-6 (11.7)**	**-12 (14.9)**	**-20 (14.2)**	**-23 (5.8)**
60	-3 (9.9)	**-6 (12.6)**	**-14 (17.0)**	**-23 (16.5)**	**-26 (5.5)**
65	-3 (10.3)	**-7 (14.1)**	**-15 (18.5)**	**-25 (18.7)**	**-30 (5.3)**
70	-3 (10.7)	**-8 (15.1)**	**-17 (20.6)**	**-28 (20.6)**	**-33 (5.0)**
75	-3 (11.2)	**-9 (16.4)**	**-19 (22.3)**	**-31 (22.4)**	**-37 (4.7)**
80	-4 (11.6)	**-9 (17.7)**	**-20 (23.6)**	**-34 (23.7)**	**-42 (4.4)**
85	-4 (12.3)	**-11 (19.3)**	**-22 (25.0)**	**-37 (24.5)**	**-47 (4.0)**
90	-4 (12.9)	**-12 (20.4)**	**-24 (26.0)**	**-39 (25.2)**	**-54 (3.4)**
95	-4 (13.4)	**-13 (21.6)**	**-25 (27.3)**	**-42 (25.2)**	**-63 (2.8)**
100	**-5 (14.1)**	**-14 (22.9)**	**-27 (27.6)**	**-43 (25.7)**	**-100 (NA)**

Table 153. Somatic growth model output for carbaryl. Scenario used to generate output was a single, 4-day exposure beginning on day 1 of the somatic growth period with 100% of the population exposed. AChE IC50=145.8, AChE slope = 0.81, Prey EC50 = 5.0, Prey Slope 5.5. Prey floor 20%. Values in bold exceed the significant percent change (one standard deviation of the percent change in growth rate, lambda) for the control matrix. S1 indicates first year survival rate.

Species	***Concentration(µg/L)***	***0.0***	***4ug/L***	***5 ug/L***	***7.5 ug/L***	***10 ug/L***	***25 ug/L***	***75ug/L***	***145.8 ug/L***
Chinook Ocean-type	percent change lambda	na	-1	-3	**-9**	**-11**	**-11**	**-11**	**-11**
	percent change lambda std	na	10	10	**9**	**9**	**9**	**9**	**9**
	lambda mean	1.09	1.09	1.07	1	0.98	0.97	0.97	0.97
	lambda std	0.08	0.08	0.08	0.07	0.07	0.07	0.07	0.07
	S1	0.00561	5.55E-03	5.13E-03	4.05E-03	3.83E-03	3.74E-03	3.75E-03	3.74E-03
7	Significant percent change								
Chinook Stream-type	percent change lambda	na	-1	**-3**	**-10**	**-11**	**-12**	**-12**	**-12**
	percent change lambda std	na	5	**4**	**4**	**4**	**4**	**4**	**4**
	lambda mean	1	0.99	0.97	0.9	0.89	0.88	0.88	0.88
	lambda std	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	S1	0.0643	6.28E-02	5.71E-02	4.25E-02	3.95E-02	3.87E-02	3.82E-02	3.79E-02
3	Significant percent change								
Sockeye	percent change lambda	na	0	-2	**-7**	**-9**	**-9**	**-9**	**-9**
	percent change lambda std	na	6	6	**6**	**6**	**5**	**6**	**5**
	lambda mean	1.01	1.01	0.99	0.94	0.92	0.92	0.92	0.92
	lambda std	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04

		S1	0.0257	2.53E-02	2.35E-02	1.87E-02	1.75E-02	1.71E-02	1.71E-02	1.71E-02
	4	Significant percent change								
Coho		percent change lambda	na	-1	-3	**-9**	**-10**	**-10**	**-10**	**-10**
		percent change lambda std	na	8	8	**7**	**7**	**7**	**7**	**7**
		lambda mean	1.03	1.02	1	0.94	0.93	0.92	0.92	0.92
		lambda std	0.06	0.06	0.06	0.06	0.05	0.05	0.06	0.06
		S1	0.0297	2.92E-02	2.74E-02	2.28E-02	2.17E-02	2.14E-02	2.13E-02	2.13E-02
	6	Significant percent change								

Table 154. Somatic growth model output for methomyl. Scenario used to generate output was a single, 4-day exposure beginning on day 1 of the somatic growth period with 100% of the population exposed. AChE IC50=213, AChE slope = 0.95, Prey EC50 = 6.7, Prey Slope 5.5. Prey floor 20%. Values in bold exceed the significant percent change (one standard deviation of the percent change in growth rate, lambda) for the control matrix. S1 indicates first year survival rate.

Species	***Concentration(µg/L)***	***0.0***	***5ug/L***	***6.7ug/L***	***10ug/L***	***25ug/L***	***50ug/L***	***100ug/L***	***213ug/L***
Chinook Ocean-type	percent change lambda	na	0	-3	**-9**	**-11**	**-11**	**-11**	**-11**
	percent change lambda std	na	10	10	**9**	**9**	**9**	**9**	**9**
	lambda mean	1.09	1.09	1.06	1	0.97	0.97	0.97	0.97
	lambda std	0.08	0.08	0.08	0.07	0.07	0.07	0.07	0.07
	S1	0.00561	5.58E-03	5.12E-03	4.07E-03	3.78E-03	3.77E-03	3.72E-03	3.74E-03
7	Significant percent change								
Chinook Stream-type	percent change lambda	na	0	**-3**	**-10**	**-12**	**-12**	**-12**	**-12**
	percent change lambda std	na	5	**5**	**4**	**4**	**4**	**4**	**4**
	lambda mean	1	1	0.97	0.9	0.88	0.88	0.88	0.88
	lambda std	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	S1	0.0643	6.35E-02	5.72E-02	4.27E-02	3.89E-02	3.83E-02	3.80E-02	3.79E-02
3	Significant percent change								
Sockeye	percent change lambda	na	0	-2	**-7**	**-9**	**-9**	**-9**	**-9**
	percent change lambda std	na	6	6	**6**	**6**	**6**	**6**	**6**

		lambda mean	1.01	1.01	0.99	0.94	0.92	0.92	0.92	0.92
		lambda std	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
		S1	0.0257	2.56E-02	2.35E-02	1.86E-02	1.72E-02	1.72E-02	1.72E-02	1.70E-02
	4	Significant percent change								
Coho		percent change lambda	na	-1	-5	**-18**	**-22**	**-22**	**-22**	**-23**
		percent change lambda std	na	8	8	**7**	**7**	**7**	**7**	**7**
		lambda mean	1.03	1.02	0.98	0.85	0.8	0.8	0.8	0.8
		lambda std	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
		S1	0.0297	2.90E-02	2.53E-02	1.66E-02	1.43E-02	1.42E-02	1.41E-02	1.38E-02
	6	Significant percent change								

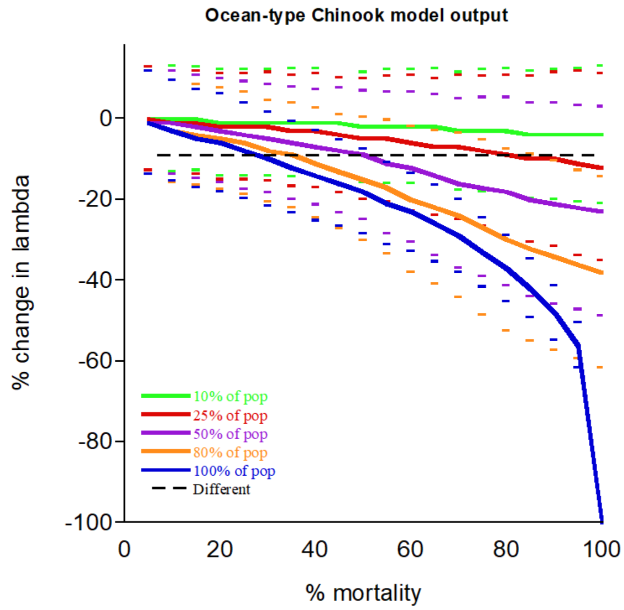

Figure 107. Percent change in population growth rate (lambda) for ocean-type Chinook for acute mortality rates from 5% to 100%. Solid lines indicate the percent of the population exposed and experiencing the acute mortality. The dotted line indicates 1 standard deviation from the baseline.

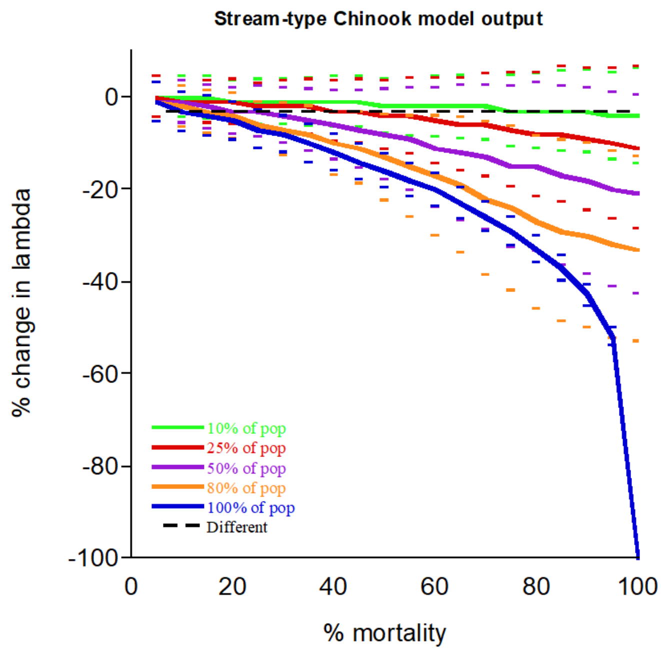

Figure 108. Percent change in population growth rate (lambda) for stream-type Chinook for acute mortality rates from 5% to 100%. Solid lines indicate the percent of the population exposed and experiencing the acute mortality. The dotted line indicates 1 standard deviation from the baseline.

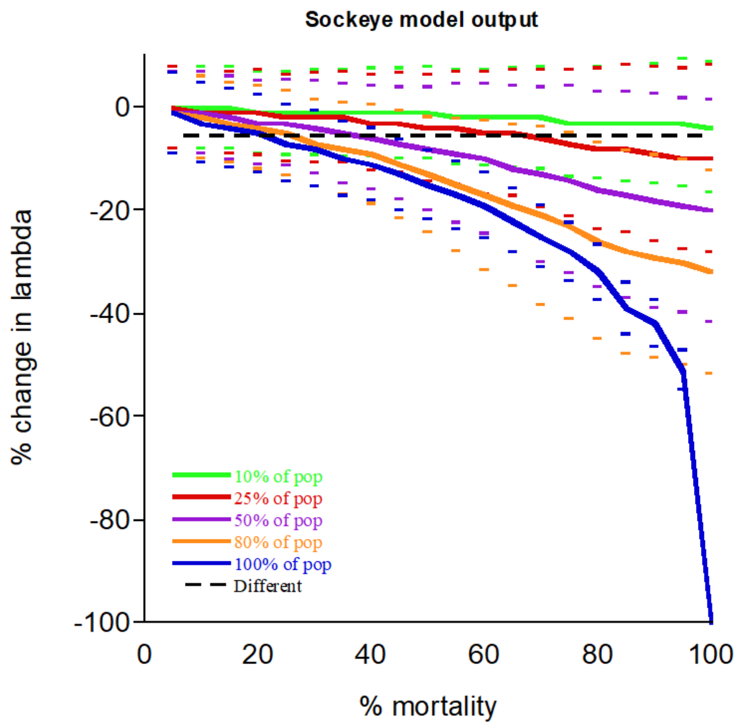

Figure 109. Percent change in population growth rate (lambda) for sockeye for acute mortality rates from 5% to 100%. Solid lines indicate the percent of the population exposed and experiencing the acute mortality. The dotted line indicates 1 standard deviation from the baseline.

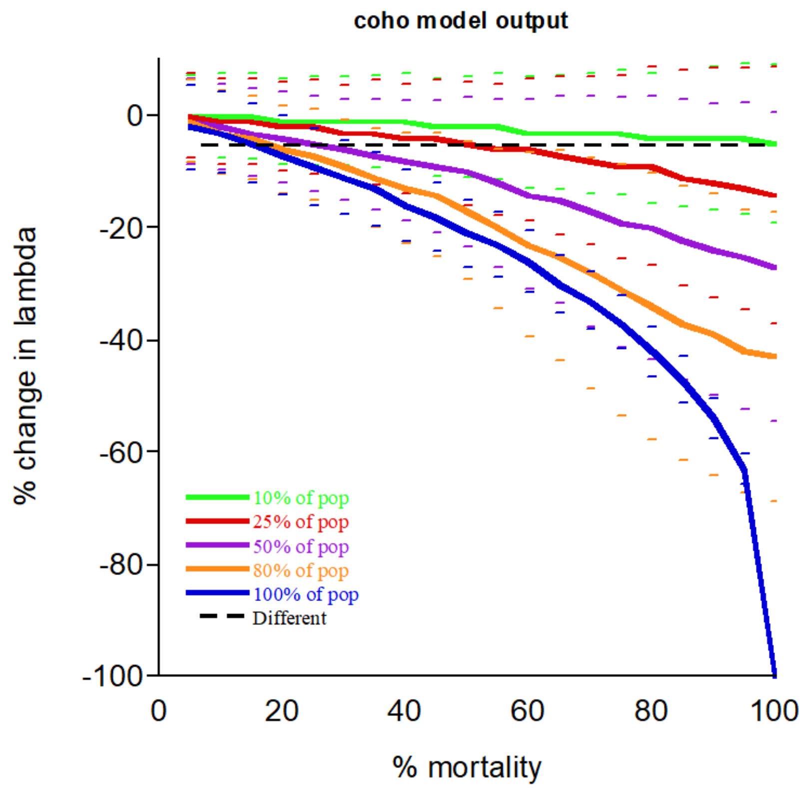

Figure 110. Percent change in population growth rate (lambda) for coho for acute mortality rates from 5% to 100%. Solid lines indicate the percent of the population exposed and experiencing the acute mortality. The dotted line indicates 1 standard deviation from the baseline.

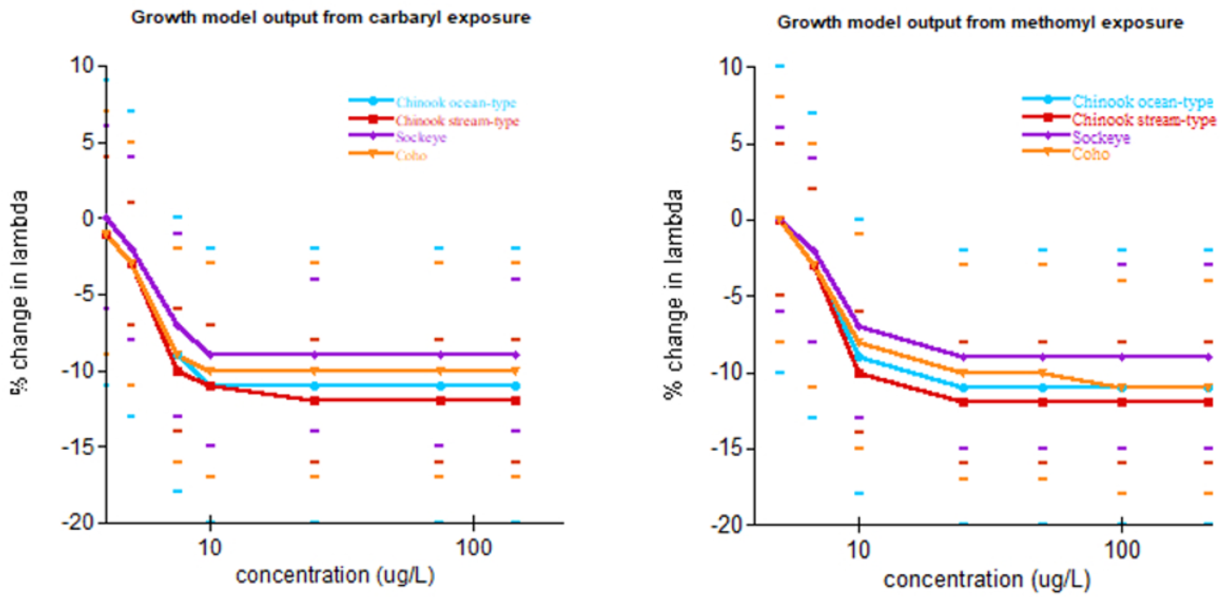

Figure 111. Somatic growth model output for carbaryl and methomyl. Scenario used to generate output was a single, 4-day exposure beginning on day 1 of the somatic growth period with 100% of the population exposed. Lines indicate mean percent change in lambda and caps show 1 standard deviation.

APPENDIX B: PHYSICAL OR BIOLOGICAL FEATURES

Table 155. Physical or biological features essential for the conservation of the species for NMFS ESA-listed species under consultation.

