United States Department of the Interior
FISH AND WILDLIFE SERVICE
Washington Fish and Wildlife Office
510 Desmond Dr. S.E., Suite 102
Lacey, Washington 98503

In Reply Refer To:
01EWFW00-2015-F-0251-R002

Daniel McNair
Director, Environmental Readiness Division
Department of the Navy
U.S. Pacific Fleet
250 Makalapa Drive
Pearl Harbor, Hawaii 96860-3131

Dear Mr. McNair:

This letter transmits the U.S. Fish and Wildlife Service's (Service) reinitiated Biological Opinion (Opinion) on the U.S. Navy's (Navy) proposed Northwest Training and Testing program that occurs in the offshore waters of northern California, Oregon, and Washington, the inland waters of Puget Sound, and portions of the Olympic Peninsula, as well as Western Behm Canal in southeastern Alaska and its effects on the bull trout (*Salvelinus confluentus*), designated bull trout critical habitat, the marbled murrelet (*Brachyramphus marmoratus*), and the short-tailed albatross (*Phoebastria albatrus*). The Opinion also addresses the U.S. Forest Service's Special Use Permit for the Navy's Pacific Northwest Electronic Warfare Range activities within the Olympic National Forest. Formal consultation on the proposed actions was conducted in accordance with Section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.) (ESA).

On October 24, 2019, the Service received your request to reinitiate formal consultation on the effects to the bull trout and the marbled murrelet, and for informal consultation on the effects to designated bull trout critical habitat and the short-tailed albatross. On March 11, 2020, the Navy informed the Service, via email, that the Navy revised its determination for short-tailed albatross to a "may affect, likely to adversely affect" determination, and requested reinitiation of formal consultation on that species. On April 3, 2020 the Service informed the Navy that the Service reinitiated formal consultation on March 11, 2020.

The Navy determined that the action will have “no effect” on designated marbled murrelet critical habitat. The determination of “no effect” to listed resources rest with the action agency. The Service has no regulatory or statutory authority for concurring with a “no effect” determination, and no consultation with the Service is required.

The enclosed Opinion is based on information provided in a biological assessment, the Draft, Final and Supplemental Environmental Impact Statements, as well as through information shared through numerous meetings, telephone conversations, letters, and emails, and through other sources cited in the Opinion. A complete record of this consultation is on file at the Service’s Washington Fish and Wildlife Office in Lacey, Washington.

If you have any questions regarding the enclosed Opinion, or our shared responsibilities under the ESA, please contact Curtis Tanner, of my staff, at (360) 753-4326, or curtis_tanner@fws.gov.

Sincerely,

BRADLEY THOMPSON
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Date: 2021.06.30 12:26:19 -07'00'

Brad Thompson, State Supervisor
Washington Fish and Wildlife Office

Enclosure(s)

cc:
DOI SOL, Washington, DC (L. Grimm)
USFWS RO, Portland, OR (L. Salata)
USFWS HQ Washington, DC (D. Laye)
USFWS HQ, Washington, DC (C. Aubrey)

Endangered Species Act – Section 7 Consultation

BIOLOGICAL OPINION

U.S. Fish and Wildlife Service Reference:
01EWFW00-2020-F-0251-R0002

Navy's Northwest Training and Testing Activities

Offshore Waters of Northern California, Oregon, Alaska, and Washington, the Inland Waters of Puget Sound, and Portions of the Olympic Peninsula

Federal Action Agency:

Department of the Navy

Consultation Conducted By:

U.S. Fish and Wildlife Service
Washington Fish and Wildlife Office
Lacey, Washington

BRADLEY
THOMPSON

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BRADLEY THOMPSON
Date: 2021.06.30
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Brad Thompson, State Supervisor
Washington Fish and Wildlife Office

Date

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ACRONYMS AND ABBREVIATIONS

ASW	Anti-Submarine Warfare
BA	Biological Assessment
CFR	Code of Federal Regulations
CI	Confidence interval
DPS	Distinct Population Segment
EOD	Explosive Ordnance Disposal
ESA	Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*)
FMO	Foraging, migration, and overwinter
ft.	foot/feet
HF	High frequency
km2	Square kilometers
LCNEP	Large-caliber non-explosive projectiles
LF	Low frequency
LMNT	Limpet Mine Neutralization Tool
m	meters
MCNEP	Medium-caliber non-explosive projectiles
MF	Mid-frequency
MOA	Military Operations Area
n/a	not applicable
NAVAIR	Naval Air Systems Command
NAVFAC	Naval Facilities Engineering Systems Command
NAVSEA	Naval Sea Systems Command
Navy	U.S. Department of the Navy
NBK	Naval Base Kitsap
NEPM	Non-explosive practice munitions
NEW	Net explosive weight
nm	nautical mile
nm2	nautical mile squared
NWFPEM	Northwest Forest Plan Effectiveness Monitoring
NWTT	Northwest Training and Testing
OCNMS	Olympic Coast National Marine Sanctuary
OEIS	Overseas Environmental Impact Statement
OPAREA	Operating area
Opinion	Biological Opinion
PCBs	Polychlorinated biphenyls
QRS	Quinault Range Site
RPM	Reasonable and Prudent Measure
SCNEP	Small-caliber non-explosive projectiles
Service	U.S. Fish and Wildlife Service
SHA	Safe Harbor Agreement
SPL	Sound pressure levels
SUS	Sound underwater signal
TORP	Torpedo
TS	Threshold shift

ACRONYMS AND ABBREVIATIONS

TTS	Temporary threshold shift
USACE	U.S. Army Corps of Engineers
VHF	Very high frequency
W-237	Warning Area 237
WDNR	Washington State Department of Natural Resources
yd.	yard

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1 INTRODUCTION

This document represents the U.S. Fish and Wildlife Service's (Service) Biological Opinion (Opinion) based on our review of the U.S. Department of the Navy's (Navy) proposed Northwest Training and Testing (NWTT) activities located in the offshore areas of northern California, Oregon, and Washington, the inland waters of Puget Sound, portions of the Olympic Peninsula, as well as part of the Western Behm Canal in southeast Alaska. The Opinion also includes the analysis for the U.S. Forest Service's Special Use Permit for the Navy's Pacific Northwest Electronic Warfare Range activities within the Olympic National Forest. We evaluated the effects of the proposed action on the bull trout (*Salvelinus confluentus*), designated bull trout critical habitat, the marbled murrelet (*Brachyramphus marmoratus*), designated marbled murrelet critical habitat, and the short-tailed albatross (*Phoebastria albatrus*) in accordance with section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*) (ESA).

The Navy has determined that the action will have "no effect" on additional listed species and designated critical habitat. The determination of "no effect" to listed resources rests with the action agency. The U.S. Fish and Wildlife Service (Service) has no regulatory or statutory authority for concurring with a "no effect" determination, and no consultation with the Service is required. The Navy should document their analysis on effects to listed species and maintain that documentation as part of the project file.

On October 24, 2019, the Service received the Navy's request for formal consultation on the effects to the bull trout and the marbled murrelet, and for informal consultation on the effects to designated bull trout critical habitat and the short-tailed albatross. On March 11, 2020, the Navy informed the Service, via email, that they were revising their determinations for short-tailed albatross and bull trout to a "may affect, likely to adversely affect" determination, and were requesting reinitiation of formal consultation on those species. On April 3, 2020 the Service informed the Navy that the Service reinitiated formal consultation on March 11, 2020.

This Opinion is a reinitiation of a 2018 Opinion (USFWS 2018), which was a reinitiation of a 2016 Opinion (USFWS 2016). Both the prior Opinions were based on information from: the January 2015 Biological Evaluation, the January 2014 Draft Environmental Impact Statement (DEIS), the December 2014 Supplement to the EIS, the October 2015 Final Environmental Impact Statement, the September 2014 Final Environmental Assessment for the Pacific Northwest Electronic Warfare Range, numerous meetings, telephone conversations and emails, as well as from other sources of information as detailed below. While we incorporate by reference and utilize information contained in the 2016 and 2018 Opinions, this Opinion supersedes and replaces the 2018 Opinion which superseded and replaced the 2016 Opinion. A complete record of this consultation is on file at the Service's Washington Fish and Wildlife Office in Lacey, Washington.

2 CONSULTATION HISTORY

The following is a summary of important events associated with this consultation:

- The Service issued an Opinion on the Navy's proposed NWTT program on July 21, 2016.
- On July 24, 2017, the Service received a request from the Navy to reinitiate consultation on the NWTT program due to the Navy's proposal to move certain training and testing activities further offshore as a conservation measure.
- The Service issued a second Opinion on the Navy's proposed NWTT program on December 11, 2018. This 2018 Opinion incorporated by reference much of the analysis from the 2016 Opinion, but revised the analysis to address training activities that were moved farther from shore. This 2018 Opinion superseded the 2016 Opinion.
- On October 24, 2019, the Service received an email from the Navy requesting reinitiation of formal consultation on the NWTT program, due to changes to some activities as well as the addition of activities.
- On November 26, 2019, the Navy informed the Service of revisions to the proposed action and that additional information would be forthcoming.
- On December 2, 2019, the Navy provided a revised version of the deconstruction matrix identifying the action's component activities and stressors.
- On December 17, 2019, the Service requested additional information from the U.S. Navy regarding range to effect estimates for underwater explosives and the Navy's effect determination for short-tailed albatross.
- On January 30, 2020, the Navy provided a portion of the additional information regarding range-to-effect estimates requested by the Service as well as a revised version of the deconstruction matrix.
- On February 2, 2020, the Navy provided the remainder of the additional information regarding their determination for short-tailed albatross, as requested by the Service.
- On February 27, 2020, the Navy provided new information to the Service regarding bull trout presence in the offshore area.
- On March 10, 2020, the Navy and the Service held a conference call to discuss the Navy's "not likely to adversely affect" determination for short-tailed albatross and the schedule for the NWTT consultation.

- On March 11, 2020, the Service received an email from the Navy stating that, as a result of the March 10, 2020 conference call, they had revised their determinations for short-tailed albatross and bull trout to "likely to adversely affect" and were requesting formal consultation on NWTT activities as of March 11, 2020.

- On April 3, 2020, the Service notified the Navy, via letter, that they had received the Navy's revised determinations for short-tailed albatross and bull trout, and had sufficient information as of March 11, 2020 to reinitiate formal consultation on the NWTT program.

- On November 16, 2020, subject matter experts (SME) from the Service and the U.S. Navy met for SME Meeting #1 – Proposed Action, to clarify details of the proposed action and how those details would be used in the exposure analysis.

- On November 19, 2020, subject matter experts from the Service and the Navy met for SME Meeting #2 – Marbled Murrelet Density Data Coordination, to discuss the Service's data sources and analytical assumptions regarding marbled murrelet population densities at sea.

- On November 23, 2020, subject matter experts from the Service and the Navy met for SME Meeting #3 – RTE Coordination, to discuss the Navy's proposed updates to ranges to effect for underwater explosives.

- On November 30, 2020, subject matter experts from the Service and the Navy met for SME Meeting #4 – Modeling Deep Dive, in which the Service presented the modeling methods used to analyze the demographic consequences of the action for marbled murrelets in Conservation Zone 2.

- On April 1, 2014, the Service provided the Navy a draft NWTT Biological Opinion for review.

- On April 14, 2021, the Navy provided comments to the Service on the draft NWTT Biological Opinion.

3 CONCURRENCES

We previously concurred in the Navy's determination that its proposed action may affect, but was not likely to adversely affect western snowy plover (*Charadrius nivosus nivosus*), streaked horned lark (*Eremophila alpestris strigata*), northern spotted owl (*Strix occidentalis caurina*), and designated bull trout critical habitat. See the 2016 Opinion (USFWS 2016, pp. 3-19), reaffirmed in the 2018 Opinion. The Navy made the same determinations for these species in this consultation. Based on our review of the updated proposed action, we find that the changes in the proposed action do not create new or different effects than those we analyzed and described in 2016 for western snowy plover, streaked horned lark, and northern spotted owl, and so we affirm our concurrence with the Navy's determinations for these species.

The action area contains designated critical habitat for bull trout. Our analysis of the effects of implementation of the NWTT program on designated bull trout critical habitat on pages 17-19 of the 2016 Opinion and unchanged in the 2018 Opinion remains valid and is incorporated here by reference. Based on that analysis, and after considering the proposed changes to the NWTT program described in the BA, the Service concurs with the Navy's determination that the proposed action is not likely to adversely affect designated critical habitat for the bull trout. Therefore, the proposed action is not likely to destroy or adversely modify critical habitat for the bull trout.

4 BIOLOGICAL OPINION

5 DESCRIPTION OF THE PROPOSED ACTION

A federal action means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies in the United States or upon the high seas (50 CFR 402.02). The proposed action includes activities that were consulted on previously and will continue, alterations to those activities, and new activities. While this Opinion will focus on the new and altered activities, the proposed action is the entire NWTT program and the Service will consider the context of the larger NWTT program in our jeopardy analysis within this Opinion. Therefore, because the duration of the action addressed in the 2016 Opinion was 20 years and the 2016 Opinion was signed on July 21, 2016, thus addressing the program through July 21, 2036, this Opinion will evaluate the on-going effects of the NWTT program for another 16 years, through July 21, 2036. At that time, the Navy would need to re-initiate consultation regarding continuation of the program. Although we are addressing the effects of the program assuming its continuation through 2036, given the history of changes in the program to date, we expect the Navy will likely have updates and changes in the program before 2036, and the Navy and the Service will consider at that time whether reinitiation of consultation is required. The following sources were relied upon to characterize the proposed action: the Northwest Training and Testing 2019 Draft Supplemental EIS-OEIS Volume 1, the October 2019 Biological Assessment, and information provided by the Navy in meetings, telephone conversations and emails.

Activities in the proposed action will occur in both the "offshore area" and "inland waters." The "offshore area" includes, in part, the air, surface, and subsurface operating areas of the Navy's offshore activities extending west from the coastline of Washington, Oregon, and Northern California to a distance of approximately 250 nautical mile (nm) in to international waters (Navy 2015, p. 2-5). The offshore area includes the coastline along the Washington coast beneath the airspace of Warning Area 237 (W-237) and the Washington coastline north of the Olympic Military Operations Areas (MOA). There is no ceiling to the airspace of the offshore area except for that described below for the Special Use Airspace.

The offshore area also includes the northern Pacific Ocean extending from the coast of Washington to the south shores of the Aleutian Islands of Alaska. This part of the action area is defined by the furthest extent that debris generated by the action will float. Main Pacific Ocean currents travel east and split along the western coast of North America. The northern Alaska current could carry material to the Subarctic Gyre and the southern California Current carries material to the North Pacific Gyre. Military debris from the Navy training and testing activities can travel both north and south. The potential effects of this aspect of the proposed action are included in this Opinion and were also discussed and analyzed in the 2016 Opinion [USFWS 2016, pp. 222-227].

The "inland waters" includes air, sea, and undersea space inland of the Pacific coastline, from buoy "J" at 48° 29.6 N, 125° W eastward, including the Strait of Juan de Fuca and Puget Sound. Within the Inland Waters are specific geographic components in which most Inland Waters training and testing occur.

As noted above, the proposed action includes those activities involving in-water acoustic or explosive sources of stressors, activities that changed since the 2018 Opinion, and new activities. In the tables below, the activities are grouped into categories based on changes in the proposed action from the 2018 Opinion: decreased activities, increased activities, activities with new components, and new activities (Note: see pgs. vi-vii of this document for a list of acronyms).

Table 1. Decreased activities, training

Activity	Change from 2018 Opinion
Air combat maneuvers (offshore area (W-237))	W-237 decreasing from 550 to 126 events per year
Gunnery exercises (surface-to-air)	Decreased from 160 to 125 events per year; Medium-caliber non-explosive rounds decreased from 9,672 to 9,660 per year and explosive rounds 6,320 to 0 per year. Large-caliber explosive rounds decreased from 230 to 0 per year. Clarified allocation of different medium-caliber rounds: 7% 40mm, 53% 25mm, 40% 20mm Adjusted area of effect for 40mm rounds since they do not create a bow shock wave. 90% of gunnery exercises occur off the Washington coast.
Missile exercises (air-to-air)	Decreased from 24 to 4 events per year
Anti-submarine warfare, helicopter tracking exercise	Decreased from 4 to 2 events per year. Decreased MF4 hours from 4 to 1. Decreased subsurface targets from 4 to 2. Reduced MF5 Sonobuoys used from 16 to 8.

Activity	Change from 2018 Opinion
Anti-submarine warfare, ship tracking exercise	Decreased from 65 to 62 events per year. Decreased MF1 hours from 141 to 117, HF6 from 80 hours to 0 hours, and Bathythermographs and targets from 65 to 62.
Anti-submarine warfare, submarine tracking exercise	Decreased HF6 from 112 hours to 0 hours
Anti-submarine warfare, maritime patrol aircraft tracking exercise	Decreased ASW2 sonobuoys from 720 to 350
Electronic warfare training, ship	Decreased from 275 to 220 events per year
Surface warfare, bombing exercise (air-to-surface)	Explosive bombs decreased from 10 E12 per year to 2 E10 per year, non-explosives decreased from 110 to 84 per year
Gunnery exercise, surface-to-surface	Non-explosive small-caliber rounds decreased from 121,200 to 121,000 per year, non-explosive medium-caliber rounds decreased from 33,492 to 16,750 per year
Mine warfare, Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises	Decreased HF4 from 384 hours to 65 hours
Submarine Sonar Maintenance	MF3 hours decreased offshore from 11 to 2 hours
Missile exercise, air-to-surface	Decreased from 4 E10 explosives to 2 per year
Maritime security operations	Decreased from 286 to 220 annual events
Personnel Insertion/ Extraction, non-submersible	Decreased from 10 to 6 annual events
Search and rescue	Decreased from 100 to 80 annual events

Table 2. Decreased NAVSEA activities, testing

Activity	Change from 2018 Opinion
Anti-submarine warfare testing	NEPM torpedoes decreased from 16 to 8 per year, TORP1 torpedoes decreased from 16 to 8 per year
Countermeasure testing (offshore)	NEPM torpedoes decreased from 123 to 12 per year, ASW3 hours decreased from 360 to 24, ASW4 sonobuoys decreased from 1,048 to 360, TORP2 torpedoes decreased from 67 to 12 per year, HF5 hours decreased from 360 to 0
Countermeasure testing (inland waters)	TORP1 torpedoes decreased from 88 to 0
Pierside sonar testing	HF1 hours decreased from 161 to 0, HF3 hours decreased from 8 to 1, M3 hours decreased from 1 to 0, MF3 hours decreased from 161 to 24, MF10 hours decreased from 60 to 7, MF9 hours decreased from 200 to 69
Torpedo (explosive) testing	TORP1 torpedoes decreased from 12 to 8, TORP2 torpedoes decreased from 18 to 8
Torpedo (non-explosive) testing (offshore)	Annual events decreased from 23 to 22
Mine Detection and Classification Testing (inland waters)	Annual events decreased from 54 to 42
Unmanned Aerial System Testing (offshore)	Annual events decreased from 20 to 2
Unmanned Surface Vehicle Testing (offshore)	Annual events decreased from 20 to 4
Unmanned Underwater Vehicle Testing (offshore)	NEPM torpedoes decreased from 27 to 24
Unmanned Underwater Vehicle Testing (inland waters)	NEPM torpedoes decreased from 107 to 72, M3 hours decreased from 766 to 256
Acoustic Component Testing	Annual events decreased from 60 to 45 LF5 hours reduced from 60 to 0 MF8 hours reduced from 40 to 0 SD1 hours reduced from 757 to 0
Cold Water Support (offshore)	DCS 1 Annual events decreased from 20 to 0
Cold Water Support (inland waters)	Annual events decreased from 65 to 4
Post-refit Sea Trial	Annual events decreased from 32 to 30
Semi-stationary Equipment Testing	Annual events decreased from 176 to 120, LF4 hours decreased from 110 to 88, MF9 hours decreased from 140 to 138

Table 3. Decreased NAVAIR activities, testing

Activity	Change from 2018 Opinion
Anti-submarine Warfare, Tracking Test, Maritime Patrol Aircraft	Annual events decreased from 49 to 4. MF5 sonobuoys decreased from 170 to 80, ASW5 sonar hours decreased from 64 to 40, and E4 sonobuoys decreased from 70 to 0.
Anti-submarine warfare, Tracking Test, Maritime Patrol Aircraft (SUS)	Annual events decreased from 5 to 4,

Table 4. Increased activities, training

Activity	Change from 2018 Opinion
Air combat maneuvers (offshore area (Olympic MOA))	Annual events increased from 550 to 700 total in offshore area (Olympic MOA increasing from 550 to 574)
Anti-submarine warfare, maritime patrol aircraft	Annual events increased from 324 to 373, MF5 sonobuoys from 896 to 926
Anti-submarine warfare, surface ship tracking exercise	Increase in ASW3 from 78 hours to 86 hours.
Gunnery exercise, surface-to-surface	Explosive medium-caliber rounds (E1) increased from 48 to 120, explosive large-caliber rounds increased from 80 to 112 per year Clarified allocation of different medium-caliber rounds: 7% 40mm, 53% 25mm, 40% 20mm Adjusted area of effect for 40mm rounds since they do not create a bow shock wave. 90% of gunnery exercises occur off the Washington coast.
Precision anchoring	Annual events increased from 10 to 40
Submarine sonar maintenance	1 hour of LF5 per year
Surface ship sonar maintenance (offshore area)	MF1 hours increased from 14 to 44 hours

Table 5. Increased NAVSEA activities, testing

Activity	Change from 2018 Opinion
Anti-submarine warfare testing:	Annual events increased from 13 to 44, ASW1 hours increased from 16 to 80, MF4 hours increased from 10 to 40, MF10 hours increased from 4 to 96, MF11 hours increased from 34 to 48, MF12 hours increased from 24 to 80, MF5 sonobuoys increased from 40 to 80
Pierside sonar testing	Annual events increased from 67 to 99
Torpedo (explosive) testing:	Annual events increased from 3 to 4, explosive torpedoes increased from 6 to 8, NEPM torpedoes increased from 6 to 16
Torpedo (non-explosive) testing (offshore)	NEPM torpedoes increased from 119 to 146, ASW3 hours increased from 4 to 177, ASW4 sonobuoys increased from 136 to 248, MF5 sonobuoys increased from 63 to 118, MF10 hours increased from 20 to 24, TORP1 torpedoes increased from 34 to 78
Torpedo (non-explosive) testing (inland waters)	Annual events increased from 41 to 61, NEPM torpedoes increased from 189 to 358, TORP1 torpedoes increased from 42 to 128, TORP2 torpedoes increased from 147 to 224
Unmanned Underwater Vehicle Testing (offshore)	Annual events increased from 28 to 39
Unmanned Underwater Vehicle Testing (inland waters)	Annual events increased from 253 to 379, SAS2 hours increased from 798 to 1,312, TORP1 torpedoes increased from 67 to 72
Cold Water Support (inland waters)	HF6 hours increased from 384 to 707
Non-acoustic Component Testing (offshore area)	Annual events increased from 6 to 8
Non-acoustic Component Testing (inland waters)	Annual events increased from 74 to 75
Post-refit Sea Trial	M3 hours increased from 608 to 736, MF10 hours increased from 79 to 759
Semi-stationary Equipment Testing	HF6 hours increased from 457 to 459, VHF2 hours increased from 35 to 135

Table 6. Increased NAVAIR activities, testing

Activity	Change from 2018 Opinion
Anti-submarine warfare, Tracking Test, Maritime Patrol Aircraft	ASW2 sonobuoys increased from 170 to 200, MF6 sonobuoys increased from 12 to 20
Anti-submarine warfare, Tracking Test, Marine Patrol Aircraft (SUS)	E1 SUS buoys increased from 0 to 8

Table 7. Activities with new component, training

Activity	Change from 2018 Opinion
DCS 2 Mine Warfare, Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises	561 hours of SAS2 per year
Gunnery exercise, surface-to-surface	130 E2 per year
Submarine Sonar Maintenance	Offshore only LF5 1 hour per year
Anti-submarine warfare, Tracking, Maritime Patrol Aircraft	NEPM Torpedoes 16 per year and ASW5 50 sonobuoys per year

Table 8. NAVSEA Activities with new component, testing

Activity	Change from 2018 Opinion
Anti-submarine warfare testing	40 ASW2 sonobuoys, 40 ASW5 hours, 48 MF1K hours
Countermeasure testing (offshore)	24 HF8 hours, 4 MF1 hours
Countermeasure testing (inland waters)	24 ASW3 hours, 720 ASW4 sonobuoys
Pierside sonar testing	1 ASW3 hours, 6 MF1 hours, 32 MF2 hours, 20 MF12 hours
Torpedo (explosive) testing	1 ASW3 hours, 2 HF1 hours, 18 HF6 hours, 1 MF1 hours, 1 MF3 hours, 1 MF4 hours, 22 MF5 sonobuoys, 16 MF6 sonobuoys
Torpedo (non-explosive) testing (offshore)	2 HF1 hours, 72 HF5 hours, 25 HF6 hours, 9 MF1 hours, 3 MF3 hours, 1 MF4 hours, 24 MF6 sonobuoys, 112 MF9 hours
Torpedo (non-explosive) testing (inland waters)	28 HF6 hours, 1 LF4 hours, 6 TORP3 torpedoes

Activity	Change from 2018 Opinion
Mine Detection and Classification Testing (inland waters)	32 BB1 hours, 32 BB2 hours, 32 LF4 hours, 960 HF4 hours
Unmanned Underwater Vehicle Testing (offshore)	8 FLS2 hours, 30 HF5 hours, 24 TORP1 torpedoes, 60 VHF1 hours
Unmanned Underwater Vehicle Testing (inland waters)	16 FLS2 hours, 130 HF5 hours, 20 HF9 hours, 260 VHF1 hours
Post-refit Sea Trial	89 HF9 hours
Semi-stationary Equipment Testing	140 HF9 hours

Table 9. New activities, training

Activity	Description	Category
Anti-submarine warfare, torpedo exercise submarine (non-explosive)	2 annual events 2 NEPM Torpedoes MF3 1 hour HF1 .5 hour	Categorized in 2015 as part of Sinking Exercise, now classified as training activity
Unmanned underwater vehicle training	60 annual events FLS2 240 hours M3 30 hours	Categorized in 2015 as testing activity, now classified as training activity

Table 10. New NAVSEA activities, testing

Activity	Description
At-sea sonar testing (offshore area)	4 annual events, 24 ASW3 hours, 6 HF1 hours, 24 HF5 hours, 96 M3 hours, 8 MF3 hours
At-sea sonar testing (inland waters)	6 max annual events, 144 ASW3 hours, 144 HF5, 24 TORP1, 24 NEPM torpedo MEM
Mine Countermeasure and Neutralization Testing (offshore area)	3 annual events (2 of which involve the use of sonar and explosives); 2 multi-day events (1-10 days per event), includes 225 HF4 hours, up to 36 E4 explosives, 5 E7 explosives per year, and a maximum of 108 E4 explosives and 15 E7 explosives over a 7-year period.
Mine Countermeasure and Neutralization Testing (inland waters)	3 annual events, 675 hours HF4
Mine Detection and Classification Testing (new in offshore area)	1 annual event: 16 BB1 hours, 16 BB2 hours, 16 LF4 hours
Kinetic Energy Weapon Testing (new activity)	4 annual events, 80 kinetic energy (in-air) explosives, 160 NEPM large-caliber projectiles
Propulsion Testing	10 annual events

Activity	Description
Undersea Warfare Testing	12 annual events, 540 ASW3 hours, 60 ASW4 sonobuoys, 8 HF4 hours, 149 MF1 hours, 9 MF4 hours, 373 MF5 sonobuoys, 172 MF6 sonobuoys, 288 MF9 hours, 18 TORP1 torpedoes, 60 TORP2 torpedoes, 78 NEPM torpedoes
Vessel Signature Evaluation (new in inland waters)	1 annual event
Acoustic and Oceanographic Research (offshore)	1 annual event, 10 LF4 hours, 10 MF9 hours
Acoustic and Oceanographic Research (inland waters)	3 annual events, 30 LF4 hours, 30 MF9 hours
Radar and Other Systems Testing (offshore)	55 high-energy laser weapons testing and laser-based optical communication systems annual events
Radar and Other Systems Testing (inland waters)	8 laser-based optical communications systems annual testing events
Simulant Testing	50 annual events

5.1 Description of New Activities

The following sections contain descriptions of new activities under the current proposed action.