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Black Abalone *Haliotis cracherodii*	76 FR 66806 10/27/2011	• Rocky substrate: Rocky benches, crevices, large boulders • Food resources: Bacterial and diatom films, algae • Juvenile settlement habitat: Rocky habitat with coralline algae and/or crevices, cryptic biogenic structures • Suitable water quality • Suitable nearshore circulation patterns
White Abalone *Haliotis sorenseni*	66 FR 29046 05/29/2001	NO DESIGNATED CRITICAL HABITAT. A designation was deemed not prudent because it was expected to increase risk of poaching
Elkhorn Coral *Acropora palmate* Staghorn Coral *A. cervicornis*	73 FR 72210 11/26/2008	Substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments
Caribbean Corals: • Lobed Star Coral *Orbicella annularis* • Mountainous Star Coral *O. faveolata* • Boulder Star Coral *O. franksi* • Pillar Coral *Dendrogyra cylindrus* • Rough Cactus Coral *Mycetophyllia ferox*	88 FR 83644 11/30/2023	The PBFs identified as essential to the conservation of each species is reproductive, recruitment, growth, and maturation habitat. Sites that support the normal function of all life stages of the corals are natural, consolidated hard substrate or dead coral skeleton free of algae and sediment at the appropriate scale at the point of larval settlement of fragment reattachment, and the associated water column. Several attributes of these sites determine the quality of the area and influence the value of the associated feature of the conservation of the species: • Substrate with presence of crevices and holes that provide cryptic habitat, the presence of microbial biofilms, or presence of crustose coralline algae; • Reefscape with no more than a thin veneer of sediment and low occupancy by fleshy and turf macroalgae; • Marine waters with levels of temperature, aragonite saturation, nutrients, and water clarity that have been observed to support any demographic function; and • Marine water with levels of anthropogenically-introduced (from humans) chemical contaminants that do not preclude or inhibit any demographic function.
Indo-Pacific Corals: • *Acropora globiceps* • *A. retusa* • *A. speciosa* • *Euphyllia paradivisa* • *Isopora crateriformis*	Proposed 85 FR 76262 11/27/2020	

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Green Turtle *Chelonia mydas* • North Atlantic DPS • South Atlantic DPS • East Pacific DPS • Central North Pacific DPS • Central South Pacific DPS • Central West Pacific DPS	Proposed 88 FR 46572 07/19/2023	The following generalized features are essential to the conservation of at least 1 DPS: • Reproductive essential feature - Mean high water line to 20 m depth of sufficiently dark and unobstructed waters adjacent to nesting beaches. • Migratory essential feature - Mean high water line to a particular depth/distance (varies by DPS) of sufficiently unobstructed corridors for transit of reproductive individuals between benthic foraging/resting areas and reproductive areas. • Benthic foraging/resting essential features - Mean high water line to 20 m depth of underwater refugia and food resources of sufficient condition, distribution, diversity, abundance, and density. • Surface-pelagic foraging/resting essential features – Oceanographic features/currents which result in concentrated components of the Sargassum-dominated drift community and currents which carry turtles to Sargassum-dominated drift communities, with sufficient food resources and refugia in at least 10 m water depth.
Hawksbill Turtle *Eretmochelys imbricata*	63 FR 46693 09/02/1998	Hawksbills depend on coral reefs for food and shelter; therefore, the condition of reefs directly affects the hawksbill's well-being. Hawksbills utilize both low- and high-energy nesting beaches in tropical oceans of the world. Within the southeastern United States they occur principally in Puerto Rico and in the U.S. Virgin Islands, with the most important sites being Mona Island in Puerto Rico and Buck Island Reef National Monument in the U.S. Virgin Islands
Leatherback Turtle *Dermochelys coriacea*	44 FR 17710 03/23/1979 77 FR 4170 01/26/2012	• Occurrence of prey species, primarily Scyphomedusae of the order Semaeostomeae (Chrysaora, Aurelia, Phacellophora, and Cyanea) of sufficient condition, distribution, diversity, and abundance to support individual as well as population growth, reproduction, and development • Migratory pathway conditions to allow for safe and timely passage and access to/from/within high use foraging areas
Loggerhead Turtle *Caretta caretta* Northwest Atlantic Ocean DPS	79 FR 39855 07/10/2014	Nearshore Reproductive Habitat • Nearshore waters directly off the highest density nesting beaches and their adjacent beaches as identified in 50 CFR 17.95(c) to 1.6 km (1 mile) offshore;

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Loggerhead Turtle *Caretta caretta* Northwest Atlantic Ocean DPS (continued)	79 FR 39855 07/10/2014	• Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water. • Waters with minimal manmade structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents. Winter Habitat • Water temperatures above 10° C from • November through April; • Continental shelf waters in proximity to the western boundary of the Gulf Stream; and • Water depths between 20 and 100 m. Breeding Habitat • High densities of reproductive male and female loggerheads; • Proximity to primary Florida migratory corridor; and • Proximity to Florida nesting grounds. Migratory Habitat • Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways; and • Passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas. Sargassum Habitat • Convergence zones, surface-water downwelling areas, the margins of major boundary currents (Gulf Stream), and other locations where there are concentrated components of the Sargassum community in water temperatures suitable for the optimal growth of Sargassum and inhabitance of loggerheads; • Sargassum in concentrations that support adequate prey abundance and cover; • Available prey and other material associated with Sargassum habitat including, but not limited to, plants and cyanobacteria and animals native to the Sargassum community such as hydroids and copepods; and

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Loggerhead Turtle *Caretta caretta* Northwest Atlantic Ocean DPS (continued)	79 FR 39855 07/10/2014	• Sufficient water depth and proximity to available currents to ensure offshore transport (out of the surf zone), and foraging and cover requirements by Sargassum for post-hatchling loggerheads, i.e., >10 meters depth.
Killer Whale *Orcinus orca* Southern Resident DPS	86 FR 41668 08/02/2021	• Water quality to support growth and development; • Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and • Passage conditions to allow for migration, resting, and foraging.
Hawaiian Monk Seal *Neomonachus schauinslandi*	51 FR 16047 04/30/1986 53 FR 18988 05/26/1988 extended CH from 10 to 20 fathoms deep around NWHI 80 FR 50925 8/21/2015	• Pupping and major hauling beaches including the vegetation immediately backing the beaches (coral sand beaches and lava benches). • Shallow protected water adjacent to the above (tide pools, inner reef waters, shoal areas, and near shore shallows). • Deeper inner reef areas and lagoon waters. • Other waters surrounding the NWHI to at least 80 fathoms. • Banks and shoals without emergent lands and pelagic waters. • Terrestrial areas and adjacent shallow, sheltered aquatic areas with characteristics preferred by monk seals for pupping and nursing • Marine areas from 0 to 200 meters in depth that support adequate prey quality and quantity for juvenile and adult monk seal foraging • Significant areas used by monk seals for hauling out, resting, or molting
Steller Sea Lion *Eumetopias jubatus* (Eastern DPS delisted, but CH still in effect; see 78 FR 66139)	58 FR 45269 8/27/1993	Terrestrial, air, and aquatic areas that support: • Reproduction • Foraging • Rest • Refuge
Atlantic Salmon *Salmo salar* Gulf of Maine DPS	74 FR 29300 6/19/2009	Spawning and Rearing • Deep, oxygenated pools and cover (e.g., boulders, woody debris, vegetation, etc.), near freshwater spawning sites, necessary to support adult migrants during summer while they await spawning in the fall.

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Atlantic Salmon *Salmo salar* Gulf of Maine DPS (continued)	74 FR 29300 6/19/2009	• Freshwater spawning sites that contain clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support spawning activity, egg incubation, and larval development. • Freshwater spawning and rearing sites with clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support emergence, territorial development and feeding activities of Atlantic salmon fry. • Freshwater rearing sites with space to accommodate growth and survival of Atlantic parr. • Freshwater rearing sites with a combination of river, stream, and lake habitats that accommodate parr's ability to occupy many niches and maximize parr production. Migration • Freshwater and estuary migratory sites free from physical and biological barriers that delay or prevent access of adult seeking spawning grounds needed to support recovered populations. • Freshwater and estuary migration sites with pool, lake, and instream habitat that provide cool, oxygenated water and cover items (e.g., boulders, woody debris, and vegetation) to serve as temporary holding and resting areas during upstream migration of adults. • Freshwater and estuary migration sites with abundant, diverse native fish communities to serve as a protective buffer against predation. • Freshwater and estuary migration sites free from physical and biological barriers that delay or prevent emigration of smolts to the marine environment. • Freshwater and estuary migration sites with sufficiently cool water temperatures and water flows that coincide with diurnal cues to stimulate smolt migration.
Smalltooth Sawfish *Pristis pectinate* U.S. DPS	74 FR 45353 09/02/2009	Juvenile Nursery Habitat • Red mangroves, and • Adjacent shallow euryhaline habitats and • the nursery area functions they provide to facilitate recruitment of juveniles into the adult population.
Gulf Sturgeon *Acipenser oxyrinchus desotoi*	68 FR 13370 03/19/2003	• Abundant food items, such as detritus, aquatic insects, worms, and/or molluscs, within riverine habitats for larval and juvenile life stages; and abundant prey items, such as amphipods, lancelets, polychaetes, gastropods,

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Gulf Sturgeon *Acipenser oxyrinchus desotoi* (continued)	68 FR 13370 03/19/2003	ghost shrimp, isopods, molluscs and/or crustaceans, within estuarine and marine habitats and substrates for subadult and adult life stages. • Riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone, or hard clay; • Riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during freshwater residency and possibly for osmoregulatory functions; • A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging, and for maintaining spawning sites in suitable condition for egg attachment, egg sheltering, resting, and larval staging; • Water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; • Sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and • Safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g., an unobstructed river or a dammed river that still allows for passage).
Green Sturgeon *Acipenser medirostris* Southern DPS	74 FR 52300 10/9/2009	Freshwater areas • Food resources. Abundant prey items for larval, juvenile, subadult, and adult life stages. • Substrate type or size (i.e., structural features of substrates) • Water flow. A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages. • Water quality. Water quality, including temperature, salinity, oxygen content, and other chemical

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Green Sturgeon *Acipenser medirostris* Southern DPS (continued)	74 FR 52300 10/9/2009	characteristics, necessary for normal behavior, growth, and viability of all life stages. • Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within riverine habitats and between riverine and estuarine habitats (e.g., an unobstructed river or dammed river that still allows for safe and timely passage). • Water depth. Deep (≥5 m) holding pools for both upstream and downstream holding of adult or subadult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or subadult fish. • Sediment quality. Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. Estuarine areas • Food resources. Abundant prey items within estuarine habitats and substrates for juvenile, subadult, and adult life stages. • Water flow. Within bays and estuaries adjacent to the Sacramento River (i.e., the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds. • Water quality. Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages. • Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats. • Water depth. A diversity of depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages. Sediment quality. Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of elevated levels of contaminants

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Green Sturgeon *Acipenser medirostris* Southern DPS (continued)	74 FR 52300 10/9/2009	Coastal Marine Areas • Migratory corridor. A migratory pathway necessary for the safe and timely passage of Southern DPS fish within marine and between estuarine and marine habitats. • Water quality. Coastal marine waters with adequate DO levels and acceptably low levels of contaminants (e.g., pesticides, PAHs, heavy metals that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon). • Food resources. Abundant prey items for subadults and adults, which may include benthic invertebrates and fish.
Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* Gulf of Maine DPS	82 FR 39160 9/18/2017	• Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 ppt range) for settlement of fertilized eggs, refuge, growth, and development of early life stages • Aquatic habitat with a gradual downstream salinity gradient of 0.5 to 30 ppt and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development • Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: (1) Unimpeded movement of adults to and from spawning sites; (2) seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and (3) staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (e.g., ≥1.2 m) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river • Water, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support: (1) Spawning; (2) annual and interannual adult, subadult, larval, and juvenile survival; and (3) larval, juvenile, and subadult growth, development, and recruitment (e.g., 13 °C to 26 °C for spawning habitat and no more than 30° C for juvenile rearing habitat, and 6 mg/L DO for juvenile rearing habitat)
Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* New York Bight DPS	82 FR 39160 9/18/2017	
Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* Chesapeake Bay DPS	82 FR 39160 9/18/2017	
Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* Carolina DPS	82 FR 39160 9/18/2017	• Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0-

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* South Atlantic DPS	82 FR 39160 9/18/2017	0.5 ppt range) for settlement of fertilized eggs and refuge, growth, and development of early life stages • Transitional salinity zones inclusive of waters with a gradual downstream gradient of 0.5-30 ppt and soft substrate (e.g., sand, mud) between the river mouths and spawning sites for juvenile foraging and physiological development • Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: (1) Unimpeded movement of adults to and from spawning sites; (2) seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and (3) staging, resting, or holding of subadults and spawning condition adults. Water depths in main river channels must be deep enough (≥1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river • Water quality conditions, especially in the bottom meter of the water column, between river mouths and spawning sites with temperature and oxygen values that support: (1) Spawning; (2) annual and inter-annual adult, subadult, larval, and juvenile survival; and (3) larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L D.O. for juvenile rearing habitat likely supports juvenile rearing habitat, whereas D.O. less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25 °C. In temperatures greater than 26 °C, D.O. greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 °C to 26 °C likely to support spawning habitat.
Eulachon *Thaleichthys pacificus* Southern DPS	76 FR 65323 10/20/2011	Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles. • Flow: A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) that supports spawning, and survival of all life stages.

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Eulachon *Thaleichthys pacificus* Southern DPS (continued)	76 FR 65323 10/20/2011	• Water Quality: Water quality suitable for spawning and viability of all eulachon life stages. Sublethal concentrations of contaminants affect the survival of aquatic species by increasing stress, predisposing organisms to disease, delaying development, and disrupting physiological processes, including reproduction. • Water Temperature: Suitable water temperatures, within natural ranges, in eulachon spawning reaches. • Substrate: Spawning substrates for eulachon egg deposition and development. Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted. • Migratory Corridor: Safe and unobstructed migratory pathways for eulachon adults to pass from the ocean through estuarine areas to riverine habitats in order to spawn, and for larval eulachon to access rearing habitats within the estuaries and juvenile and adults to access habitats in the ocean. • Flow: A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) that supports spawning migration and outmigration of larval eulachon from spawning sites. • Water Quality: Water quality suitable for survival and migration of spawning adults and larval eulachon. • Water Temperature: Water temperature suitable for survival and migration. • Food: Prey resources to support larval eulachon survival. Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival. • Food: Prey items, in a concentration that supports foraging leading to adequate growth and reproductive development for juveniles and adults in the marine environment. • Water Quality: Water quality suitable for adequate growth and reproductive development.

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Puget Sound / Georgia Basin Rockfish: • Yelloweye *Sebastes ruberrimus* • Boccacio *Sebastes paucispinis*	79 FR 68041 11/13/2014	Adult bocaccio and adult/juvenile yelloweye rockfish. Benthic habitats and sites deeper than 30 m (98 ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat. Site attributes: • Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities, • water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities, and • the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance. Juvenile boccacio. Juvenile settlement habitats located in the nearshore with substrates such as sand, rock and/or cobble compositions that support kelp. Site attributes: • Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities; and • water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities.
West Coast Salmon: Chum Salmon (*O. keta)* • Columbia River ESU • Hood Canal summer-run ESU Sockeye Salmon (*O. nerka)* • Ozette Lake ESU • Snake River ESU Chinook Salmon (*O. tshawytscha)* • Lower Columbia River ESU • Puget Sound ESU • Upper Columbia River spring-run ESU • Upper Willamette River ESU Steelhead Trout (*O. mykiss*) • Lower Columbia River DPS • Middle Columbia River DPS • Upper Columbia River DPS	70 FR 52629 09/02/2005 70 FR 52630 09/02/2005 58 FR 68543 12/28/1993	Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development; Freshwater rearing sites with: • Water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; • Water quality and forage supporting juvenile development; • Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting

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West Coast Salmon: Chum Salmon (*O. keta)* • Columbia River ESU • Hood Canal summer-run ESU Sockeye Salmon (*O. nerka)* • Ozette Lake ESU • Snake River ESU Chinook Salmon (*O. tshawytscha)* • Lower Columbia River ESU • Puget Sound ESU • Upper Columbia River spring-run ESU • Upper Willamette River ESU Steelhead Trout (*O. mykiss*) • Lower Columbia River DPS • Middle Columbia River DPS • Upper Columbia River DPS (continued)	70 FR 52629 09/02/2005 70 FR 52630 09/02/2005 58 FR 68543 12/28/1993	juvenile and adult mobility and survival; Estuarine areas free of obstruction and excessive predation with: • Water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh & saltwater; • Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; • Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. Nearshore marine areas free of obstruction and excessive predation with: • Water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and • Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Coho Salmon (*O. kisutch)* • Central California Coast ESU • Southern Oregon / Northern California Coast ESU	64 FR 24049 05/05/1999	Within the range of both ESUs, the species' life cycle can be separated into 5 essential habitat types: Juvenile summer and winter rearing areas; juvenile migration corridors; areas for growth and development to adulthood; adult migration corridors; and spawning areas. Essential features of coho critical habitat include adequate: (1) substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) water velocity, (6) cover/shelter, (7) food, (8) riparian vegetation, (9) space, and (10) safe passage conditions.

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Steelhead *Oncorhynchus mykiss* Puget Sound DPS Coho Salmon *Oncorhynchus kisutch* Lower Columbia River ESU	81 FR 9251 03/25/2016	• Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development. • Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. • Freshwater migration corridors free of obstruction with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. • Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. • Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. • Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Steelhead (*O. mykiss)* • Northern California DPS • California Central Valley DPS • Central California Coast DPS • South-Central California Coast DPS • Southern California DPS	70 FR 52629 9/2/2005	• Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development. • Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Steelhead (*O. mykiss)* • Northern California DPS • California Central Valley DPS • Central California Coast DPS • South-Central California Coast DPS • Southern California DPS	70 FR 52629 9/2/2005	beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. • Freshwater migration corridors free of obstruction with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. • Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. • Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. • Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Coho Salmon *Oncorhynchus kisutch* Oregon Coast ESU	73 FR 7816 2/11/2008	• Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation, and larval development. • Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. • Freshwater migration corridors free of obstruction with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Coho Salmon *Oncorhynchus kisutch* Oregon Coast ESU	73 FR 7816 2/11/2008	• Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. • Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. • Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Chinook Salmon (*O. tshawytscha)* • Snake River fall-run ESU • Snake River spring/summer-run ESU	58 FR 68543 12/28/1993 64 FR 57399 10/25/1999	Juvenile rearing areas include adequate: • Spawning gravel • Water quality • Water quantity • Water temperature • Cover/shelter • Food • Riparian vegetation • Space Juvenile and adult migration corridors are the same as for Snake River sockeye salmon. Critical habitat for chinook salmon Snake river spring/summer-run ESU is designated as all areas currently accessible to the species within the range of the ESU.
Sockeye Salmon *Oncorhynchus nerka* Snake River ESU	58 FR 68543 12/28/1993	Spawning and juvenile rearing areas: • Spawning gravel • Water quality and Water quantity • Water temperature • Food • Riparian vegetation • Access Juvenile migration corridors: • Substrate • Water quality

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Sockeye Salmon *Oncorhynchus nerka* Snake River ESU (continued)	58 FR 68543 12/28/1993	• Water quantity • Water temperature • Water velocity • Cover/shelter • Food • Riparian vegetation • Space • Safe passage conditions Adult Migration corridor has the same Essential Features, excluding "food".
Nassau Grouper *Epinephelus striatus*	89 FR 126 01/02/2024	Recruitment and developmental habitat. Areas from nearshore to offshore necessary for recruitment, development, and growth of Nassau grouper containing a variety of benthic types that provide cover from predators and habitat for prey, consisting of the following: • Nearshore shallow subtidal marine nursery areas with substrate that consists of unconsolidated calcareous medium to very coarse sediments (not fine sand) and shell and coral fragments and may also include cobble, boulders, whole corals and shells, or rubble mounds, to support larval settlement and provide shelter from predators during growth and habitat for prey. • Intermediate hardbottom and seagrass areas in close proximity to the nearshore shallow subtidal marine nursery areas that provide refuge and prey resources for juvenile fish. The areas include seagrass interspersed with areas of rubble, boulders, shell fragments, or other forms of cover; inshore patch and fore reefs that provide crevices and holes; or substrates interspersed with scattered sponges, octocorals, rock and macroalgal patches, or stony corals. • Offshore Linear and Patch Reefs in close proximity to intermediate hardbottom and seagrass areas that contain multiple benthic types, for example, coral reef, colonized hardbottom, sponge habitat, coral rubble, rocky outcrops, or ledges, to provide shelter from predation during maturation and habitat for prey. • Structures between the subtidal nearshore area and the intermediate hardbottom and seagrass area and the offshore reef area including overhangs, crevices, depressions, blowout ledges, holes, and other types of formations of varying sizes and complexity to support juveniles and adults as movement corridors that include