5.1.1 Anti-Submarine Warfare, Torpedo Exercise Submarine (non-explosive) and Unmanned Underwater Vehicle Training

The actions associated with these two activities were categorized as testing activities in the 2016 and 2018 Opinions. Under the proposed action for this Opinion, both activities (and the actions they include) will be carried out as both training and testing activities in similar manners. Overall, non-explosive torpedo exercises will increase from 281 to 478 annual events and unmanned underwater vehicle exercises will increase from 64 to 85 annual events.

5.1.2 At-sea Sonar Testing

The actions associated with at-sea sonar testing were classified as training activities in the 2016 and 2018 Opinions. Under the proposed action for this Opinion, both activities (and the actions they include) will be carried out as both training and testing activities in similar manners, in both the offshore area and inland waters. The annual amounts of acoustic hours associated with this activity are listed in Table 10.

5.1.3 Mine Countermeasure and Neutralization Testing

Mine countermeasure and neutralization testing involves the use of air, surface, and subsurface vessels to neutralize mines and mine-like objects that pose a threat. This testing activity will occur in both the offshore area and inland waters portion of the action area, with three events per year in each area (two of which involve the use of sonar and explosives (explosives will not be used in the inland waters area)) and each event occurring over a maximum of ten days. All actions associated with this activity will occur only during daytime hours.

5.1.3.1 Offshore Area

Mine countermeasure and neutralization testing activities will occur closer to shore than other activities previously analyzed in the 2016 and 2018 Opinions that involve the use of in-water explosives in the offshore area. Mine countermeasure and neutralization testing involving the use of explosives will consist of two events per year, in waters 3 nm or greater from shore at the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nm or greater from shore elsewhere in the offshore area (note: mine countermeasure and neutralization testing will not occur off the California coast). Explosives will only be used in the water column (i.e., not on the sea floor), with E4 explosives used at approximately 20 ft. or greater below the water surface and half greater than 4.6 nm from shore and half at least 3 nm from shore; and E7 explosives used at approximately 100 ft. or greater below the water surface and at least 7.5 nm from shore. The maximum depth for explosive testing will be approximately 1,000 ft.

5.1.3.1.1 Conservation Measures

The Navy will not conduct explosive mine countermeasure and neutralization testing within 350 yd. of live hard bottom, artificial reefs, and shipwrecks (Seafloor Resource Mitigation Areas).

Explosives will only be used in the daytime in Beaufort Sea State of less than or equal to 3.

Explosive testing will not be conducted within the Olympic Coast National Marine Sanctuary Mitigation Area (continuation from the 2016 and 2018 USFWS Opinions).

During explosive Mine Countermeasure and Neutralization Testing, the Navy will not use explosives in bin E7 closer than 7.5 nm from shore in the Quinault Range Site, and at least half of E4 explosions will occur farther than 4.6 nm from shore.

Within the Juan de Fuca Eddy Marine Species Mitigation Area, the Navy will not conduct explosive Mine Countermeasure and Neutralization Testing activities.

5.1.3.1.2 Inland Waters

Mine countermeasure and neutralization testing activities will not involve the use of explosives in the inland waters area. Testing in inland waters will involve nonexplosive aspects of mine countermeasure and neutralization testing, including the placement of nonexplosive targets and the operation of unmanned underwater vehicles and associated systems and sensors. Non-explosive mine countermeasure and neutralization testing will occur in the following portions of the inland waters area: NBK Bremerton, Carr Inlet Operations Area, Crescent Harbor Explosive Ordnance Disposal (EOD) Range, Dabob Bay Range Complex, Hood Canal EOD Range, Naval Station Everett, Keyport Range Site, Naval Magazine Indian Island, and NAVY 3 OPAREA. Manned aircraft will not be used in the inland waters except within the area above Navy 3 OPAREA and operate per FAA regulations.

5.1.3.1.3 Conservation Measures

The Navy will not place mine shapes, anchors, or mooring devices (except in designated areas) within 350 yd. of live hard bottom, artificial reefs, and shipwrecks (Seafloor Resource Mitigation Areas).

5.1.4 Mine Detection and Classification Testing (new in offshore area)

Mine detection and classification testing activities have not previously been conducted in the offshore area, and therefore are a new activity in this Opinion. The activities may occur anywhere within the Quinault Range Site.

Mine detection and classification testing activities involve the use of high frequency or very high frequency sonars to locate mines and other small objects. Higher frequencies allow for greater resolution and, due to their greater attenuation, are most effective over shorter distances. Mine detection sonar can be deployed (towed or vessel hull-mounted) at variable depths on moving platforms (ships, helicopters, or unmanned vehicles) to sweep a suspected mined area. Sonars used for imaging are usually used in close proximity to the area of interest, such as pointing downward near the seafloor.

Mine detection and classification testing events will occur once per year. Each event will occur over a maximum of 24 days, with up to 12 hours of acoustic activity each day. All activities will occur during daylight hours only.

Mine-like targets and temporary anchored devices may be deployed for the duration of a single test event or may be left in place for up to 12 months to support multiple events; all devices and their anchors will be recovered. Bottom anchors will not be deployed in known sensitive shallow water benthic habitats such as eelgrass beds.

5.1.4.1 Conservation Measures

The Navy will not place mine shapes, anchors, or mooring devices (except in designated areas) within 350 yd. of live hard bottom, artificial reefs, and shipwrecks (Seafloor Resource Mitigation Areas).

5.1.5 Kinetic Energy Weapon Testing

The kinetic energy weapon will be tested aboard surface vessels, firing explosive and non-explosive projectiles at air- or sea-based targets. The system uses stored electrical energy to accelerate the projectiles, which are fired at supersonic speeds over great distances. Explosive rounds are designed to detonate above the surface target.

Testing events will occur 4 times per year, with each event lasting one day. Up to 25 percent of testing may occur at night. An annual total of 80 kinetic energy (in-air) explosives and 160 NEPM large-caliber projectiles will be used. One target will be expended per event.

5.1.5.1 Conservation Measures

In the Marine Species Coastal Mitigation Area, the Navy will conduct explosive large-caliber exercises greater than 50 nm from shore, and non-explosive large-caliber exercises greater than 20 nm from shore.

5.1.6 Propulsion Testing

Propulsion testing involves ships running at high speeds in various formations. Surface ships may not travel in a straight line. Surface ships will operate in the offshore area at least 10 nm from shore, across the full spectrum of capable speeds.

Propulsion testing events will occur up to 10 times per year, with each event lasting up to 5 days (surface ships will not be conducting test constantly for the entire duration). Up to 50 percent of propulsion testing activities could occur at night.

5.1.7 Undersea Warfare Testing

Undersea warfare testing involves ships demonstrating the capabilities of countermeasure systems and underwater surveillance, weapons engagement, and communications systems in order to test ships' ability to detect, track, and engage undersea targets.

Undersea warfare testing events will occur up to 12 times per year, with each event lasting up to 10 days (ships will not be conducting tests constantly over the 10 day period). The annual amounts of sonar hours, sonobuoy hours, and torpedoes associated with this activity are listed in Table 10. Undersea warfare testing events will take place in the offshore area and Quinault Range Site, at depths greater than 300 ft.

5.1.7.1 Conservation measures

In the Olympic Coast National Marine Sanctuary, 20 nm from shore in the Marine Species Coastal Mitigation Area, and the Juan de Fuca Eddy Marine Species Mitigation Area, the Navy will limit MF1 hours to 33 hours combined annually for testing.

5.1.8 Vessel Signature Evaluation Testing

Vessel signature evaluation testing had not previously occurred in the inland waters area, and therefore is a new activity in this Opinion. This activity involves assessments of surface ship, submarine, and auxiliary system signatures, which includes electronic, radar, acoustic, infrared and magnetic signatures.

Vessel signature evaluation testing events will occur once per year, with each event lasting up to 20 days. Fifty percent of activities may occur at night.

5.1.9 Acoustic and Oceanographic Research Testing

Acoustic and oceanographic research testing activities will occur in both the offshore and inland waters areas, and will involve research using active transmissions from sources deployed from ships, aircraft, and unmanned underwater vehicles. Research sources can be used as proxies for current and future Navy systems.

In both the offshore and inland water areas, testing will occur during daytime hours only. In the offshore area (Quinault Range Site), testing events will occur once per year, with each event lasting up to 14 days. In the inland waters area (DBRC, Keyport Range Site), testing events will occur three times per year, with each event lasting up to 14 days. The annual amounts of sonar hours for this activity are listed in Table 10.

5.1.10 Radar and Other Systems Testing

The radar and other systems testing activity contains a new component—high energy laser weapons testing—that was not analyzed in either the 2016 and 2018 Opinions. All other components of the activity were analyzed in the 2016 and 2018 Opinions. Those analyses are incorporated into this Opinion by reference.

High-energy laser weapons will be employed from surface ships, helicopters, manned or unmanned underwater vehicles and will involve the use of directed energy at small surface and airborne targets. The high-energy laser will be used at short ranges (i.e., line-of-sight).

High-energy laser weapons testing will occur 54 times per year, with each event typically lasting 12 hours per day over a seven day period. Up to 25 percent of testing may occur at night. All high-energy laser weapons testing will occur in the offshore area, greater than 12 nm from shore, only in open-ocean locations (i.e., not close to land areas).

The following safeguards will be employed on high-energy laser platforms in order to reduce the probability of the laser striking the water:

- The high energy laser platform will have provisions that prevent misfiring (i.e., firing when not intended) that will all but eliminate the possibility of misfire and ensure that the system will only fire when the operator pulls the trigger.

- The high-energy laser platforms will have built-in constraints that only permit firing when it is locked onto a target, and automatically interrupts firing if the target track on a target is lost.
- The operators will be trained to stop firing when the laser aim point moves off of the selected target.

The Navy will also test a high-energy laser-based optical communication system. During one event each year, either in inland waters or the offshore area, the Navy will emit lasers through seawater to contact a receiver.

5.1.11 Simulant Testing

Simulant testing involves testing the capability of surface ship defense systems to detect and protect against chemical and biological attacks. Only chemical simulants with low toxicity to humans and the environment, (e.g., glacial acetic acid and triethyl phosphate) and Biosafety Level 1 organisms (e.g., spore-forming bacteria, non-spore-forming bacteria, the protein ovalbumin, MS2 bacteriophages, and the fungus *Aspergillus niger*) will be used during this activity. Simulant testing will occur up to 50 times per year in the offshore area, at least 3 nm from shore.

5.2 Standard Operating Procedures

The standard operating procedures in the proposed action that are relevant to the species and habitats considered in this Opinion are identical to those consulted on in the 2018 and 2016 Opinions (Navy 2019, p. 2-29), and are incorporated here by reference. Additionally, the proposed action includes new standard operating procedures for target deployment and retrieval safety procedures, and underwater detonation safety procedures that were not part of the 2016 proposed action.

5.3 Conservation Measures

The conservation measures in the proposed action that are relevant to the species and habitats considered in this Opinion are identical to those consulted on in the 2018 and 2016 Opinions, which are incorporated here by reference, with the following exceptions (and those described for each new activity category in the sections above):

- After the completion of activities involving medium-caliber projectiles, the Navy will, when practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred, and if any injured or dead ESA-listed species are observed, follow established incident reporting procedures.
- The Navy will not conduct explosive activities (with the exception of explosive mine countermeasure and neutralization testing activities) within 50 nm from shore within the marine species coastal mitigation area.

6 ACTION AREA

The proposed action occurs within, and does not expand, the action area analyzed in the 2016 and 2018 Opinions. The action area remains based on the geographic extent of underwater and in-air sound and the distance that floating debris (specifically plastics), generated by the proposed action, will travel. Note: Figure 1, below, depicts the locations of proposed training and testing activities, and not the entire action area. See the 2016 Opinion (USFWS 2016, pp. 48-53) for a more detailed description.

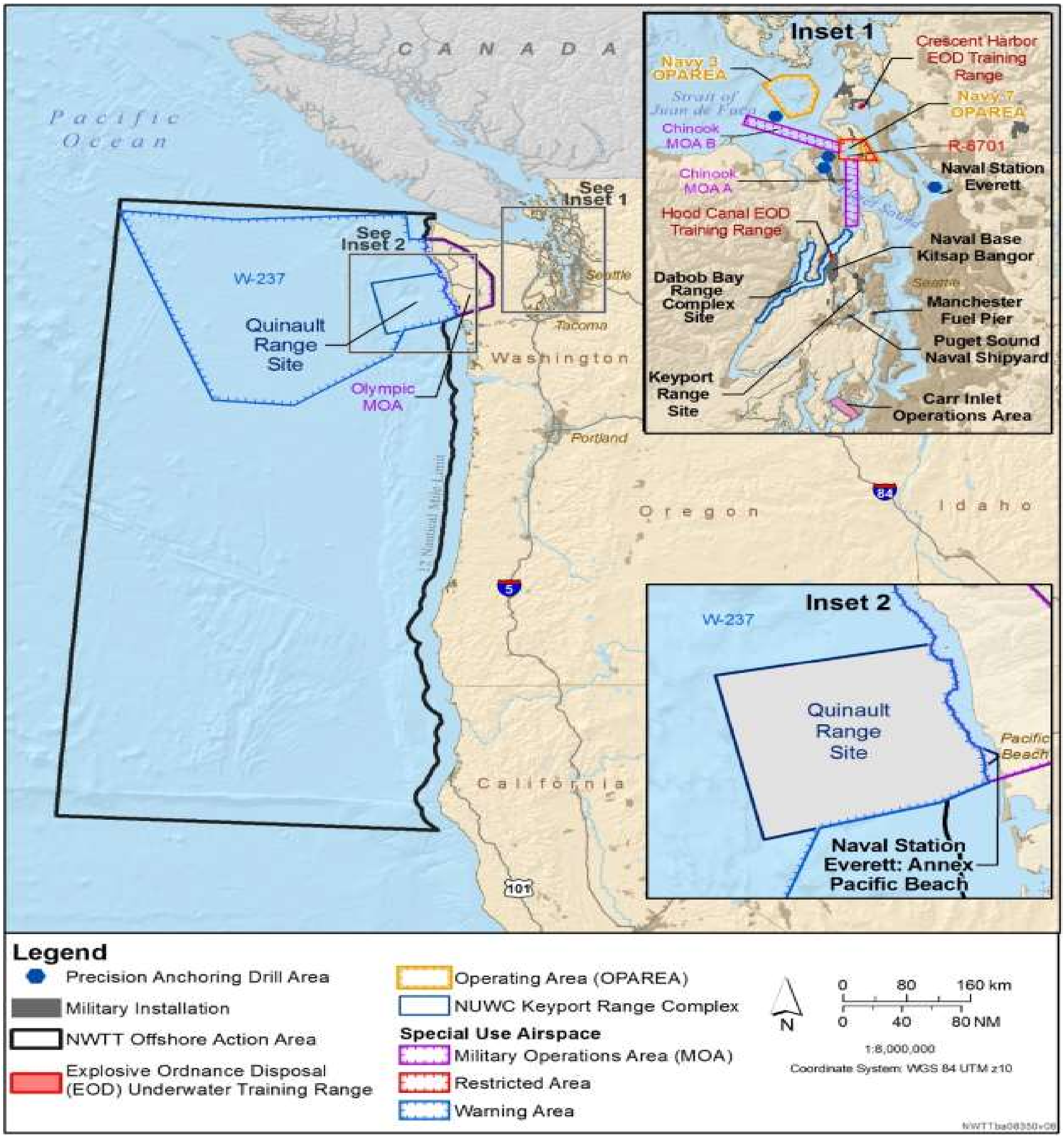

Figure 1. Proposed Training and Testing Locations as depicted by the Navy in their Biological Assessment.
(Navy, 2019)

7 ANALYTICAL FRAMEWORK FOR THE JEOPARDY DETERMINATIONS

7.1 Jeopardy Determination

In accordance with regulation (see 84 FR 44976), the jeopardy determination in this Opinion relies on the following four components:

1. The *Status of the Species*, which evaluates the species' range-wide condition relative to its reproduction, numbers, and distribution, the factors responsible for that condition, and its survival and recovery needs; and explains if the species' current range-wide population is likely to persist while retaining the potential for recovery or is not viable;

2. The *Environmental Baseline*, which evaluates the condition of the species in the action area relative to its reproduction, numbers, and distribution absent the consequences of the proposed action; the factors responsible for that condition; and the relationship of the action area to the survival and recovery of the species;

3. The *Effects of the Action*, which evaluates all future consequences to the species that are reasonably certain to be caused by the proposed action, including the consequences of other activities that are caused by the proposed action, and how those impacts are likely to influence the conservation role of the action area for the species; and

4. *Cumulative Effects*, which evaluates the consequences of future, non-federal activities reasonably certain to occur in the action area on the species, and how those impacts are likely to influence the survival and recovery role of the action area for the species.

In accordance with policy and regulation, the jeopardy determination is made by evaluating the consequences of the proposed federal action in the context of the species' current range-wide status, taking into account any cumulative effects, to determine if implementation of the proposed action is likely to cause an appreciable reduction in the likelihood of both the survival and recovery of the species in the wild by reducing its reproduction, numbers, or distribution. The key to making this finding is clearly establishing the role of the action area in the conservation of the species as a whole, and how the effects of the proposed action, taken together with cumulative effects, are likely to alter that role and the continued existence (i.e., survival) of the species.

The range of the bull trout in the coterminous United States is divided into six biologically-based recovery units, described in final recovery plan. For the marbled murrelet, conservation zones were defined in the final recovery plan. In the section 7 process, these zones are recognized as recovery units. Pursuant to Service policy, when an action impairs or precludes the capacity of a recovery unit from providing both the survival and recovery function assigned to it, that action may represent jeopardy to the species. When using this type of analysis, the biological opinion describes how the action affects not only the recovery unit's capability, but the relationship of the recovery unit to both the survival and recovery of the listed species, in terms of its numbers, reproduction, and distribution. The analysis in the following sections applies the above approach and considers the relationship of the action area to the recovery unit and the relationship of the recovery unit to both the survival and recovery of the bull trout and marbled murrelet as a whole as the context for evaluating the significance of the effects of the Federal action, taken together with cumulative effects, for purposes of making the jeopardy determination.

8 STATUS OF THE SPECIES RANGE-WIDE

8.1 Bull Trout

The bull trout was listed as a threatened species in the coterminous United States in 1999. Throughout its range, the bull trout is threatened by the combined effects of habitat degradation, fragmentation, and alteration (associated with dewatering, road construction and maintenance, mining, grazing, the blockage of migratory corridors by dams or other diversion structures, and poor water quality), incidental angler harvest, entrainment, and introduced non-native species (64 CFR 58910 [Nov. 1, 1999]). Since the listing of bull trout, there has been very little change in the general distribution of bull trout in the coterminous United States, and we are not aware that any known, occupied bull trout core areas have been extirpated (USFWS 2015b, p. iii).

The 2015 recovery plan for bull trout identifies six recovery units of bull trout within the listed range of the species (USFWS 2015b, p. 34). Each of the six recovery units are further organized into multiple bull trout core areas, which are mapped as non-overlapping watershed-based polygons, and each core area includes one or more local populations. Within the coterminous United States, we currently recognize 109 currently occupied bull trout core areas, which comprise 600 or more local populations (USFWS 2015b, p. 34). Core areas are functionally similar to bull trout metapopulations, in that bull trout within a core area are much more likely to interact, both spatially and temporally, than are bull trout from separate core areas.

The Service has also identified a number of marine or mainstem riverine habitat areas outside of bull trout core areas that provide foraging, migration, and overwinter (FMO) habitat that may be shared by bull trout originating from multiple core areas. These shared FMO areas support the viability of bull trout populations by contributing to successful overwintering survival and dispersal among core areas (USFWS 2015b, p. 35).

For a detailed account of bull trout biology, life history, threats, demography, and conservation needs, refer to Appendix A in this Opinion.

8.2 Marbled Murrelet

The marbled murrelet (murrelet) was listed as a threatened species in Washington, Oregon, and California in 1992 under the federal Endangered Species Act. The primary reasons for listing included extensive loss and fragmentation of old-growth forests which serve as nesting habitat for murrelets and human-induced mortality in the marine environment from gillnets and oil spills (57 FR 45328 [Oct. 1, 1992]). Although some threats such as gillnet mortality and loss of nesting habitat on Federal lands have been reduced since the 1992 listing, the primary threats to species persistence continue (USFWS 2019a, p. 65).

The most recent population estimate for the entire Northwest Forest Plan area in 2019 was 21,200 murrelets (95 percent confidence interval [CI]: 16,400 to 26,000 birds) (McIver et. al 2021, p. 3). The long-term trend derived from marine surveys for the period from 2001 to 2018 indicate that the murrelet population across the entire Northwest Forest Plan area has increased at a rate of 0.5 percent per year (McIver et. al 2021, p. 4). While the overall trend estimate across this time period is slightly positive, the confidence interval is fairly tight around zero (95% CI -0.5 to 1.5 percent), leading to the conclusion that there is no directional trend (McIver et. al 2021, p. 4).

Murrelet population size and marine distribution during the summer breeding season is strongly correlated with the amount and pattern (large contiguous patches) of suitable nesting habitat in adjacent terrestrial landscapes (Falxa and Raphael 2016, p. 109). The loss of nesting habitat was a major cause of murrelet decline over the past century and may still be contributing as nesting habitat continues to be lost to fires, logging, and wind storms (Miller et al. 2012, p. 778). Monitoring of murrelet nesting habitat within the Northwest Forest Plan area indicates nesting habitat has declined from an estimated 2.53 million acres in 1993 to an estimated 2.23 million acres in 2012, a total decline of about 12.1 percent (Falxa and Raphael 2016, p. 72). The largest and most stable murrelet subpopulations now occur off the Oregon and northern California coasts, while subpopulations in Washington declined at a rate of approximately -3.9 percent per year for the period from 2001 to 2019 (McIver et al. 2021, p. 4). Rates of nesting habitat loss have also been highest in Washington, primarily due to timber harvest on non-Federal lands (Falxa and Raphael 2016, p. 37), which suggests that the loss of nesting habitat continues to be an important limiting factor for the recovery of murrelets.

Factors affecting murrelet fitness and survival in the marine environment include: reductions in the quality and abundance of murrelet forage fish species, harmful algal blooms, toxic contaminants; murrelet by-catch in gillnet fisheries; murrelet entanglement in derelict fishing gear; oil spills, and human disturbance in marine foraging areas (USFWS 2019a, pp. 29-61). While these factors are recognized as stressors to murrelets in the marine environment, the extent that these stressors affect murrelet populations is unknown. As with nesting habitat loss, marine habitat degradation is most prevalent in the Puget Sound area where anthropogenic activities (e.g., shipping lanes, boat traffic, and shoreline development) are an important factor influencing the distribution and abundance of murrelets in nearshore marine waters (Falxa and Raphael 2016, p. 106).

Detailed accounts of murrelet biology, life history, threats, demography, and conservation needs are presented in the *Recovery Plan for the Marbled Murrelet* (USFWS 1997), and in the *Northwest Forest Plan—The first 20 years (1994-2013): Status and Trend of Marbled Murrelet Populations and Nesting Habitat* (Falxa and Raphael 2016) as well as Appendix B in this Opinion. A peer-reviewed, pre-print draft of the *Status and Trend of Marbled Murrelet Populations in the Northwest Plan Area, 2000 to 2018* (McIver et al., in press) is available at: https://www.fs.fed.us/r6/reo/monitoring/marbled-murrelet.php.

8.3 Short-Tailed Albatross

The range-wide population of the short-tailed albatross has been growing steadily. Based on surveys at the breeding colonies on Torishima Island, Japan, the three-year running average of the population growth rate between 2017 and 2019 is estimated at 8.9 percent (USFWS 2020b, p. 6). To date, conservation efforts have largely focused on addressing the threats of habitat alteration and loss due to catastrophic events and commercial fishing. Less effort has been invested to alleviate threats to short-tailed albatross from climate change, ocean regime shift, and contaminants including plastics.

Over three-quarters of the breeding population of short-tailed albatross nest on Torishima Island (USFWS 2020b, p. 7). There have been volcanic eruptions on Torishima that have killed large numbers of birds and destroyed nesting habitat (Austin, Jr. 1949, p. 288). It was estimated that a volcanic eruption on Torishima in the near future could kill as much as 54 percent of the world's population of short-tailed albatross (USFWS 2008a, p. 17). Conservation strategies for short-tailed albatross emphasize the importance of establishing breeding colonies on other islands to hedge against losing a large proportion of short-tailed albatross from a single catastrophic event (USFWS 2008a). By-catch of short-tailed albatross by commercial fisheries continues to be a major conservation concern; efforts to address the threat are primarily focused on raising awareness and use of seabird deterrents in the industry (USFWS 2014, p. 15).

The Service has not designated critical habitat for the short-tailed albatross. For a detailed account of short-tailed albatross biology, life history, threats, demography, and conservation needs, refer to Appendix C in this Opinion.

9 ENVIRONMENTAL BASELINE

Regulations implementing the ESA (50 CFR 402.02) define the environmental baseline as the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed 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 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.

9.1 Status of the Bull Trout in the Action Area

The listed entity of bull trout is divided into six recovery units (which may have regions within them) and each recovery unit is broken down into core areas consisting of core habitats with core populations. Bull trout from the Coastal Recovery Unit (RU) are divided into two geographic regions within the action area: Puget Sound and Olympic Peninsula. The Puget Sound and Olympic Peninsula geographic regions are entirely within Washington. The Puget Sound geographic region contains eight core areas, and the Olympic Peninsula geographic region contains six core areas. Within the Washington core areas (which includes Lower Columbia bull trout which are outside of the action area), there are 71 local populations, 4 potential local populations, 2 local populations located in both Canada and Washington, and 4 local populations in both Washington and Oregon (USFWS 2015a, pp. A-148 to A-151).

Bull trout core areas within Washington support anadromous, fluvial, adfluvial, and resident life history forms. The Puget Sound and Olympic Peninsula geographic regions contain the anadromous life history form. Two core areas within these regions, the Upper Skagit River and Chester Morse, are isolated above one or more dams and only contain fluvial, adfluvial, and resident life history forms.

9.1.1 Bull Trout Abundance

Since listing in 1999, bull trout abundance has been identified as relatively stable range-wide (USFWS 2015b, p. 8). The NatureServe status assessment tool (Faber-Langendoen et al. 2009, as cited in USFWS 2015b, p. 8), indicated that the Coastal RU was "vulnerable to extirpation" with a NatureServe Rank Score of 3.31. The NatureServe Rank Scores are based on nine factors: linear distance of occupancy; number of occurrences, or local populations; adult population size; environmental specificity; intrinsic vulnerability; short-term trend; long-term trend; threat scope; and threat severity. NatureServe scores range from 0 to 5.5, with low scores being the least robust, most threatened RU, and high scores being the most robust, least threatened RU. When a factor is unavailable for a core area, NatureServe is capable of running the model with an "unknown" value for that factor. No specific bull trout abundance or density estimates have been established for any Coastal RU core areas within Washington. The status of core areas in the Coastal RU, including those in Oregon, is variable (USFWS 2015a, p. A-6).

9.1.1.1 Puget Sound Geographic Region

The Puget Sound geographic region has two core areas that were considered bull trout population strongholds, the Lower Skagit and Upper Skagit, and two core areas, the Puyallup River and Stillaguamish River, that were identified as having small population sizes (USFWS 2015a, p. A-27). During NatureServe ranking efforts, bull trout abundance information was lacking due to sampling variability, differences in methods used to estimate abundance and in some core areas, lack of data (USFWS 2008b, p. 14). Some core areas have long-term data that can be used to provide information on the status of bull trout, with that information potentially extrapolated to other core areas when information is lacking.

Bull trout within individual core areas within the Coastal RU are monitored or surveyed at different levels and frequency. Based on bull trout monitoring surveys through 2019, the Puget Sound geographic region of the Coastal RU is showing unstable and declining bull trout numbers. Once considered a stronghold, populations of bull trout in the Lower Skagit core area are decreasing based on numbers of bull trout redds and juveniles being caught in the downstream screw trap. More data is needed in the Upper Skagit core area to make any determination of bull trout abundance. The Stillaguamish River core area is also showing extremely low numbers of bull trout redds with only one redd observed between 2017 and 2019. Based on redd counts and number of adults passed over Sunset Falls (85 percent decline since

2006), trend information for bull trout in the Snohomish/Skykomish River core area also appears to be declining. The Puyallup River core area, considered to have small population numbers has had increased numbers of bull trout passed over Mud Mountain Dam since 2008 and an increase in bull trout redd numbers.

The following provides an overview of the status of individual core areas within the Puget Sound geographic region.

The Chester Morse Lake core area is a closed watershed with little to no anthropogenic impacts. Activities within the watershed are covered under the City of Seattle's Habitat Conservation Plan. The City of Seattle has not conducted formal bull trout monitoring since 2015, and any bull trout monitoring that has occurred does not provide any population trend information (Mesa 2008; USFWS 2008b; Barnett 2013; West Fork Environmental 2019; Thompson 2020, Pers. comm.). Information is also unavailable regarding bull trout population trends within the Chilliwack River core area.

The Nooksack River core area appears to be stable. Within the Nooksack core area, where long-term bull trout survey data are very limited, the number of bull trout observed in Thompson Creek during salmon spawning surveys has been stable or slightly increasing (Appendix A Table 1 and Figure 1) (WDFW 2011-2021). More survey data are needed in the Nooksack River core area to make any specific short-term trend on abundance of bull trout.

Bull trout redd and spawning surveys have only been conducted in the Upper Skagit core area since 2017 (Appendix A Table 2) (Majeske and Hugh 2020). All spawning locations are not known and the occurrence of Dolly Varden (*S. malma*) complicates survey data as visually the surveyor cannot distinguish the difference between a bull trout and a Dolly Varden. Stetattle Creek, a tributary to the Gorge Reservoir, has very limited spawning habitat, and redds may be superimposed by other spawning fish species. Native char were observed on the spawning habitat, but only a few redds were found. Thunder Creek, a tributary to Diablo Reservoir, has spawning habitat used by native char. The local population within Thunder Creek appears to be stable. Few redds within the tributaries to Ross Lake have been found. Redd surveys within other tributaries to Ross Lake are needed to determine the status of these local populations.

Bull trout redd counts have been conducted since 2002 in the Lower Skagit River core area (Appendix A Table 3 and Figure 2) (Fowler 2019). Peak number of redds occurred in 2006 (855 redds) and 2014 (1010 redds). Between 2015 and 2019, the number of bull trout redds has decreased to the lowest number (175 redds observed in 2019) since 2002. Similarly, the 5-year mean shows a decline in bull trout redd numbers between 2014 and 2019. Redd numbers have decreased in most streams, with lowest numbers being found in Illabot Creek (7 redds), SF Sauk River (20 redds), and Downey Creek (21 redds). In the Cascade River, there were lower number of bull trout redds in 2019, but a slight increase in redds in 2017 and 2018. West Fork Bacon Creek had 4 redds found in 2017, but increased numbers in 2018 (29) and 2019 (30).

With the decrease in redd numbers in the Lower Skagit River core area, a similar decrease has been observed in captures of juvenile bull trout in the lower Skagit River screw traps (Appendix A Table 3 and Figure 3) (WDFW 2011-2021). Peak numbers of juveniles were caught in 2012

(204 juveniles) and in 2015 (174 juveniles). Juveniles caught in the screw trap declined between 2015 and 2019 with a low of 53 juveniles in 2019. The 5-year mean shows a decline in redd numbers since 2016. Similarly, the number of bull trout observations during spawning surveys have gone down since a peak in 2015 (690 bull trout), but has not had a steady decline (Appendix A Table 3 and Figure 4) (WDFW 2011-2021). There was an increase in observed bull trout from 2016 to 2017. The 5-year mean shows bull trout observations of bull trout during spawning surveys is stable.

The Baker River Hydroelectric Facility captures adult bull trout for transportation above the dams as well as juvenile bull trout for downstream passage. Upstream passage of adult bull trout has declined from 2015 to 2019, but the low number transported in 2019 (10 adults) is similar to those transported upstream in 2006 and 2007 (Appendix A Table 3 and Figure 5) (PSE 2019, 2020). Downstream captures of juveniles at Upper Baker Reservoir have also declined from 2015 (129 juveniles) to 2019 (32 juveniles), but juvenile numbers captured in Lower Baker Reservoir had the second highest captured in 2018 (28 juveniles) since 2003 (Appendix A Table 3 and Figure 6) (PSE 2019, 2020). In 2015, 81 juvenile bull trout were captured in Lower Baker Reservoir.

The Lower Skagit River core area was considered a bull trout stronghold, but redd numbers, screw trap numbers, and observations of adult bull trout during spawning surveys, all indicated a downward trend in bull trout abundance.

The Stillaguamish River core area, a core area identified as having low population abundance, has had only one bull trout redd identified between 2016 and 2019 (Appendix A Table 4 and Figure 7) (Fowler 2019). In 2014 and 2015, ten bull trout redds were found in the Stillaguamish River. The highest number of bull trout redds were found in 2006 with 67 redds, and in 2008 with 64 redds. Bull trout redd numbers within the Stillaguamish River have always been low, with less than 30 redds occurring 12 times between 2002 and 2019. Bull trout abundance within the Stillaguamish River core area is inferred to be extremely low based on redd counts.

Bull trout abundance in the Snohomish and Skykomish River core area is declining based on numbers of redds, bull trout observed during spawning surveys, and adult bull trout transported over Sunset Falls on the North Fork Skykomish River. The peak number of adult bull trout being transported above Sunset Falls occurred in 2010 (97 adult bull trout) (Appendix A Table 5 and Figure 8) (WDFW 2011-2021). Between 2011 and 2019, bull trout numbers have been fluctuating with a low of 26 adults in 2013, an increase to 67 adults in 2014, then decreased to 9 adults in 2017, and a very gradual increase in 2018 (10 adults) and 2019 (13 adults). The 5-year mean of adult bull trout transported over Sunset Falls shows a decline in adult bull trout. Bull trout observed during spawning surveys show yearly fluctuating numbers with a stable 5-year mean (Appendix A Table 5 and Figure 9) (WDFW 2011-2021). Bull trout redd counts have fluctuated in the NF Skykomish River, with peak numbers of redds in 2004 and 2005 with 247 redds, 2008 with 195 redds, and in 2015 with 141 redds (Appendix A Table 5 and Figure 10) (Fowler 2019). The fewest number of bull trout redds found was 12 redds in 2016 and then numbers have increased to 88 redds in 2018 and 86 in 2019. With incomplete surveys conducted in four years, the 5-year mean of redd numbers in the North Fork Skykomish River shows a decline in redd numbers, but stable numbers between 2013 and 2019.

The Puyallup River core area was considered to have small population sizes, but survey and fish passage numbers show the population of bull trout is increasing (Appendix A Table 6) (USACE 2020; Marks, et al 2020). Bull trout are passed above Mud Mountain Dam and numbers have increased since 2008 when only 14 bull trout, the lowest recorded, were transported over the dam (Appendix A Table 6 and Figure 11) (USACE 2020). Since 2008, the number of bull trout transported over the dam increased to 406 in 2014, declined to 222 in 2015 and increased to 388 in 2019. With the increasing number of bull trout transported over Mud Mountain Dam, a similar increase has been seen in the number of bull trout redds within the White River (Appendix A Table 6 and Figure 12) (Marks, E. L., et al 2020). Except for Fryingpan Creek, bull trout redd numbers have increased in other tributaries to the White River. Peak number of redds in the White River occurred in 2014 (406 redds), dropped to 222 redds in 2017, and then increased to 388 in 2019.

9.1.1.2 Olympic Peninsula Geographic Region

The Olympic Peninsula geographic region has two core areas, the Dungeness River and Skokomish River, which were identified as having small population sizes (USFWS 2015a, p. A-27). The Quinault River core area was identified as the one stronghold in this geographic region (USFWS 2015a, p. A-3). Similar to the Puget Sound geographic region, abundance information is lacking for many of the core areas. Scattered surveys including redd counts, bull trout observations, snorkel surveys, etc. were conducted in many of the core areas within the Olympic Peninsula geographic region. However, long-term surveys or surveys after 2010 did not occur, so it is not possible to assess bull trout abundance trends through 2019.

Given the lack of data for the Olympic Peninsula geographic region, population trends and abundance are unclear for most of the core areas, including the Queets, Quinault, and Hoh. Data available for the Skokomish River and Dungeness core areas indicate relatively stable bull trout populations. Screw trap data in the Dungeness River core area between 2007 and 2019 show that the highest number of juvenile bull trout caught was 147 in 2014, but the number has decreased since, with only 2 juvenile bull trout caught in 2019 (Appendix A Table 7 and Figure 13) (WDFW 2011-2021). Although, the 5-year mean shows a relatively stable number of bull trout caught in the screw trap, large numbers caught in 2008 and 2014 indicate a decreasing trend. The Skokomish River core area appears to be more stable than the Dungeness. Also, more bull trout were observed during spawning surveys and redd counts in the NF Skokomish River than in the SF Skokomish River (Appendix A Table 8 and Figure 14) (USFS 2020a; 2020b, unpublished data). Bull trout abundance is expected to increase in the Elwha River given the removal of two dams that blocked passage.

9.1.2 Climate Change

Within Western Washington, predicted environmental changes as a result of climate change include increased air temperature, reduced snow accumulation, increased frequency and intensity of rain events, and declining summer precipitation (Mauger, et al 2015). These changes result in increases in river water temperatures, winter flood risk, higher flood flows, and decreased river baseflows. Within Washington State, the air temperature has increased approximately 1.5 °F (0.8 °C) between 1900 and 2019 (NCICS 2021).

All life stages of bull trout rely on cold water. The onset of bull trout spawning is determined by stream temperatures falling below 48 °F (9 °C). Within the Puyallup, Lower Skagit River and Chester Morse core areas, spawning surveys have shown that bull trout spawning begins approximately two weeks after such temperature drops occur (Mesa 2008; Barkdull 2020, Pers. comm.; Thompson 2020, Pers. comm.). While specific reasons for the later onset of spawning has not been determined, increased stream temperatures as a result of climate change may be a cause. In addition, bull trout spawning streams are being altered by changes in flows. A spawning stream will be dry one day and after a rain, the stream is drastically altered due to high flows (Barkdull 2020, Pers. comm.). With changes in stream water temperatures and flows, bull trout may be utilizing new waters for spawning that are not surveyed. These streams may have the cold water required for bull trout spawning.

The long-term consequences of delays in bull trout spawning are unknown. The period of egg incubation to emergence of fry may take up to 210 days (7 months). If winter and spring water temperatures are also warmer due to climate change, larval development will be accelerated. However, it is unknown if that accelerated development will result in deformities, mortality, or a lack of synchronicity with critical prey resources.

9.1.3 Threats

9.1.3.1 Puget Sound Geographic Regions

There are multiple threats to bull trout in the Puget Sound geographic region (Appendix A Table 10 and Table 11) (USFWS 2015a, pp. A-11 to A-16). Within many core areas in this region, development and related impacts, flood control, flood plain disconnection, bank armoring, channel straightening, loss of instream habitat complexity, and connectivity issues are common along mainstem river corridors (USFWS 2015a, p. A-9). Within every core area, forest management, flood control, recreational mining, development, and fish passage issues are the major threats to bull trout and their prey base (Appendix A Table 10) (USFWS 2015a, pp. A-11 to A-15).

Puget Sound nearshore and estuarine habitats have been severely degraded due to development (Appendix A Table 11) (USFWS 2015a, p. A-16). Residential and industrial development have resulted in increased bank armoring, and expansions of marinas, piers, and docks. These habitat impacts have resulted in impacts to bull trout, but also their prey species. Juvenile salmon migration and foraging have been impacted and marine forage fish spawning has been lost or altered. Many of these primary threats continue, with restoration projects, such as estuary restoration and fish passage improvements associated with the culvert lawsuit, providing some benefits in selected areas of the geographic region.

Within the Puget Sound geographic region, there are no physical barriers to bull trout migrating between core areas that enter into Puget Sound. Bull trout are known to migrate from one core area to another core area, a non-core area (smaller rivers that enter into Puget Sound), or foraging, migration, and overwintering areas (Duwamish River, Lake Washington, etc.). For example, bull trout have been observed migrating from the Snohomish River core area down to the Duwamish River and then returning (Goetz, et al 2012).

Migration is impeded within many of the core areas in the Puget Sound geographic region by dams or diversions. Chester Morse and the Upper Skagit River core areas are isolated above dams that do not have any fish passage. The Baker River, the Lower Skagit River core area, Skykomish River, and the White River in the Puyallup River core area have dams that have trap and haul facilities that limit bull trout movement between local populations within the core area. The Stillaguamish River core area has a diversion dam in the lower mainstem river. The fish ladder around the diversion dam was replaced in 2020 because the previous fish ladder was not functioning as designed.

9.1.3.2 Olympic Peninsula Geographic Region

In the Olympic Peninsula geographic region, angling or harvest of bull trout was identified as the primary threat in four bull trout core areas: Hoh, Queets, Quinault, and Skokomish River core areas (Appendix A Table 12) (USFWS 2015a, pp. A-17 to A-20). Reduced prey abundance is also a threat. With the removal of two dams on the Elwha River, bull trout and their prey base (primarily juvenile salmonids) have been improving as access to more than 30 miles of pristine spawning and rearing habitat was restored. Transportation networks, and both improved and unimproved forest roads, have caused significant impacts in this region. Many roads within this region are adjacent to streams and have numerous stream crossings that have direct impacts to the stream banks, habitat, and channels as the roads periodically fail. Road maintenance results in a continuous supply of sediments that reduces spawning habitat.

Within shared FMO, residential development and urbanization is a primary threat along the Strait of Juan de Fuca. Along the Pacific Coast and tributaries, legacy forest management is a primary threat (Appendix A Table 13) (USFWS 2015a, p. A-21).

Connectivity between core areas in this geographic region is naturally low due to the geographic distance between them: one core area is located in Hood Canal, two are in the Strait of Juan de Fuca, and three are along the Washington Coast (USFWS 2015a, p. A-17). However, bull trout can migrate between nearby core areas, or can migrate to non-core areas or FMO habitat (Humptulips, Chehalis, Moclips, Raft, etc.). Removal of the Elwha River dams provides unimpeded passage for bull trout migration throughout the core area from spawning areas in the headwaters to the marine water in the Strait of Juan de Fuca.

9.2 Status of the Marbled Murrelet in the Action Area

Ongoing surveys have provided new data on the status of murrelet in the action area since the 2016 and 2018 Opinions (see Tables 11 in those documents). The action area, as described in the 2016 Opinion (USFWS 2016, pp. 48-53), encompasses the listed range of murrelet. Therefore, the Status of the Species (Appendix B) provides the best description of the status of the murrelet in the action area.

9.3 Status of the Short-Tailed Albatross in the Action Area

The training and testing area along the west coast of the United States is used by juvenile and sub-adult short-tailed albatross. As birds age they appear to spend more time in other parts of the species range, especially in the marine waters of Alaska and the Aleutian Islands. The action area does not include any current breeding habitat for short-tailed albatross.

Aside from the population increasing at a slightly faster rate than was assumed previously, the status of short-tailed albatross in the action area has not changed significantly since the 2016 and 2018 Opinions; see the 2016 Opinion (USFWS 2016, pp. 85-90) for a detailed description.

9.4 Climate Change

We have not gained information that changes our discussion of climate change in the environmental baseline since the 2016 and 2018 Opinions, except as detailed in Appendix B for murrelets. For a detailed description of climate change as part of the environmental baseline for the proposed action, see the 2016 Opinion (USFWS 2016, pp. 90-94).

9.5 Previously Consulted-on Effects

Within the action area, the Service has consulted on the effects of many federal proposed actions and the effects of those actions are part of the environmental baseline considered here. The federal proposed actions include those involving:

- harbor expansions
- shoreline armoring
- ferry terminal upgrades
- aquaculture activities
- discharges from wastewater treatment plants
- construction of piers, ramps, and floats
- bridge, road, pier, and wharf maintenance and upgrades

The effects to murrelets and bull trout associated with these projects are similar to one another, and are related to a combination of exposure to increased sound pressure levels from pile driving, decreased water quality due to increased turbidity and the introduction and circulation of contaminants, and adverse impacts to forage fish populations.

The Service has recently consulted on the continued Treaty and non-Treaty salmon fisheries throughout Puget Sound, which affect murrelets directly through net entanglements. Over the twenty years of the fisheries Opinion and across all types of salmon fisheries, the Service expects 273 murrelets to be captured and for those captures to result in the death of 137 adult, sub-adult, and unfledged murrelets (USFWS 2017). While the Puget Sound fisheries occur within Zone 1 (see Appendix B for descriptions of the murrelet conservation zones), much of the fishing activity occurs outside of the murrelet breeding season, and we expect that seasonal migrants to

Zone 1, associated with breeding season populations of Conservation Zones and Canada, will also be among those affected.

The Service also issued an Incidental Take Permit associated with a Habitat Conservation Plan (HCP) for the Skookumchuck Wind Energy Project, which was subject to consultation (USFWS 2019b), and included removal of lost or derelict fishing gear in Puget Sound, as well as nesting habitat conservation, as measures to mitigate for murrelet mortality at inland wind turbines. Individuals killed at these wind turbines may be associated with Zone 1 or Zone 2, those saved by derelict net removal will be a similar population to that described above for Puget Sound fisheries, and those assisted by nesting habitat conservation will be from Zone 2.

The Service has also consulted on other military training activities. This includes Growler airfield operations at NAS Whidbey Island Complex that have been ongoing since 2012 with expansion of Growler training in 2020. Thirty years of Growler training based at NAS Whidbey Island Complex affect murrelets by exposing birds to over three million exposures to overflights annually, disrupting normal behaviors such as foraging and resting (USFWS 2020a, p. 113). This action occurs within Zone 1, though winter migrants from other populations will also be affected.

The Service has consulted on some large terrestrial actions intersecting the terrestrial portion of the action area. For example, the Service recently completed consultation with the U.S. Forest Service addressing all routine activities, including commercial thinning, within the Olympic National Forest, which spans parts of Zones 1 and 2. Over the ten-year term of the action, the Service expects that this action will reduce nesting success at 118 murrelet nests, due to edge effects, increases in corvid density due to recreational uses, and noise and visual disturbance.

The Service also approved an HCP amendment to the Washington State Department of Natural Resources (WDNR) to incorporate their Long Term Conservation Strategy for murrelets into the existing State Lands HCP. This HCP is in effect until 2067, and during this time, murrelet conservation will be concentrated in 20 "special habitat areas" that encompass over 46,000 acres on WDNR-managed lands, all known Occupied Sites will be protected, and most existing murrelet habitat on WDNR-managed lands will be conserved. The long-term strategy allows for approximately 38,000 acres of potential murrelet nesting habitat to be available for harvest over a period of two decades, but overall, the amount of murrelet nesting habitat on WDNR-managed lands was projected to increase from approximately 207,000 acres in 2019 to over 272,000 acres in 2067. Both harvest and conservation of habitat will affect Zones 1 and 2.

The Service has consulted on some large terrestrial actions outside of the action area, but that will affect murrelet populations also expected to be present within the marine portion of the action area. These include the revision of several Bureau of Land Management Resource Management Plans for western Oregon in 2016, in portions of Zones 3 and 4. Consultations on these management plans took the form of framework programmatic consultations, and the quantification of adverse effects was deferred until individual projects are proposed, at which time those adverse effects will be accounted for in the baseline.

10 EFFECTS OF THE ACTION

10.1 Introduction

Effects of the action are all consequences to listed species or designated critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed 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 (See § 402.17).

We expect the effects of the action with the proposed changes to be similar to the effects described in the 2016 and 2018 Opinions (USFWS 2016, pp. 97-247; USFWS 2018, pp. 12-14) with the addition of two new stressors, i.e., high-energy lasers, and chemical and biological simulants, as well as an increase in the number of underwater explosions and occurrence of underwater explosions closer to shore in the offshore area.

In our original analysis of the 2016 proposed action, we analyzed both the response and exposure of each listed species to each of the stressors associated with the proposed activities. For those stressors that would result only in insignificant effects, we did not conduct a detailed exposure analysis. For other stressors, we assessed the likelihood of exposure for each species, and categorized exposure as discountable, not reasonably certain to occur, or reasonably certain to occur. Discountable exposures were those that were extremely unlikely to occur during the term of the action. In this context we considered exposure to be reasonably certain when we assessed that a species was more likely to be exposed than not to be exposed. We concluded that exposure was not reasonably certain to occur when the likelihood of exposure was greater than discountable but less than reasonably certain. We continued using this categorization in our 2018 Opinion, and we maintain these same categories in our current analysis.

After reviewing the 2016 and 2018 Opinions and the stressors and activities that were determined to result in insignificant, discountable, and/or not reasonably certain to occur effects, we have concluded that, using the best available scientific information, the changes in those activities and stressors under the current proposed action do not warrant changes to those determinations. Therefore, our determinations for those activities and effects remain valid and are incorporated here by reference (USFWS 2016, pp. 97-247; USFWS 2018, pp. 12-13). In the following sections we will analyze the effects of new activities, new stressors, and activities/stressors that were determined to be reasonably certain to result in adverse effects in the 2016 and 2018 Opinions.

10.2 High-energy Laser Testing

High-energy lasers will be tested for two applications, weapons and communications.

10.2.1 High-energy Laser Weapons Testing

High-energy laser weapons will be employed from surface ships or helicopters and will involve the use of directed energy at small surface and airborne targets. The high-energy laser will be used at short ranges (i.e., line-of-sight). High-energy laser weapons testing will occur 54 times per year, with each event typically lasting 12 hours per day over a seven day period (lasers will be energized for short periods during these events). Up to 25 percent of testing may occur at night. All high-energy laser weapons testing will occur in the offshore area, greater than 12 nm from shore, only in open-ocean locations (i.e., not close to land areas).

The following safeguards will be employed on high-energy laser platforms in order to reduce the probability of the laser striking the water:

- The high energy laser platform will have provisions that prevent misfiring (i.e., firing when not intended) that will all but eliminate the possibility of misfire and ensure that the system will only fire when the operator pulls the trigger.
- The high-energy laser platforms will have built-in constraints that only permit firing when it is locked onto a target, and automatically interrupts firing if the target track on a target is lost.
- The operators will be trained to stop firing when the laser aim point moves off of the selected target.