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Nassau Grouper *Epinephelus striatus* (continued)	89 FR 126 01/02/2024	temporary refuge that reduces predation risk as Nassau grouper move from nearshore to offshore habitats. Spawning Habitat. Marine sites used for spawning and adjacent waters that support movement and staging associated with spawning.
Beluga *Delphinapterus leucas* Cook Inlet DPS	76 FR 20179 04/11/2011	• Intertidal and subtidal waters of Cook Inlet with depths less than 30 ft (MLLW)(9.1 m) and within 5 miles (8 km) of high and medium flow anadromous fish streams. • Primary prey species consisting of 4 species of Pacific salmon (Chinook, sockeye, chum, and coho), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole. • Waters free of toxins or other agents of a type and amount harmful to Cook Inlet beluga whales. • Unrestricted passage within or between the critical habitat areas. • Waters with in-water noise below levels resulting in the abandonment of critical habitat areas by Cook Inlet beluga whales.
Northern Right Whale *Eubalaena glacialis*	81 FR 4837 01/27/2016	Gulf of Maine – Georges Bank; *Unit 1.* The physical and biological features essential to the conservation of the North Atlantic right whale, which provide foraging area functions in Unit 1 are: The physical oceanographic conditions and structures of the Gulf of Maine and Georges Bank region that combine to distribute and aggregate *C. finmarchicus* for right whale foraging, namely prevailing currents and circulation patterns, bathymetric features (basins, banks, and channels), oceanic fronts, density gradients, and temperature regimes; low flow velocities in Jordan, Wilkinson, and Georges Basins that allow diapausing *C. finmarchicus* to aggregate passively below the convective layer so that the copepods are retained in the basins; late stage *C. finmarchicus* in dense aggregations in the Gulf of Maine and Georges Bank region; and diapausing *C. finmarchicus* in aggregations in the Gulf of Maine and Georges Bank region. Southeast U.S. Coast; *Unit 2.* The physical features essential to the conservation of the North Atlantic right whale, which provide calving area functions in Unit 2, are: • Sea surface conditions associated with Force 4 or less on the Beaufort Scale,

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Northern Right Whale *Eubalaena glacialis* (continued)	81 FR 4837 01/27/2016	• Sea surface temperatures of 7 °C to 17 °C, and • Water depths of 6 to 28 m, where these features simultaneously co-occur over contiguous areas of at least 231 nmi2 of ocean waters during the months of November through April. When these features are available, they are selected by right whale cows and calves in dynamic combinations that are suitable for calving, nursing, and rearing, and which vary, within the ranges specified, depending on factors such as weather and age of the calves.
North Pacific Right Whale *Eubalaena japonica*	90-Day Finding 87 FR 41271 07/12/2022 73 FR 19000 05/08/2008	Prey species of large zooplankton in areas where right whale are known or believed to feed. In particular, these are the copepods *Calanus marshallae*, *Neocalanus cristatus*, and *N. plumchrus*, and a euphausiid, *Thysanoessa raschii*, whose very large size, high lipid content, and occurrence in the region likely makes it a preferred prey item for right whales. Areas of concentration where right whales feed are characterized by certain physical and biological features which include nutrients, physical oceanographic processes, species of zooplankton described above, and long photoperiod due to the high latitude.
Humpback whale *Megaptera novaeangliae* Central America DPS	86 FR 21082 04/21/2021	Prey species, primarily euphausiids (Thysanoessa, Euphausia, Nyctiphanes, and Nematoscelis) and small pelagic schooling fishes, such as Pacific sardine (*Sardinops sagax*), northern anchovy (*Engraulis mordax*), and Pacific herring (*Clupea pallasii*), of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth.
Humpback whale *Megaptera novaeangliae* Western North Pacific DPS	86 FR 21082 04/21/2021	Prey species, primarily euphausiids (Thysanoessa and Euphausia) and small pelagic schooling fishes, such as Pacific herring, capelin (*Mallotus villosus*), juvenile walleye pollock (*Gadus chalcogrammus*) and Pacific sand lance (*Ammodytes personatus*) of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth.
Humpback whale *Megaptera novaeangliae* Mexico DPS	86 FR 21082 04/21/2021	Prey species, primarily euphausiids (Thysanoessa, Euphausia, Nyctiphanes, and Nematoscelis) and small pelagic schooling fishes, such as Pacific sardine, northern anchovy, Pacific herring, capelin, juvenile walleye pollock, and Pacific sand lance of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth.

Species	FR Notice Date	Physical or Biological Features Essential for the Conservation of the Species
Ringed seal *Phoca (pusa) hispida* Arctic subspecies	87 FR 19232 04/01/2022	• Sea ice habitat suitable for the formation and maintenance of subnivean birth lairs used for sheltering pups during whelping and nursing, which is defined as seasonal landfast (shorefast) ice, or dense, stable pack ice, that has undergone deformation and contains snowdrifts at least 54 cm deep. • Sea ice habitat suitable as a platform for basking and molting, which is defined as sea ice of 15% or more concentration. • Primary prey resources to support Arctic ringed seals, which are defined to be small, often schooling, fishes, in particular, Arctic cod, saffron cod, and rainbow smelt; and small crustaceans, in particular, shrimps and amphipods.
Pacific bearded seal *Erignathus barbatus nauticus* Beringia DPS	87 FR 19180 04/01/2022	• Sea ice habitat suitable for whelping and nursing, which is defined as areas with waters 200 m or less in depth containing pack ice of at least 25% concentration and providing bearded seals access to those waters from the ice. • Sea ice habitat suitable as a platform for molting, which is defined as areas with waters 200 m or less in depth containing pack ice of at least 15% concentration and providing bearded seals access to those waters from the ice. • Primary prey resources to support bearded seals: Waters 200 m or less in depth containing benthic organisms, including epifaunal and infaunal invertebrates, and demersal fishes.

APPENDIX C: RISK-PLOT GENERATION

To provide an aid in the Risk Characterization section, NMFS developed a plot (referred to as a 'Risk-plot') consolidating the various sources of data (i.e., exposure, response, and use) available as part of the consultation (e.g., EPA's BEs and risk assessments and NMFS's analyses) into a single visualization.

The Risk-plots are generated using the R programming language: R Core Team (2017). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/. While the R code used by NMFS is not included in this opinion, all of the data used to generate a Risk-plot is provided in Attachment 2 of this document.

This Appendix consists of several sections with information on the Risk-plot process:

- **Risk-plot Process Overview:** An overview of the Risk-plot process
- **Example Risk-Plot:** An example of a Risk-plot
- **Directories and Files:** A list of the directories and file used by the Risk-plot process and provided in Attachment 2 of this document

Risk-plot Process Overview

The following is a brief overview of the Risk-plot process. The data displayed on the Risk-plots comes from several sources. For the Risk-plot process, these sources appear as various files that the R code relies on for generating the plots. A list of the files is provided in the **Directory and Files** section. A summary of the sources is detailed here:

1) Toxicity information for a species gathered from the available literature. For sublethal endpoints, such as growth, this is typically a range of LOECs or EC25s across the available studies. For endpoints such as mortality, this is can be a range of percent mortalities using an LC50 and slope chosen based on a species sensitivity distribution. For the Risk-plot process this information is provided by a tab-delimited text file selected by the user (see the Toxicity Tables in Attachment 2 of this document).
2) Data on the species range, critical habitat, and areas of concern (e.g., a list of HUC-12s), the uses of the pesticide (e.g., Vegetable and Ground Fruit), and their overlap (by HUC-12). For the Risk-plot process, this information is organized into various files. The relevant HUC-12s for a Risk-plot (e.g., species range, habitat, or area) are specified in Species_HUC_d.xlsx (see Attachment 2 of this document). The HUC12_Overlap_Data.csv in Attachment 2 provides a list of uses and their overlaps for each of the HUC-12s.
3) Exposure estimates generated using existing exposure models for each crop and use category (e.g., annual EECs for lettuce crop within the Vegetable and Ground Fruit use category). For the runoff, index reservoir, and farm pond EECs the Pesticide Water Calculator (PWC) was used to generate thirty years of EECs for each HUC-2 (see Attachment 2 of this document for the PWC batch files). For the drift EECs, NMFS generated EECs using EPA provided equations based on AgDRIFT deposition curves, the Tier 1 Rice Model, and aquatic degradation (see Attachment 2 of this document for the

spreadsheets). For the Risk-plot process, the EECs are combined in the Carbamate_eecs_111622.csv provided in Attachment 2 of this document.

The process of generating a Risk-plot typically starts with selecting a chemical and species. The selection of species determines which HUC-12s are extracted from the data files. The selection of chemical determines the relevant EECs and uses for the list of HUC-12s. The list of HUC-12s determines which HUC-2s are needed from the EEC data.

The plot displays a single EEC range for a specific crop and use category (e.g., lettuce as a Vegetable and Ground Fruit use) in a HUC2. If the list of HUC-12s spans multiple HUC-2s, 1 point will be displayed for each HUC2. Differences between HUC2s in precipitation, soil, etc. will lead to slight differences in the EECs for the same crop and use.

The R code compiles these sources of information into a single plot. For each Risk-plot, an associated table with the same information is also created and provided in Attachment 2 of this document. An example of a Risk-plot is shown in Figure 112. The plot consists of five parts.

1) The upper portion displays the information present in the selected toxicity data file (i.e., file in the Toxicity Tables folder of Attachment 2 of this document). This consists of multiple rows of endpoints each with a set of labeled markers. The meaning of each marker is up to the user (e.g., a LOEC, percent morality, etc.). The markers are positioned along the concentration axis below.
2) The center of the plot displays all the EEC data associated with the selected chemical, HUCs and relevant scenarios (i.e., from the pesticide's EEC file in the EEC Files folder of Attachment 2 of this document). For each crop (e.g., lettuce) of a use category (e.g., Vegetables and Ground Fruit) there will be a point for each averaging period (different rows for 1-day, 1-day, 21-day, and 60-day time weighted averages) and each EEC type (different symbols for drift, runoff, index reservoir, and farm pond). For runoff, index reservoir, and farm pond EECs, each point represents the median peak annual EEC for 1 averaging period for a specific PWC scenario. Error bars around the point indicate the 5% and 95 percentile of the distribution of thirty years of data. For the drift EECs, each point represents the concentration after accounting for aquatic degradation and soil sorption. Error bars show the range of increased concentrations without those processes. The EEC data is positioned using the same concentration axis as the toxicity data to allow direct comparison of exposure and effects.
3) The left side of the plot (i.e., the left Y-axis labels) list the use categories associated with the species range (or list of HUC12s) in order of their area (largest area at the top). The total area the HUC12s is denoted at the bottom.
4) The right side of the plot (i.e., the right Y-axis labels) shows the acres within the HUC12s associated with the use category and the percent of the total area represented by each use.
5) The bottom of the plot has 4 lines of text that identify the specific information presented in the Risk-plot. The first line shows the chemical (i.e., which EEC file was used) and toxicity data file (i.e., which Toxicity Table file was used). The second line shows which averaging periods were plotted. The third line lists the HUC-2s and the EECs being displayed ('r' for runoff, 'd' for drift, 'ir' for index reservoir, and 'fp' for farm pond). The 4th line shows the species and info on the source of HUC-12s (e.g., range, habitat, or list)

and the number of HUC-12s. 'Range', 'Habitat', and 'List' denotes which set of HUC-12s was used for the plot (i.e., the worksheet and column in the Species_HUC_d.xlsx spreadsheet).

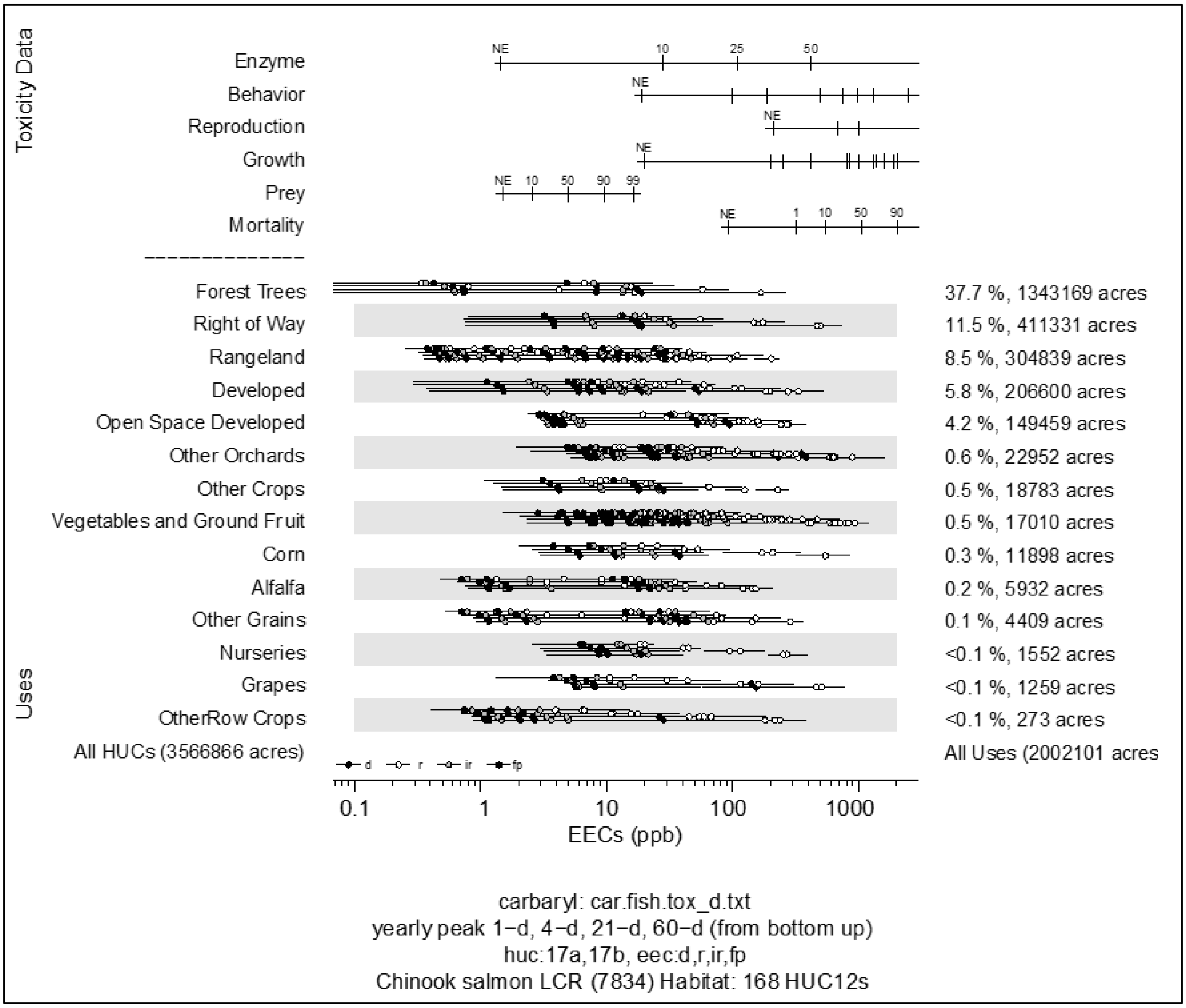


Figure 112. Example of an aquatic Risk-plot.

Directory and Files

List directories and files associated with generating a Risk-plot. These files are provided in Attachment 2 of this document.

Folder	**Comments**
UDL Overlaps	CSV files with overlap data by UDL and HUC12
HUC12_Overlap_Data.csv	
HUC12 Info	Files with HUC12 lists used in the Risk-plots
Species_HUC_d.xlsx	
huc_convert.csv	File specifying the HUC2 EEC data associated with a HUC12
EEC Files	EEC data used to generate a Risk-plot
Carbamate_eecs_111622.csv	
Toxicity Tables	Tab-delimited files with taxa-specific toxicity data
Species Info	Files for converting species to EntityIDs used in other tables
Risk-plot Tables	Folders with CSV Risk-plot tables (one per Risk-Plot)

APPENDIX D: CONSIDERATION OF USAGE INFORMATION

Introduction

It is the policy of NMFS to evaluate all scientific and other information used in opinions to ensure that it is reliable, credible, and represents the best scientific and commercial data available. In addition to usage information, NMFS must consider all relevant factors relating to the nature of all the available data and the inferences it can support to evaluate effects to ESA-listed species and their designated or proposed critical habitats over the 15 year duration of the action. Based on the data quality considerations (e.g., NOAA's Information Quality Guidelines) and the standards of the Administrative Procedure Act (APA), NMFS considers the uncertainty and reliability of all the available information.

Sources of information, and uncertainties associated with them, are discussed elsewhere in the opinion. For example, NMFS's assessment of the toxicity information (e.g., LC50 data) included evaluating the extent and quality of available toxicity studies and their ability to serve as surrogate data for ESA-listed species. Assessing exposure information (e.g., EECs) included evaluating the available models (e.g., PWC), their inputs, and how well they modeled specific habitats. The degree to which the opinion relied on specific information depended on NMFS's assessment of its uncertainty and reliability. A summary of NMFS's assessment of the available usage information is presented below.

The term "use" describes the authorized parameters (e.g., application rate, frequency, crop type, etc.) of pesticide application as described on the FIFRA label. EPA authorizes the FIFRA label that describes when, where, and how pesticide products can legally be applied. Therefore, the label defines the Federal action and is the subject of the analysis in the "Effects of the Action" portion of this opinion. "Usage" describes parameters (e.g., rate, frequency, percent treated) related to the ways in which a particular pesticide is applied. In short, use describes how pesticides are authorized to be applied, whereas usage describes how pesticides have been applied in the past or predictions of how they might be applied in the future. Usage can change over time. When we assess whether EPA is able to insure that their registration is not likely to jeopardize species or destroy or adversely modify critical habitat, we consider the effects of the registration action over the duration of the action (15-year registration review period).

The primary source of information we base our assessment of the effects of the action on is the FIFRA label. We consider the potential applications that the label allows, and whether or not the labels contain directions that are sufficient to insure that species will not be jeopardized by the allowed uses over the duration of the action. Other sources of information that inform our effects analysis take into consideration questions about future usage including regulatory actions, monitoring data and available reports on past usage (e.g., California DPR Pesticide Use Reporting and market research estimates by Kynetec).

NMFS evaluated many sources of information regarding usage (e.g., use reports, survey based estimates, regulatory bans/restrictions, and pesticide detections in water quality monitoring) in the development of this opinion. NMFS evaluated the data sources using the following 4 criteria: Transparency; Accuracy; Spatial precision; and Predictive/representative of future usage.

We applied these criteria taking into account the specific context of NMFS species and their aquatic habitats and life histories. For example, spatial precision is important because risk is driven by the proximity of pesticide application to individuals of the species. Below we briefly describe sources of usage information considered, provide an assessment based on the criteria we use to evaluate them, and describe the application of usage information in the opinion.

Sources of usage data In assessing whether EPA is able to insure that their action is not likely to jeopardize a species or destroy or adversely modify critical habitats, NMFS considered a number of sources of information potentially relevant to past or future usage. Many sources were summarized in EPA's Revised National and State Use and Usage Summary Reports (SUUMs; Attachment 1 of this document). Overall, the available information fell into several forms discussed briefly below that inform their limitations in understanding future usage.

The available FIFRA labels define where, how, and when pesticides are authorized to be applied, so the labels must inform assessment of future pesticide usage. Use Data Layers discussed in the opinion (e.g., CDL and other non-agricultural GIS information) define where use sites are located and inform where future usage could occur within a species range. Detections of a pesticide in water quality monitoring data provides evidence of past usage that can inform predictions or assumptions about future usage. State and national regulatory pesticide bans can also serve to predictably limit future usage, as would regulatory revocations of food tolerances for a pesticide.