In the event that a laser beam misses a boat target, the beam may strike the water somewhere between 200 m (219 yd.) and 6.5 km (3.5 nm) or more from the laser, assuming a range of 200 m (219 yd.) to 5 km (2.7 nm) between the laser and target. At these ranges, the low angles to the water will reflect most of the laser energy. Underwater, the laser will lose a significant amount of energy within only a few centimeters from the surface (Navy 2019, p. 5-24). Therefore, only water in the immediate vicinity of the laser beam and a few centimeters under the surface will be affected, and any resulting hot water will quickly mix with the cooler surrounding water. As a result, striking the ocean with a high-energy laser beam is not expected to be a hazard to underwater marine life, except in the immediate vicinity of the laser beam, just below the ocean surface.

High-energy laser weapon testing will occur far enough from shore that exposure to bull trout, which remain near the coast, is discountable. However, both short-tailed albatross and murrelets could potentially be struck by the high-energy laser beam at or near the water's surface, with extended exposure potentially resulting in injury or death due to traumatic burns from the beam. Should the laser strike the sea surface, individual birds in the immediate vicinity of the laser beam could be exposed. Marine birds would be exposed to the beam if it missed the target or if a bird flew between the source and the target.

10.2.1.1 Marbled Murrelet

High-energy laser activities will occur in the offshore area at least 12 nm from shore. In this area, we expect that murrelets will be present at extremely low densities during the breeding season (roughly April through September). Therefore, exposure of murrelets during the breeding season is discountable.

During winter, murrelets are more often found farther offshore, up to approximately 50 nautical miles (see USFWS 2016, Appendix A, pp. 9-11). To determine the likelihood of murrelet exposure to high-energy laser beams, we made the following assumptions (similar to those in the 2016 Opinion (see USFWS 2016, Appendix A): 1) on average, half (27) of the laser tests will be conducted during the October through March timeframe; 2) these laser tests will be distributed equally throughout the offshore area, from Washington to California and from 12 nm to 200 nm; 3) as a consequence of assumption 2, 17 percent of the laser tests will occur in the area between 12 nm to 50 nm, which makes up 17 percent of the offshore area; and 4) the maximum area in which a pair of murrelets may be affected by the laser beam is 6.5 km long, because lasers missing their target may hit the water 6.5 km away, and 1 m wide, to account for the width of two murrelets flying abreast (while the width of the laser beam is presumably much narrower than 1 m, at least one of a pair of murrelets flying within 1 m of the laser beam's path may be struck).

We used the methods outlined in Section *10.6.1.1 Offshore Non-breeding Marbled Murrelet Density Model 1* to estimate murrelet density within 50 nm of shore, where we expect murrelet distribution during the non-breeding season to overlap with laser testing. We used methods outlined in the appendices of the 2016 opinion (USFWS 2016, Appendix A and Appendix G) to calculate the probability of murrelet exposure to laser beams during testing. Since murrelets typically travel in pairs, we assessed the probability of exposure to a group of two individuals traveling together, as in our 2016 analysis (USFWS 2016, Appendix A). Assuming that on average, lasers are fired no more than 5 times per year (27 times * 17 percent, rounded up to the next whole number) within 50 nm of shore during October through March, and that the area in which murrelets may be affected by each laser firing is no more than 0.0065 km^2 (6.5 km x 1 m), we expect that the likelihood of a pair of murrelets being struck by the laser beam is less than 3 percent. Because most lasers are expected to strike targets 200 m away, rather than striking the water 6.5 km away, 3 percent is likely to be a high estimate of the probability of murrelet exposure, as long as the other assumptions listed above are accurate.

Because murrelets are extremely unlikely to encounter high-energy laser beams during any time of year, any effects to murrelets from high-energy laser are expected to be discountable.

10.2.1.2 Short-Tailed Albatross

The pelagic range of short-tailed albatross overlaps with the area within the offshore area where high-energy laser testing activities will occur. However, short-tailed albatross in the open ocean are unlikely to be exposed to high-energy lasers due to: 1) the expected low numbers of short-tailed albatrosses in the areas where high-energy laser testing activities will occur (projected to rise to 0.0008 birds/km^2 by the end of the action); 2) the low number of events relative to the area in which they will occur (54 per year throughout the offshore area); 3) the small potential impact area of the laser beam (conservatively estimated as 0.0065 km^2), and 4) the low density of short-tailed albatross throughout the action area. Together, these factors make a direct strike of a short-tailed albatross at the water's surface or within the beam path unlikely. Therefore, effects to short-tailed albatross from high-energy laser testing activities are expected to be discountable.

10.2.1.3 High-energy Laser-based Optical Communication Systems

High-energy lasers used for communication could cause injury if they were to come in contact with eyes. However, these lasers will expose an extremely small area due to the low transmission of energy through the water. Eighty-six percent of blue-green light from these lasers will be lost within 10 m of the laser source, and more than eighty-six percent of other light wavelengths will be lost over the same distance (Navy 2019, p. 5-25). Due to the small area of exposure, the small area of potential injury (the eyes), and the low number of activities (one each year), exposure of bull trout, marbled murrelet, and short-tailed albatross to potentially injurious light from communication lasers is extremely unlikely to occur.

10.3 Underwater Explosions

The following analyses of the effects of underwater explosives focuses on two NWTT activities: mine neutralization/EOD disposal training, and mine countermeasure and neutralization testing.

The Navy's mine neutralization/EOD disposal training involves detonating up to 18 Limpet Mine Neutralization Tool (LMNT) charges E0 with charge weight less than 0.1 lb. NEW) and three larger charges (E3 with charge weight up to 2.5 lb. NEW) at the Crescent Harbor EOD Training Range site, annually. The Navy will also detonate up to 18 LMNT (E0) and three E3's in the Hood Canal EOD Training Range site, annually.

Mine countermeasure and neutralization testing involves the use of up to 36 E4 explosives and five E7 of underwater explosives and will occur approximately two times per year (two multi-day events with 1-10 days per event), in waters 3 nm or greater from shore at the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nm or greater from shore elsewhere in the offshore area (note: mine countermeasure and neutralization testing will not occur off the California coast). A maximum of 108 E4 and 15 E7 explosives will be used over a seven year period and a maximum of 246 explosives (216 E4, 30 E7) will be used through July 21, 2036. Explosives will be used in the water column (i.e., not on the sea floor), with E4 explosives used at 20 ft or greater below the water surface and E7 explosives used at 100 ft or greater below the water surface. Half of the E4 explosives will be detonated at distances at least 4.6 nm from shore and half at least 3 nm from shore; and E7 explosives will be detonated at least 7.5 nm from shore. The maximum sea floor depth for explosive testing will be approximately 1,000 ft.

10.3.1 Thresholds

10.3.1.1 Bull Trout

Underwater explosions can affect fish behavior in a manner that reduces their fitness or survival. For fish that are close enough, the blast can physically injure or kill them (Nedwell and Edwards 2002; Nedwell et al. 2003).

The principal mechanism by which pressure waves from blasts cause physical injuries to organisms is through oscillations of body tissues and sudden compression and expansion of air-filled organs. Most blast injuries in marine animals involve damage to air- or gas-containing organs (Yelverton and Richmond 1981). For example, fish with swim bladders (including salmonids) are vulnerable to the effects of explosives, while fish without swim bladders (sand lance, flatfish, sharks, and rays) and invertebrates are much more resistant (Yelverton and Richmond 1981; Young 1991). When exposed to shock waves, the swim bladder oscillates and may rupture, in turn causing hemorrhages in nearby organs. Fish that have thick-walled swim bladders that are close to the body wall and away from the kidneys are more resistant to blast injury than are fish with thin-walled swim bladders that touch the kidneys.

Several authors have described methods for calculating the theoretical kill or injury zones around underwater explosions (e.g., Gaspin 1975; O'Keeffe and Young 1984; Young 1991). However, a more common metric to use for a single acoustic event that accounts for both the negative and positive pressure wave is sound exposure level (SEL) (Hastings and Popper 2005). The SEL is the time-integrated sound pressure-squared and is expressed in dB referenced to 1 micropascal-squared-second ($1 \mu Pa^2 s$).

In our previous 2016 and 2018 Opinions, the Service used the best available information on the effects of underwater detonations to determine thresholds (impulse levels) for injury to fish, including applying the findings from Yelverton et al. 1975 to bull trout. These thresholds were based on the mass (size and weight) of the experimental fish. Hastings and Popper (2005) used the Yelverton et al. (1975) data to derive an SEL-based threshold where injury was not observed.

The Service established injury and mortality thresholds for fish from explosives (Table 11).

Table 11. The Service's injury or mortality thresholds for bull trout from explosives.

Mortality (dB SPL_{peak} re: $1\mu Pa$)	**Injury** (dB SPL_{peak} re: $1\mu Pa$)	**Threshold Shift** (dB SEL re: $1 \mu Pa^2$-sec)
229	206	186

After establishing the thresholds, the Service asked the Navy to calculate the ranges to effect to those thresholds. Since exposure to sound exceeding any of these thresholds could injure fish, the Service uses the threshold with the largest range to effect for analyzing effects to bull trout. The resulting calculated range to effects (for peak sound level associated with injury [sounds exceeding 206 dB_{peak}]) for E0 and E3 (inland) are provided in Table 12 and have been carried forward for the effects analyses of those explosives in this Opinion.

Table 12. Onset of injury ranges to effect for bull trout from explosions.

Explosive Bin	Location	Onset of Injury (Sound exceeds 206 dB_{peak})
E0	Crescent Harbor	661
	Hood Canal	427
E3	Crescent Harbor	1,674
	Hood Canal	1,210

10.3.1.2 Marbled Murrelet and Short-tailed Albatross

Underwater detonations are known to have negative physiological and neurological effects on a wide variety of vertebrate species; these effects include coronary air emboli, lung hemorrhaging, ruptured livers, hemorrhaged kidneys, ruptured air sacs, and ruptured and scarred eardrums (Cudahy and Ellison 2002; Gisiner et al. 1998; Hastings and Popper 2005; Yelverton et al. 1973; Yelverton and Richmond 1981). Experiments using underwater explosives found that rapid change in underwater sound pressure levels (SPLs) resulted in internal hemorrhaging and mortality in submerged mallards (*Anas platyrynchos*) (Yelverton et al. 1973, p. 49). Death from barotrauma can be instantaneous, occurring within minutes after exposure, or several days later (Abbott et al. 2002). Several birds exposed to explosions survived and appeared uninjured, but upon necropsy two weeks later there was evidence of liver blood clots and lung and kidney injuries (Yelverton et al. 1973, p. 51).

There are no published studies specific to explosions and their physiological effects on murrelets or short-tailed albatross. However, there are some data specific to other birds from evaluations of the effects of underwater blasting and seismic testing (Cooper 1982; Flint et al. 2003; Lacroix et al. 2003; Stemp 1985; Yelverton and Richmond 1981, p. 3). During seismic explorations, it has been noted that seabirds were attracted to fishes killed as a result of the seismic work (Fitch and Young 1948; Stemp 1985). Fitch and Young (1948) found that diving cormorants were consistently killed by seismic blasts, and pelicans were frequently killed when they were exposed when their heads were below water. For exposure of fish and mammals to impulses underwater, Yelverton and Richmond (1981) and Yelverton et al. (1973) found a correlation between the size of animal and the impulse level needed to elicit an injury. While Yelverton did not do this analysis for birds, we reason that this correlation was independent of the organism's taxonomic classification and thus it also applies to birds (for underwater explosions). In the absence of controlled studies specific to seabirds, we considered evaluations of the effects of other types of blast impulses on a variety of vertebrate species, including birds, for evaluating the effects of explosions on murrelets and short-tailed albatross.

Detonating explosives can result in a variety of injuries to organisms. Important biological variables that influence the degree to which an animal is affected include size, anatomical variation, and location of the organism relative to the explosive source in the water column (Gisiner et al. 1998). Studies of explosives by Yelverton and Richmond (1981), Yelverton et al. (1973) and Damon et al. (1974) identified injury thresholds in relation to the size of the charge, the distance from the animal at which the charge was detonated, and the mass of the animal exposed. Much work has been done to assess impacts to avian hearing from in-air sound (Dooling 1980; Dooling 1982; Dooling et al. 2000; Brittan-Powell and Dooling 2002; Dooling and Dent 2002; ; Dooling and Brittan-Powell 2005; Dooling and Popper 2007;; ; Ryals et al. 1999; Ryals and Dooling 2001; Saunders and Dooling 1974; Saunders and Henry 1989); most of this work assessed avian hearing range and hearing loss from over-exposure to in-air sound. The principal mechanism by which blast pressure waves cause physical injuries to organisms is through oscillations of body tissues and sudden compression and expansion of gas-filled organs. Most blast injuries in marine animals involve damage to gas-containing organs (e.g., lungs, gastrointestinal tract, bowels); however, injuries also occur to liver, kidneys, ears, and coronary arteries (Cudahy and Ellison 2002; Gisiner et al. 1998; Hastings and Popper 2005; Yelverton et al. 1973; Yelverton and Richmond 1981).

Injuries from high underwater pressure waves occur over a continuum of potential effects, ranging from mortality to sub-lethal physical effects including Threshold Shift (TS, decreased hearing ability at specific thresholds) and gastrointestinal tract lesions, to non-injurious effects that might result in significant disruption of normal behaviors. At the most severe end of the spectrum, direct mortality or obvious injuries can occur. At the least severe end of the spectrum of injurious effects, there may be temporary hearing shifts or small burst blood vessels.

Several authors have described methods for calculating the theoretical kill or injury zones around underwater explosions (Gaspin 1975; O'Keeffe and Young 1984; Young 1991). A common metric used for a single acoustic event that accounts for both the negative and positive pressure wave is sound exposure level (SEL) (Hastings and Popper 2005). An impulse, measured in Pascal seconds (Pa-sec), is the best way to describe and measure the effects of the explosion on organisms because it captures all the forces occurring with a fast-acting explosion over time. Impulse values better reflect the complex components of the pressure wave associated with an explosion, such as over pressure and under pressure, and the peak SPL. If we used a single component to describe the effects to murrelets, such as peak SPL, or SEL, we may not adequately account for the energy from the shock wave or the over pressure. These components contain significant energy, so by accounting for that energy we have increased confidence that the distances to effect for barotrauma or injury are comprehensive.

The Service established thresholds for onset of injury to murrelets from underwater explosions (Table 13). The Service requested that the Navy calculate the ranges to effect (i.e., the area in which we expect injury of murrelets and short-tailed albatross to occur) for underwater explosions based on information provided for mallards in Yelverton et al. (1973). The ranges to effect for murrelets and short-tailed albatross have been revised since 2016 to reflect a better understanding of the information available in Yelverton et al. (1973).

Table 13. The Service's injury or mortality thresholds for murrelets and short-tailed albatross from underwater explosions.

Bird Species	Auditory Injury dB SEL re: 1 µPa^{2}s	Barotrauma (Pa-sec)	Mortality (Pa-sec)
Marbled Murrelet	212	36	138
Short-tailed Albatross	212	94	361

The range to effect for barotrauma represents the largest area of effect and also encompasses other effects from exposure, including auditory injury and mortality. On that basis, we consider this distance as the threshold for the onset of injury to murrelets caused by underwater explosions, and modeled the probability of murrelet exposure and injury based on this distance. The ranges to effect account for the masses of a murrelet and short-tailed albatross, and for the depths of the explosives and the birds. Range to effect calculations for explosives used in inland waters (E0 and E3) assumed murrelets would be at the same depth as the explosive. This assumption is conservative, as the radius of effects will be largest at the depth of the explosive, but ensures that all exposure is accounted for. See Table 14 below for ranges to effect for inland explosions. Ranges to effect for offshore explosions were calculated at two murrelet diving depths: 27 and 47 m beneath the surface. To be conservative and avoid failing to account for effects of the action, we applied the radius of effects associated with the murrelet diving depth that is closest to the depth of the explosive. Ranges to effect for short-tailed albatross, which feed near the surface, were calculated at a dive depth of 2 m. See Table 15 below for ranges to effect for offshore explosions.

Note on exposure: Short-tailed albatross are extremely unlikely to occur in inland waters, therefore Table 14 does not include ranges to effect for albatross for explosives that only occur in Puget Sound. Murrelets are likely to be present at extremely low densities beyond 12 nm from shore in the summer and are not reasonably certain to be present beyond 50 nm from shore in the winter. For clarity, ranges to effect for actions that occur farther than these distances, and are therefore not reasonably certain to expose murrelet to stressors, are not shown in Table 15.

Table 14. Ranges to effect (onset of barotrauma injury; in meters) for marbled murrelet from inland underwater explosions.

	Marbled Murrelet	
Location	**Crescent Harbor**	**Hood Canal**
Explosive Bin	**≥ 36 Pa-sec at 22.9 m diving depth**	**≥ 36 Pa-sec at 12.2 m diving depth**
E0	31	31
E3	238	183

Table 15. Ranges to effect (onset of barotrauma injury; in meters) for marbled murrelet and short-tailed albatross from offshore underwater explosions.

	Marbled Murrelet		**Short-tailed Albatross**
Explosive Bin	**≥ 36 Pa-sec at 27 m diving depth**	**≥ 36 Pa-sec at 47 m diving depth**	**≥ 94 Pa-sec at 2 m diving depth**
E1	*	*	3†/20††
E2	*	*	3†
E3	*	*	63
E4	347	*	92
E7	*	1,338	252
E8	*	*	304
E10	*	*	13†
E11	*	*	625

* Marbled murrelet exposure not reasonably certain to occur, explosions only occur farther than 50nm from shore in the winter and farther than 12 nm from shore in the summer

† Surface detonation

†† E2 explosions will also detonate 12.2 m below the surface for Tracking Test – Maritime Patrol Aircraft (SUS)

The Navy conducts a variety of activities that include underwater detonations. Based on the distribution and density of murrelets, the location of detonations, and the range to effects values (Tables 14 & 15), we calculated the cumulative probability that a murrelet would be exposed to, and injured from, underwater detonations. A comprehensive description of the assumptions made in our exposure analysis is provided in the 2016 Opinion (Appendices A and G), but a number of the assumptions regarding murrelet density, details of the proposed action, and ranges to effect have been modified as noted above and below.

10.3.2 Explosive Ordnance Disposal

The Navy's Mine Neutralization/EOD disposal training involves detonating up to 18 LMNT charges (E0 with charge weight less than 0.1 lb. NEW) and three larger charges (E3 with charge weight up to 2.5 lb. NEW) at the Crescent Harbor EOD Training Range site, annually. The Navy will also detonate up to 18 LMNT (E0) and three E3's in the Hood Canal EOD Training Range site, annually.

10.3.2.1 Effects to Bull Trout

10.3.2.1.1 Exposure of Bull Trout in Hood Canal

Based on historic observations (1980's) in the Duckabush, Quilcene, and other nearby rivers and estuaries entering Hood Canal from the west, we expect that very few bull trout occur near the Hood Canal EOD Training Range site (Brenkman and Corbett 2005; Brenkman and Corbett 2007; Goetz et al. 2004; Goetz et al. 2007). These rivers are approximately 12.9 km (8 miles) west of the Hood Canal EOD Training Range site. The closest population of bull trout in Hood Canal is in the Skokomish River located 53.1 km (33 miles) to the south of Hood Canal EOD

Training Range site. Hood Canal has been identified as an important foraging, migration, and overwintering habitat for bull trout and would likely be used as the Skokomish River core population increases in abundance (USFWS 2004, Volume II, p. 66).

Fluvial and, potentially, anadromous bull trout are present in the South Fork Skokomish River local population. Although there may be a residual expression of anadromy in the South Fork population, there are currently no indications or data that suggests that individuals are entering the marine environment. The North Fork Skokomish River local population has been isolated above Cushman No.1 and No 2 dams for over a century, but as a result of a recent settlement agreement, Tacoma Power is restoring fish passage to the North Fork. Bull trout pass the dams on a very limited basis, but there is potential for the anadromous life history form of bull trout to become more prevalent in the future. However, habitat degradation of nearshore foraging, migration, and overwintering habitat from natural and human sources (Brennan 2007; Goetz et al. 2004; PSAT 2007; PSP 2008; Puget Sound Water Quality Action Team 2002) and the distance from the Skokomish River, is still likely to limit bull trout occurrence near the Hood Canal EOD Training Range site.

10.3.2.1.2 Response of Bull Trout Exposure in Hood Canal

The Hood Canal EOD Training Range site is located on the eastern shore of Hood Canal at Naval Base Kitsap Bangor. The radius of effect for E0 and E3 explosives is 427 m and 661 m, respectively. Any bull trout that would be exposed to increased SPLs associated with underwater detonations would be injured or killed. Considering the low numbers of bull trout and their expected infrequent use of the Hood Canal EOD Training Range site, we anticipate the risk of exposure to underwater detonations to be extremely low.

10.3.2.1.3 Conclusion for Bull Trout Exposure to EOD in Hood Canal

Bull trout exposure to EOD activities at Hood Canal EOD Training Range site is extremely unlikely, and therefore, discountable.

10.3.2.1.4 Exposure of Bull Trout in Crescent Harbor

Given the effects of underwater explosives on bull trout, the extensive distance that the underwater acoustic environment can be influenced, and the expected presence of anadromous bull trout at the Crescent Harbor EOD Training Range site, individual bull trout are at high risk of being exposed to increased SPLs associated with underwater detonations. The marine areas around Whidbey Island and Crescent Harbor play a critical role in the anadromous life-cycle of bull trout. Larger-sized juveniles, sub-adult, and adult bull trout use marine water for feeding, migrating, and overwintering. The Service expects that large juvenile, sub-adult, and adult bull trout will be present in the Crescent Harbor portion of the action area. Larger juveniles and sub-adult bull trout are present in marine waters throughout the year and adults typically enter marine waters each year in December and January following spawning in freshwater. The adults typically remain in marine waters until July and August, when they leave and migrate to freshwater streams to spawn. Bull trout abundance is expected to vary daily and seasonally as a function of several interacting factors, including the proximity of core areas, abundance/availability of forage, distance from shore, and the time of year (life-cycle stage).

Bull trout exposure is expected at Crescent Harbor because there are three bull trout core areas in relatively close proximity. We assume bull trout presence at the Crescent Harbor EOD Training Range site will be predominately from the Lower Skagit River core area. This core area has one of the highest populations of bull trout and the Skagit River flows directly into the marine waters near the Crescent Harbor EOD Training Range site. We expect bull trout from the Snohomish/Skykomish and Stillaguamish core areas will also be present, though to a much lesser degree due to the farther distance and smaller population sizes.

Crescent Harbor is located near the Skagit River estuary and the shallowness of Skagit Bay allows large juveniles, sub-adults and adults to migrate towards Whidbey Island and Crescent Harbor. The Crescent Harbor EOD Training Range site is no closer than 1,000 meters from shore to minimize increased underwater exposure levels to salmonids. While there is evidence that bull trout tend to stay near the shore in shallow marine water (less than 4 m deep; Hayes et al. 2011, pp. 394, 403-404), radio-tagged bull trout have been documented crossing areas of Puget Sound that are more than 183 m (600 ft) deep (Goetz et al. 2012). Adult bull trout have been caught within Crescent Harbor and the surrounding marine waters from April through July, all in shallow water near shore. However, since bull trout have been documented crossing waters deeper than 600 ft, bull trout habitat is not limited to near-shore shallow waters. When adult bull trout return to spawn in the freshwater in July and August, bull trout density decreases in the marine environment during the period of August through November each year. The remaining large juveniles and sub-adult bull trout likely will be concentrated near the estuaries and lower reaches of large river systems. However, we assume the use of explosives could occur any month of the year (with a limit of one EOD exercise during the winter period). Therefore, we have determined that bull trout are likely to be in habitat sufficiently close to explosions (see 10.3.2.1.5 below) to be exposed to the Navy's use of high explosive ordnance for underwater mine detonations at the Crescent Harbor EOD Training Range site.

The risk of exposure to these stressors varies annually, with highest risk occurring between December to August as adult bull trout inhabit the marine environment and lowest risk occurring between August and November when most adult bull trout are in the fresh water environment. Exposure will also be greater if/when stressors occur in shallow water or, as in the case of underwater detonations, high SPLs reach shallow nearshore habitat where bull trout occur in higher abundance.

10.3.2.1.5 Response of Bull Trout Exposure in Crescent Harbor

We expect that bull trout will be exposed at Crescent Harbor to the effects of underwater detonations in exceedance of our established thresholds. We expect bull trout to be injured or killed as a result of these exposure. The Service estimated the number of bull trout that may be injured or killed based on the number of detonations, the detonation site, and the month of the detonations provided by the Navy. At the Crescent Harbor EOD Training Range site, the Navy proposes to detonate up to eighteen E0 and three E3 charges per year. The E0 explosives are a highly focused single charge consisting of less than 0.1 lb. of explosive. The Navy calculated the distances to the Service-established bull trout injury and mortality thresholds for both explosives (see Table 12). Based on those distances, we expect bull trout to be injured when they are within 661 m of E0 explosions and within 1,674 m of E3 explosions.

For E0 charges, using a 661 m radius, the amount of bull trout habitat where injury to bull trout could occur is approximately 1,372,628 m^2 (1.4 km^2/0.4 nm^2) for each E0 detonation. Using a radius of 1,674 m, we determined the amount of bull trout habitat exposed to injurious pressure waves will be approximately 8,803,610 m^2 (8.8 km^2/2.6 nm^2) for each E3 detonation.

Bull trout in habitat within these ranges (661 m for E0 and 1,674 m for E3) when the explosions occur will be injured and may be killed by the pressure caused by the explosion. A total of approximately 826 km^2 (241 nm^2) of bull trout habitat (i.e., the sum of all individual areas of effect) will be affected by these detonations over 16 years.

10.3.2.1.6 Conclusion for Bull Trout Exposure to EOD in Crescent Harbor

Because injured or dead bull trout are hard to detect, we used the area of effect for E0 and E3 detonations to determine where bull trout will be injured or killed. The Service expects bull trout within 661 m of E0 detonations and 1,674 m of all E3 detonations will be injured or killed as a result of increased SPLs resulting from underwater detonations at the Crescent Harbor EOD Training Range site.

10.3.2.2 Marbled Murrelet Exposure

The area of exposure for underwater explosions is defined by the distance from the explosion source at which injury of a murrelet is likely to occur (i.e., the range to effect). That distance is related to the specific net explosive weight of the charge and is discussed above in section 10.3.1.2.