Other sources considered by NMFS provide information on past usage, with EPA's SUUM reports being the primary summary of several sources. Broadly, the information on past usage consists of 3 categories:

- Sales of pesticide products – NMFS did not receive this type for the carbamate consultation.
- Survey estimates of pesticide usage - NMFS received estimates of pesticide usage that were based on surveys of end-user applications at the state level for some pesticide uses (e.g., Kynetec data provided in EPA SUUM reports). The extent and methods of the surveys informs the likelihood that estimates of past usage are accurate, and influences the reliability that they will be predictive of future pesticide usage. Important questions include; how many uses were surveyed and what percent of total growers responded to surveys.
- Required reporting of pesticide usage - NMFS reviewed several sources of pesticide usage reports that are required for certain uses and locations. Perhaps the best example of this is the California Department of Pesticide Regulations Pesticide Use Reporting (PUR). The PUR data consists of publicly available reporting by most commercial applicators in California of each use of a pesticide every year. Unfortunately, comparable data is not available elsewhere.

Analyses of usage data

NMFS considered available usage data using the 4 criteria mentioned above in the context of NMFS species and their habitats and life histories. For each criterion some analyses and conclusions are presented below.

Transparency: how transparent are the usage data?

NOAA's Information Quality Guidelines and the standards of the APA direct us to assess the utility of the information we rely on and to be transparent about the uncertainties and limitations. Almost all of the usage data are proprietary estimates of usage. Where the data and methods are classified as proprietary, how these estimates are derived is not fully transparent. To the extent possible, we conduct rigorous robustness checks when we rely on confidential information. This entails considering the precision of the estimates and characterization of the robustness for scientific use in ecological risk assessment. Without further information on the sources of usage data (e.g., the margin of error associated with specific usage estimates), such characterization of the usage data is not currently possible.

The majority of the usage data provided for states outside of California are from proprietary sources. An example for agricultural uses is Kynetec. According to materials provided by the company, Kynetec data is "designed to address market questions asked most often by senior executives, and those involved in product development, sales, and marketing." Surveys are designed to reach a particular percentage of the total crop grown at a national level, though statistics are reported at the state and Crop Reporting District (CRD) level when sample size is adequate. The data provided is lacking the statistical foundation to understand the robustness at the state level or any geographic specificity at the sub-state level. NMFS has not received any detailed information (e.g., how many applicators responded to the survey, how many acres are represented by the survey at the state level), nor any standards used to determine an adequate sample size at these levels, nor the minimum threshold required for reporting these values. Our understanding is that this varied on a case-by-case basis, according to the surveyor, crop, and state. Usage estimates for non-agricultural applications were based on sales information (manufacturer and retail) and end-user surveys, though neither sources nor methodologies were identified for individual estimates.

Accuracy: how accurate are the usage data?

NMFS recognizes that the usage estimates may be sufficiently accurate for use in other contexts (e.g., marketing decisions). However, the available sources of usage data were not designed for the purpose of assessing risk to threatened and endangered species. To determine if they are robust enough to characterize exposure in the context of a section 7 ESA consultation, NMFS must consider the accuracy and completeness of the available usage estimates. In doing so, NMFS identified several concerns that include:

Few states require reporting of actual pesticide usage (with California as a notable exception). National-level data are estimates based on surveys which: do not cover many non-agricultural uses; do not cover all authorized crop uses; and do not cover all states. Extrapolating to fill these data gaps would introduce substantial uncertainties.

Usage data are at geographic scales that limit their utility for our biological opinions. In assessing risks to NMFS species, the primary concern is a pesticide application in direct proximity to species habitat; the existing data and methods do not presently allow for an assessment at this scale.

Comparisons of available usage data to other sources (e.g., water quality monitoring) suggests that usage estimates can fail to detect actual past usage.

As an example of the last point, prometryn was detected in water samples from rivers and streams in Oregon and Western Washington but no use was reported in those regions (Figure 113). Prometryn has no non-agricultural uses and is only registered for use on several vegetables known to be grown in Oregon and Washington (e.g., carrots). Similar detections exist in other states (e.g., Nebraska) where registered uses are presumably also present.

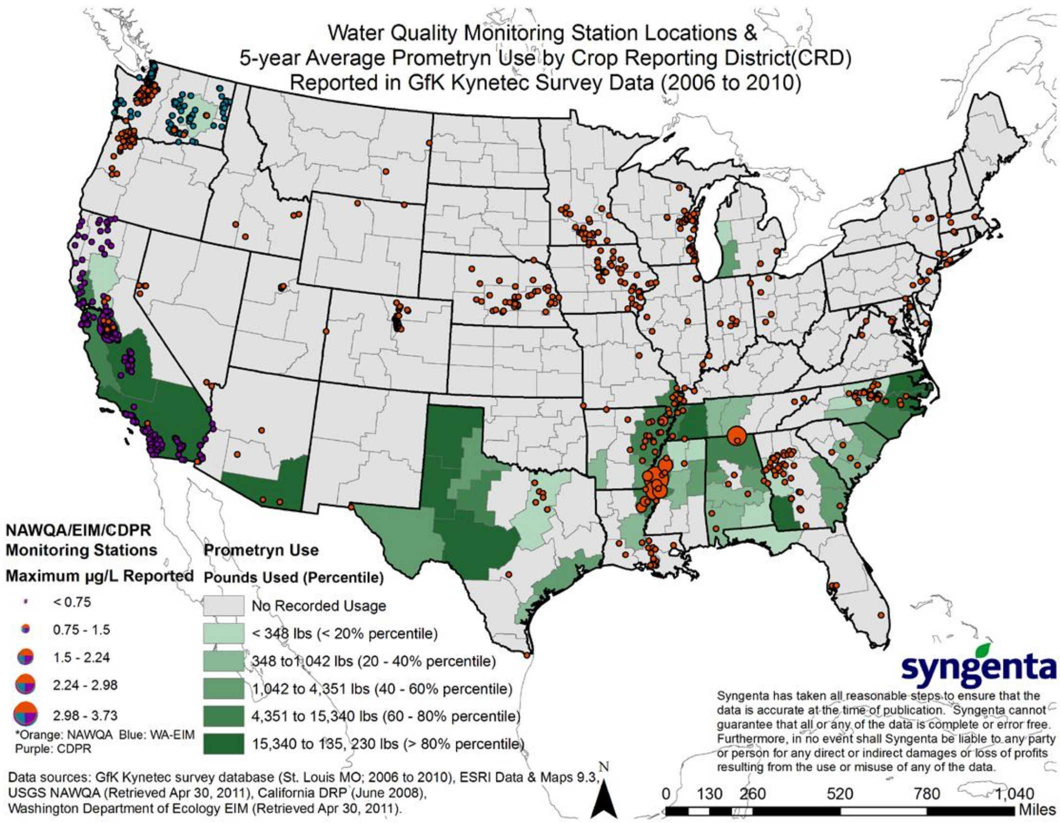


Figure 113. From Figure 15 of Prometryn Case Study, Syngenta Crop Protection, LLC. Minor Crop Farmer Alliance (MCFA) Endangered Species Assessment Workshop Denver, CO May 24 – 25, 2011. https://ccqc.org/wp-content/uploads/2011/07/Prometryncasestudy.pdf.

With respect to survey-based usage data such as the Kynetec data, reports of actual use of pesticides in California reveal that the survey estimates sometimes fail to detect actual use. For example, in the case of cotton, EPA's summary of Kynetec estimates indicated that malathion use was surveyed but no usage was reported. However, on average, actual use reported for the state averaged over 22,000 pounds annually, with annual applications averaging more than 10,000 acres.

Table 156. Comparisons of the Kynetec survey-based data for CA (EPA 2018 Malathion Report) to CA DPR required use reporting shows that the surveys sometimes fail to detect use that may be small but still consequential. For 2011-2015,

the majority of malathion uses for which surveys reported no use had actual use reported to CA DPR.

Kynetec AMRD		CA DPR	min lbs	max lbs	avg lbs
Beans (succulent)	surveyed but no usage reported	BEAN, SUCCULENT	3.2	718.2	175.1
Cotton	surveyed but no usage reported	COTTON	707.1	54826.5	22410.7
Cucumbers	surveyed but no usage reported	CUCUMBER	37.1	133.6	93.1
Peaches	surveyed but no usage reported	PEACH	0.0	72.0	14.7
Pears	surveyed but no usage reported	PEAR	0.0	45.1	10.2
Potatoes	surveyed but no usage reported	POTATO	0.0	774.7	352.7
Rice	surveyed but no usage reported	RICE	0.0	3384.1	720.4
Watermelons	surveyed but no usage reported	WATERMELON	0.0	473.0	144.1
Lima Beans	surveyed but no usage reported	Not Found	--	--	--
Corn, Field	surveyed but no usage reported	CORN (FORAGE - FODDER)	362.2	1122.0	602.2

One approach taken to address the uncertainty associated with uses where no usage was reported or no surveys were conducted is to assume a non-zero area treated (e.g., a 2.5% acres treated). NMFS considered the accuracy associated with all usage estimates. For example, how accurate is any given estimate of percent acres treated for a specific crop, state, and year? If an estimate of 0% treated could be off by as much as 2.5%, could an estimate of 3% actually represent 5.5%? Due to the lack of transparency discussed above, for most usage data NMFS lacks the information with which to assess the accuracy of a usage estimate, e.g., the magnitude of the uncertainties. The degree to which these estimates can be relied on for conclusions that are conservative and protective for a species depends on understanding the magnitude of these uncertainties.

Spatial precision: how spatially precise are the usage data?
Specifically, are the existing usage data available at a geographic scale that is useful for ESA assessments? In assessing risks to NMFS's listed species and the designated or proposed critical habitats of those species, the primary concern is a pesticide application in direct proximity to aquatic species habitat (i.e., within 300 m of ESA-listed species aquatic habitats). However, the existing usage data and methods do not currently allow for an assessment at this scale. This is because most usage estimates are only available at the state-level (with the California (PUR) data a notable exception). For some non-agricultural applications, the information was reported at a national scale. While informative, state-based usage estimates do not allow us to evaluate whether or not pesticide applications are likely to occur within direct proximity to species habitat, an event which occurs at the scale of an individual operation or application. The usage data for a crop may provide information on the percent of a crop's area that has been treated in the past, but it does not provide information on specifically where application occurred. Many of NMFS's species occupy large areas with individuals moving over the entire area during their life history. Even small areas of pesticide application may have biological significance to a large portion of the population (e.g., salmon juveniles rearing and out-migrating past application sites). Because of this, NMFS does not equate a low percent of the species range treated (whether by use overlap or use plus usage) with a low percent of the individuals exposed. The degree to which a small area of application impacts a population will depend on precisely where that application occurs within the range of the species. The usage data does not provide the spatial precision to inform whether specific small areas of a species range will be treated in the future.

Predictive/representative of future usage: how well are the usage data able to represent use over the next 15 years?
Pesticide labels tell us exactly how these compounds are authorized to be used. Data on how pesticides were used in the past do not necessarily provide a complete picture of future usage unless we assume that future usage will follow known past usage. However, other evidence indicates that usage is often highly variable over time. Variables that influence usage include:

- Changing pest pressures
- Emergence of new pests
- Development of pest resistance
- Regulatory changes to products
- Market changes

To evaluate the degree to which pesticide use can change over a longer period of time, NMFS evaluated California PUR data for 1990 to 2017 (https://www.cdpr.ca.gov/docs/pur/purmain.htm). Whereas most usage data sets are merely estimates based on surveys, California PUR data represents the most comprehensive, locally scaled, reliable usage information available, as the state requires reporting for all agricultural applications, and all applications by certified applicators, in California. Therefore, there is greater certainty that these data reflect actual usage across a wide range of application sites. This analysis is not meant to be a risk assessment for any specific pesticide or application site. Every pesticide and site will present a unique situation of the various factors driving changes in usage. Assessing trends for every specific pesticide or site would be highly complex and introduce numerous other uncertainties (e.g., potentially pesticide-specific changes in pest pressures). Similarly, the selection of criteria below (e.g., a 2-fold increase) is not meant to represent a level of change that represents an unacceptable risk to ESA-listed species. Rather, the analysis below is meant to capture and describe trends in usage across a broad range of pesticides and sites.

Custom code was used to import the data and perform the analyses in R (https://www.r-project.org). EPA's SUUM reports summarize pesticide usage on the 5 most recent years of data available. We implemented methods to select pesticides that were used over a 20-year duration to evaluate how usage changes over time and to evaluate how well 5-years in usage data may predict 15-years of future usage (the duration of the action). The pesticides were selected based primarily on their frequency of use in either 1998 or 2017 (for 2017, see https://www.cdpr.ca.gov/docs/pur/pur17rep/top_100_ais_acres_2017.htm). Additional neurotoxic pesticides were included (the organophosphates, carbamates, neonicotinoids, and pyrethroids) if they weren't already on the list. We excluded adjuvants and oils from the list because these ingredients are generally not considered a.i.s. All sites for which there was at least 1 use of any of the pesticides were included. In all, NMFS selected 153 pesticides and 248 use sites to examine (see Attachment 2 of this document).

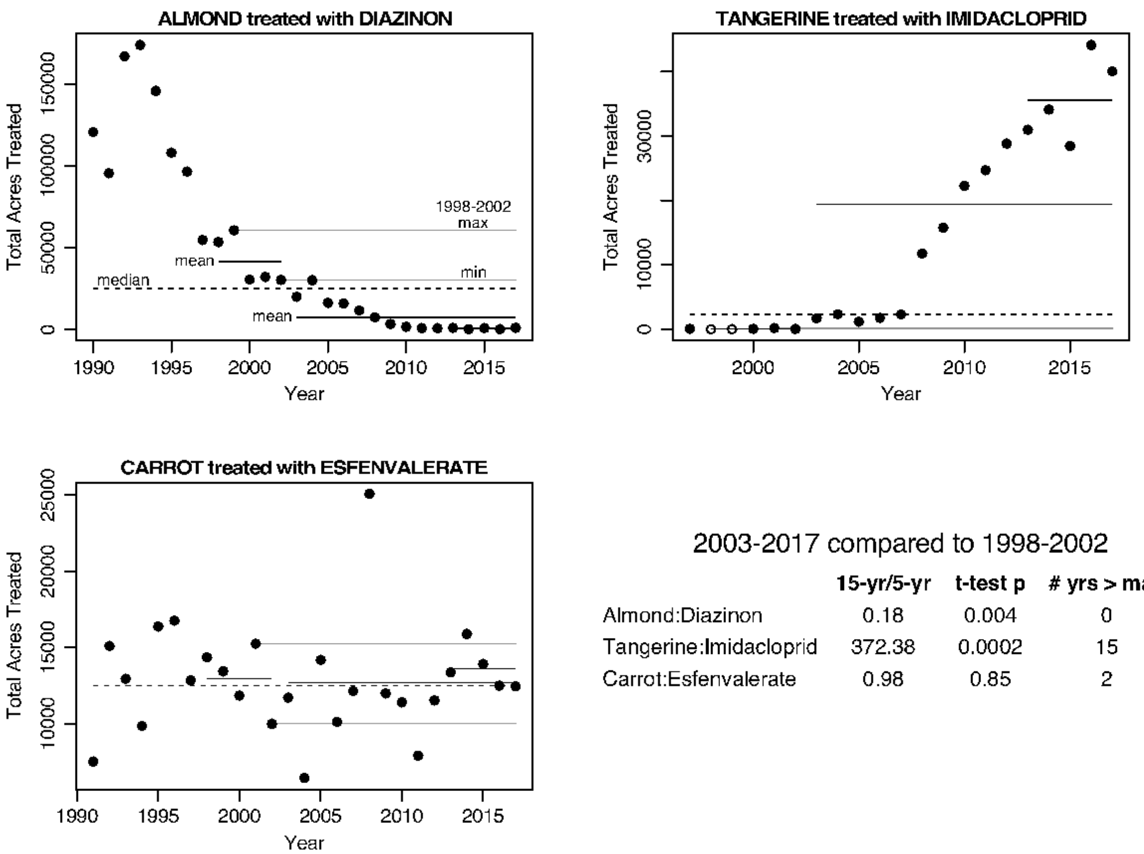
We assessed all possible combinations of a pesticide and a site, focusing on those with at least 20 years of data. A site:pesticide combination (e.g., use on almonds of diazinon) needed to have a recorded use 1) on or before 1998 to make sure the pesticide was in use at the start of the 20-year period and 2) used at least once in the last 5 years (2013-2017) to make sure that the pesticide was still in use at the end of the 20-year period. The procedure netted a large number of samples for evaluation; a total of 3,269 site:pesticide combinations met these criteria. Preliminary analyses of these combinations are shown below. This analysis is only for California, due to the source of information. However, NMFS assumes that the various factors influencing usage will lead to similar changes in usage over time in other states. California, like many of the coastal states where NMFS species occur, cultivates a large number of "minor crops." Use on these crops will appear in the CA DPR PUR data, but estimates of usage on minor crops are frequently not available elsewhere.

NMFS chose to use total acres treated for this analysis of trends. Acres treated is a reliable value provided for every recorded use in the CalDPR PUR data. NMFS recognizes this is not the same metric as Percent Crop Treated (PCT) reported in the SUUM. This analysis is not meant to evaluate trends in PCT. Unlike PCT, total acres treated will additionally include repeat applications to the same field and is subject to change if total acres grown changes. Nonetheless, changes in total acres treated also are a measure of changes in usage and an appropriate, while different, indicator of changes in risk to ESA-listed species over time. For example, regardless of the reason for an increase in total acres treated, the change represents an increase in risk to ESA-listed species. An increase in the number of applications to the same field is an increase in risk.

Figure 114 shows annual data on total acres treated for 3 combinations of a single use site and a single pesticide. For example, the first graph shows the application of diazinon to almonds with each point representing the total acres of almonds in California treated by diazinon in a given year. For some years, no use may have been reported, but given the required nature of the

California PUR reporting NMFS represented that year's use with a zero (e.g., the open circles on the tangerine:imidacloprid graph). The dashed lines represent the median of all of the years of data. Solid black lines represent the mean of the data for the years spanned by the line (e.g., 1998-2002 and 2002-2017). Gray lines show the minimum and maximum use for the 1998-2002 period.

As seen, application of a pesticide to a site can vary quite dramatically over a 20-year period and in different ways. For example, in treatment of almonds with diazinon there was a marked decline in use, while for treatment of tangerines with imidacloprid there was a dramatic increase. Finally, treatment of carrots with the insecticide esfenvalerate was variable, but did not show an obvious shift up or down over the 20-year period.



2003-2017 compared to 1998-2002

	15-yr/5-yr	t-test p	# yrs > max
Almond:Diazinon	0.18	0.004	0
Tangerine:Imidacloprid	372.38	0.0002	15
Carrot:Esfenvalerate	0.98	0.85	2

Figure 114. Changes in usage over time.

To quantify these changes in use over the 20-year period for all 3,269 site:pesticide combinations, NMFS calculated 3 metrics (shown above for the 3 specific combinations). First, the mean of the recent 15-year period was compared to the previous 5-year period. Second, we performed a t-test to determine if the mean of the 15-year period was significantly different from the mean of the previous 5-year period. Finally, the number of years for which the 15-year period was above the maximum of the previous 5-year period was calculated. This would assess how well the maximum of a 5-year would protect from future increases in use.