The Navy will detonate EOD explosives on-command once pre-detonation surveys are completed. The pre-detonation surveys will reduce, but not eliminate, the possibility of murrelets being within the area where injury may occur. There is no quantitative information on the effectiveness of the Navy's pre-detonation surveying efforts. Without that information it is impossible to accurately determine a percentage-based level of effectiveness for implementation of their monitoring effort. For context, the Service does have an established protocol for monitoring for murrelet presence during impact pile driving projects (USFWS 2013, entire). That protocol requires certified observers and its development was informed by research on observer effectiveness in similar situations. When that pile-driving protocol is applied as intended, we expect that it is 78 percent effective at detecting murrelets within the survey area. Murrelets are small and cryptic and thus, difficult to detect – especially amongst any waves, chop, or glare, or at distances greater than 50 m. In the absence of effectiveness data on the Navy's pre-detonation survey approach in the marine environment, and based on a comparison between their method and our established protocol for monitoring for murrelets during pile driving projects, we estimate that the Navy's monitoring efforts are 50 percent effective. We expect that the Navy will observe half of the birds actually present during surveys while monitoring according to their current survey methodology.

We previously determined that murrelet exposure to <E1 detonations (now categorized as E0) was not reasonably certain to occur and is therefore not an effect of the action. After reviewing the current information regarding the range to effect for these detonations, and updated information regarding murrelet density, we have concluded that this determination remains valid.

E3 explosives will be detonated at a depth of 22.9 m in Crescent Harbor, creating a radius of effect where injury of murrelets may occur that extends 238 m from the source of the explosion. Therefore, each detonation of E3 explosives in Crescent Harbor will expose 177,952 m^2 (0.18 km^2/0.05 nm^2) of murrelet habitat to an injurious pressure impulse. Detonations in Crescent Harbor will occur annually within the same general area (Figure 2). This area is approximately 1,200 m wide and 2,400 m long (total area of approximately 2.88 km^2/0.84 nm^2) and is illustrated by the shaded rectangle in Figure 2. We assume that the detonations may occur anywhere within this rectangular area and the effects may extend a maximum of 238 m from the outer limits for the largest explosion (up to 2.5 lb. for E3). Therefore, all effects of this stressor will be geographically restricted to the zone where these detonations may occur, plus a 238-m buffer from its edges, a total area of 3.79 km^2/1.11 nm^2. At this location, there will be three detonations per year, and assuming that they do not occur in exactly the same location every time, as much as 0.53 km^2/0.16 nm^2 of habitat (three times the area of effect for a single detonation) may be exposed to these stressors each year. Over the entire 16 year period, the total cumulative area affected at Crescent Harbor (i.e., the sum areas of effect for every individual explosion) will be 8,541,696 m^2 (8.5 km^2/2.5 nm^2). As this total exposed area is larger than the area in which explosions will be detonated it is likely that at least portions of the area will be exposed repeatedly to impulses from E3 detonations.

Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community

Figure 2. Location of explosions within Crescent Harbor will occur within the same area every year (shaded polygon, estimated based on information provided by the Navy).

E3 explosives will be detonated at a depth of 12.2 m in the Hood Canal EOD Training Range, creating a radius of effect where injury of murrelets may occur that extends 183 m from the source of the explosion. Therefore, each detonation of E3 explosives in Hood Canal will expose 105,209 m^2 (0.11 km^2/0.03 nm^2) of murrelet habitat to an injurious pressure impulse. Detonations in Hood Canal will also occur within the same general area annually (Figure 3), and this area is smaller and the detonation location is more precise than in Crescent Harbor. This area is a circle, approximately 300 m radius (total area of the circle is approximately 282,600 m^2/0.28 km^2/0.08 nm^2) and is illustrated by the yellow dot in Figure 3. We assume that the detonations may occur anywhere within this area and the effects may extend a maximum of 183 m from the outer limits for the largest explosion, for a maximum of 483 m radius circle (2.5 lb. for E3). Therefore, all stressors associated with these detonations will be geographically limited to a 483 m radius circle, with an area of 732,899 m^2 (0.73 km^2/0.21 nm^2). Annually, there are three detonations and the exact location would vary slightly, but assuming that all are detonated within the 300 m radius circle, the areas of effect are likely to overlap. Therefore, within a given year, the geographic area exposed to effects will be less than 0.33 km^2/0.09 nm^2 of habitat (three times the area of effect of a single detonation). Over the entire 16 year period, the total cumulative area affected in Hood Canal (i.e., the sum areas of effect for every individual explosion) will be 5,050,032 m^2 (5.1 km^2/1.5 nm^2). As this total exposed area is much larger than the area in which explosions will be detonated it is likely that portions of the area will be exposed repeatedly to impulses from E3 detonations.

Figure 3. Location of explosions within the Bangor EOD site will occur within the same area every year, shown by the yellow dot.

Murrelet density in Inland Waters was determined using data from the Northwest Forest Plan Effectiveness Monitoring (NWFPEM) effort (McIver et al. 2021, pp. 11-17) and murrelet monitoring performed by the Washington Department of Fish and Wildlife (WDFW) (Pearson and Lance 2016, p. 12; Pearson and Lance 2017, p. 12; Pearson and Lance 2018, p. 13; Pearson and Lance 2019, p. 14; Pearson and Lance 2020, p. 14). Based on the location, frequency, and duration of the E3 EOD detonations in Inland Waters, using the threshold distances discussed above, the assumed effectiveness of pre-detonation monitoring, and the densities of murrelets in the areas, we estimated the number of murrelets likely to be exposed to, and injured from, E3 EOD detonations in the Inland Waters. The expected number of murrelets expected to be exposed to stressors from E3 EOD explosions is shown below in Table 16. Note that these are numbers of individual murrelets, not groups, which we expect to be exposed.

Table 16. Expected direct exposure of marbled murrelet individuals to injurious impulses from explosions in the Inland Waters over the entire proposed action.

Explosive Bin	Location	Expected Marbled Murrelets Directly Exposed
E3	Crescent Harbor	2.4
E3	Hood Canal	1.0

10.3.2.3 Marbled Murrelet Response to EOD Exposure

E3 underwater detonations could potentially injure or kill adult and subadult murrelets by exposing them to underwater impulses. We expect that if birds are exposed to E3 EOD detonations, the detonations will affect adult sub-adult, or fledged juvenile murrelets through impulse-related stressors (i.e., blast waves, elevated SPLs, overpressures and underpressures, etc.), resulting in auditory injury, barotrauma, or mortality.

One type of auditory injury that individual murrelets may experience from exposure to explosions is damage to the hair cells in their inner ears, causing “threshold shift” (TS) when there is decreased hearing capability at specific thresholds. Threshold shift can last for hours or be permanent. Murrelets experiencing threshold shift may not be able to detect biologically relevant sounds such as approaching predators or prey, and/or hear their mates attempting to communicate. Birds with reduced hearing sensitivity are at increased risk of predation and reduced foraging efficiency. Some birds may regain some or all of their hearing sensitivity; however, they are still temporarily at risk while experiencing TS.

Individual murrelets exposed to explosions may experience lethal or non-lethal injuries. Non-lethal injuries may include TS, scarred or ruptured eardrums, or gastrointestinal tract lesions. Individual murrelets may survive their exposure to the explosions; however, we expect such individuals to have a reduced level of fitness and reproductive success, and a higher risk of predation by reducing their ability to detect and/or evade predators. Exposed individuals may also experience lethal injuries that occur instantaneously or manifest over time, such as direct mortality, lung hemorrhaging, ruptured liver, hemorrhaged kidney, ruptured air sacs, and/or coronary air embolisms. Death from barotrauma can be instantaneous, occurring within minutes after exposure, or several days later (Abbott et al. 2002). Several birds exposed to explosions survived and appeared uninjured, but upon necropsy two weeks later there was evidence of liver blood clots and lung and kidney injuries (Yelverton et al. 1973, p. 51).

Individual murrelets that are beyond the area of injurious pressure impulses (beyond the ranges to effect listed above and therefore not injured or killed), may still exhibit a startle response, flushing, or avoidance (i.e., diving, or leaving the area). In uninjured individuals, these responses would be short term and we would not expect significant disruptions to their normal behavior that would create a likelihood of injury. However, if several detonations occurred per day, it may result in significant disruptions to a murrelet’s normal foraging behavior, potentially reducing individual fitness or their ability to feed a chick. For underwater detonations at the Hood Canal and Crescent Harbor EOD Training Range sites, we do not expect significant disruptions to normal behaviors because the associated stressors are of short-duration and do not occur frequently in a day or for an extended period of time. We expect that if a murrelet is not injured or killed by the detonation, they will return to normal behaviors in a short period of time.

Therefore, based on our exposure analysis and the fact that these detonations will occur over the next 16 years, we expect that 3.3 murrelets are reasonably certain to be exposed to, and injured or killed by, E3 underwater detonations.

10.3.2.4 Effects to Eggs and Chicks Resulting from Exposure of Nesting Adults

Nestlings and chicks are not found at sea, and will not be affected by the explosions themselves, but we expect that nestlings and chicks will be affected whenever a breeding adult is killed or injured as the result of an explosion. Both parents are needed to incubate murrelet eggs and bring food to murrelet chicks. The death of either parent will lead to the egg or chick not receiving adequate thermoregulation or food, and the egg or chick will die.

If one parent is injured, even if the parent later recovers from the injury, we expect that this parent will have decreased ability to fly inland safely, to forage for itself, or to forage for a nestling. When an incubating adult is injured, we expect that the injured bird will be less able to complete its 24-hour incubation shifts, which will result in the egg being left unattended for periods of time. Eggs that are left unattended are exposed to cooling, heating, drying, and predation, all of which can cause mortality. If injury occurs during the nestling stage, we expect that it will lead to reduced feedings of nestlings. Depending on the degree of reduction in feeding, nestlings will experience delayed fledging, stunting, or death.

During chick rearing, adults feed the young 1 to 8 times per day (mean = 3.2 ±1.3 SD) (Nelson and Hamer 1995, p. 61). With an average of 3.2 feedings per day, a single aborted feeding would often constitute a loss of 33 percent of that day's food and water intake for the nestling. Fish-eating alcids (e.g., murrelets, *Brachyramphus spp*.; and puffins, *Fratercula spp*.) exhibit wide variations in nestling growth rates, and murrelet chicks grow rapidly compared with other alcids (Nelson and Hamer 1995, p. 60). The nestling stage of murrelet development can vary from 27 to 40 days before fledging (DeSanto and Nelson 1995, p. 45). Young murrelets that receive multiple daily feedings grow faster and fledge earlier than those with lower provisioning rates. Early fledging helps minimize nest mortality (Nelson and Hamer 1995, p. 66).

The variations in alcid development are attributed to constraints on feeding ecology, such as specialized foraging behaviors, unpredictable and patchy food distributions, and great distances between feeding and nesting sites (Oyan and Anker-Nilssen 1996, p. 830). Food limitation often results in poor growth, delayed fledging, increased mortality of chicks, and nest abandonment by adults (Oyan and Anker-Nilssen 1996, p. 836). Growth rates of body mass and skeletal elements in alcids are strongly affected by rates of food intake; and low rates of daily food intake result in a significant increase in the duration of chick development time (Kitaysky 1999, p. 466). Some alcids respond to reduced provisioning by slowing their metabolic rates and allocating growth to the head and wings to facilitate successful fledging (Oyan and Anker-Nilssen 1996, p. 830; Kitaysky 1999, p. 470). Murrelets also exhibit this adaptive behavior by prioritizing wing and bill growth in the nest and delaying the development of fat stores to post-fledging development

(Janssen et al. 2011, p. 859). This is believed to be an adaptive strategy to reduce the length of the nestling period while maintaining a high probability of successful fledging and survival immediately after fledging (Janssen et al. 2011, p. 866). However, murrelets may already use this developmental flexibility to its limit in responding to a shift in prey availability.

Contemporary studies of murrelet diets in the Puget Sound–Georgia Basin region indicate that Pacific sand lance (*Ammodytes hexapterus*) (sand lance) now comprise the majority of the murrelet diet (Gutowsky et al. 2009, p. 251). Historically, energy-rich fishes such as herring (*Clupea pallasii*) and northern anchovy (*Engraulis mordax*) comprised the majority of the murrelet diet (Becker and Beissinger 2006, p. 470; Gutowsky et al. 2009, p. 247). This is significant because sand lance have the lowest energetic value of the fishes that murrelets commonly feed on. For example, a single northern anchovy has nearly six times the energetic value of a sand lance of the same size (Gutowsky et al. 2009, p. 251), so a chick would have to eat six sand lance to get the equivalent energy of a single anchovy. Lower caloric value food resources increases the significance of missed feeding events. Assuming nestlings receive an average of three single-fish feedings per day (Nelson and Hamer 1995, p. 61), a nestling being fed a low-quality diet comprised primarily of sand lance may be on the edge of its energetic needs for successful development. Nestlings have minimum daily energetic demands to sustain life and development, and mortality from starvation occurs when nestlings do not receive sufficient food (Kitaysky 1999, p. 471).

A study conducted over 2004 to 2008 of 157 radio-tagged murrelets in Washington found that of 20 confirmed nesting attempts, only 4 nests were successful, indicating a very low nesting rate and low nesting success (Lorenz et al. 2017, p. 310). When information regarding the cause of nest failure was available, failures during incubation typically occurred when the adults abandoned the egg, and failures during the nestling state were typically consistent with starvation (Lorenz et al. 2019, Appendix 2). These findings indicate that murrelets in Washington are not initiating nesting or are abandoning their nests during incubation or chick rearing, possibly in response to poor foraging conditions. For those murrelets that do initiate nesting and begin chick rearing, the implications of missed feedings are significant, because each missed feeding represents a delay in the development of the chick, prolonging the time to fledging and increasing the risk of predation, accidental death from falling off the nest, or abandonment by the adults. As outlined above, chicks may frequently already be affected by inadequate nutrition, simply due to baseline conditions, and in this case additional reductions in feeding due to the injury of a parent would increase the likelihood of mortality from starvation.

In a recent population viability analysis, Washington murrelet population trends could be simulated when 40 percent of the simulated population consisted of breeding adults, each of which attempted breeding in 9 out of 10 years (Peery and Jones 2019, pp. 9, 14, 26). In other words, 36 percent of the population would make a nesting attempt in any given year. Of the expected 3.4 individuals injured or killed by E3 EOD detonations, we expect that 2.5 individuals will be injured or killed during the breeding season. Combining this expectation with the likelihood of nesting attempts, our average expectation is that that 0.90 eggs or chicks would be affected over 16 years by the injury or mortality of a parent, and that these eggs or chicks are likely to die or suffer from some combination of stunting, delayed fledging, and increased risk of predation.

10.3.2.5 Conclusions of Effects of EOD to Marbled Murrelet

Based on our exposure analysis, 3.4 murrelets will be exposed to, and injured or killed by, E3 underwater detonations in the Inland Waters over the next 16 years. Because we expect that some of this injury and mortality will affect adults during the breeding season, we expect that 0.90 eggs or chicks will die or suffer developmental injuries over the next 16 years following the injury or death of a parent.

Our model included explicit assumptions about the seasonal distribution of murrelets and the extent of the potential effects. The areas of effect for each individual detonation are 177,952 m^2 (0.18 km^2/0.05 nm^2) in Crescent Harbor and 105,209 m^2 (0.11 km^2/0.03 nm^2) in Hood Canal. Over the next 16 years, the total cumulative area affected (i.e., the sum of all individual areas of effect) will be 13,591,746 m^2 (13.6 km^2 or 3.96 nm^2). However, all detonations occur within the same general location within the each of the Hood Canal and Crescent Harbor EOD Training Range sites; therefore, the effects will be confined to a geographic area of 4.52 km^2 (1.32 nm^2) across two sites, and many portions of this geographic area will be affected repeatedly.

10.3.3 Mine Countermeasure and Neutralization Testing

A new testing activity, mine countermeasure and neutralization testing, will occur closer to shore than other activities previously analyzed in the 2016 and 2018 Opinions that involve the use of in-water explosives in the offshore area. Mine countermeasure and neutralization testing involving the use of explosives will occur approximately two times per year (two multi-day events with 1-10 days per event), in waters 3 nm or greater from shore at the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nm or greater from shore elsewhere in the offshore area (note: mine countermeasure and neutralization testing will not occur off the California coast). Up to 36 E4 explosives and five E7 explosives will be used per year, with a maximum of 108 E4 and 15 E7 explosives used over a seven year period and a maximum of 246 explosives (216 E4, 30 E7) used through July 21, 2036. Explosives will be used in the water column (i.e., not on the sea floor), with E4 explosives used at 20 ft or greater below the water surface and E7 explosives used at 100 ft or greater below the water surface. Half of the E4 explosives will be detonated at distances greater than 4.6 nm from shore and half 3 nm or greater from shore; and E7 explosives will be detonated 7.5 nm or greater from shore. The maximum sea floor depth for explosive testing will be approximately 1,000 ft.

10.3.3.1 Bull Trout Exposure

In 2019, researchers from the Northwest Fisheries Science Center captured and tagged (with active sonic tags) 17 bull trout (6 from the Hoh River and 11 from Kalaloch Creek). In 2020, one of the tagged bull trout from Kalaloch Creek was detected by a receiver, 5 nm from shore in the area between the Queets and Quinault Rivers (Smith and Huff 2020, p. 3). Based on this data, the Service expects that large juvenile, sub-adult, and adult bull trout bull trout are likely to be present during mine countermeasure and neutralization testing activities that occur within 6 nm of shore in the offshore area. Larger juveniles and sub-adult bull trout are present in marine waters throughout the year and adults typically enter marine waters each year in December and January following spawning in freshwater. The adults typically remain in marine waters until

July and August, when they leave and migrate to freshwater streams to spawn. Bull trout abundance is expected to vary daily and seasonally as a function of several interacting factors, including the proximity of core areas, abundance/availability of forage, distance from shore, and the time of year (life-cycle stage).

The effects to bull trout from mine countermeasure and neutralization testing activities will be similar to those considered and analyzed in the 2016 Opinion (pgs. 126-132, 144-154, 206, 243), with the notable difference being the use of underwater explosives in waters 3 nm or greater from shore at the Quinault Range Site under the current proposed action. Because the use of underwater explosives in this area was not specifically considered in the 2016 Opinion, we will apply the same concepts, assumptions, and analytical framework used to analyze the effects to bull trout from underwater explosives from that Opinion (pgs. 144-154) to the use of underwater explosives in the area where mine countermeasure and neutralization testing activities will occur (i.e., in waters 3 nm or greater from shore at the Quinault Range Site). The difference between the 2016 Opinion and the new mine countermeasure and neutralization testing activities in the offshore area relates to what we consider an independent exposure event. In the 2016 Opinion we considered groups of explosions to be independent events exposing bull trout to stressors. However, because underwater explosives during mine countermeasure and neutralization testing events are larger than what was analyzed in the 2016 Opinion, in the current analysis we considered each individual explosion to be an independent event that would expose bull trout to stressors.

As in the 2016 and 2018 Opinions, we assumed that all use of explosives will occur during the nine month period when bull trout density in the marine environment is highest (December through August). We also assumed that one large juvenile, sub-adult, or adult bull trout will be present within the radius of the detonation site for each E4 explosive event. Additionally, we utilized the maximum estimated ranges to injury (TTS included) and mortality for E4 and E7 explosives provided in the Navy's Biological Assessment (Navy 2019, pp. 5-90 - 5-91).

Specifically, we assume that one large juvenile, sub-adult, or adult bull trout will be present within the 1,829 m (6,001ft) radius of the detonation site for each E4 detonation. Bull trout are not expected to be within the radii of E7 detonations because those detonations will all occur at least 7.5 nm from shore and the range to injury for E7 explosives (4,259 m) will not extend to 5 nm from shore where bull trout have been detected. Given a maximum of 216 total E4 mine countermeasure and neutralization testing detonations through July 21, 2036, we estimate that over the next 16 years, a total of 2,270 km^2 (661 nm^2) of bull trout FMO habitat (i.e., the sum of all individual areas of effect) will be affected by these detonations over the next 16 years.

10.3.3.1.1 Conclusion

Because injured or dead bull trout are hard to detect, we used the area of effect for E4 detonations to determine when bull trout will be injured or killed. The Service expects bull trout within 1,829 m of E4 detonations will be injured or killed as a result of increased SPLs resulting from underwater detonations at the Quinault Range Site.

10.3.3.2 Marbled Murrelet Exposure

The nature of the effects to murrelets from mine countermeasure and neutralization testing activities will be similar to those considered and analyzed in the 2016 Opinion (pp. 154-158, 164-168, 206, 243); however, the level of exposure will differ in the current proposed action, due to the proposed location of these activities within the Quinault Range Site in waters >3 nm from shore. The use of underwater explosives in this area was not specifically considered in the 2016 Opinion.

In estimating murrelet exposure to injurious pressure levels from underwater explosives associated with mine countermeasure and neutralization testing, we assumed explosives use would occur: 1) at the maximum frequency, 2) in the area of highest murrelet density, within the area where explosives would be used, and 3) during the murrelet breeding season. The maximum number of explosives to be used each year is 41 (36 E4, 5 E7), with a maximum of 123 (108 E4, 15 E7) in a 7-year period. By July 21, 2036, we anticipate 16 seasons of mine countermeasure and neutralization testing, including of a total of 246 explosive detonations (216 E4, 30 E7). This is equivalent to one 7-year period with maximum 7-year usage, plus an additional 9-year period with no more than the maximum 7-year usage (a maximum of 123 detonations, 108 E4 and 15 E7). As described above, half of the E4 explosives may be detonated as close as 3 nm from shore, while the other half will be detonated at least 4.6 nm from shore in the Quinault Range Site. As the action is currently described (Navy 2020, in litt., p. 1), E7 explosives are detonated in water at least 175 ft deep, which can be found no closer than 10 nm from shore in the Quinault Range Site; however, the Navy has also described this activity as occurring at least 7.5 nm from shore. Therefore, we analyze effects of E7 explosions as if all occurred at 7.5 nm from shore. The Navy has stated that most mine countermeasure and neutralization testing activity will take place between July 1 and September 30 (Navy 2020, p. K-8), substantially overlapping with the murrelet breeding season. Although it is possible that some mine countermeasure and neutralization testing will occur during the non-breeding season, it is also reasonably likely that all of these activities will occur during the breeding season, especially given the need for relatively calm seas (Beaufort sea state of 3 or less) during these activities (Navy 2020, p. K-34).

As in our 2016 analyses of exposure to underwater explosives, we estimated the number of murrelets likely to be exposed to injurious pressure levels based on the total number of explosions, the area affected by each explosion, the estimated population density of murrelets in those areas, and the proportion of murrelets likely to be underwater at the time of the explosion. We used updated range-to-effect distances based on Yelverton 1973 (Equations 2-5, p. 15) taking into account the weight of the explosive, depth of explosive, depth of murrelet dives, and the 36 Pa-sec barotrauma threshold for murrelets. We expect murrelets to experience barotrauma if they are within 347 m of an E4 explosion or within 1,338 m of an E7 explosion. In our 2016 analysis, we previously calculated an exposure probability of 0.99 for E4 explosives use in the offshore area. Since this calculation applied to a similar number of explosions in an area of much lower murrelet density, we assume that murrelets will have at least a 99 percent chance of becoming exposed to E4 and E7 explosions associated with mine countermeasures and neutralization testing.

We estimated murrelet densities separately for E4 explosions detonated at least 3 nm from shore, E4 explosions detonated at least 4.6 nm from shore, and E7 explosions, since these explosions take place at different distances from shore. Within 8 km (4.3 nm) of shore, NWFPEM at-sea surveys offer the best available information regarding murrelet densities during the breeding season (McIver et al. 2021, pp. 11-17). Off the outer coast of Washington, approximately 95 percent of murrelets are expected to be present within 8 km of shore (Bentivoglio et al. 2002, pp. 29-31). Outside of this area, up-to-date information regarding murrelet density is not available, and instead a model of murrelet density (Menza et al. 2015, p. 20) provides the best available information regarding murrelet densities during the breeding season. This model of murrelet marine habitat use is highly sensitive to the survey methodology used to collect the data on which the model is based (Menza et al. 2015, pp. 16, 20-21, 49), and therefore may not represent an unbiased map of true murrelet distribution in the marine environment; however, we lack better information regarding murrelet density in areas farther than 8 km (4.3 nm) from shore.

To estimate murrelet density 3 nm (5.6 km) from shore, we averaged Zone 2 murrelet density estimates from the last five years of surveys (2013-2015, 2017, and 2019) to reach a mean estimated density of 1.22 murrelets/km^2 (McIver et al. 2021, pp. 15-16). We considered using only the Zone 2, Stratum 2 estimates to make our average, since the portion of the Quinault Range Site that falls outside of the Olympic Coast National Marine Sanctuary (OCNMS) and within 8 km of shore is within Stratum 2. Stratum 1 of Zone 2, which is mainly within the OCNMS, has consistently higher densities of murrelets than Stratum 2, and at first glance, using an average from the entire Zone could lead us to overestimate the exposure of murrelets to mine countermeasure and neutralization testing activities. However, for the following reasons, we determined that the average density for all of Zone 2 offered the more appropriate density estimate for this area: 1) Within Stratum 2, murrelet density is lower in the south and higher in the north, and the mine countermeasure and neutralization testing activities will take place within the northern portion of Stratum 2. Therefore, using a Stratum 2 density to estimate murrelet exposure to explosions is likely to underestimate effects. 2) Although we do not have year-by-year NWFPEM murrelet density estimates at scales finer than the Stratum, an analysis of the survey data from 2001-2012 shows estimates in the range of 0.9-2.4 murrelets/km^2 in this area (Raphael et al. 2015, p. 21). Murrelet density in these areas was less than 1 murrelet /km^2, averaged across 2009 through 2014 (MMEMM 2015), the same was true of the density estimate for all of Zone 2, averaged across the same years (McIver et al. 2021, pp. 14-15). 3) Murrelet density estimates in Stratum 2 vary, year-to-year, over three orders of magnitude, and the last three years of survey include both the second-highest and second-lowest Stratum 2 estimates (McIver et al. 2021, pp. 11-17). Additionally, many of the estimates have wide confidence intervals. This uncertainty and variability reflect actual year-to-year variation in the population size, as well as sampling error associated with the patchy distribution of murrelets at sea and the relatively low intensity of survey efforts (McIver et al. 2021, pp. 4; Raphael et al. 2015, pp. 21-22). For all of these reasons, we conclude that the best available information regarding murrelet density in the portion of the Quinault Range Site nearest to shore comes from the estimates of Zone 2 density as a whole, rather than from estimates specific to Stratum 2. At 3 nm from shore we expect a murrelet density of 1.22 murrelets/km^2.

To estimate murrelet density at 4.6 and 7.5 nm (11.1 km) from shore, we examined a Geographic Information System (GIS) representation of the murrelet density map modeled by Menza and others (2015, p. 20). Within the Quinault Range Site, at approximately 4.6 nm (11.1 km) from shore, and south of the OCNMS, this map depicts a murrelet density of approximately 0.26 murrelets/km^2. In that same area, but 7.5 nm from shore, Menza et al. (2015, p. 20) depicts a density of 0.07 murrelets/km^2.

In our 2016 analysis of effects that were reasonably certain to occur, we anticipated that the murrelet population would decline steadily from 2013 population estimates. However, in Zone 2, murrelet population estimates have exceeded the 2013 population estimate for every year since then (McIver et al. 2021, pp. 15-16; and see 2016 Opinion, Appendix A, pp. 12-13). Given the high year-to-year variability in the Zone 2 murrelet population, as well as the uncertainty associated with population trends over time, we risk underestimating exposure if we assume that the murrelet population will steadily decline, because the number of individuals exposed increases with the population density. Therefore, our analysis is based on the assumption that average population density over the next 16 years will be similar to the 2013-2019 average population density.

We expect that these explosions will cause significant effects only to those birds that are underwater at the time of the explosion. As in our 2016 analysis, we assume that half of the murrelets in the area of the explosion will be underwater at any given time. This is a conservative assumption accounting for the amount of time spent resting on the water over the course of a day, the amount of time spent diving during active foraging, and the possibility of avoidance diving in the presence of Naval vessels (USFWS 2016, Appendix A, p. 13). In keeping with this assumption, we divide the density estimates by half to represent the density of murrelets that will be affected by each explosion.

We calculate our estimates of the number of murrelets exposed by multiplying the area affected by each explosion by the underwater density of murrelets in that area, and then multiplying this value by the number of detonations anticipated before July 21, 2036. We anticipate that during this period, 24.9 murrelets will be exposed to an underwater impulse causing barotrauma as a result of E4 explosions detonated no closer than 3 nm from shore. We expect 5.3 murrelets to be exposed to barotrauma-inducing impulses from E4 explosions detonated no closer than 4.6 nm from shore. E7 explosions farther than 7.5 nm from shore will expose 5.9 murrelets to impulses causing barotrauma. In all, 36.1 individual murrelets at sea will be underwater within the range to effect of an E4 or E7 explosive associated with mine countermeasure and neutralization testing activities. See Table 17 below for a summary of these results.

Table 17. Summary of marbled murrelet exposure to mine countermeasure and neutralization explosions.

	Radius of barotrauma effect (m)	Area of barotrauma effect (km^2)	Total explosions (16 years)	Total area exposed (km^2)	Marbled murrelet density (birds / km^2)	At-sea murrelets exposed (16 years)
E4 ≥ 3 NM	347	0.378	108	40.85	1.22	24.9
E4 ≥ 4.6 NM	347	0.378	108	40.85	0.26	5.3
E7 ≥ 7.5 NM	1,338	5.624	30	168.72	0.07	5.9
Totals				**250.43**		**36.1**

We expect that exposure to underwater explosions will affect males, females, and murrelets of all life stages, in proportion to their density at sea. As noted in Section 10.3.2.4, nestlings and chicks are not found at sea, and will not be affected by the explosions themselves, but we expect that nestlings and chicks will be affected whenever a breeding adult is injured or killed as a result of an explosion. As explained above, we estimate that 36 percent of the population will make a nesting attempt in any given year. Applying this assumption to the 36.1 individuals injured or killed by mine countermeasure and neutralization testing activities, we estimate that 13 eggs or chicks would be affected. As explained above, we expect that affected eggs would be exposed to cooling, heating, drying, predation, or parental abandonment, any of which can cause mortality. We expect that affected nestlings will experience reduced feedings, resulting in delayed fledging, stunting, or death from predation or starvation. Given that most mine countermeasure and neutralization testing activities will occur in July through October, it is likely that a proportion of the young of the year will already have fledged by the time the activities occur. We do not explicitly account for this proportion of already-fledged young in our estimate of affected eggs and nestlings, because doing so would require making further assumptions about the timing of mine countermeasure and neutralization testing activities, and detailed calculations based on what is known about the timing of murrelet nesting phases (USFWS 2012a), but it is not clear that these efforts would improve the accuracy of our estimate. Our estimate of 13 eggs or chicks affected by the injury or death of a parent is a based on breeding rates estimated in a recent, geographically-relevant study, which we regard as the best available locally-relevant information at this time. However, empirical estimates of the proportion of murrelets breeding each year range from 13 percent, measured in Washington (Lorenz et al. 2017, p. 316), to 84 percent, measured in southeastern Alaska (Barbaree et al. 2014, p. 177). Variation in these estimates is partly due to differences in measurement technique: estimates based on inland flight patterns observed through radiotelemetry are lower than estimates based on the presence of a well-developed brood patch at the time of capture, for example. The actual rates (and timing) of nesting attempts are likely to vary from year to year and from place to place. Therefore, our estimate of 13 eggs or chicks may be too high, if fewer than 36 percent of murrelets attempt nesting, or if many have completed their nesting attempts before mine countermeasure and neutralization testing activities occur in a given year; or it may be too low, if more than 36 percent of murrelets attempt nesting. Rates of breeding are discussed in greater detail in Appendices B and D.

Based on those assumptions and estimates, and given a maximum of 246 total mine countermeasure and neutralization testing detonation events (216 E4, 30 E7) through July 21, 2036, we estimate that over a 16-year period, a total of 36.1 individual murrelets are expected to be injured or killed by these detonations. An additional 13 eggs or chicks at the nest will be injured or killed due to of the injury or death of a parent resulting from one of these detonations. A total of 250.4 km^2 of marine murrelet habitat (i.e., the sum of all individual areas of effect) will be affected by these detonations over 16 years.

10.3.3.2.1 Conclusion

The expected value of 36.1 murrelets represents the number of individuals that are reasonably certain to be injured or killed from E4 and E7 underwater explosives within a cumulative area (i.e., the sum of all individual areas of effect) of 250.4 km^2 (73.0 nm^2) before July 21, 2036. This expected value includes 30.2 individuals within a cumulative area (i.e., the sum of all individual areas of effect) of 81.7 km^2 (23.8 nm^2) for E4 underwater explosives, and 5.9 individuals in within a cumulative area (i.e., the sum of all individual areas of effect) of 168.7 km^2 (49.2 nm^2) for E7 underwater explosives. Additionally, we expect that effects to 36.1 individual murrelets will in turn cause injury or mortality of 13 eggs or chicks.

10.3.3.3 Short-tailed Albatross Exposure

The effects to short-tailed albatross from mine countermeasure and neutralization testing activities will be similar to those considered and analyzed in the 2016 Opinion (pgs. 168-173), with the notable difference being the use of underwater explosives in waters 3 nm or greater from shore at the Quinault Range Site under the current proposed action. Because the use of underwater explosives in this area was not specifically considered in the 2016 Opinion, we will apply the same concepts, assumptions, and analytical framework used to analyze the effects to short-tailed albatross from underwater explosives from that Opinion (pgs. 168-173) to the use of underwater explosives in the area where mine countermeasure and neutralization testing activities will occur (i.e., in waters 3 nm or greater from shore at the Quinault Range Site).

As in the 2016 Opinion, we assumed that short-tailed albatross could be anywhere in the offshore portion of the action area at any time of year, (p. 171). Additionally, based on the distribution and density of short-tailed albatross, the location of detonations, and the calculated range to effects values for E1-E12 explosives, the probability that short-tailed albatross would be exposed to underwater explosions in the offshore area was estimated to be less than 0.10 for all explosives categories in the 2016 Opinion (p. 173). Finally, short-tailed albatross are primarily surface feeders, spending relatively little time with their heads underwater (USFWS 2016, p. 172). Given the above factors, short-tailed albatross are extremely unlikely to be exposed to effects from mine countermeasure and neutralization testing activities.

10.3.3.4 Conclusion

Based on density and distribution of short-tailed albatross in the offshore area, the extremely low likelihood of short-tailed albatross exposure to effects, and the fact that short-tailed albatross are primarily surface feeders, the effects to short-tailed albatross from mine countermeasure and neutralization testing activities are expected to be discountable.

10.4 Kinetic Energy Weapon Testing

The kinetic energy weapon will be tested aboard surface vessels, and involves firing explosive and non-explosive projectiles at air or sea-based targets.

All kinetic energy weapon testing activities will occur in the offshore area, greater than 50 nm from shore. Testing events will occur 4 times per year, with each event lasting one day. Up to 25 percent of testing may occur at night. An annual total of 80 kinetic energy (in-air) explosives (comparable to large-caliber explosive rounds) and 160 NEPM large-caliber projectiles will be used. One target will be expended per event.

10.4.1 Marbled Murrelet

Explosives and non-explosive projectiles associated with kinetic energy weapons testing will only occur farther than 50 nm from shore. We are not aware of records of murrelet occurrence in the action area, greater than 50 nm, off the coast of Washington, Oregon or California. Since it would be speculative to assume they occur at that distance we are not reasonably certain of murrelet presence farther than 50 nm from shore. Therefore, we conclude that murrelet exposure to effects associated with kinetic energy weapons testing is extremely unlikely, and therefore discountable.

10.4.2 Short-Tailed Albatross

Stressors from kinetic energy weapon testing are comparable to the large-caliber explosive rounds and the large-caliber NEPM activities that were specifically analyzed in the 2016 and 2018 Opinions. The effects to short-tailed albatross from kinetic energy weapons testing were therefore analyzed by adding occurrences of kinetic energy weapon testing to the corresponding activity (large-caliber explosive rounds and large-caliber NEPM) in the offshore area greater than 50 nm from shore. Therefore, as was concluded for the comparable activities, we conclude that short-tailed albatross will be exposed to stressors associated with kinetic energy weapon testing (physical strike or pressure wave) and that exposed short-tailed albatross will be injured or killed.

10.5 Simulant Testing

Simulant testing involves testing the capability of surface ship defense systems to detect and protect against chemical and biological attacks. Only chemical simulants with low toxicity to humans and the environment, (e.g., glacial acetic acid and triethyl phosphate) and Biosafety Level 1 organisms (e.g., spore-forming bacteria, non-spore-forming bacteria, the protein ovalbumin, MS2 bacteriophages, and the fungus *Aspergillus niger*) will be used during this activity. Simulant testing will occur up to 50 times per year in the offshore area, at least 3 nm from shore.

Biosafety Level 1 organisms are described by the Center for Disease Control as microbes that "are not known to consistently cause disease in healthy adults and present minimal potential hazard to laboratorians and the environment" (CDC 2020). Because simulant testing activities will only use Biosafety Level 1 organisms and chemical simulants with low toxicity, and given the low number of events relative to the area in which they will occur (50 per year throughout offshore area) the effects to bull trout, murrelets, and short-tailed albatross from implementation of simulant testing are expected to be insignificant.

10.6 Non-Explosive Gunnery Exercises

The effects of gunnery exercises are similar to those described in the 2016 Opinion, but refinements to the description of gunnery exercises using medium-caliber non-explosive projectiles, along with updated information regarding murrelet densities within the areas where these exercises will occur, alter our analysis from the previous Opinion.

In particular, the Navy provided new information indicating that 90 percent of Gunnery Exercises are proposed take place off of Washington, 5 percent off of Oregon, and 5 percent off of northern California (Kunz 2020a, in litt.). The Navy also provided new information indicating that only ten Surface-to-Surface Gunnery Exercises and five Surface-to-Air Gunnery Exercises are proposed within 50 nautical miles from shore during the October through March period each year (Kunz 2020a, in litt.; Kunz 2020b, in litt). Additionally, the Navy provided additional information regarding the sizes and types of medium-caliber non-explosive projectiles to be used (Kunz 2020a, in litt.).

10.6.1 Updates to Marbled Murrelet Density Estimates

Due to the updated information regarding the locations in which Gunnery Exercises will take place, it was not clear that our previous assumptions were adequate to estimate effects to murrelets in each Conservation Zone related to non-explosive projectiles. We previously assumed that Gunnery Exercises would occur with equal probability at any location within the offshore area, and furthermore that murrelets from all Conservation Zones would form a mixed population during the non-breeding season.

As a result of the new information regarding the locations of Gunnery Exercises, we were concerned that our previous assumption of a fully mixed population would lead us to underestimate exposure of individuals associated with the breeding season population of Zone 2. Therefore, we generated an alternative set of assumptions about the non-breeding season distribution of murrelets from Conservation Zones 2 through 6. However, we were also concerned that this alternate set of assumptions might lead us to underestimate the overall exposure of murrelets to stressors in the offshore area. As detailed below, these two sets of assumptions represent the two ends of a spectrum of possibilities regarding non-breeding season murrelet distribution, whereas the reality is almost certainly somewhere in between the two extremes. Therefore, we used model averaging, a technique widely used in the scientific literature when there is uncertainty about which model is best, or when multiple models capture different aspects of a system but no model is expected to be entirely accurate in itself (for example, climate models). We used the model average, as detailed below, to estimate murrelet exposure to each size of non-explosive projectile.

10.6.1.1 Offshore Non-breeding Marbled Murrelet Density Model 1

Model 1 (Table 18) uses assumptions similar to our previous assumptions regarding murrelet distribution and density within the offshore area (USFWS 2016, Appendix A). We assumed that all murrelets associated with the breeding season populations of Conservation Zones 2, 3, and 4, along with ten percent of the breeding season population of both Conservation Zones 5 and 6, would be present during the non-breeding season within 50 nm of shore off of Conservation Zones 2, 3, and 4. We assumed that they would form a mixed population, and we made no specific assumption about how murrelets would be distributed within this area, but rather calculated the average density by calculating the number of birds, then dividing by the total area. This model differs from that used in 2016 in that we averaged murrelet densities estimated from the last five years of NWFPEM surveys (McIver et al. 2021, pp. 11-17), corrected for the five percent of the population assumed to be outside of the survey area during surveys (Bentivoglio et al. 2002, pp. 22, 29, 34, 40) and assumed that the population size would remain more or less stable during the term of the action, rather than assuming a constant rate of population increase or decline.

As in 2016, we assumed that the population would be randomly distributed between the coastline and 50 nautical miles from the coast, and we continued this assumption. We have obtained some new information providing a small amount of support for the hypothesis that murrelet density is greater closer to shore than farther away during the non-breeding season (Pearson 2019, p. 5). However, due to the small number of murrelet observations even at distances nearer the shore, these surveys do not provide conclusive information that can be used to model a gradient of murrelet densities. We also obtained new information including several previously unavailable murrelet observations at distances between 12 and 40 nm from shore (Pearson 2019, p. 5, Drew and Piatt 2020). This new information is not detailed or abundant enough to allow us to model murrelet densities at different distances from shore during the non-breeding season.

Table 18. Calculations for Model 1.

Zone	**Average Population Size**	**Corrected for 5% not in survey area**	**Number in (or inshore of) offshore area**	**Area (km^2, nm^2)**	**Density (birds/km^2, birds/nm^2)**
2	2,013	2,119	2,119	n/a	n/a
3	8,061	8,486	8,486	n/a	n/a
4	7,023	7,393	7,393	n/a	n/a
5	235	247	25	n/a	n/a
6	441	464	46	n/a	n/a
Total	17,773	18,709	18,069	92,604 km^2, 26,888 nm^2	0.195/km^2, 0.67/nm^2

10.6.1.2 Offshore Non-breeding Marbled Murrelet Density Model 2

For Model 2 (Table 19), we assume that murrelets do not generally move north or south during the non-breeding season, but rather remain at the latitudes associated with their breeding season Conservation Zone. In other words, we assumed that birds associated with the breeding season population of Conservation Zone 2 would be present year-round in or offshore of Conservation Zone 2, birds associated with the breeding season population of Conservation Zone 3 would be present year-round in or offshore of Conservation Zone 3, and the breeding season population of Conservation Zone 4 would be present year-round in or offshore of Conservation Zone 4. According to the assumptions of Model 2, individuals associated with the breeding season populations of Conservation Zones 5 and 6 would not be present in (or shoreward of) the Navy's offshore area, which extends approximately as far south as the southern coastal boundary of Conservation Zone 4. In contrast to Model 1, Model 2 generated separate density estimates for each of the Conservation Zones 2 through 4. In all other ways, the assumptions of Model 2 were similar to those of Model 1.

Table 19. Calculations for Model 2.

Zone	Average Population Size	Corrected for 5% not in survey area	Area (km^2, nm^2)	Density (birds/km^2, birds/nm^2)
2	2,013	2,119	25,696 km^2, 7,492 nm^2	0.08/km^2, 0.28/nm^2
3	8,061	8,486	31,152 km^2, 9,082 nm^2	0.27/km^2, 0.93/nm^2
4	7,023	7,393	36,933 km^2, 10,768nm^2	0.20/km^2, 0.69/nm^2

10.6.1.3 Model Averaging

Model 1 assumes that the murrelet population is perfectly well-mixed during the non-breeding season, whereas Model 2 assumes no mixing whatsoever between the breeding season populations of different Conservation Zones. In reality, there is evidence of post-breeding-season movements to the north or south, toward more protected inland waters, and farther out to sea (Drew and Piatt 2020; Pearson 2019, p. 5; USFWS 2016, Appendix A, pp. 9-11), but it seems likely that at least some members of each Conservation Zone's breeding season population do remain in the same Zone, especially given observations of murrelets visiting nesting habitat during the winter (Naslund 1993, p. 596). Therefore, the reality of murrelet non-breeding season distributions likely lies somewhere between the distributions described in Model 1 and Model 2. To better reflect reality, we model murrelet exposure to offshore area stressors separately using the assumptions of Model 1 and Model 2, then average the expected number of birds exposed from each Conservation Zone. The exposure estimates for murrelets given below reflect this averaging.

We note that both Model 1 and Model 2 represent a uniform distribution of murrelets between the coast and 50 nm offshore, whereas in most locations, murrelets population density is generally assumed to be higher closer to shore (Nelson 1997, p. 3). However, we still lack certainty or sufficient information to support any specific model of a gradient between higher densities nearshore and lower densities offshore.

10.6.2 Small-caliber Non-explosive Projectiles

Murrelet and short-tailed albatross exposure to small-caliber non-explosive projectiles (SCNEPs) used in surface-to-surface gunnery exercises changed very slightly due to the Navy decreasing the number of rounds decreasing from an annual total of 121,200 rounds to an annual total of 121,000.

We repeated the effects analysis as described in the 2016 Opinion using the new information on the number of rounds and updated murrelet densities and short-tailed albatross population data. For more detail on those methods, refer to the 2016 Opinion (Section 10.4.5.4), except for the alterations discussed above.

10.6.2.1 Small-caliber Non-Explosive Projectile Effects to Marbled Murrelet

Gunnery exercises with SCNEPs will only occur farther than 12 nm from shore, where murrelet exposure during the breeding season is extremely unlikely to occur. Murrelets will be exposed to stressors from SCNEP gunnery exercises when those activities occur in the offshore area within 50 nm of shore during the winter. Of the 121,000 SCNEPs fired annually, we expect that 6,050 SCNEPs will be used in the winter (October – March) within 50 nm from shore. The Navy informed the Service that projectiles fired during gunnery exercises are typically fired in groups and assuming five projectiles per burst would approximate the grouping of projectiles (see USFWS 2016, Appendix A). Assuming bursts of five SCNEPs are a distinct opportunity for exposure, there will be 1,210 instances when murrelets could be struck by SCNEPs. As described in the 2016 Opinion (see Section 10.4.5.4 and Appendix A), each instance of SCNEP gunnery exercises will expose 0.004 km^2 (0.0011 nm^2) of murrelet habitat to stressors. In total over the 16 years of the proposed action, 70.8 km^2 (20.6 nm^2) of murrelet habitat will be exposed to SCNEPs. Using the model averaging explained above, we expect 5.2 murrelets will be struck by SCNEPs (also assuming birds will not be struck underwater). Of those 5.2 murrelets, we expect 1.7 to be from the Conservation Zone 2 population, 1.9 to be from the Conservation Zone 3 population, and 1.6 to be from the Conservation Zone 4 population. As a result of being struck by a SCNEP, murrelets will be injured or killed. We do not expect indirect effects to eggs or chicks from this exposure since this exposure will only occur in the winter when murrelets are not breeding.

10.6.2.2 Small-caliber Non-Explosive Projectile Effects to Short-tailed Albatross

Short-tailed albatross have the potential to be exposed to all gunnery exercises with SCNEPs. Assuming bursts of five SCNEPs are a distinct opportunity for exposure, there will be 24,200 instances when short-tailed albatross could be struck by SCNEPs. As described in the 2016 Opinion (see Section 10.4.5.4 and Appendix A), each instance of SCNEP gunnery exercises will expose 0.011 km^2 (0.003 nm^2) of short-tailed albatross habitat to stressors. In total over the 16 years of the proposed action, 4,259.2 km^2 (1,241.8 nm^2) of short-tailed albatross habitat will be exposed to SCNEPs. Updating the analysis used in the 2016 Opinion to include short-tailed albatross population data from the 2020 5-year review (USFWS 2020b, p. 4), the proposed action will result in 2.6 short-tailed albatross being struck by SCNEPs. The expected consequence of being struck by a SCNEP is that the short-tailed albatross will be injured or killed.

10.6.3 Medium-caliber Non-explosive Projectiles

The effects of medium-caliber non-explosive projectiles (MCNEPs) are similar to those described in the 2016 Opinion with a few adjustments. First, while the 2016 Opinion assumed that MCNEPs would be evenly allocated between four sizes of projectiles, the Navy has clarified that of all MCNEPs 7 percent will be 40mm, 53 percent will be 25mm, and 40 percent will be 20mm. Secondly, the Navy clarified that 40mm MCNEPs do not travel at supersonic speeds and they will therefore not generate a bow shock wave. We adjusted the area of effect for the 40 mm MCNEP accordingly. Lastly, the number of MCNEPs decreased from 9,672 to 9,660 used for surface-to-air exercises and from 33,492 to 16,750 used for surface-to-surface exercises.

We repeated the effects analysis as described in the 2016 Opinion using this new information and updated murrelet densities and short-tailed albatross population data. For more detail on those methods, refer to the 2016 Opinion (Section 10.4.5.4), and the alterations discussed above.

10.6.3.1 Medium-caliber Non-Explosive Projectile Effects to Marbled Murrelet

Gunnery exercises with MCNEPs will only occur farther than 12 nm from shore, where murrelet exposure during the breeding season is extremely unlikely to occur. Murrelets will be exposed to stressors from MCNEP gunnery exercises when those activities occur in the offshore area within 50 nm of shore during the winter. As with SCNEPs, we assume that a group of 5 projectiles will be a distinct opportunity for exposure to stressors from MCNEPs. Table 20 below summarizes the proportion of MCNEP activities that will expose murrelets to stressors.

Table 20. Annual marbled murrelet exposure to medium-caliber non-explosive projectiles.

Gunnery Exercise	Total number of MCNEPs	MCNEPs by caliber	MCNEPs < 50 nm from shore in winter	Instances of marbled murrelet exposure	Area of effect for each instance (km^2)	Total marbled murrelet habitat exposed annually (km^2)
Surface-to-air	9,660	20mm 3,864	155	31	0.004	0.42
		25mm 5,120	205	41	0.006	0.81
		40mm 677	27	6	0.002	0.01
Surface-to-surface	16,750	20mm 6,700	335	67	0.013	3.04
		25mm 8,878	444	89	0.018	5.43
		40mm 1,173	59	12	0.001	0.04

Over the 16 years of the proposed action, a total of 20.0 km^2 (5.8 nm^2) of murrelet habitat will be exposed to stressors associated with surface-to-air MCNEP gunnery exercises and 136.3 km^2 (39.7 nm^2) of murrelet habitat will be exposed to stressors from surface-to-surface MCNEP gunnery exercises. Using the model averaging outlined above, we expect 11.4 murrelets will be exposed to stressors from MCNEPs (also assuming birds will not be struck underwater). Of those 11.4 murrelets, we expect 3.8 to be from the Conservation Zone 2 population, 4.1 to be from the Conservation Zone 3 population, and 3.5 to be from the Conservation Zone 4 population. As a result of that exposure to a MCNEP, murrelets will be injured or killed. We do not expect indirect effects to eggs or chicks from this exposure since this exposure will only occur in the winter when murrelets are not breeding.

10.6.3.2 Medium-caliber Non-Explosive Projectile Effects to Short-tailed Albatross

Short-tailed albatross have the potential to be exposed to all gunnery exercises with MCNEPs. Assuming bursts of five MCNEPs to be a distinct opportunity for exposure, the 9,660 surface-to-air projectiles and 16,750 surface-to-surface projectiles will create 1,933 (surface-to-air) and 3,350 (surface-to-surface) instances (the number of projectiles divided by 5) when short-tailed albatross could be struck my MCNEPs. Table 21 below summarizes the MCNEP activities that will expose short-tailed albatross to stressors.

Table 21. Annual short-tailed albatross exposure to medium-caliber non-explosive projectiles.

Gunnery Exercise	**Total number of MCNEPs**	**MCNEPs by caliber**	**Instances of short-tailed albatross exposure**	**Area of effect for each instance (km^2)**	**Total short-tailed albatross habitat exposed annually (km^2)**
Surface-to-air	9,660	20mm 3,864	773	0.004	3.09
		25mm 5,120	1,024	0.006	6.14
		40mm 677	136	0.002	0.27
Surface-to-surface	16,750	20mm 6,700	1,340	0.013	17.42
		25mm 8,878	1,776	0.018	31.97
		40mm 1,173	235	0.001	0.24

Over the 16 years of the proposed action, a total of 152.1 km^2 (44.3 nm^2) of short-tailed albatross habitat will be exposed to stressors associated with surface-to-air MCNEP gunnery exercises and 793.9 km^2 (231.5 nm^2) of short-tailed albatross habitat will be exposed to stressors from surface-to-surface MCNEP gunnery exercises. In that habitat over the entire proposed action, we expect 0.59 short-tailed albatross to be exposed to stressors from MCNEPs. As a result of that exposure to a MCNEP, short-tailed albatross will be injured or killed.