Figure 115 compares the means for all 3,269 combinations analyzed. Using a criterion of a 2-fold change in mean we found that 29% of the combinations had an increase in mean use and 34% of the combinations had a decrease. Only 37% of the combinations showed a change in mean within 2-fold (up or down) of the 5-year mean. As mentioned above, the selection of 2-fold is just meant to provide a summary of the distribution in magnitudes, not a statement about risk. This summary does include ‘minor’ crops (e.g., with mean total acres treated <100 acres) that could be considered a source of bias. For these crops, a small absolute change in usage (e.g., from 10 to 20 acres) would be the same relative increase (2-fold increase) as a large absolute increase in a ‘major’ crop (e.g., from 3,000 to 6,000 acres). It is worth noting that changes in usage in the ‘minor’ crops could, nonetheless, collectively represent a consequential change in risk (e.g., they form a substantial contribution to the Vegetable and Ground Fruit UDL). NMFS did assess the extent of a ‘minor’ crop bias by performing an analysis omitting site:pesticide combinations with <100 mean total acres for both time periods (i.e., they were 'minor' crops during both periods). Of the 2,303 combinations for which at least 1 period had ≥100 mean total acres treated, 25% of those combinations showed a 2-fold increase in mean.

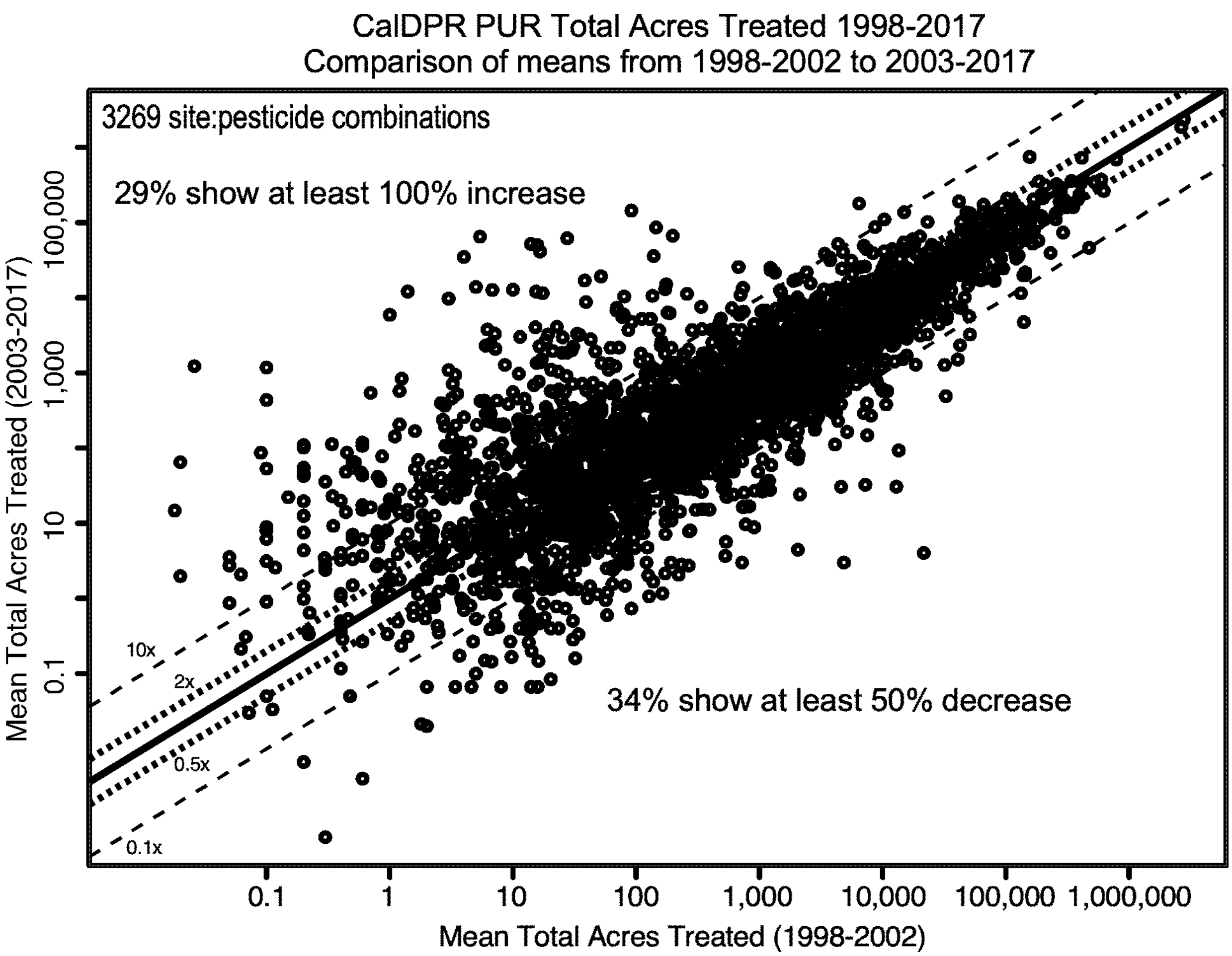


Figure 115. Mean of the first 5-year period versus the mean of the subsequent 15-year period for each site:pesticide combination. Figure 116 shows summary histograms of the results of the comparisons to the 5-year max (left) and the t-tests (right). Both metrics show that a substantial percentage of

the combinations of site and pesticide show a measurable change in use over a 20-year period. For 27% of the combinations at least 5 years of the 15-year period had more use than that of the maximum of the preceding 5-year period. For 36.5% of the combinations there was a significant difference in the mean of the 15-year period relative to the previous 5-year period.

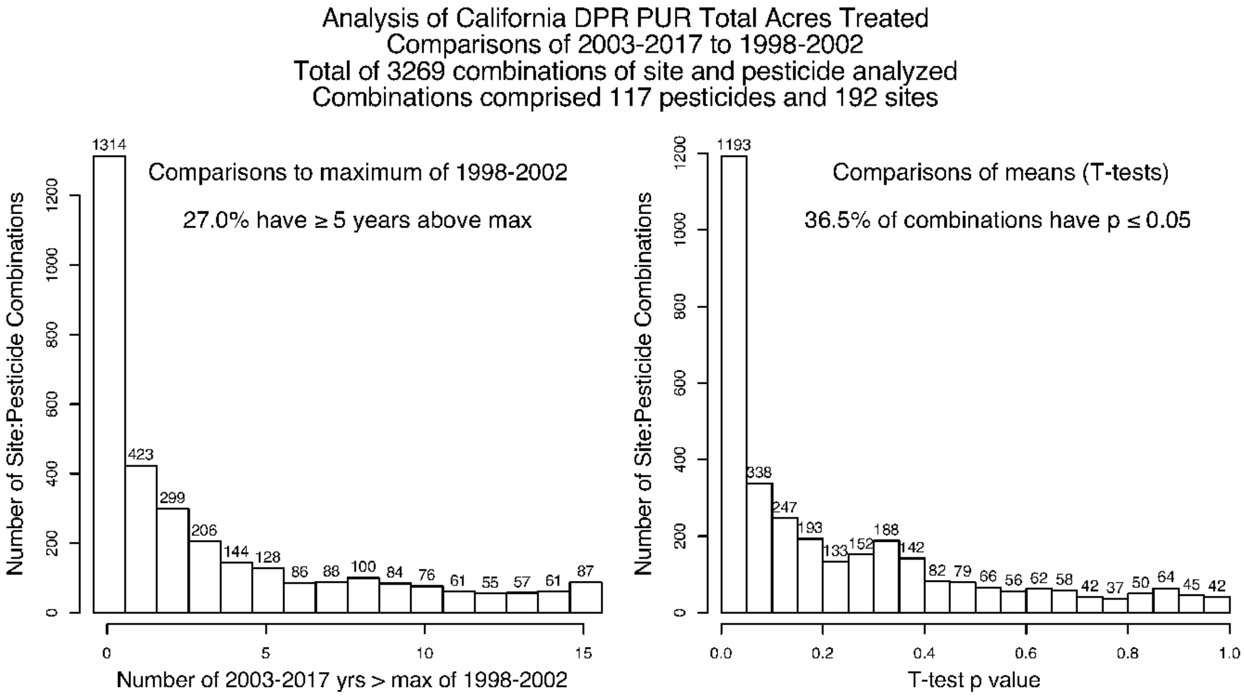


Figure 116. Histograms of the distributions of comparisons of the maximum of the 5-year period to subsequent 15-years (left) and of t-tests comparing the means of the preceding 5-year period to the subsequent 15-years (right). For example, every year of the 15-year period was greater than the maximum of the preceding 5-years for 87 of the site:pesticide combinations (left) and the t-test p value was ≤ 0.05 for 1193 of the combinations (right).NMFS's evaluation of over 3,200 combinations of pesticides and crop combinations in California indicate that usage patterns often undergo significant changes over time periods consistent with EPA's re-registration period (15-duration). If we assumed that the likelihood of exposure to ESA-listed species is primarily a function of the usage of pesticides within the species range and that usage will remain consistent with the most recent 5-year average over the 15 subsequent years of the action, then we would underestimate the actual usage of pesticide by more than 100%, approximately 29% of the time. Further, we found that using the maximum usage of the most recent 5 years would not be protective of approximately 27% of future pesticide use (see Figure 4). This analysis demonstrates that relying on the most recent 5 years of data does not accurately predict future pesticide use.

Approach to incorporating usage data

The analyses presented above represent specific examples of NMFS's consideration of available information to assess future usage of pesticides for the purposes of this opinion. Based on the uncertainties in the available information discussed above, we incorporated usage data into the opinion in 3 general areas: 1) the environmental baseline; 2) the effects analysis; and 3) the Reasonable and Prudent Alternative (RPA). Below is a brief description of each area.

Environmental Baseline

In the environmental baseline we assess, among other things, the presence of pesticides within species aquatic habitats. This includes an evaluation of pesticide environmental mixtures, which may increase risk to listed-species because of additive or synergist effects. The 3 primary sources of information assessed for evidence of past pesticide contamination are usage data, land use categories, and surface water monitoring data. EPA has provided NMFS with national and state use and usage summaries for carbaryl and methomyl. The usage information within these reports comes from both direct pesticide usage reporting (e.g., California PUR) as well as usage estimates from proprietary surveys (e.g., from Kynetec USA, Inc). Land use categories were evaluated within each species range by performing an overlap analysis with the National Land Cover Database information. The United States Geological Survey's National Water-Quality Assessment reports document trends between pesticide concentration and land use for both agricultural and urban applications. Monitoring data was accessed from national sources (e.g., EPA's Water Quality Portal, USGS National Water-Quality Assessment), as well as state programs (e.g., The Washington State Department of Agriculture – Natural Resources Assessment Section surface water monitoring).

Effects Analysis

In the effects analysis, we assess the anticipated impacts to species and habitat associated with the stressors of the action, over the next 15-year duration. The primary source of information on which to base assumptions of future usage is the FIFRA label. The registration of an active ingredient creates a potential for exposure by authorizing application at certain rates, times, and locations (i.e., labeled directions for use). When we assess whether EPA is able to insure that their registration is not likely to jeopardize species or destroy or adversely modify critical habitat, one thing considered is the potential usage that the label allows, and whether or not the labels contain directions that are sufficient to insure species will not be jeopardized or critical habitats will not be destroyed or adversely modified over the duration of the action. Substantial population-level impacts to species are not only possible via large-scale impacts across the entire species range, but could occur via smaller scale impacts to essential sub-populations, or essential life stages.

The potential for exposure is realized when applications are made directly to aquatic habitats where listed-species are present, or pesticides are transported to these habitats from applications which occur in proximity. Many pesticides are authorized for use directly adjacent to aquatic habitats, and some are authorized for direct application to aquatic habitats. These situations can be problematic because pesticides are inherently toxic and therefore, exposure may result in take. For a given chemical and species, the degree to which risk is anticipated is a function of the extent and frequency of these exposure scenarios over the entire range of the species.

The extent and frequency of future usage is driven largely by market forces and the collective choices of individual end-users. Temporal variation in the extent and frequency is driven by variables such as: changing pest pressures, emergence of new pests, development of pest resistance, regulatory changes to products, market changes, and the choices of individual end-users. We did examine information on past use ("usage") available to NMFS (e.g., survey information on agricultural uses provided by EPA and use reporting from California PUR). Given the degree of uncertainty and speculation associated with these factors, and usage information generally, we determined that, in most cases, we cannot rely on them to construct assumptions about the exposure potential and whether the response to exposure will result in fitness consequences to individuals of the listed species. We were, however, able to utilize usage information of some kinds to incorporate into our risk assessment a degree of confidence regarding whether future usage would be minimal.

The following describes our method for considering certain usage information as informing the confidence in our effects determinations. We consider and track the underlying uncertainties when evaluating the risk hypotheses by describing levels of confidence in our effect of exposure, likelihood of exposure, and in our overall risk determination for each risk hypothesis. When assessing the confidence we have in the likelihood of exposure characterization, 1 factor considered is the evidence we have that future usage of the a.i. to the use category being assessed will be minimal over the next 15 years. By minimal, we mean that the amount of acres treated or pounds applied are such that even if application is made in proximity to species habitat we would anticipate the exposure level would not result in an appreciable reduction in the reproduction, numbers or distribution of the species or in the value of designated or proposed critical habitat for that species. To make an assessment that usage will be minimal, we seek sufficient evidence, by which we mean that there is enough certainty in the evidence that we can rely on it as part of an analysis (along with the status of the species and critical habitat, the environmental baseline, and cumulative effects) that ultimately is required by the ESA to insure that the species is not likely to be jeopardized over the 15-year duration of the action.

Information used to assess whether future usage is likely to be minimal can include regulatory actions (e.g., state regulatory bans), monitoring data and available usage data, for example. In determining whether the evidence available was sufficient or not to find that future usage will be minimal, we considered criteria including those discussed above. In this way we were able to utilize usage information of some kinds to incorporate into our risk assessment a degree of confidence regarding whether future usage would be minimal. The confidence levels associated with all use categories were then used to inform our overall risk characterization.

Mitigations

In developing the conservation measures with EPA and the registrants, NMFS incorporated usage information by including an option for mitigation based on actual use rate and method applied (pesticide usage) rather than assuming application at the maximum-labeled rate (pesticide use). The process involves 3 steps which will be specified on EPA's Bulletin's Live website:

- Step 1. The end-user determines whether any mitigation is needed based on the location of the application. Is pesticide application to be made within 300 meters of ESA-listed species habitat? If yes, go to step 2.
- Step 2. The end-user determines the number of drift and runoff mitigation points needed for the pesticide application based on the application rate and method employed.
- Step 3. The end user chooses mitigation options to implement from a pick-list of options. Mitigation options can be added together, based on their point values.

APPENDIX E: SUMMARY OF ALL CARBARYL & METHOMYL USES

Introduction

The summary tables below contain selections of each use for both carbaryl and methomyl that are presumed to represent the highest risk by considering maximum application rates and application methods. See the description of the action chapter for more details regarding the action assessed in this consultation.

Summary of all carbaryl uses

The summary of all carbaryl uses presented below contains selections of each use that are presumed to represent the highest risk by considering maximum application rates and application methods. To summarize the current carbaryl action, NMFS integrated the mitigation agreed to as part of the PID, as well as the conservation measures discussed in the Description of Action. Based on these considerations, the current single maximum application rate for carbaryl is 12 lbs a.i./A for red scale treatments in California citrus. However, the maximum single use rate is limited to ≤ 2 lbs a.i./A in most agricultural crops. The maximum annual use rate for carbaryl on agricultural use sites is 20 lbs a.i./A in citrus, and approximately 33 lbs a.i./A in residential ornamental plantings (up to 8.36 lbs a.i./A with 4 applications).

Table 157. Summary of All Carbaryl Uses

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
ASPARAGUS	Preharvest	Ground, Chemigation	FlC	1	3	3 (NS[5])	7	19713-49
ASPARAGUS	Pre/Postharvest	Ground	WP	2.04	4	5.3125	7	19713-363
BEANS - edible podded, dried shelled, and plant parts used for feed and Soybeans	Foliar	Ground, Chemigation	FlC	1.53	4	6.1194	7	19713-49
BEANS - covers dried shelled beans and peas	Soil	Ground	G	1.4	4	6	7	9198-146
BEANS - stringbeans, dry beans, peas, lentils, pole beans, and soybeans	Foliar	Duster	D	1.5	NS	6.1	5	19713-53
BERRIES - covers caneberries and other berries - Subgroups 13-07A and 13-07B	Foliar.	Ground, Chemigation	FlC/WP	2.04	4	8	5	19713-49, 19713-363
BERRIES - covers caneberries and other berries - Subgroups 13-07A and 13-07B	Foliar.	Ground	WP	2.04	4	8	5	

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
BERRIES - covers blackberries, boysenberries, dewberries, loganberries, raspberries, and blueberries.	Foliar	Duster	D	2	4	8	5	19713-53
BRASSICA Subgroup 5A (broccoli, Chinese broccoli, Brussels sprouts, cabbage, Chinese cabbage (napa), Chinese mustard cabbage (gai choy), cauliflower, cavalo broccolo, and kohlrabi; BRASSICA Subgroup 5B (broccoli raab (rapini), Chinese cabbage (bok choy), collards, kale, mizuna, mustard greens, mustard spinach, rape greens, and turnip greens.	Foliar.	Ground, Chemigation	FlC	2.04	4	6.1194	7	19713-49
BRASSICA Subgroup 5A (broccoli, Chinese broccoli, Brussels sprouts, cabbage, Chinese cabbage (napa), Chinese mustard cabbage (gai choy), cauliflower, cavalo broccolo, and	Foliar.	Ground	WP, WSP	2.04	4	5.95	7	19713-363

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
kohlrabi; BRASSICA Subgroup 5B (broccoli raab (rapini), Chinese cabbage (bok choy), collards, kale, mizuna, mustard greens, mustard spinach, rape greens, and turnip greens.								
BRASSICA Subgroup 5A (broccoli, Chinese broccoli, Brussels sprouts, cabbage, Chinese cabbage (napa), Chinese mustard cabbage (gai choy), cauliflower, cavalo broccolo, and kohlrabi; BRASSICA Subgroup 5B (broccoli raab (rapini), Chinese cabbage (bok choy), collards, kale, mizuna, mustard greens, mustard spinach, rape greens, and turnip greens.	Ground	Spreader or shank applicator	G	2	3	6	7	19713-627
BRASSICA - covering Brussels sprouts, broccoli, cabbage, cauliflower, and kohlrabi	Foliar	Duster	D	2	NS	NS	5	19713-53

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
CARROT (INCLUDING TOPS & ROOTS)	Foliar	Duster	D	2	NS	NS	NS	19713-53
CITRUS, Crop Group 10	Foliar.	Ground, Chemigation	FlC	5	1	5 (NS)	NA	19713-49
CITRUS, Crop Group 10	Foliar.	Ground	FlC, WSP	5	4	20.398	NS	19713-49
CITRUS, Crop Group 10	Foliar.	Ground, Chemigation	FlC	5.1	4	20.398	NS	19713-51
CITRUS, Crop Group 10	Foliar.	Ground	WP, WSP	5	4	5 (NS)	NS	19713-363, 432-1226
CITRUS, Crop Group 10	Foliar	Ground	WP, WSP	5	4	19.975	NS	19713-363, 432-1226
CITRUS, Red Scale treatments in California	Foliar	Ground	all	12	1	20 for all carbaryl uses combined	Maintain 14 day interval before & after other carbaryl treatment s	NS.
COMMERCIAL/INSTITU TIONAL/INDUSTRIAL	When needed.	Hand bulb duster.	D	0.016	NS	NS	NS	36272-14

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
PREMISES/EQUIPMENT (OUTDOOR)								
COMMERCIAL/INSTITUTIONAL/INDUSTRIAL PREMISES/EQUIPMENT (OUTDOOR)	Perimeter	Spray, aerial application prohibited	FIC	0.08	4	NS	7	19713-49
COMMERCIAL/INSTITUTIONAL/INDUSTRIAL PREMISES/EQUIPMENT (OUTDOOR)	Perimeter	Spray, aerial application prohibited	WP	1	4	NS	7	19713-363
CORN (FIELD & POP)	Foliar	Ground, Chemigation	FIC	2.04	4	8.1592	14	19713-49
CORN (FIELD & POP)	Foliar	Ground	WP	2.04	4	7.99	14	19713-363
CORN (FIELD & POP)	Ground	Spreader or shank applicator	G	2	4	8	14	19713-627
CORN (SWEET)	Foliar.	Ground, Chemigation	FlC	2.04	4	8	3	19713-49
CORN (SWEET)	Ground	Spreader or shank applicator	G	2	4	8	3	19713-627
CORN (SWEET)	Foliar	Ground	WP	2.04	4	8	14	19713-363

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
CORN (SWEET)	Foliar	Duster	D	4	4	8	5	19713-53
CRANBERRY	Foliar.	Ground, Chemigation	FlC	2.04	4	8	7	19713-49
CUCURBITS - covers Crop Group 9	Foliar.	Ground	WP	1.02	4	4	7	19713-363
CUCURBITS - covers Crop Group 9	Foliar.	Ground, Chemigation	FlC	1.02	4	4	7	19713-49
CUCURBITS - covers Crop Group 10	Ground	Spreader or shank applicator	G	1	4	4	7	19713-627
CUCURBITS - covers cantaloupes, cucumber, melon, pumpkin, and squash	Foliar	Duster	D	1	4	4	5	19713-53
FLAX	Foliar.	Ground, Chemigation	FlC	1.53	2	3.0597	14	19713-49
FLAX	Foliar.	Ground	WP	1.53	2	2.975	14	19713-363
FORAGE CROPS: alfalfa, birdsfoot trefoil, and clover	Foliar, Stubble	Ground, Chemigation	FlC	1.53	4	6	56	19713-49
FORAGE CROPS: alfalfa, birdsfoot trefoil, and clover	Foliar, Stubble	Ground	WP	1.53	4	6	56	19713-363

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
FORAGE CROPS: alfalfa, clover, forage grasses, and pasture	Foliar	Duster	D	1.5	NS	NS	7	19713-53
FOREST TREES (ALL OR UNSPECIFIED), covers forested areas and rangeland trees	Foliar	Sprayer. Aerial application prohibited	FlC	1.02	2	N/A	7	19713-49
FOREST TREES (ALL OR UNSPECIFIED), covers forested areas and rangeland trees	Foliar	Sprayer. Aerial application prohibited	WP, WSP	1.02	2	NS	7	19713-363, 432-1226
FOREST TREES (ALL OR UNSPECIFIED), covers forested areas and rangeland trees	Direct Trunk Treatment	Sprayer. Aerial application prohibited	FlC	**2***	2	NS	NS	19713-49
FOREST TREES (ALL OR UNSPECIFIED), covers forested areas and rangeland trees	Direct Trunk Treatment	Sprayer. Aerial application prohibited	WP, WSP	**2***	2	NS	NS	19713-363, 432-1226
FRUITING VEGETABLES Crop Group 8	Foliar.	Ground, Chemigation	FlC	2.04	4	8.1592	7	19713-49
FRUITING VEGETABLES Crop Group 8	Foliar.	Ground	WP, WSP	2.04	4	7.99	7	19713-363, 432-1226

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
FRUITING VEGETABLES Crop Group 8	Ground	Spreader or shank applicator	G	2	4	8	7	19713-627
FRUITING VEGETABLES - covers tomato, pepper, and eggplant	Soil	Ground	G	2	4	8	7	9198-146
FRUITING VEGETABLES - covers tomato, pepper, and eggplant	Foliar	Duster	D	2	4	8	5	19713-53
GOLF COURSES	Broadcast	Ground	WSP	5.0	2	10	7	432-1226, 432-1227
GOLF COURSES	Broadcast	Ground	WP	5.0	2	10.0	7	19713-363
GRAPES	Foliar.	Ground, Chemigation	FlC	2.04	4	10.199	7	19713-49
GRAPES	Foliar.	Ground	WP	2.04	4	9.9875	7	19713-363
GRAPES	Foliar	Duster	D	2	4	10.2	5	19713-53
GRASSHOPPERS (all crops / sites)	Foliar or mature grasshoppers	Sprayer.	WSP	1.5	NS	NS	NS	432-1226

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
GRASSHOPPERS (all crops / sites)	Foliar or mature grasshoppers	Sprayer.	FIC	1.5	NS	NS	NS	19713-49
GRASSHOPPERS (all crops / sites)	Foliar or mature grasshoppers	Sprayer.	WP	1.53	NS	NS	NS	19713-363
IMPORTED FIRE ANTS (all crops / sites)	Mound treatment	Drench	FIC	0.05	NS	NS	30	19713-49
IMPORTED FIRE ANTS (all crops / sites)	Mound treatment	Drench	WP	0.05	NS	NS	NS	19713-363
IMPORTED FIRE ANTS (all crops / sites)	Outdoor, growing media	Sprayer. Aerial application prohibited	FIC	0.05	1	NS	NS	19713-49
IMPORTED FIRE ANTS (all crops / sites)	Outdoor, growing media	Sprayer. Aerial application prohibited	WP	1.5	1	NS	NS	19713-363
LEAFY VEGETABLES - Leaf petioles subgroup 4B, dandelion, endive escarole,	Foliar.	Ground, Chemigation	FlC	2.04	4	6.1194	7	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
lettuce, parsley, and spinach.								
LEAFY VEGETABLES - Leaf petioles subgroup 4B, dandelion, endive escarole, lettuce, parsley, and spinach.	Foliar.	Foliar	WP, WSP	2.04	4	5.95	7	19713-363, 432-1226
LEAFY VEGETABLES	Foliar	Broadcast, Fixed wing, Helicopter	EC	2	4	6	7	61842-37, 61842-38
LEAFY VEGETABLES - covering collards, garden beets (tops), kale, lettuce (head & leaf), mustard greens, spinach, and turnip (tops)	Foliar	Duster	D	2	4	6.1	7	19713-53
LEAFY VEGETABLES - garden beets (tops), turnip (tops)	Ground	Spread or shank applicator	G	2	3	6	7	19713-627
NON-CROPLAND USES: covers Conservation Reserve Program (CRP), Set Aside Program Acreage,	Spray	Ground, Chemigation	FIC	1.02	2	3.059	14	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
Wasteland, Rights of Way, Hedgegrows, ditchbanks, Roadsides.								
NON-CROPLAND USES: covers Conservation Reserve Program (CRP), Set Aside Program Acreage, Wasteland, Rights of Way, Hedgegrows, ditchbanks, Roadsides.	Spray	Ground	WSP	1	2	3	14	432-1226
NON-CROPLAND USES: covers Conservation Reserve Program (CRP), Set Aside Program Acreage, Wasteland, Rights of Way, Hedgegrows, ditchbanks, Roadsides.	Spray	Ground	WP	1.02	2	2.975	14	19713-363
OKRA	Foliar.	Ground	FlC	1.53	4	6.1194	6	19713-49
OKRA	Foliar.	Ground	WP	1.53	4	5.95	6	19713-363
OKRA	Foliar	Duster	D	2	NS	6.1	5	19713-53
OLIVE	Foliar.	Ground, Chemigation	FlC	5	2	10	14	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
OLIVE	Foliar.	Ground	WP, WSP	5	2	10	14	19713-363, 432-1226
ORNAMENTALS (UNSPECIFIED): covers trees and plants, woody shrubs and vines	Foliar	Sprayer. Aerial application prohibited	FlC	1.02	4	4	7	19713-49
ORNAMENTALS (UNSPECIFIED): covers trees and plants, woody shrubs and vines	Foliar	Sprayer. Aerial application prohibited	WP, WSP	1.2	4	4	7	19713-363, 432-1226
ORNAMENTALS (UNSPECIFIED): covers trees and plants, woody shrubs and vines	Direct Trunk Treatment	Sprayer.	FlC	**2***	2	4	6	19713-49
ORNAMENTALS (UNSPECIFIED): covers trees and plants, woody shrubs and vines	Direct Trunk Treatment	Sprayer.	WP, WSP	**2***	2	4	6	19713-363, 432-1226
ORNAMENTAL LAWNS & TURF	Soil	Ground	G	5	4	10	7	9198-146
ORNAMENTAL SOD FARM (TURF) and LAWNS AND TURF	Foliar.	Pressure sprayer.	FlC	5	2	10	7	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
ORNAMENTAL SOD FARM (TURF) and LAWNS AND TURF	Broadcast	Ground	WSP	5	2	10	7	432-1226
ORNAMENTAL SOD FARM (TURF) and LAWNS AND TURF	Broadcast	Ground	WP	5	2	10	7	19713-363
PASTURES, grasses grown for hay and/or seed	Foliar.	Ground, Chemigation	FlC	1.53	2	3.0597	14	19713-49
PASTURES, grasses grown for hay and/or seed	Foliar.	Ground	WP, WSP	1.53	2	2.975	14	19713-363, 432-1226
PASTURES, grasses grown for hay and/or seed	Ground	Spreader or shank applicator	G	1.5	2	3	14	19713-627
PEANUTS	Foliar.	Ground, Chemigation	FlC	2.04	4	8.1592	7	19713-49
PEANUTS	Foliar.	Ground	WP	2.04	4	7.99	7	19713-363
PISTACHIOS	Foliar, Dormant / Delayed dormant	Ground	WP, WSP	5.02	4	14.96	7	19713-363, 432-1226
PISTACHIOS - nonbearing	Ground	Spreader or shank applicator	G	2	NS	10	7	19713-627

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
PISTACHIOS - all states	Foliar, Dormant / Delayed dormant	Ground, Chemigation	EC	5	4	15	7	61842-37
PISTACHIOS - all states	Foliar, Dormant / Delayed dormant	Ground, Chemigation	FIC	5.1	4	15.2985	7	19713-49
POME FRUIT (Group 11): covers apples, pears and others	Foliar, Fruit Thinning	Ground, Chemigation	FlC	3.06	4	12	14	19713-49
POME FRUIT (Group 11): covers apples, pears and others	Foliar, Fruit Thinning	Ground	WP, WSP	2.975	4	12	14	19713-363, 432-1226
POTATO	Ground	Spreader or shank applicator	G	2	3	6	7	19713-627
POTATO	Foliar	Duster	D	2	NS	6	5	19713-53
PRICKLYPEAR CACTUS PADS	Foliar.	Ground	FIC	2.04	4	6.1194	7	19713-49
PRICKLYPEAR CACTUS PADS	Foliar.	Ground	WP	2.04	4	5.95	7	19713-363
RANGELAND	Foliar	Ground, Chemigation	FIC	1.02	1	1.0199	NA	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
RANGELAND	When needed.	Ground	G	1	1	1	NA	19713-630, 19713-627
RANGELAND	Foliar	Ground	WP, WSP	1.02	1	1.2	NA	19713-363, 432-1226
RANGELAND	Ground	Spreader or shank applicator	G	1	1	NS	NA	19713-627
ROOT CROP VEGETABLES - covering garden beets (roots), carrots, horseradish, parnsips, radish, rutabaga, salsify, and turnip (roots)	Broadcast	Spreader.	G	2	3	6	7	19713-627
ROOT CROP VEGETABLES - covering garden beets (roots), radish, rutabaga, and turnip (roots)	Foliar	Dust Applicator	D	2	NS	6	7	19713-53
ROOT CROP VEGETABLES - covering garden beets (roots), horseradish, radish, parsnip, rutabaga, salsify, and turnip (roots)	Ground	Spreader or shank applicator	G	2	3	6	7	19713-627

Use	**Application Timing/Target**	**Application Method**	**Formulation [1]**	**Maximum Single rate (lbs a.i./A)**	**Max #/ Yr[2]**	**Max Annual rate[3]**	**MRI (Days) [4]**	**Label**
ROOT & TUBER CROPS - Crop Group 1 except sugar beets and sweet potatoes	Foliar	Ground, Chemigation	FIC	2.04	6	6.1194	7	19713-49
ROOT & TUBER CROPS - Crop Group 1 except sugar beets and sweet potatoes	Foliar	Ground	WP	2.04	6	5.95	7	19713-363
SHRIMP PONDS, COMMERCIAL	When needed.	Sprayer.	FlC	8.01	NS	8.01164	NS	TX020007
SORGHUM	Foliar.	Ground, Chemigation	FlC	2.04	4	6.1194	7	19713-49
SORGHUM	Foliar.	Ground	WP	2.04	4	5.95	7	19713-363
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Delayed Dormant or Dorman	Ground, Chemigation	FlC, WSP	5.1	1	14.2786	**7**	19713-49, 432-1226
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Foliar	Ground, Chemigation	FlC, WSP	4.08	3	14.2786	**7**	19713-49, 432-1226
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Foliar	Ground, Chemigation	FlC	3.06	3	14.2786	**7**	19713-49

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Foliar	Ground, Chemigation	WP, WSP	3	3	14	**7**	19713-50, 432-1226
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Foliar,	Ground, Chemigation	WP	4	3	14	**7**	19713-50
STONE FRUIT (Group 12): covers apricot, cherries, nectarines, peaches, plums, plumcot, and prunes	Delayed Dormant or Dorman	Ground, Chemigation	WP, WSP	5	1	14	7	19713-50, 432-1226
STRAWBERRIES	Foliar.	Ground, Chemigation	FlC	2.04	4	8	7	19713-49
STRAWBERRIES	Soil	Ground	G	2	4	8	7	9198-146
STRAWBERRIES	Foliar	Duster	D	2	4	8	5	19713-53
SUGAR BEET	Foliar.	Ground, Chemigation	FlC	1.53	2	3.0597	14	19713-49
SUGAR BEET	Foliar.	Ground	WP	1.53	2	2.975	14	19713-363
SUGAR BEET	Ground	Spreader or shank applicator	G	1.5	2	3	14	19713-627

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
SUNFLOWER	Foliar.	Ground, Chemigation	FlC	1.53	2	3.0597	7	19713-49
SUNFLOWER	Foliar.	Ground	WP	1.53	2	2.975	7	19713-363
SWEET POTATO	Foliar.	Ground, Chemigation	FlC	2.04	4	4	7	19713-49
SWEET POTATO	Foliar.	Ground	WP	2.04	4	4	14	19713-363
Ticks (all crops/sites)	Perimeter	Ground	WSP	1	4	4	7	432-1226
Ticks (all crops/sites)	Perimeter	Spray	FIC	1	4	NS	As Needed	19713-49
TOBACCO	Foliar.	Ground, Chemigation	FlC	2.04	4	8.1592	7	19713-49
TOBACCO	Foliar.	Ground	WP	2.04	4	7.99 lb / a	7	19713-363
TREE NUTS Crop Group 14	Foliar, Dormant or Delayed Dormant	Ground, Chemigation	FlC	5.1	4	15.2985	7	19713-49
TREE NUTS Crop Group 14	Foliar, Dormant or Delayed Dormant	Ground, Chemigation	EC	5	4	15	7	61842-37

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
TREE NUTS Crop Group 14	Foliar, Dormant or Delayed Dormant	Ground	WP, WSP	5	4	14.96	7	19713-363, 432-1212
TREE NUTS Crop Group 14	Foliar, Dormant or Delayed Dormant	Ground	WSP	5	4	15	7	432-1226
RESIDENTIAL USES								
ASPARAGUS - preharvest (spears)	Ground	Broadcast	G	2	3	6	3	432-1212
ASPARAGUS - postharvest (ferns)	Ground	Broadcast	G	2	2	10 - pre & post	7	432-1212
ASPARAGUS	Foliar	Spray	EC	0.023	3	NS	As needed	19713-89
BEANS	Foliar	Spray	EC	0.023	4	NS	NS	19713-89
BEANS	Ground	Broadcast	G	1.5	4	6	7	432-1212
BLUEBERRIES	Soil	Bait	WP	0.415	3	NS	14	8119-5
BRASSICA: covering broccoli, Brussels sprouts, cabbage, cauliflower, Chinese cabbage, collards,	Ground	Broadcast	G	2	4*	6	7	432-1212

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
kale, mustard greens, and Kohlrabi								
BRASSICA / COLE CROPS: covering broccoli, Brussels sprouts, cabbage, and cauliflower	Ground	Broadcast	G	1.96	3	NS	7	829-285
BRASSICA / COLE CROPS: covering broccoli, Brussels sprouts, cabbage, cauliflower, cavalo broccolo, collards, kale, kohlrabi, mizuna, mustard greens, and rape greens.	Soil	Bait	WP	0.415	3	NS	14	8119-5
BRASSICA / COLE CROPS: covering broccoli, Brussels sprouts, cabbage, cauliflower, chinese cabbage, kohlrabi, collards, kale, mustard greens, mustard spinach, and turnip greens	Foliar	Spray	EC	0.023	4	NS	NS	19713-89
BRASSICA / COLE CROPS, Sub-Group 5 A	Ground	Broadcast	G	2	4	6.0	7	34704-289

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days)[4]	Label
CANEBERRIES & OTHER BERRIES	Soil	Bait	WP	0.415	3	NS	14	8119-5
CANEBERRIES & OTHER BERRIES: covering blackberries, blueberries, dewberries, grapes, raspberries, and strawberries	Foliar	Spray	EC	0.046		NS	7	19713-89
CITRUS	Foliar	Spray	EC	0.046	8	NS	7	19713-89
CORN (SWEET)	Foliar	Spray	EC	0.023	8	NS	NS	19713-89
CORN (SWEET)	Ground	Broadcast	G	2	4	8	7	34704-289
CUCURBITS: Covering cucumber, melons, and squash	Ground	Broadcast	G	0.98	6	NS	7	829-285
CUCURBITS: Covering cucumber, melons, and squash	Foliar	Spray	EC	0.023	6	NS	NS	19713-89
CUCURBITS: Covering cucumber, melons, and squash	Grounds	Broadcast	G	1	6	6	7	432-1212

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
FRUITING VEGETABLES: Covering tomato, pepper, eggplant	Ground	Broadcast	G	2	4	8	7	432-1212
FRUITING VEGETABLES: Covering eggplant, peppers, and tomato	Ground	Broadcast	G	1.96	4	NS	7	829-285
FRUITING VEGETABLES: Covering eggplant, groundcherry, okra, peppers, and tomatoes	Foliar	Spray	EC	0.023	7	NS	NS	19713-89
FRUITING VEGETABLES - Crop Group 8	Ground	Broadcast	G	2	4	8	7	34704-289
HOUSEHOLD/DOMESTIC DWELLINGS OUTDOOR PREMISES	Ground	Broadcast	G	7.84	3	NS	7	432-1212
HOUSEHOLD/DOMESTIC DWELLINGS OUTDOOR PREMISES	Ground	Broadcast	G	1.87	3	NS	7	8378-36
HOUSEHOLD/DOMESTIC DWELLINGS OUTDOOR PREMISES	Ground	Broadcast	G	8.36	4	NS	7	9198-146