10.6.4 Large-caliber Non-explosive Projectiles

Murrelet and short-tailed albatross may be exposed to large-caliber non-explosive projectiles (LCNEPs) used in surface-to-air and surface-to-surface gunnery exercises. The number of projectiles did not change since the 2016 Opinion, but more of the exercises will occur farther than 50 nm from shore.

We repeated the effects analysis as described in the 2016 Opinion using the new information on the location of the exercises and updated modeling of murrelet densities and short-tailed albatross population data. For more detail on those methods, refer to the 2016 Opinion (Section 10.4.5.4) and the information above.

10.6.4.1 Large-caliber Non-Explosive Projectile Effects to Marbled Murrelet

Gunnery exercises with LCNEPs will only occur farther than 20 nm from shore, where murrelet exposure during the breeding season is extremely unlikely to occur. Murrelets will be exposed to stressors from LCNEP gunnery exercises when those activities occur in the offshore area within

50 nm of shore during the winter. As with other non-explosive projectiles, we assume that a group of 5 projectiles will be a distinct opportunity for exposure to stressors from LCNEPs. Table 22 below summarizes the proportion of MCNEP activities that will expose murrelets to stressors.

Table 22. Annual marbled murrelet exposure to large-caliber non-explosive projectiles.

Gunnery Exercise	**Total number of LCNEPs**	**LCNEPs 20-50 nm from shore in winter**	**Instances of marbled murrelet exposure**	**Area of effect for each instance (km^2)**	**Total marbled murrelet habitat exposed annually (km^2)**
Surface-to-air	80	4	1	0.46	0.46
Surface-to-surface	2,720	136	28	0.94	26.42

Over the 16 years of the proposed action, a total of 7.3 km^2 (2.1 nm^2) of murrelet habitat will be exposed to stressors associated with surface-to-air LCNEP gunnery exercises and 422.7 km^2 (123.2 nm^2) of murrelet habitat will be exposed to stressors from surface-to-surface LCNEP gunnery exercises. Using the model averaging outlined above, we expect 30.9 murrelets will be exposed to stressors from LCNEPs (also assuming birds will not be struck underwater). Of those 31.4 murrelets, we expect 10.5 to be from the Conservation Zone 2 population, 11.3 to be from the Conservation Zone 3 population, and 9.7 to be from the Conservation Zone 4 population. As a result of that exposure to a LCNEP, murrelets will be injured or killed. We do not expect indirect effects to eggs or chicks from this exposure since this exposure will only occur in the winter when murrelets are not breeding.

10.6.4.2 Large-caliber Non-Explosive Projectile Effects to Short-tailed Albatross

Short-tailed albatross have the potential to be exposed to all gunnery exercises with LCNEPs. Assuming bursts of five LCNEPs to be a distinct opportunity for exposure, the 80 surface-to-air projectiles and 2,720 surface-to-surface projectiles will create 16 (surface-to-air) and 544 (surface-to-surface) instances when short-tailed albatross could be struck my LCNEPs. Table 23 below summarizes the LCNEP activities that will expose short-tailed albatross to stressors.

Table 23. Annual short-tailed albatross exposure to large-caliber non-explosive projectiles.

Gunnery Exercise	Total number of LCNEPs	Instances of short-tailed albatross exposure	Area of effect for each instance (km^2)	Total short-tailed albatross habitat exposed annually (km^2)
Surface-to-air	80	16	0.46	7.36
Surface-to-surface	2,720	544	0.94	511.36

Over the 16 years of the proposed action, a total of 117.8 km^2 (34.3 nm^2) of short-tailed albatross habitat will be exposed to stressors associated with surface-to-air LCNEP gunnery exercises and 8,299.5 km^2 (2,419.8 nm^2) of short-tailed albatross habitat will be exposed to stressors from surface-to-surface LCNEP gunnery exercises. In that habitat over the entire proposed action, we expect 5.1 short-tailed albatross to be exposed to stressors from LCNEPs. As a result of that exposure to a LCNEP, short-tailed albatross will be injured or killed.

10.6.5 Conclusion

Gunnery exercises using non-explosive projectiles will expose murrelets and short-tailed albatross to stressors that will lead to injury and death of birds. The expected exposure to these activities is summarized in Tables 24 and 25 below.

Table 24. Summary of Expected Exposure of Marbled Murrelets and Short-tailed Albatross to Non-explosive Gunnery Exercises.

Gunnery Exercise	Marbled Murrelet		Short-tailed Albatross	
	Habitat (km^2/nm^2)	Birds	Habitat (km^2/nm^2)	Birds
Small-caliber	70.8 / 20.6	5.2	4,248.5 / 1,238.7	2.6
Medium-caliber	156.3 / 39.7	11.4	946.1 / 275.8	0.6
Surface-to-air	*20.0 / 5.8*		*152.0 / 44.3*	
Surface-to surface	*136.3 / 39.7*		*794.1 / 231.5*	
Large-caliber	430.0 / 125.3	31.4	8,333.3 / 2,429.6	5.1
Surface-to-air	*7.3 / 2.1*		*116.7 / 34.0*	
Surface-to surface	*422.7 / 123.2*		*8,216.6 / 2,395.6*	
Totals		**48.0**		**8.3**

Table 25. Expected Exposure of Marbled Murrelets to Non-explosive Gunnery Exercises by Population.

Gunnery Exercise	**Marbled Murrelet Conservation Zone**			
	2	**3**	**4**	**Total**
Small-caliber	1.7	1.8	1.6	**5.1**
Medium-caliber	3.8	3.9	3.5	**11.2**
Large-caliber	10.5	10.8	9.7	**30.9**
Total	**16.0**	**16.5**	**14.8**	**47.2**

10.7 Helicopter Use

10.7.1 Bull trout Exposure to Helicopter Use in Inland Waters

Bull trout may be present in the water beneath where helicopters are operating in Inland Waters. Helicopters create noise that could disrupt the normal behaviors of exposed individuals. Recent research on jet noise suggests that aircraft noise will reach depths of up to 30 m below the surface of the water (Kuehne et al. 2020, pp.4-5, 7). However, helicopters are not nearly as loud as the jets that caused the noise analyzed in Kuehne et al. 2020 (Navy 2020, p. 3-20). As such, we do not expect noise from helicopters to exceed the 150 dB_{RMS} re: 1µPa level used as guidance for considering potential behavioral responses underwater. Furthermore, the majority of activities that include helicopter use in Inland Waters will be relatively short in duration (2 to 3 hours), so that even if, bull trout were to hear and respond to helicopter noise, short duration and intermittent exposures would not result in significant effects to individuals.

10.7.2 Marbled Murrelet Exposure to Helicopter Use in Inland Waters

Based on best available information regarding murrelet occurrence in Inland Waters (McIver et al. 2021, pp. 11-17), murrelets are likely to be present in areas of helicopter use. We adjusted our previous estimates of murrelet exposure to helicopter use based on the most recent information regarding murrelet densities, as well as the additional details regarding proposed helicopter use outlined below.

Stressors associated with helicopter use include rotor strikes, elevated in-air Sound Pressure Levels (SPLs; i.e., noise), water plumes, flying debris, and rotorwash (downdraft). Exposure to these stressors can result in injury, mortality, displacement, missed feedings, disturbance, and reduced fitness.

Helicopters (rotary-wing aircraft) will be used for four activities in Inland Waters. Helicopters produce lower-frequency sound and vibration at a higher intensity than fixed-wing aircraft (Richardson et al. 1995). Helicopter sounds contain dominant tones from the rotors that are generally below 500 Hz (the lower hearing range of murrelets, is approximately 480 Hz). Helicopters often radiate more sound forward than backward (Navy 2015, p. 3.0-37). Helicopter use is typically limited to approximately four-hour durations due to fuel capacity; therefore, the total hours described for each activity are expected to be intermittent based on the limited capacity of helicopter use and the inherent travel time associated with their departures/arrivals from where they are stored.

Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises are conducted at various ports and harbors to support homeland defense/security. Helicopters are used to tow mine sweeping/detecting devices at any time of year, for up to 24 hours over a several day period. Based on the information provided by the Navy, murrelets may be exposed to stressors related to helicopter use in the Inland Waters for 24 hours of helicopter flight time per event. Each activity creates 6 opportunities for murrelet exposure to stressors from helicopter use in seven possible locations: Port Angeles, Indian Island, Naval Base Kitsap (NBK) Bangor, Everett, NBK Bremerton, the Manchester Fuel Pier, and the Port of Seattle. These events will predominantly occur during the murrelet breeding season of April – September with one event planned every other year for 16 years. Due to the uncertainty in location, the possibility of impacts being focused in areas of higher murrelet density, and the variability of murrelet density annually, we conservatively analyzed these events as if they occur every year for 16 years.

Naval Special Warfare Personnel Insertion/Extraction (Non Submersible) training exercises are conducted at Crescent Harbor and Navy 7 (R6701) to train personnel to approach or depart using various means. Training personnel are inserted into the water via low, slow-flying helicopters from which personnel jump. These activities occur year round, for 2 to 8 hours, at any time of day. Based on the information provided by the Navy, we estimate that murrelets may be exposed to stressors related to helicopter use in Crescent Harbor and Navy 7 (R6701) for 3 events per year, for 8 hours per event, for 16 years (Kunz 2020a, in litt.).

Search and Rescue operations are conducted in Crescent Harbor and at Navy 7 training areas. Helicopters fly below 3,000 ft elevation to train in rescuing personnel. These activities occur year round, for 2 to 3 hours, at any time of day. Based on the information provided by the Navy, we estimate that murrelets may be exposed to stressors related to helicopter use in Crescent Harbor and Navy 7 for a total of 1,920 hours over a 16-year period in each location (20 events per year at each site, for 3 hours per event).

For all exercises involving helicopter use, we expect that helicopters will transit to the activity area at around 500 ft above ground level (AGL), and will fly slowly (~1 knot) and low (10 – 20 ft AGL) when participating in the training activity. Although the maximum duration of each event is between 2 to 8 hours, we expect the maximum duration that a helicopter would remain hovering in a particular area would be less than two hours. We expect helicopters would likely deploy from the nearest Navy air field, travel to the activity area, perform the exercise and return to the air field, or temporary waiting location, and would only remain within a particular area for only as much time as it would take to complete the exercise.

10.7.3 Marbled Murrelet Response to Helicopter Use in Inland Waters

Murrelets are expected to be exposed to stressors from helicopter use in Inland Waters (Puget Sound and Strait of Juan de Fuca). These stressors include rotor strikes, elevated in-air SPLs, water plume, flying debris, and helicopter rotor wash (downdraft). The close proximity of an operating helicopter to a murrelet will expose that murrelet to all of these stressors. The discussion below does not attempt to differentiate between these stressors, but assumes that exposure to a helicopter results in exposure to all of these stressors. Exposure to these stressors can result in injury, mortality, displacement, missed feedings, disturbance, and reduced fitness.

We expect that murrelets will perceive an approaching helicopter as an aerial threat and their primary response will be to dive. The length and distance of the murrelet dive may not be sufficient to completely evade helicopter downwash as the craft hovers. Depending on how long the helicopter hovers, the murrelet may dive and re-surface several times to evade the downwash. The area of effect where murrelets may be exposed is based on the assumption that rotor downwash extends three times the diameter of the rotor (FAA 2014, p. 7-3-6), which yields an area of effect of approximately 0.02 km^2 each time, and a total of 44 events per year will include helicopters. Over 16 years, we expect that a cumulative total of 14.1 km^2 (i.e., the sum of all individual areas of effect) of murrelet habitat within the Inland Waters will be exposed to helicopter downwash. We expect that murrelets exposed to helicopter downwash may not always be able to evade it and may experience a significant disruption of their normal behaviors.

We expect that when murrelets are exposed to helicopters performing training activities, murrelet foraging bouts and resting attempts will be interrupted. They are likely to abandon use of these areas until the helicopters are no longer present. We anticipate murrelet energy expenditure will be increased above normal when they flush, relocate out of the area, increase their diving effort to replace lost foraging opportunities, and escape from perceived predators (i.e., helicopters). During the nestling stage of the breeding cycle, nesting adults spend some proportion of their time at sea holding fish caught for the purpose of nestling feeding; these fish are typically larger and may be more difficult to find than the fish adult murrelets eat for their own nutrition (Burkett 1995, pp. 241-242; Carter 1984, p. 23; Carter and Sealy 1987, p. 289; Carter and Sealy 1990, p. 100; Nelson 1997, p. 8). As part of the startle response, fish-holding murrelets are likely to drop or swallow this fish (Speckman et al. 2004, p. 33), and may not be readily able to find a replacement. Given that murrelets have high energetic demands and must consume a large percentage of their body weight every day, we expect that these responses in the context of the duration, frequency, and affected areas represent a significant disruption of normal behaviors that creates a likelihood of injury.

We do not expect murrelets will collide with, or be struck by, helicopters. Collision is extremely unlikely because murrelets are expected to flush (flying low over the water) or dive, to avoid being struck.

Based on the average murrelet densities over the last five survey years, the area of effect for each instance of helicopter hovering outlined above, and the proposed frequency, location, and timing of helicopter use, we estimate that 8.4 individual murrelets will be exposed to stressors associated with hovering helicopters during the 16-year term of the action.

10.7.4 Effects to Eggs and Chicks as a Consequence of Adult Exposure to Helicopters

When nesting adult murrelets experience increased energetic demands as a consequence of their behavioral response to helicopter exposure, we expect that this will result in effects to their eggs or chicks. During the incubation phase, breeding adults have less than 24 hours of foraging at sea to meet all of their nutritional needs for a 48-hour period, including the energy needed for a round-trip inland flight to the nest. Interruptions in foraging, along with increased energy expenditure during attempts to evade hovering helicopters, are likely to lead to delays in incubation exchanges, leaving eggs exposed to heating, cooling, and predation risk. During the nesting phase, these interruptions in foraging, increased energy expenditure, and especially loss of fish caught for the purpose of nestling feedings are likely to lead to delayed or missed nestling feedings. Missed nestling feedings result in delayed development, stunting, and, when the nestling is already in poor nutritional condition, may lead to mortality. For more information, see discussion in Section 10.3.2.4. Given our expectation that, on average, 36 percent of murrelets at sea would be nesting in any one breeding season, behavioral disruption of 8.4 murrelets, all during the breeding season, would lead to injury and mortality of 3.0 chicks over the 16-year duration of the action.

10.7.5 Conclusions of Effects of Inland Helicopter Use to Marbled Murrelet

In Inland Water areas, such as Navy 7, and ports and harbors where Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection exercises occur, murrelets will be exposed to downwash from helicopters. The area of exposure is approximately 0.02 km^2 for each instance. Over 16 years, we expect that a cumulative total of 14.1 km^2 (i.e., the sum of all individual areas of effect) of murrelet habitat within the Inland Waters will be exposed to helicopter downwash. Given the location and number of events using helicopters and the area of effect explained above, and the densities of murrelets in these locations, we are reasonably certain that 8.4 murrelets will be directly exposed over 16 years. Exposed murrelets are expected to respond by diving repeatedly and or by vacating the area. Based on the high energetic demands of murrelets, coupled with the duration of this exposure, we expect that this disruption of normal behaviors will result in reduced foraging efficiency to the extent that there are measurable effects to individuals that create a likelihood of injury. Due to exposure of breeding adults to helicopter stressors we expect injury or death of an additional 3.0 murrelet eggs or chicks. In all, helicopter use in Inland Waters will expose 11.5 murrelets to stressors that will either disrupt their behavior creating a higher likelihood of injury or cause injury.

10.8 Ingestion of Debris

The proposed activities will introduce debris into the ocean that could exacerbate threats to seabirds through direct ingestion of plastics/debris, indirect ingestion via prey, or bioaccumulation of toxins through the food chain. While some materials will quickly sink beyond the reach of seabirds, murrelet and short-tailed albatross may ingest pieces of debris that remain at or near the surface of the water. Unrecovered materials from the Navy's training and testing activities that that could float at or below the surface include chaff fibers, plastic end caps and pistons from flares, plastic end caps and pistons from chaff cartridges, fragments of missiles (rubber, carbon, or Kevlar fibers), (Navy 2015, pp. 3.1-61, 3.4-299 - 300), and fragments of targets. Plastic end caps and pistons from flares and chaff cartridges may float for some period of time (Navy 2015, pp. 3.1-61, 3.5-66). The exposure and response of murrelets to marine debris introduced by the proposed action are analyzed in the 2016 NWTT Biological Opinion (USFWS 2016, pp. 222-227), and those analyses remain applicable to the current proposed action. While the number of activities that introduce debris into the marine environment have changed since the 2016 Opinion, our conclusion remains the same. Murrelet and short-tailed albatross will be exposed to debris caused by the proposed action and that exposure is likely to adversely affect individuals. However, given the range of responses to debris ingestion, the Service is not reasonably certain that murrelet or short-tailed albatross will be directly or indirectly injured by debris resulting from the proposed action.

10.9 Summary of Effects to Bull Trout, Marbled Murrelets, and Short-Tailed Albatross

The 2016 Opinion has a summary of the expected adverse effects to bull trout, murrelets, and short-tailed albatross (USFWS 2016, pp. 245-247), and the 2018 Opinion (p. 13) summarizes the changes to those effects between the 2016 and 2018 Opinions. As noted in the *Description of the Proposed Action*, the proposed action includes activities that have been consulted on previously and will continue, alterations to those activities, and new activities. As a result, the proposed action is the entire NWTT program and the Service will consider the context of the larger NWTT program in our jeopardy analysis within this Opinion. Therefore, the following tables (Table 26, Table 27, and Table 28) list and summarize all expected adverse effects stemming from the activities under consultation in this proposed action, as well as those from the 2016 and 2018 Opinions that are expected to continue.

Additionally, as part of our analysis for this Opinion we considered the most current information regarding the status of murrelets. Our previous analysis included an assumption that murrelet populations would continually decline at the same rate estimated in 2015 from at-sea survey data (Falxa et al. 2015, p. 8). Murrelet populations have not continually declined at the same rate, and murrelet population densities have recently been higher than we anticipated in the 2016 opinion. Because the estimated numbers of murrelets exposed to stressors depends in part on murrelet densities, increased density may result in increased numbers or murrelets exposed to some stressors. Therefore, we recalculated the number of murrelets reasonably certain to be exposed for each stressor, assuming that murrelet population densities will remain similar to those observed over the last few years. Our updated exposure modeling also included updated ranges to effect for underwater explosions, and new information clarifying details of the proposed action even for activities may not have changed in frequency. The results of the updated modeling are shown in Table 27. Therefore, although some activities have not changed or have been reduced, the estimated number of murrelets reasonably certain to be exposed may increase.

If there was no new information on stressor exposure (such as a new activity, revised range to effect, or updated species density), the revised estimates of individuals and areas affected due to changes under the current proposed action were generated by revising the previous effects estimates in proportion to the change in the stressor/activity (e.g., a 10% change in the stressor/activity results in a 10% change in the number of individuals and area affected).

Table 26. Summary of changes to reasonably certain adverse effects to bull trout and FMO habitat between the 2018 Opinion and the current proposed action

Stressor/ Activity	2018 Bull Trout	2018 Total Area Affected (km^2/ nm^2)	Change in Stressor/ Activity Under Current Proposed Action	Current Proposed Action Total Habitat Affected (km^2/nm^2)
<E1/E3 underwater explosives (mine neutralization-EOD, Crescent Harbor Range)	120	16 / 5	New range to effect	826 / 241
E4/E7 underwater explosives (mine countermeasure and neutralization testing)	n/a	n/a	New activity	2,270 / 661
Totals	120	16 / 5		**3,096 / 902**

Table 27. Summary of changes to reasonably certain adverse effects to murrelets and murrelet habitat between the 2018 Opinion and the current proposed action.

Stressor/ Activity	2018 Marbled Murrelets *	2018 Total Area Affected (km^2/ nm^2)	Change in Stressor/ Activity Under Current Proposed Action	Current Proposed Action Marbled Murrelets	Current Proposed Action Total Area Affected (km^2/nm^2)
Sonar, inland waters: MF8	2.4	0.44 / 0.13	Decreased from 40 to 0 hours per year	0	0
Helicopter rotor wash (multiple activities)	13.2	26.8 / 7.8	n/a	11.5 (all CZ 1)	14.1 / 4.1
E3 underwater explosives (mine neutralization-EOD, Bangor-Hood Canal and Crescent Harbor Ranges)	6.4	25.5 / 7.4	n/a	4.3 (all CZ 1)	13.6 / 3.96
Small-caliber non-explosive projectiles, offshore area (surface-surface)	2.34	124 / 36	Decreased from 121,200 to 121,000 rounds per year	5.16 (CZ 2: 1.72) (CZ 3: 1.85) (CZ 4: 1.59)	70.8 / 20.6

Stressor/ Activity	2018 Marbled Murrelets *	2018 Total Area Affected (km^2/ nm^2)	Change in Stressor/ Activity Under Current Proposed Action	Current Proposed Action Marbled Murrelets	Current Proposed Action Total Area Affected (km^2/nm^2)
Medium-caliber non-explosive projectiles (surface-air and surface-surface)	18.2	965 / 281.4	Decreased from 43,164 to 26,410 rounds per year	11.4 (CZ 2: 3.8) (CZ 3: 4.09) (CZ 4: 3.52)	156.3 / 39.7
Large-caliber non-explosive projectiles (surface-air, surface-surface)	14.2	754 / 220	n/a	31.44 (CZ 2: 10.46) (CZ 3: 11.29) (CZ 4: 9.67)	430.0 / 125.3
E4/E7 underwater explosives (mine countermeasure and neutralization testing)	n/a	n/a	New activity	49.1 (all CZ 2)	250.4 / 73.0
Totals	**56.74**	**1,922.8 / 560.6**		**112.9** (CZ 1: 15.8) (CZ 2: 65.1) (CZ 3: 17.2) (CZ 4: 14.8)	**936.8 / 267.2**

*Note: Numbers of adversely affected marbled murrelets were expressed in "groups" in the 2016 and 2018 Opinions, but are expressed as individual marbled murrelets in this Opinion.

Table 28. Summary of changes to reasonably certain adverse effects to short-tailed albatross and short-tailed albatross habitat between the 2018 Opinion and the current proposed action.

Stressor/ Activity	2018 Short-Tailed Albatross	2018 Total Area Affected (km^2/ nm^2)	Change in Stressor/ Activity Under Current Proposed Action	Current Proposed Action Short-Tailed Albatross	Current Proposed Action Total Area Affected (km^2/nm^2)
E1 and E2 medium-caliber explosive projectiles, offshore area (surface-surface gunnery exercises)	5.5	30,437/ 8,874	Decreased from 6,368 to 258 rounds per year	0.8	1,217 / 355

Stressor/ Activity	2018 Short-Tailed Albatross	2018 Total Area Affected (km^2/ nm^2)	Change in Stressor/ Activity Under Current Proposed Action	Current Proposed Action Short-Tailed Albatross	Current Proposed Action Total Area Affected (km^2/nm^2)
E3 and E5 large-caliber explosive projectiles, offshore area (surface-air, surface-surface, kinetic energy weapon)	1.3	7,409/ 2,160	Decreased from 310 to 192 rounds per year	2.8	4,594 / 1,339
Small-caliber non-explosive projectiles, offshore area (surface-surface)*	0.79	5,319/ 1,551	Decreased from 121,200 to 121,000 rounds per year	2.6	4,259 / 1,243
Medium-caliber non-explosive projectiles (surface-air and surface-surface)	2.1	13,834 / 4,033	Decreased from 43,164 to 26,410 rounds per year	0.6	946/ 276
Large-caliber non-explosive projectiles (surface-air, surface-surface, kinetic energy weapon)	1.6	10,414 / 3,036	Increased from 2,800 to 2,960 rounds per year	5.1	8,300/ 2,420
Totals	**11.3**	**67,413 / 19,654**		**11.9**	**19.316 / 5,633**

* Level of decrease in activity insufficient to decrease amount of effects

Based on the estimates in Tables 26-28, the overall net change in adverse effects to bull trout, murrelets, and short-tailed albatross from implementation of the proposed action and the overall NWTT program are as follows:

- Bull trout adversely affected through July 21, 2036, increased from 120 to 336.
- Bull trout FMO habitat adversely affected through July 21, 2036, increased from 16 km^2 to 172 km^2.

- Murrelets adversely affected through July 21, 2036, increased from 56.74 to 112.9.

- Murrelet habitat adversely affected through July 21, 2036, decreased from 1,922.8 km^2 to 936.8 km^2.

- Short-tailed albatross adversely affected through July 21, 2036, increased from 11.3 to 11.9.

- Short-tailed albatross habitat adversely affected through July 21, 2036, decreased from 67,413 km^2 to 19,316 km^2.

11 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state or private activities, not involving federal activities that are reasonably certain to occur in the action area considered in this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Within Puget Sound, all State, tribal, local, and private actions are required to obtain a U.S. Army Corps of Engineers (USACE) permit for work conducted in, over, or under navigable waters under the authority of Section 10 of the Rivers and Harbors Act and/or for the discharge of dredged or fill material under Section 404 of the Clean Water Act. Therefore such actions will require section 7 consultation with the Service.

However, bull trout and murrelets will continue to be affected by ongoing activities within Puget Sound and along rivers and streams draining into Puget Sound, including those without a federal nexus. Threats to Puget Sound habitat quality include population growth, shoreline development and armoring, urbanization that increases the amount of impervious surfaces, pressures on water supplies, filling of wetlands, and water and air pollution (WDOE 2015). Within the next 5 years, the population in the Puget Sound region is estimated to grow by 700,000 people.

Population increases results in higher levels of toxic chemicals entering Puget Sound from surface runoff, groundwater discharges, and municipal and wastewater outfalls. These contaminants include oil, grease, PCBs, and heavy metals. Many areas surrounding Puget Sound are highly urbanized with development spreading to the surrounding areas and converting agriculture and forested lands to impervious surfaces. The increase in impervious surfaces increases storm water runoff, which carries contaminants into the action area (WDOE 2006; WDOE and King County 2011, p. 30). Air pollution increases due to increased urbanization also lead to the increased deposition of contaminants such as polybrominated diphenyl ethers (PBDEs, used as flame retardants) into the marine environment (WDOE and King County 2011, p. 32).