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
HOUSEHOLD/DOMESTIC DWELLINGS OUTDOOR PREMISES	Foliar	Spray	EC	0.046	NS	NS	NS	19713-89
LEAFY BRASSICA - Sub-Group 5 B	Ground	Broadcast	G	2	4	6	7	34704-289
LEAFY VEGETABLES: Covering cardoon, celery, celtuce, Florence fennel, dandelion, endive, lettuce (head, leaf), parsley, rhubarb, spinach, and Swiss chard	Foliar	Spray	EC	0.023	5	NS	NS	19713-89
LEAFY VEGETABLES: covering garden beets (tops) and turnip (tops)	Ground	Broadcast	G	2	3	6	7	432-1212
LEAVES of ROOT & TUBER VEGETABLES: covering beet, garden (tops) and turnip (tops)	Ground	Broadcast	G	2	3	6	7	34704-289
ORNAMENTAL PLANTINGS	Ground	Broadcast	G	4	3	NS	7	432-1212
ORNAMENTAL PLANTINGS	Ground	Broadcast	G	1.96	3	NS	7	829-285

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
ORNAMENTAL PLANTINGS	Soil	Bait	WP	0.415	6	NS	21	8119-5
ORNAMENTAL PLANTINGS	Ground	Broadcast	G	4	3	NS	7	8378-36
ORNAMENTAL PLANTINGS	Ground	Broadcast	G	4	4	NS	7	9198-146
ORNAMENTAL PLANTINGS	Foliar	Spray	EC	0.023	6	NS	NS	19713-89
POTATO	Ground	Broadcast	G	2	3	6	7	432-1212
RESIDENTIAL LAWNS	Foliar.	Spot	FlC	5	2	10	7	19713-49
RESIDENTIAL LAWNS	Foliar.	Spot	FlC	5	2	10	7	19713-49
RESIDENTIAL LAWNS	Ground	Spot	WSP	5	2	10	7	432-1226
ROOT CROP VEGETABLES: Covering garden beets (roots), carrots, horseradish, radishes, parsnips, rutabaga, salsify, turnip (roots)	Ground	Broadcast	G	2	6	6	7	432-1212
ROOT CROP VEGETABLES: Covering garden beets (roots),	Ground	Broadcast	G	1.96	3	NS	2	829-285

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
potatoes, carrots, radishes, and turnip (roots)								
ROOT CROP VEGETABLES except sugarbeets and potatoes	Foliar	Spray	EC	0.023	6	NS	NS	19713-89
ROOT & TUBOR VEGETABLES - Crop Group 1	Ground	Broadcast	G	2	3	6.0	7	34704-289
STRAWBERRIES	Ground	Broadcast	G	2	5	10	7	432-1212
STRAWBERRIES	Soil	Bait	WP	0.415	3	NS	14	8119-5
STRAWBERRIES	Ground	Broadcast	G	2	4	8.0	7	34704-289
SWEET POTATO	Foliar	Spray	EC	0.023	8	NS	NS	19713-89
SWEET POTATO	Ground	Broadcast	G	2	3	6.0	7	34704-289
TOMATOES	Soil	Bait	WP	0.415	3	NS	14	8119-5
TREE FRUIT: Covering apples, apricots, cherries, nectarines, peaches, pears, plums, and prunes	Foliar	Spray	EC	0.046	8	NS	7	19713-89

Table 157 Footnotes

1- Formulation: flowable concentrate (FlC), emulsifiable concentrate (EC), wettable powder (WP), water soluble powder (WSP), bait, granular (G), or dust (D)
2- Max #/Yr = Maximum number of applications per year
3- Max Annual rate = Maximum lbs a.i./A/year

Use	Application Timing/Target	Application Method	Formulation [1]	Maximum Single rate (lbs a.i./A)	Max #/ Yr[2]	Max Annual rate[3]	MRI (Days) [4]	Label
4- Minimum Reapplication Interval (MRI) 5- Not Specified (NS)								

Summary of all methomyl uses

The summary of all methomyl uses presented below contains selections of each use that are presumed to represent the highest risk by considering maximum application rates and methods proposed by the action. To summarize the current methomyl action, NMFS integrated the mitigation agreed to as part of the PID, as well as the conservation measures discussed as discussed in the Description of Action. In general, current single maximum methomyl application rates do not exceed 0.9 lb a.i./A nationwide for flowable formulations; however, a single application rate of 1.5 lb a.i./A is currently permitted for corn and sweet corn use patterns for granular formulation. The maximum annual rate of methomyl that may be applied to certain crop sites is 13 lb a.i./A (e.g., sweet corn). Fly baits, recommend frequent reapplication (*e.g.,* every 2-5 days; registrations 2724-274 and 7319-6) and do not specify a maximum annual rate.

Table 158. Methomyl Master Use Summary

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Alfalfa	post-cutting to harvest	air/ground/overhead chemigation	SP[B] LV[C]	0.9	18	13	5	61842-52, 61842-55
Anise (fennel)	emergence to harvest	air/ground	SP[B] LV[C]	0.9	5 10 (CA)	4.5 9 (CA)	5	61842-52, 61842-55
Apple	pre-bloom to harvest	Ground only	SP[B] LV[C]	0.9	5	4.5	7	61842-52, 61842-55
Asparagus	emergence to harvest	air/ground	SP[B] LV[C]	0.9	5	4.5	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Avocado	post-bloom to harvest	air/ground	SP[B] LV[C]	0.9	1	0.9	5	61842-52, 61842-55
Bean, Succulent	emergence to harvest	air/ground/ overhead chemigation	SP[B] LV[C]	0.9	5 10 (AZ, CA, FL, NC, SC)	4.5 9 (AZ, CA, FL, NC, SC)	5	61842-52, 61842-55
Bean, dry	emergence to harvest	air/ground/ overhead chemigation	SP[B] LV[C]	0.9	5 10 (AZ, CA)	4.5 9 (AZ, CA)	5	61842-52, 61842-55
Beet, table	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4 8 (CA, TX)	3.6 7.2 (CA, TX)	5	61842-52, 61842-55
Bermudagrass pasture	post-cutting to harvest	air/ground	SP[B] LV[C]	0.9	9	8.1	5	61842-52, 61842-55
Blueberry	green-up to harvest	Ground only	SP[B] LV[C]	0.9	4	3.6	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Broccoli	emergence to harvest	air/ground	SP[B] LV[C]	0.9	7-14 21 (AZ)	NS[H]	2	61842-52, 61842-55
Brussels Sprouts	emergence to harvest	air/ground	SP[B] LV[C]	0.9	6 12 (CA, DE, GA, MD, NJ, TX)	5.4 10.8 (CA, DE, GA, MD, NJ, TX)	2	61842-52, 61842-55
Cabbage	emergence to harvest	air/ground	SP[B] LV[C]	0.9	8 16 (DE, FL, GA, MD, NJ, NC, PA, SC, TX, VA, WI) 24 (AZ, CA)	7.2 13 (DE, FL, GA, MD, NJ, NC, PA, SC, TX, VA, WI, AZ, CA)	2	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Carrot	emergence to harvest	air/ground	SP[B] LV[C]	0.9	7 14 (CA)	6.3 12.6 (CA)	5	61842-52, 61842-55
Cauliflower	emergence to harvest	air/ground	SP[B] LV[C]	0.9	8-16 24 (AZ)	7.2-13	2	61842-52, 61842-55
Celery	emergence to harvest	air/ground	SP[B] LV[C]	0.9	7-14 21 (AZ)	6.3-12.6 13 (AZ)	5	61842-52, 61842-55
Chicory	emergence to harvest	air/ground	SP[B] LV[C]	0.9	2	1.8	5	61842-52, 61842-55
Chinese cabbage	emergence to harvest	air/ground	SP[B] LV[C]	0.9	8-16 24 (CA)	7.2-13	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Collards	emergence to harvest	air/ground	SP[B] LV[C]	0.9	6-12 18 (CA, FL)	5.4-10.8 13 (CA, FL)	5	61842-52, 61842-55
Corn, Field, Popcorn and seed	emergence to harvest	air/ground	SP[B] LV[C]	0.45	5 10 (CA, HI, TX)	2.25 4.5 (CA, HI, TX)	5	61842-52, 61842-55
	leaf stage until tasseling	Ground (banded)	G[D]	0.15	10 [E]	1.50	NS[H]	57242-2
Corn, sweet	emergence to harvest	air/ground	SP[B] LV[C]	0.45	14-28 42 (AZ)	6.3-12.6 13 (AZ)	1	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
	leaf stage until tasseling	Ground (banded)	G[D]	0.15	10 [E]	1.50	NS	57242-2
Cotton	emergence to harvest	air/ground	SP[B] LV[C]	0.675	2	1..35	3 or 5	61842-52, 61842-55
Cucumber	emergence to harvest	air/ground	SP[B] LV[C]	0.9	6 12 (AZ, CA, FL, GA, MD, NC, PA, SC, TX, VA)	5.4 10.8 (AZ, CA, FL, GA, MD, NC, PA, SC, TX, VA)	5	61842-52, 61842-55
Eggplant	emergence to harvest	air/ground	SP[B] LV[C]	0.9	5-10 15 (GA)	4.5-9 13 (GA)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Endive, escarole	emergence to harvest	air/ground	SP[B] LV[C]	0.9	5-10 15 (CA)	4.5-9 13 (CA)	5	61842-52, 61842-55
Garlic	emergence to harvest	air/ground	SP[B] LV[C]	0.45	6	2.7	5	61842-52, 61842-55
Grapefruit	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	3	2.7	5	61842-52, 61842-55
Horseradish	emergence to harvest	Ground only	SP[B] LV[C]	0.45	4	1.8	5	61842-52, 61842-55
Leafy Green Vegetables (Beet tops, dandelions, kale, mustard greens, parsley, Swiss chard, turnip greens)	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4-8 12 (AZ, CA)	3.6-7.2 10.8 (AZ, CA)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Lemon	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	3	2.7	5	61842-52, 61842-55
Lentils	emergence to harvest	air/ground	SP[B] LV[C]	0.9	1	0.9	5	61842-52, 61842-55
Lettuce, head	emergence to harvest	air/ground	SP[B] LV[C]	0.9	7-12 21 (AZ, CA)	6.3-12.6 13 (AZ, CA)	2	61842-52, 61842-55
Lettuce, leaf	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4-12 16 (AZ)	3.6-10.8 13 (AZ)	2	61842-52, 61842-55
Melon	emergence to harvest	air/ground	SP[B] LV[C]	0.9	6-12 18 (AZ)	5.4-10.8 13 (AZ)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Mint (Peppermint and Spearmint)	green-up to harvest	air/ground	SP[B] LV[C]	0.9	4	3.6	5	61842-52, 61842-55
Nectarine	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	3	2.7	5	61842-52, 61842-55
Onion (Green)	emergence to harvest	air/ground/ overhead and drip chemigation	SP[B] LV[C]	0.9	6-12 18 (AZ, CA)	5.4-10.8 13 (AZ, CA)	5	61842-52, 61842-55
Onion (Dry Bulb)	emergence to harvest	air/ground/ overhead and drip chemigation	SP[B] LV[C]	0.9	4 8 (TX, WI)	3.6 7.2 (TX, WI)	5	61842-52, 61842-55
Orange	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	3	2.7	5	61842-52, 61842-55
Peach	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	6	5.4	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Peanut	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4	3.6	5	61842-52, 61842-55
Pear	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	2	1.8	5	61842-52, 61842-55
Pea, succulent	emergence to harvest	air/ground/ overhead chemigation	SP[B] LV[C]	0.9	3-6 9 (CA, GA)	2.7-5.4 8.1 (CA, GA)	3	61842-52, 61842-55
Pecan	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	7	6.3	5	61842-52, 61842-55
Pepper	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4-8 12 (AZ)	3.6-7.2 10.8 (AZ)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Pomegranite	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	2	1.8	5	61842-52, 61842-55
Potato	emergence to harvest	air/ground/ overhead chemigation	SP[B] LV[C]	0.9	5 10 (CA, TX)	4.5 9 (CA, TX)	5	61842-52, 61842-55
Sorghum	emergence to harvest	air/ground	SP[B] LV[C]	0.45	2	0.9	5	61842-52, 61842-55
Soybean	emergence to harvest	air/ground	SP[B] LV[C]	0.45 - 0.9	3	1.35 2.7 (AR,LA)	5	61842-52, 61842-55
Spinach	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4-12 16 (AZ, CA)	3.6-10.8 13 (AZ, CA)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Sugar beet	emergence to harvest	air/ground/overhead chemigation	SP[B] LV[C]	0.9	5	4.5	5	61842-52, 61842-55
Summer squash	emergence to harvest	air/ground	SP[B] LV[C]	0.9	6-12 18 (AZ, CA, FL, GA)	5.4-10.8 13 (AZ, CA, FL, GA)	5	61842-52, 61842-55
Tangelo, tangerine	pre-bloom to harvest	air/ground	SP[B] LV[C]	0.9	3	2.7	5	61842-52, 61842-55
Tobacco	emergence to harvest	air/ground	SP[A] LV[B]	0.45	5	2.25	5	61842-52, 61842-55
Tomato	emergence to harvest	air/ground	SP[B] LV[C]	0.9	7 14 (AZ, southern CA, FL, GA, TX)	6.3 12.6 (AZ, southern CA, FL, GA, TX)	5	61842-52, 61842-55

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
Tomatillo	emergence to harvest	air/ground	SP[B] LV[C]	0.9	5 10 (AZ, southern CA, FL, GA, TX)	4.5 9 (AZ, southern CA, FL, GA, TX)	5	61842-52, 61842-55
Turf (sod farms)	emergence to harvest	air/ground	SP[B] LV[C]	0.9	4	3.6	5	61842-52, 61842-55
Wheat in ID, OR & WA only	emergence to harvest	air/ground/overhead chemigation	SP[B] LV[C]	0.45	4	1.8	5	61842-52, 61842-55
Supplemental labels								
Broccoli raab - CA	emergence to harvest	air/ground	SP[B]	0.9	16	13	5	61842-52
Chinese broccoli - CA.	emergence to harvest	air/ground	SP[B]	0.9	10	9	2	61842-52

Use	Application Timing	Application Method	Formulation	Maximum Single Application Rate (lbs ai / A)	Maximum Number of Applications/Year	Maximum Annual Application Rate (lbs ai /A/Year)	MRI [A] (days)	Label
NonBearing fruit, nut, grape - CA.	post-bloom	air/ground	SP[B]	0.9	5	4.5	5	61842-52
Pumpkin - CA.	emergence to harvest	air/ground	SP[B]	0.9	3	2.7	5	61842-52
Radish - CA	emergence to harvest	air/ground	SP[B]	0.9	8	7.2	5	61842-52
Radish - FL	emergence to harvest	air/ground	SP[B] LV[C]	0.45	32	13	5	61842-52, 61842-55
Sweet Potato - CA	emergence to harvest	air	SP[B]	0.9	3	2.7	5	61842-52
Commercial facilities, Agricultural, Non-Agricultural [G]								
Fly Baits [D, E, f, G]	Presence of active flies	Not Applicable	See [F]	0.22	NS[H]	NS	1-5	2724-274, 7319-6

Table 158 Footnotes

[A] Minimum Reapplication Interval (MRI)

[B] SP-Water soluble powder in water soluble package

[C] LV-Water soluble liquid

[D] Granular

[E] Based on maximum dose/year

[F] Non-flowable, solid formulation (granular, bait, or slurry painted on outside walls)

[G] Additional information on the fly baits:

- Not to be used inside or around homes, or any other place where children or pets are likely to be present.
- Place scatterbait in areas inaccessible to livestock. Keep children and pets out of treated areas. Do not place scatterbait around commercial dumpsters that are not enclosed.
- Bait stations should be at least 4' above ground and in areas not accessible to children, pets, and livestock.
- Brush paste on outside of structures so that it is inaccessible to children, pets, and livestock.

For use:

- Outside of commercial facilities, such as, canneries, beverage plants, meat and poultry processing plants, food processing plants, commercial refuse dumpsters which are enclosed, feedlots, and livestock housing.
- Stables, outside of milking parlors, kennels, fast food establishments, restaurants, commissaries, bakeries, supermarkets, warehouses, feedlots, livestock housing, food processing plants, beverage plants, meat and poultry processing plants, fenced dumpsters
- Livestock housing (outside)
- Inside on walkways in caged poultry layer houses,
- Inside of caged poultry layer houses
- Initially daily to control fly population than decrease reapplications to 2-5 days depending on fly control

[H] Not Specified (NS)

APPENDIX F: RUNOFF MITIGATION

Introduction
This appendix contains a copy of EPA's proposed descriptions for conservation measures in the runoff picklist. Transmitted from EPA to NMFS on 12-07-2023.

EPA Proposed Descriptions for Conservation Measures in the Runoff Pick List

This appendix describes the runoff mitigation pick list measures referenced earlier in section **Error! Reference source not found.**. These descriptions identify the minimum requirements (indicated in **bold** text) for each measure. The descriptions do not provide the prescriptive design elements for these measures. To better understand the descriptions, it may be useful for individuals to first understand the basics of sheet flow or concentrated flow. Sheet flow is when water flows in a thin layer. The greater the distance that water must flow (and based on field topography), the more that sheet flow will become concentrated flow, which can lead to significant sediment erosion.

Because implementation of specific mitigation measures varies by crop and location, pesticide users adopting one or more of these measures would be encouraged to consult with local specialists experienced in planning, building, and maintaining these mitigation measures. Additionally, some measures may have specific state and/or local laws and regulations that must be followed.

The descriptions of the mitigation measures included in this appendix are adapted from the National Pollutant Discharge Elimination System (NPDES) Permit Writers' Manual for CAFOs and information from the open literature. For further discussion and consideration of the application of these mitigation measures, see EPA's webpage on non-point source pollution reduction in agriculture and National Management Measures to Control Nonpoint Pollution from Agriculture (Chapter 4).

Vegetative Filter strips (on-field)

Filter strips are managed on-field areas of grass or other permanent herbaceous vegetation that intercept and disrupt flow of runoff, trap sediment, and reduce pesticide concentrations in water. Generally, a filter strip can vary in width (typically 20 to 120 feet wide). However, minimal distances for effective vegetative filter strips are 30 feet for water runoff. Filter strips are usually planted with native grasses and perennial herbaceous plants. Nutrients, pesticides, and soils in the runoff water are filtered through the grass, potentially adsorbed by the soil, and potentially taken up by the plants. The effectiveness of filter strips to reduce pesticide loading into an adjacent surface water body depends on many factors, such as topography, field conditions, hydrologic soil group, antecedent moisture conditions, rainfall intensity, properties of the pesticide, application methods, width of the filter strip, and types of vegetation within. Therefore, risk reductions obtained from the use of filter strips may vary. Its use can support or connect other buffer practices within and between fields.

Establish and maintain vegetative filter strips such that the area immediately upslope must eliminate or substantially reduce concentrated flow and promote surface sheet flow runoff. The design and maintenance must consider a lifespan sufficient for the expected duration of the use of the product across multiple growing seasons. Where there is concentrated flow, structural elements must be added within the field to prevent erosion and promote sheet flow across the filter strip.

This may be most easily achieved by aligning rows as closely as possible so that they are perpendicular to the slope. Use of water bars or berms to break up the concentrated flow and divert concentration flow back into the field is another useful tool to promote sheet flow. Reduced tillage practices, especially near the field border strip, will result in less sediment loading and the best performance of a vegetative filter strip.

Permanent filter strip vegetative plantings must be harvested or mowed as appropriate (producers enrolled in conservation programs need to follow specific mowing and maintenance restrictions) to encourage dense growth and maintain upright growth.

The maintenance program must keep vegetation tall in spring and early summer to help slow runoff flow, maximize disruption of concentrated flow, and reduce the chance of structural damage. Regular maintenance must also include inspection after major storms, removal of excess trapped sediment, and repair of eroding areas.

Grassed Waterways (on-field and off-field)

Grassed waterways are natural or constructed vegetated channels designed to direct surface water, flowing at non-erosive velocities, to an outlet that is not likely to erode (e.g., another vegetated channel, an earth ditch). Grassed waterways are used to prevent significant erosion. In concentrated flow areas, grassed waterways can act as an important component of erosion control by slowing the flow of water and filtering sediment.

Other benefits of grassed waterways include the safe disposal of runoff water, improved water quality, improved wildlife habitat, reduced damage associated with sediment, and an improvement in overall landscape aesthetics. Grassed waterways are usually planted with perennial grasses, preferably native species where possible. Some common grass species used in waterways are Timothy, tall fescue, perennial ryegrass and Kentucky bluegrass.

The user must establish a maintenance program to maintain waterway capacity, vegetative cover, and outlet stability. Do not damage vegetation by machinery, herbicides, or erosion. Grassed waterways must be inspected regularly, especially following heavy rains. Any damage or disruptions must be repaired immediately by filling, compacting, and reseeding. Sediment deposits must be removed to maintain capacity of grassed waterway. Maintain a healthy, dense, and functional grass strip. Runoff outflow must be directed to a system such

as another grassed waterway, an earthen ditch, a grade-stabilization structure, a filter strip, water or sediment basin, or other suitable outlet with adequate capacity to handle the runoff and prevent significant erosion.

Field Border (off-field)

A field border is defined as a strip of permanent vegetation established at the edge or around the perimeter of a field. A field border can reduce runoff-based erosion and protect soil and water quality, when down slope of a crop field, by slowing the flow of water, dispersing concentrated flow, and increasing the chance for soil infiltration.

Use of a field border can support or connect other buffer practices within and between fields.

Establishment and maintenance of the field border and any land immediately upslope of the border must aim to eliminate or significantly reduce concentrated water flow and promote surface sheet flow runoff.

To prevent significant erosion within a field border, **concentrated flow must be broken up or redirected**. This may be achieved by aligning the field border and planting rows as closely as possible in a direction that is perpendicular to the slope. Use of water bars or berms to divert concentrated flow back into the field is another useful tool to break up the concentrated flow and promote sheet flow into the border.

A field border must have a minimum width 30 feet for the purpose of reducing pesticides in runoff and be composed of a permanent dense vegetative stand. This stand must be composed of stiff upright grasses. Non-woody flowering plants may also be included in a well-managed border.

Reduced tillage practices, especially near the field border strip, will result in less sediment loading and the best performance of the field border in reducing runoff.

Inspect field borders after major storms and repair eroding areas.

Cover Crop (on-field)

A cover crop is a close-growing crop that temporarily protects the ground from wind and water erosion. Common cover crops include cereal rye, oats, clover, crown vetch, and winter wheat or combinations of those crops. Cover crops are most often used when low residue-producing crops are grown on erodible land. Cover crops increase soil stability, reduce runoff, and reduce erodibility of field soils.

The cover crop must be planted and remain on the field up to the field preparation for planting the crop.

Crop insurance allows for cover crop flexibilities and producers should be mindful of those flexibilities and guidelines.

Planting directly into a standing terminated, mowed, or rolled cover crop will provide the greatest benefit for reducing runoff. Cover crops may be used in conjunction with reduced tillage practices to further reduce surface runoff from production fields.

Contour Buffer Strips (on-field)

Contour buffer strips are strips of permanent herbaceous vegetation, primarily of perennials such as grass, alternated with wider cultivated strips that are farmed on the contour. Contour buffer strips help to manage runoff and trap sediment. Because the vegetated buffer strip is established on the contour, runoff flows evenly across the entire surface of the strip, reducing water and sediment erosion. The vegetation slows runoff, helping the water to soak into the soil and reducing erosion. Sediment, nutrients, and other pollutants are filtered from the runoff as it flows through the strip, thereby improving surface water quality.

The specific recommendations for establishing buffers vary from site to site.

Contour buffer strip widths must be a minimum of 15 feet. Wider distances may be appropriate based on variables such as slope, soil type, field conditions, climate, and erosion potential. Contour buffer strips are unsuitable in fields where irregular, rolling topography makes following a contour impractical.

To ensure maximum performance, **the integrity of the buffer must be maintained for the entire width and length, including:**

- The contour buffer must be harvested or mowed, reseeded, and fertilized as necessary to maintain plant density and vigorous plant growth.
- Vegetation must be kept tall in spring and early summer to help slow runoff flow, maximize disruption of concentrated flow, and reduce the chance of structural damage.
- Regular maintenance must also include inspection after major storms, removal of trapped sediment, and repair of eroding areas.

Contour Farming (on-field)

Contour farming is the use of ridges and furrows formed by tillage, planting, and other farming operations following the contour to change the direction of runoff from directly downslope to across the slope. The disruption of downslope flow slows the runoff velocity and allows for more time for runoff to infiltrate the field soils, thereby reducing runoff.

The effectiveness of contour farming to reduce soil erosion and increase infiltration of runoff is dependent on several factors including the amount of rainfall, the grade and height of row ridges, the steepness and length of the slope, the crop residue and surface roughness, and the soil hydrologic group.

Contour farming is an option on slopes between 2% and 10%, with a minimum ridge height of 1 inch, in areas with 10-year rain events less than 6.5 inches/24 hours, **and with a length of slope between 100 and 400 feet**.

In areas with heavier rainfall events, and/or fields with steeper or longer slopes, the function of the ridges to hold back the runoff is lessened and may result in structural failure along the contour. In those cases, the efficacy of this practice is potentially compromised.

Establish and maintain the direction of rows as close to the angle of the contour as possible.

Coupling the practice with reduced tillage practices will result in the best performance of contour farming.

Contour Strip Cropping (on-field)

In contour strip cropping, a field is managed with planned rotations of row crops, forages, small grains, or fallow in a systematic arrangement of equal width strips following the contour across a field. Crops are typically arranged so that a strip of grass or forage crop (low erosional risk because of their fibrous root system) is alternated with a strip of row crop (high erosional risk; e.g., corn). The crops are planted across the slope of the land, as in contour buffer strips. This practice differs from contour buffer strips in that it allows for crops to be planted across 100% of the field area.

Plant row crops on less than half the field and, at a minimum, 50% of the slope must be planted with low erosional risk plants (e.g., grass plants because of their fibrous root system).

The low erosional risk crops reduce erosion, slow runoff water, and trap sediment entering through runoff from upslope areas. This practice combines the benefits of contouring and crop rotation.

Contour strip cropping is not as effective if the row crop strips are too wide and **are an option on slopes of ≤10%**. **Establish and maintain the rows as close to the contour as possible.**

Coupling the practice with reduced tillage practices will result in the best performance of contour strip cropping.

Terrace Farming (on-field)

Terraces are described as a stair-stepping technique of creating flat or nearly flat crop areas along a gradient. They can be constructed as earth embankments or a combination of ridge and channel systems. A terrace is an earthen embankment that is built across a slope to intercept and store water runoff. Some terraces are built level from end to end to contain water used to grow crops and recharge groundwater. Others, known as gradient terraces, are built with some slope or grade from one end to the other and can slow water runoff. Both help to reduce soil erosion by slowing the velocity of runoff and increasing the time for water infiltration. On the field, terraces can be used as a part of an overall system based on the topography of the land. Additionally, an earthen ridge or terrace can be constructed across the slope upgrade from a field area to prevent runoff from entering the area or to direct runoff from one area of production to a common runoff collection area. Reduced tillage practices will result in less sediment loading and the best performance of a terraced farming system.

The ends of terraces, including turnrows, must be structured and maintained to prevent concentrated flow from damaging the function of the terrace. If runoff outflows are necessary, the runoff must be directed to a system such as a grassed waterway, a grade-stabilization structure, a filter strip, water or sediment basin, or other suitable outlet with adequate capacity to handle the runoff and prevent gully formation.

Strip Cropping (on-field)

In strip cropping, a field is managed with planned rotations of row crops, forages, small grains, or fallow in a systematic arrangement of equal width strips. Crops are typically arranged so that a strip of grass or forage crop (low erosional risk because of their fibrous root system) is alternated with a strip of row crop (high erosional risk; e.g., corn). This practice differs from contour strip cropping in that rows do not need to be planted along a contour, which allows strip cropping to be used on land without a contour.

Alternate strips of high-erosion-risk crops and low-erosion-risk crops. A minimum of 50% of the field must be planted with low erosional risk crops or sediment trapping cover.

The low erosional risk crops reduce erosion, slow runoff water, and trap sediment entering through runoff.

Strip cropping is not as effective if the row crop strips are too wide and **must only be implemented on slopes ≤10% slope**.

Coupling the practice with reduced tillage practices will result in the best performance of strip cropping.

No Tillage/Reduced Tillage (on-field)

This category of practices includes conservation tillage practices such as no-till, strip-till, ridge-till, and mulch-till.

Each of these involves year-round management of the amount, orientation and distribution of crop and other plant residue on the soil surface, while limiting the soil-disturbing activities used to grow and harvest crops in systems where the field surface is tilled, raked, or left undisturbed prior to planting. **For each tillage practice below, more than 30% of the surface must remain covered with plant residue.**

- No-till/strip till: In these systems, the soil is left undisturbed from harvest to planting. Planting or drilling is accomplished using disc openers, coulter(s), and row cleaners. Weeds are controlled primarily with crop protection products.
- Strip till: In these systems, the soil is left undisturbed from harvest to planting except for strips up to one-third of the row width. (The strips could involve only residue disturbance or could include soil disturbance.) Planting or drilling is accomplished using disc openers, coulter(s), row cleaners, in-row chisels, or rototillers; cultivation can be used for emergency weed control. Other common terms used to describe strip-till, include row-till, and slot-till.
- Ridge-till: Ridge-till is a system in which seeds are planted into a seedbed prepared by scraping off the top of the ridge. The scraped-off ridge usually provides an excellent environment for planting. Ridges are formed during cultivation of the previous year’s crop. Ridge-till operations consist of planting in the spring and at least one cultivation to recreate the ridges for the next year. Rows remain in the same place each year and any crop residue on the ridges at planting is pushed between the rows.
- Mulch-till: This system uses full-width tillage involving one or more tillage trips, which disturbs the entire soil surface but leaves a uniform layer on crop residue on the soil surface and is done before or during planting. Tillage tools such as chisels, field cultivators, discs, sweeps, or blades are used. Weeds are controlled with crop protection products or cultivation or both.

Vegetative Barriers (on-field)

Vegetative barriers are narrow, permanent strips of stiff-stemmed, erect, tall and dense vegetation established in parallel rows on the contour of fields to reduce soil erosion and sediment transport. These buffers function similar to contour buffer strips and may be especially effective in dispersing concentrated flow, thus increasing sediment trapping and water infiltration. Because the vegetative barrier, typically comprised of grasses, is established on the contour, runoff is restricted, reducing sheet flow and erosion from concentrated flow. The grass slows runoff, helping the water soak into the soil and reducing erosion. The specific recommendations for establishing the vegetative barrier vary from site to site.

Barrier widths are determined by variables such as slope, soil type, field conditions, climate, and erosion potential but **must be a minimum of 3 feet wide**. To ensure maximum performance, the pesticide user **must maintain the integrity of the barrier for the entire width and length, including:**

- The barrier must be harvested, mowed, reseeded, and fertilized as necessary to maintain plant density and vigorous plant growth.
- The maintenance schedule must keep vegetation tall in spring and early summer to help

slow runoff flow, maximize disruption of concentrated flow, and reduce the chance of structural damage.

- Regular maintenance must also include inspection after major storms, removal of trapped sediment, and repair of eroding areas.

Vegetated Ditch Banks (off-field)

A vegetated ditch bank is a sloped channel, planted with vegetation (grass or otherwise) that transports surface water at such a rate that it does not erode soil to an outlet that is not likely to erode.

- The bottom width of the (trapezoidal) vegetated ditch bank must be less than 100 ft.
- The side slope of the vegetated ditch bank must be flatter than a ratio of 2:1 horizontal: vertical.
- The depth/capacity of the vegetated ditch bank must accommodate peak runoff volume expected from a 10-year frequency, 24-hour duration storm.
- Vegetation must be selected such that the vegetation will achieve an adequate density, height, and vigor, and is stable to peak runoff volume expected from the 10-year frequency, 24-hour duration storm.

Maintenance must include ensuring a healthy grassed or vegetative surface within the vegetated ditch bank, inspections after major storms and repair to damaged areas, as well as removal and redistribution of excess sediment back to the field.

Riparian buffers (herbaceous and forest buffers; off-field)

These buffers are similar in that they reduce erosion and, at minimum, maintain water quality. **Vegetation for both buffers must be tolerant to intermittent flooding and saturated soil and be managed until established in the transitional zone between a field and an aquatic habitat. Herbaceous buffers must consist of non-woody vegetation and must have a minimal width of 2.5 times the width of the stream (based on the horizontal distance between bank-full elevations) or 35 feet if adjacent to a larger water body. Forest buffers must be planted to trees and shrubs and must have a minimal width of 35 feet from the waterbody.** Riparian buffers should only be used where channel and stream bank stability is adequate to support this practice.

Management of Surface and Subsurface Water on the Field: Water and Sediment control Basins, Ponds, and Constructed Wetlands

There are several conservation practices that involve management of surface and subsurface water on the field. However, for any of these practices to be an acceptable runoff mitigation strategy, **a sediment basin must be used in conjunction with practices managing surface and subsurface runoff (described below). Growers who wish to use any of these practices must follow all state and local laws and regulations and adhere to any requirements associated with conservation programs in which they are participating.**

Sediment basins: Sediment basins are used to capture runoff (with sediment) leaving the field, such that sediment has adequate time to settle out of the water column. Sediment basins are constructed by creating an embankment, excavating a dugout, or both such that the basin has an outlet. Basins are not stand-alone practices and should be used in conjunction with other runoff/erosion practices like:

- Subsurface drainage: This is a practice where an underground pipe is installed to collect and move excess water from a field.
- Tailwater recovery systems: These systems are intended to collect, move, and temporarily store runoff water so that it can be reused later.
- Drainage water management: This conservation practice involves managing the flow of surface and subsurface drainage systems by changing the elevation of outflow.

Water and sediment control basins: This practice is effective for managing runoff, trapping sediment, and reducing gully erosion. Basins are described as an earthen embankment or basin, or a combination ridge and channel, constructed across the slope of a minor drainage area in a field. Control basins must also have an outlet so that water can be released in a manner that does not lead to damage.

Ponds are similar in function to sediment basins, as they can allow time for the sediment to settle from sediment-laden runoff drained from a field. They are also similar in design to sediment basins but have a dam as an outlet.

Constructed wetlands: Water-tolerant vegetation is used to create a manmade wetland that can provide for the biological treatment of water to improve water quality.

Maintenance of basins and ponds must include the following: ensuring a healthy vegetative surface to maintain the structural integrity of the basin/pond; inspections after major storms, repair to damaged areas, and removal of any obstructions that interfere with flow around inlets; and removal and redistribution of excess sediment back to the field.

Mulching with Natural Materials (on-field)

This practice is used to reduce runoff and erosion. Natural mulches should be applied such that mulch provides **a minimum of 70 percent ground cover**. The **minimum depth of mulch must be 2 inches** such that the mulch will remain during heavy rain or winds. Vegetation-based mulches must have a carbon:nitrogen ratio greater than 20:1. If mulch needs to be held in place, appropriate measures must be used (e.g., tacking, crimping) so that the mulch remains on the field. **The mulch must be periodically inspected to ensure that the mulch is intact and repair/reinstall mulch as needed**.

Alley Cropping (on-field)

Alley cropping is effective at reducing surface water runoff and erosion. This practice involves trees or shrubs being planted in single or multiple rows where other commodities (i.e., agronomic or horticultural crops or forages) are planted in the alleys of the trees or shrubs. **Trees or shrubs must be planted on or near the contour. The vegetation in the alleys must be established in conjunction with the trees/shrubs** to be effective against water erosion. For wind erosion, tree/shrubs must be planted perpendicular to erosive wind patterns. Additionally, the species of trees/shrubs planted must have deep root systems that assist in water infiltration and rapid growth rates. **During the period of establishment, tree/shrubs must be maintained/replaced as needed. Soil erosion must be controlled by vegetative or other means until the alley cropping design is fully functional.**

Attachment 1 National and State Use and Usage Summaries

For the following documents, see the "Attachment 1" folder.

- Carbaryl (056801) National and State Summary Use and Usage Summary
  (February 27, 2020)

- Carbaryl Usage Data Source Summary-Crosswalk
  (February 27, 2020)

- Methomyl (090301) National and State Summary Use and Usage Matrix
  (February 27, 2020)

- Methomyl Usage Data Source Summary-Crosswalk
  (February 27, 2020)

Attachment 2 Supplemental Files

For the following documents, see the "Attachment 2" folder.

- **Risk-plot Tables**: Folders organized by taxa with CSV tables (one per Risk-plot) corresponding to data displayed on the Risk-plots.
- **HUC12 Info**: A folder containing HUC12 lists for species ranges and habitats used to calculate overlap information displayed in the Risk-plots. Requires knowing a species EntityID.
- **Species Info**: A folder with information useful for converting a species to an EntityID number for use with other tables.
- **Toxicity Tables**: A folder with taxa-specific toxicity data displayed on the Risk-plots.
- **UDL Overlaps**: A folder with overlap data by HUC12 and UDL.
- **EEC Files**: A folder with inputs used for the exposure models and resulting EECs.
- **PWC+ Methods and Analysis**: A folder with PWC+ analyses for each of the listed salmonid species, as well as a methods document (MRID 51902803).
- **RiskPlottingCode.zip**: Zipped file including the R code and associated files/folders.

Attachment 3 Species and Habitat Effects Analysis & Integration and Synthesis

For the following documents, see the "Attachment 3" folder.

- Chapter 11. Species Effects Analysis: Carbaryl
- Chapter 12. Species Effects Analysis: Methomyl
- Chapter 13. Critical Habitat Effects Analysis: Carbaryl
- Chapter 14. Critical Habitat Effects Analysis: Methomyl
- Chapter 16. Species Integration and Synthesis: Carbaryl
- Chapter 17. Species Integration and Synthesis: Methomyl
- Chapter 18. Habitat Integration and Synthesis: Carbaryl
- Chapter 19. Habitat Integration and Synthesis: Methomyl

ATTACHMENT 4 EPA LABEL TABLES FOR CARBARYL & METHOMYL

Information on EPA's proposed label changes related to this consultation for carbaryl and methomyl can be reviewed in the attachment folder. The label table for carbaryl was sent from EPA to NMFS on 1-18-24. The label table for methomyl was sent from EPA to NMFS on 1-09-24.