Degraded water quality results in metabolic stress; avoidance responses which prevent or discourages free movement, reduced locomotor performance, and impaired olfactory responsiveness which may compromise growth, long-term survival, and reproductive potential. Contaminants have been found in murrelet prey species within the action area at levels that may affect prey health and reproductive success (USFWS 2009, p. 39-40; Liedtke et al. 2013, p. 5). These contaminants increase in concentration as they move up the food chain (Borgå et al. 2001, pp. 191-196). Such contaminants have been shown to cause developmental abnormalities, wasting, disruption of thyroid function, immunosuppression, and decreased reproductive success in fish-eating birds (reviewed in Luebke et al. 1997, pp. 7-10; Rolland 2000, pp. 615, 620-626).

Oil tanker and barge traffic is increasing within the Salish Sea (Felleman 2016, p. 27; Etkin et al. 2015, p. 271). In particular, the Canadian Trans Mountain pipeline expansion project, now owned by the Canadian government as a federal Crown corporation, is expected to be complete in late 2022 (Trans Mountain 2021). This expansion is expected to lead to approximately one additional oil tanker per day departing Burnaby, British Columbia, and traveling through the action area (Felleman 2016, pp. 37-38; Kinder Morgan 2016; Van Dorp et al. 2014, pp. 38, 52), and tanker and tug traffic related to the expansion are projected to increase vessel traffic through the Georgia, Haro, and Juan de Fuca Straits by approximately 7 to 14 percent over 2012 traffic rates (NEB 2019, p. 363).

Increases in oil transportation within the Salish Sea raise the likelihood of an oil spill affecting the action area. A major oil spill here would likely kill murrelets, as has been documented as a result of previous oil spills in other areas (reviewed by Carter and Kuletz 1995, entire). Oil spills may also cause sublethal injury to murrelets and may affect forage fish populations (Carter and Kuletz 1995, p. 264). Oil spill remediation may also be damaging to forage fish populations (Penttila 2007, p. 19).

Within the Olympic Military Operations Area, non-federal lands are managed primarily for timber production. Two HCPs cover management activities on state trust lands managed by WDNR, and on private lands where the Forest Practices Act applies, respectively. Therefore, effects of land management under these HCPs are not cumulative effects, as defined above. However, murrelets are not a covered species under the Forest Practices HCP. The Service determined that the covered activities of this HCP were likely to adversely affect murrelets, but were not likely to jeopardize their continued existence, based on the protection of known occupied murrelet nesting sites required by the Forest Practices Rules.

Murrelets and short-tailed albatross in the offshore area are threatened by continued overfishing, pollution, shipping, and oil and gas development (World Wildlife Federation 2015). Many of these actions are currently present, but are expected to increase in the future. Approximately 80 percent of the world's fisheries are already fully exploited or overexploited (UN 2010, p.1). Marine transport of cargo through the action area is expected to increase over the term of the proposed action. Increased shipping through the action area will also increase risk of exposing seabirds to stressors associated with shipping including disturbance, contaminant spills, and debris.

In Washington, the tonnage and ton-miles of cargo transported by marine vessel in Washington are expected to increase annually by 0.8 and 0.9 percent, respectively, between 2015 and 2035 (WSDOT 2017, p. 8). Some increase in tonnage may be associated with increasing size of ships, but increases in vessel traffic are likely to account for some of the increase as well. In the inland waters, oil traffic is likely to increase, as noted above, while other types of vessel traffic may increase or decrease, depending on economic conditions (WDOE 2019, pp. 43-49). Vessel traffic through the OCNMS has been steady or slightly declining in recent years, with the majority of vessels staying outside of mapped Area To Be Avoided (NOAA 2019, p. 7). In contrast, vessel traffic into Grays Harbor has increased in recent years, mainly due to increases in cargo and passenger ships (Robertson et al. 2019, p. 6). Vessel traffic to and from ports along the Columbia and Snake Rivers may also travel through marine waters off of Washington before entering the mouth of the Columbia.

In Oregon, the tonnage of imports arriving and exports departing by marine shipping is expected to increase annually by 2.9 and 2.7 percent, respectively, between 2007 and 2040 (Hernandez 2016, pp. 56, 58). Vessels arriving at Oregon ports may travel through marine waters off of Washington or Oregon before entering the Columbia to reach the Port of Portland, or along Oregon's coast to reach other ports. The projected increase in tonnage implies that vessel size, vessel traffic, or both will also increase.

Freight shipped by water to, from, or within California is expected to increase 44 percent, by weight, between 2015 and 2045 (Caltrans 2020, pp. 159-162). However, it is not clear how this increase will be apportioned between the northern portion of the state, within the area where the proposed action will be carried out, versus the southern portion of the state, which includes the large ports in Los Angeles, Long Beach, and San Diego. The northernmost port, the Port of Humboldt Bay in Zone 4, currently receives little shipping traffic, due to the lack of rail access at this port, which is not expected to change (Caltrans 2020, p. 139).

Some increases in shipping may be associated with a federal nexus (e.g., construction of new terminals or expansion of existing terminals that would require permitting by the USACE), and therefore would not constitute cumulative effects addressed here but would be addressed in the consultation for the permitting. Any increase that relies on existing terminal capacity would not be likely to involve a federal nexus and would result in cumulative effects.

These increases in vessel traffic are most likely to affect murrelets within Washington and Oregon (Zones 1 through 3, and the northern portion of Zone 4). Murrelets in California (southern Zone 4, and Zones 5 and 6) are less likely to be affected by the increase due to the locations of ports relative to concentrations of murrelets at sea. Waters off of the southern region of California, where several of the largest ports are located, are within the non-breeding season range of the murrelet, but are south of the southernmost Conservation Zone. Most of the remaining ports near the boundary between Zones 5 and 6, an area that receives little use by murrelets.

Increased shipping results in an elevated probability of oil spills, whether from oil cargo or from the vessels' own fuel oil, as well as an increase in the number of opportunities for ships to disrupt murrelet foraging, resting, or staging activities. Effects of vessel disturbance are varied, and include temporary decreases in foraging, increases in energy expenditure, and missed feedings to chicks (see USFWS 2020a, pp. 72-82 for a review). At times when prey are abundant and easily accessible, these effects to individuals, especially non-breeding individuals, may be insignificant. Decreased energy intake and increased energy expenditure are likely to have more consequence to all murrelets during poor forage years, and in all years to breeding individuals, given that breeding activities already likely require these individuals to operate at their maximum energy intake and output levels.

For short-tailed albatross, contaminants and floating plastics and debris will continue to pose a threat to their recovery as both affect survival through reduced growth, decreased reproduction, and egg and chick survival, thereby limiting their population growth. Bull trout and murrelets will continue to have direct and indirect effects to the species and their designated critical habitat from human population growth and its associated urbanization and development through habitat degradation, fragmentation, degraded water quality, and impacts to marine forage fish. These effects, especially in the Puget Sound area (Zone 1), will likely adversely influence reproduction and abundance of murrelets, and the distribution and abundance of bull trout.

12 INTEGRATION AND SYNTHESIS

The Integration and Synthesis section is the final step in assessing the risk posed to species and designated critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action and cumulative effects to the environmental baseline and, in light of the status of the species and designated critical habitat, formulate the Service's opinion as to whether the action is likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of designated critical habitat.

12.1 Bull Trout

The proposed action contains both a continuation of adverse effects to bull trout that were considered in the 2016 Opinion, and new adverse effects to bull trout that were not considered in the 2016 Opinion. Adverse effects from the use of E3 and <E1 (now E0) explosives in the Crescent Harbor site were analyzed in the 2016 Opinion, whereas adverse effects from the use of E4 explosives greater than 3 nm from shore in the offshore area are new under this Opinion.

12.1.1 Effects to Bull Trout Populations

A qualitative evaluation of the effects to bull trout populations is provided, because demographic data are not available to support a quantitative analysis.

12.1.1.1 Numbers

The use of E4 explosives 3 nm or greater from shore at the QRS in the offshore area from implementation of mine countermeasure and neutralization testing activities is expected to result in injury to all bull trout within 1,829 m of each of the 216 explosions through July 21, 2036 (14 per year on average). Explosive Ordnance Disposal activities are expected to result in injury to all bull trout within 661 m of 288 (average 18 per year) E0 explosions and within 1,674 m of 48 (average 3 per year) E3 explosions through July 21, 2036. Adverse effects are expected within an area of exposure totaling 3,096 km^2 and through July 21, 2036 (826 km^2 attributable to Explosive Ordnance Disposal and 2,270 km^2 attributable to mine countermeasure and neutralization testing activities).

The proposed action will injure or kill bull trout from two regions within the bull trout Coastal Recovery Unit which spans western Oregon and Washington. Mine countermeasure and neutralization testing activities will injure or kill bull trout from the Olympic Peninsula geographic region within the Coastal Recovery Unit. The Olympic Peninsula region is divided into six core areas; bull trout from three of those six core areas (Hoh, Queets, and Quinault Rivers) have the potential to be exposed to stressors from mine countermeasure and neutralization testing activities. There are no current efforts to evaluate or estimate bull trout population sizes on the western coast of the Olympic Peninsula, but the core area closest to mine countermeasure and neutralization testing activities, the Quinault River core area, is considered a population stronghold as one of the most stable and robust populations in the recovery unit (USFWS 2015a, p. A-3; USFWS 2015b, p. 79).

Explosive ordnance disposal activities will injure or kill bull trout from the Puget Sound geographic region within the Coastal Recovery Unit. The best information we have about the size of bull trout populations in the Puget Sound geographic region comes from surveys of bull trout redds. These data represent the minimum number of redds in each year since surveys will not count all redds due to limitations of sampling. Furthermore, not all bull trout spawn each year, and each redd represents more than one bull trout, so actual bull trout populations will be larger than the number of redds counted during surveys. Recent redd counts occurred in three basins (Skagit, Stillaguamish, and Snohomish) of the eight Puget Sound core areas. Over the past five years those surveys counted an average of 408 redds in those rivers (which are only a portion of the region).

Due to the apparent stability and health of the Quinault core area, and the apparent productivity of the Puget Sound region, the expected effects of the proposed action will not appreciably reduce the number of bull trout in the Coastal Recovery Unit. As numbers will not be appreciably reduced at the scale of the Coastal Recovery Unit, they will also not be appreciably reduced at the scale of the listed entity.

12.1.1.2 Reproduction

Bull trout typically spawn in low-gradient streams that have loose, clean gravel, and are fed by springs or other sources of cold groundwater. The proposed action will not affect these areas and will therefore not decrease the function of spawning areas or bull trout access to them. The proposed action will injure or kill sub-adult and adult bull trout in marine environments used as foraging, migratory, and overwintering habitat. We expect that bull trout that are killed and injured will be precluded from reproduction. This is a conservative assumption, but reasonable due to the long distances between marine FMO habitat and headwater streams where bull trout spawn. Injured bull trout are less likely to survive the migration to spawning areas when those injuries make them more susceptible to predation or disease. However, the proposed action does not exacerbate the primary threats to the species which limit the reproductive capacity of bull trout habitat, and removing the expected number of bull trout from the population will not have a significant effect on the reproduction of bull trout at the core area scale. Since the proposed action will not have a significant effect on reproduction at the core area scale, it will also not appreciably affect reproduction at the larger recovery unit or species scales.

12.1.1.3 Distribution

The proposed action will not permanently alter bull trout FMO habitat or prevent bull trout from using marine areas for foraging, migration, or overwintering. The bull trout Coastal Recovery Unit comprises three major geographic regions: Olympic Peninsula, Puget Sound, and Lower Columbia River. Populations of bull trout within those geographic regions are organized into core areas. Bull trout injured or killed by the proposed action will be primarily, if not completely, from the Olympic Peninsula and Puget Sound portions of the Coastal Recovery Unit that spans western Washington and north-central Oregon. The only bull trout core areas currently supporting anadromous populations of bull trout are located within the Puget Sound and Olympic Peninsula regions (USFWS 2015a, p. A-3). Underwater detonations associated with mine countermeasure and neutralization testing activities will occur in the QRS, which is adjacent to the Olympic Peninsula. The bull trout affected by implementation of mine countermeasure and neutralization testing activities originate from the Olympic Peninsula region. However, most bull trout core areas within the Olympic Peninsula Regions support anadromous life history forms, and the core areas along the Pacific Coast of Washington (where the QRS is located) likely have the best demographic status in the Olympic Peninsula region (USFWS 2015a, p. A-7). The Quinault River core area is the closest core area to mine countermeasure and neutralization testing activities that will affect bull trout from the Olympic Peninsula region. While recent demographic trends in the Quinault River are unclear, that core area has been identified as population stronghold of the Olympic Peninsula region. Therefore, the core populations of bull trout within the Olympic Peninsula region are considered to be sufficiently resilient to the loss of 14 individuals per year.

As noted above, the use of E0 and E3 explosives in the Crescent Harbor site will injure or kill bull trout within a cumulative 826 km^2 of bull trout habitat through July 21, 2036. While fish from other Puget Sound core areas may be exposed to explosives in Crescent Harbor, bull trout originating from rivers close to Crescent Harbor are more likely to be exposed to stressors there. The Lower Skagit River, Stillaguamish River and Snohomish/Skykomish Rivers core areas are closest the Crescent Harbor and have shown a recent overall, but not uniform, declining trends in reproduction. However, the proposed action will not exacerbate the primary threats that limit bull trout use of FMO habitat in the Puget Sound region. Therefore bull trout use of FMO habitat in Puget Sound not be affected by instantaneous exposures of 1.4 km^2 or 8.8 km^2 to explosions causing the injury or loss of individuals from the Puget Sound region and will not appreciably limit the distribution of bull trout at the scale of the listed entity. In the 2016 Opinion, the Service (p. 249) concluded that the "the effects of past and ongoing activities in the Crescent Harbor action area would maintain the existing habitat conditions" for bull trout. Because the type and frequency of the activities and stressors in those areas are unchanged under the current proposed action, that conclusion and its rationale remain valid (USFWS 2016, pp. 249-251).

Since bull trout FMO habitat will not be significantly affected (longer than the instantaneous explosions) by the proposed action, and bull trout spawning habitat will not be impacted, the action will not alter the range or distribution of bull trout.

12.1.2 Effects to Survival of Bull Trout

The proposed action will injure or kill bull trout within a cumulative 194 km^2 area of bull trout habitat each year over 16 years from the Coastal Recovery Unit. This unit is vulnerable to extirpation, but not imperiled (USFWS 2015b, pp. 8-9). The loss of those bull trout will decrease the breeding population of bull trout, but that habitat (even cumulatively) represents a small fraction of the FMO habitat used by bull trout and will not diminish spawning habitat or greatly reduce bull trout reproduction at a population scale. The proposed action will not permanently prevent FMO habitat from providing foraging, migration, or overwintering opportunities to bull trout, nor will it create gaps in, or constrict the range of bull trout. Bull trout populations will survive despite the proposed action due to the bull trout population still being able to reproduce and because of the enduring functionality of spawning as well as foraging, migration, and overwintering habitat.

For these reasons, the proposed action will not appreciably reduce the likelihood of bull trout survival at the core area scale and will therefore not reduce the likelihood of survival at the larger recovery-unit scale or across the range to the listed entity through a reduction in bull trout numbers, reproduction, or distribution.

12.1.3 Effects to Recovery of Bull Trout

The bull trout recovery plan (USFWS 2015b, entire) and coastal recovery unit implementation plan (USFWS 2015a, entire) identify the threats to bull trout that must be ameliorated for the species to recover. The primary threats to bull trout in the coastal recovery unit are: loss/degradation of habitat (including fragmentation); demographic factors (including lost connectivity, incidental catch, and forage availability); and interaction with nonnative species (including hybridization) (USFWS 2015a, pp. A-10 – A-21; USFWS 2015b, pp. IV, 11). The proposed action will not exacerbate these primary threats nor will it hinder efforts to ameliorate them. The proposed action will reduce bull trout numbers, but the core areas most likely to be affected include a population stronghold and core area with a stable 5-year mean of observations during spawning surveys, and will not have an appreciable impact on reproduction at a recovery unit scale. By not affecting spawning habitat and injuring or killing bull trout in a small portion of FMO habitat will allow the population to replace lost individuals and benefit from conservation actions building toward recovery. The proposed action will therefore not appreciably reduce the likelihood of recovery at the recovery unit scale or across the range of the listed entity.

12.2 Marbled Murrelet

12.2.1 Status of the Species and Environmental Baseline

The marine portion of the action area is large, including all areas ensonified by underwater and in-air explosions and all areas that receive floating marine debris resulting from the Navy's testing and training activities at sea. Therefore, the action area encompasses much or all of the marine range associated with the listed Distinct Population Segment (DPS) (see Appendix B) of the murrelet. The terrestrial action area is smaller, limited to portions of the murrelet's range within Washington. Therefore, to avoid redundancy, our integration and synthesis presents a combined summary of the status of the species and environmental baseline in the action area.

12.2.1.1 Numbers

In 2018, the latest year for which there is an estimate, the murrelet population in the Northwest Forest Plan area (Conservation Zones 1 through 5) was estimated to include 21,200 individuals (McIver et al. 2021, p. 3). For the period from 2001 through 2019, the population trend in this area was estimated to be close to flat, with a confidence interval fairly closely bracketing 0, indicating that there is no substantial trend. Populations in Washington (Zones 1 and 2) declined over that time period, with greatest confidence and the steepest decline in the Salish Sea (Zone 1). Populations in Oregon and Northern California (Zones 3 and 4) increased over the same time period, with the steepest increase in Southern Oregon and Northern California (Zone 4). Because of the infrequency of surveys, the trend in Zone 5 is inconclusive, with extremely wide confidence intervals. Outside of the Northwest Forest Plan area, the Central California murrelet population (Zone 6) was estimated to consist of 404 individuals in 2019, the last year for which an estimate is available (Felis et al. 2020, p. 7). Population trends have not been calculated for Zone 6.

12.2.1.2 Reproduction

In spite of the trend estimate described above, which potentially indicates near-stability of the murrelet population rangewide, a wide variety of estimates of murrelet productivity throughout the range indicate that individuals are not currently able to replace themselves. These estimates include ratios of juveniles to older birds (adults and subadults), which can be calculated from at-sea survey data, and are available in long-term, multi-year datasets for the majority of the range. Additional information is available from estimates of breeding propensity and nest success, which must be gathered via radiotelemetry, examinations of captured birds, nest monitoring, or combinations of these three methods. Regardless of the method used, none of the estimates of murrelet productivity, made within the listed range, indicate that reproduction is occurring at levels that would offset mortality and support population stability.

Ratios of juveniles to older birds (hereafter called "juvenile ratios") provide a direct index of productivity, and can be converted to a measure of fecundity by accounting for the proportion of the population consisting of subadults. Surveys to measure juvenile ratios are conducted before the breeding season is entirely finished, so observed ratios are corrected to account for juveniles that have not yet fledged at the time of the survey. With adult survival rates in the range of 85 to 90 percent, juvenile ratios must minimally be between 0.176 and 0.279 to sustain population stability (Beissinger and Nur 1997, p. B-13). Historical murrelet juvenile ratios were likely around 0.297 (Beissinger and Peery 2007, p. 300). In contrast, within the listed range, multi-year averages of juvenile ratios range from 0.032 in Central California (Peery et al. 2007, p. 234) to 0.089 in Washington's inland waters (Beissinger and Peery 2003, p. 19). More recent research showed that, during the years from 1995-2012, the corrected juvenile ratio in Washington's San Juan Islands showed no significant directional trend, and averaged 0.07 (Lorenz and Raphael 2018, p. 206). Outside of the listed range, one study in British Columbia reported a multi-year average juvenile ratio of 0.13 (Lougheed et al. 2002b, p. 314), which is the highest multi-year ratio that has been observed across the range of the species. Juvenile ratios for single years are often higher than those listed above, but even for a single year, the maximum corrected ratio reported within the listed range is 0.157, observed in Northern Oregon in 1995 (Beissinger and Nur 1997, p. B-26). All of these estimates indicate that mortality is likely outpacing reproduction, and murrelets are not replacing themselves.

Fecundity estimates generated from studies of breeding propensity and nest success show a similar pattern. High estimates between 0.19 and 0.23 juveniles per breeding-aged adult have been reported from British Columbia, but the study authors noted that these estimates were based on breeding propensities between 0.8 and 0.95, whereas the observed breeding propensity in this study was 0.65 (Bradley et al. 2004, pp. 323-324), which would reduce the fecundity estimate to approximately 0.16. In Central California, combining breeding propensity and nest success rates led to an estimate of 0.027 juveniles per breeding-aged adult (Peery et al. 2004, p. 1094). In a radiotelemetry study in Washington, encompassing parts of Zone 1 and Zone 2, breeding propensity was between 0.131 and 0.2, and nest success was 0.2 (Lorenz et al. 2017, pp. 313, 316), which would combine to produce a fecundity estimate between 0.0131 and 0.02. In Oregon, recent fecundity estimates are not available, but preliminary data from an ongoing radiotelemetry study indicate extremely low breeding propensities, averaging less than 0.06 over the period from 2017 through 2019 (Adrean et al. 2021, p. 2), which would result in a fecundity estimate of less than 0.03 even in the unlikely event that all nests were successful. These fecundity estimates are extremely low, and while radiotelemetry may negatively influence the tagged individual's breeding propensity or nest success, the high estimate from British Columbia also came from a radiotelemetry study. However, it is not clear that even the high fecundity estimate from British Columbia would result in population stability or growth. Juvenile ratios will always be lower than fecundity estimates, because juvenile ratios represent the number of juveniles divided by the number of subadults and adults, whereas fecundity represents the number of juveniles per adult. In other words, the numerator is the same in both ratios, but the denominator is larger for the juvenile ratio than for fecundity. Therefore, a fecundity of 0.16 would translate to a juvenile ratio smaller than 0.16, which is lower than the minimum value likely to result in population stability. Other studies have separately examined either breeding propensity or nesting success, and the results of these studies are consistent with the results discussed above (see Appendices A and D for a more in-depth review).

We created a demographic model of Zone 2 to inform our analysis of the baseline condition and population-level effects of the action (see Appendix B). This model provides an illustration of how survival and reproductive rates combine to contribute to population growth or decline. We selected survival and reproductive rates for this model to correspond to an average rate of population change of -2.2 percent per year, in keeping with the best available estimate of the Zone 2 population trend (McIver et al. 2021, p. 20). The survival and fecundity rates corresponding to this rate of decline, 0.9 and 0.1554, respectively, are near the higher end of empirical estimates made within the entire range of the species, and are higher than any empirical estimates made recently within the listed range. This implies either that murrelet productivity is underestimated by all of the various empirical measures used to estimate it, or that the true rate of population change in the breeding population of Zone 2 is a much steeper decline than the estimated -2.2 percent per year. For example, the at-sea population numbers in Zone 2 could be bolstered by the presence of non-breeding visitors (see Appendix D for further discussion). However, we regard the –2.2 percent annual trend as the best available information regarding the Zone 2 trend, because other estimates of the trend are not available.

12.2.1.3 Distribution

The listed DPS occurs at the southern end of the entire species range. The southernmost population, Zone 6 in Central California, is small and geographically distant from the neighboring population in Zone 5. Nesting habitat in Zone 6 is restricted to relatively small, discrete patches in State and County Parks and on private lands, and a large proportion of nesting habitat within Zone 6 burned in 2020 (Singer 2021, in litt.). The Zone 6 population is also genetically differentiated from those in the rest of the range, likely due to geographic separation from other murrelet populations resulting from historical habitat loss (Hall et al. 2009, p. 5078; Peery et al. 2010, p. 703). Although migrants from other populations are sometimes present, genetic testing shows that few, if any, of these migrants join the Zone 6 breeding population (Peery et al. 2010, p. 702; Vásquez-Carrillo et al. 2013, p. 177). To the north of Zone 6, the Zone 5 population is even smaller, and its nesting habitat is even more restricted to small patches on National, State, and County park lands (McShane et al. 2004, p. 4-14).

In contrast, murrelet populations in Northern California through Central Oregon (Zone 4 and the southern portion of Zone 3) are large and well-distributed within the region. In spite of apparently low productivity throughout the listed range (see above), populations in Zones 3 and 4 have increased since 2001. These increases may result from some combination of local increases in productivity, migration from other parts of the DPS or full species range, and changes in murrelet behavior such that a larger proportion of individuals are present at sea at the time of surveys (McIver et al., in press, pp. 38, 42-43). Although the murrelet populations in this portion of the range currently appear to be robust and well-distributed at sea, nesting habitat in the Klamath region of Southern Oregon is prone to large wildfires; for example, between 1993 and 2012, 62 percent of all habitat loss from reserved federal lands resulted from a single, large wildfire in this region (Raphael et al. 2016, p. 75). Since then, a number of other wildfires have also burned in this region, but a quantification of the loss of murrelet nesting habitat is not yet available. The Zone 3 population also includes Northern Oregon, which is populated more sparsely both at sea and inland, in keeping with the relatively smaller amounts of nesting habitat present in this region.

Adjacent areas of Southwestern Washington (southern portion of Zone 2) are also sparsely populated by murrelets, both at sea and inland. The 1997 Recovery Plan (USFWS 1997, p. 126) identifies a gap in the range within Zone 2, between northern and southern subpopulations along Washington's outer coast. In 1997, murrelets could reliably be found flying inland in large numbers in some areas of Southwestern Washington, but this is no longer the case. It appears that the gap mentioned in the Recovery Plan has widened to encompass a relatively large area with relatively little murrelet presence in Southwestern Washington and Northern Oregon.

Murrelet populations in Washington are declining. The population decline is steepest and most statistically significant in Washington's inland waters (Zone 1), but there is some evidence for a declining population in Zone 2 as well. These declines may be due simply to the low reproductive rates discussed above, to migratory movements to the south, or both. It is unlikely that murrelets from Washington are moving northward in large numbers, given that populations in British Columbia are also declining, with the steepest declines close to the border (Bertram et al. 2015, pp. 9, 11-12).

In summary, the distribution of murrelets within the listed range consists of small, fragmented populations to the south; large, well-distributed populations in the center of the range; and large but declining populations to the north, separated from the central populations by a sparsely-populated gap that appears to have grown since the time of listing.

12.2.2 12.2.2 Factors Relevant to the Condition of the Species

Several past, and present, factors affect the condition of the species, including the loss of nesting habitat, changes in forage quality and availability, and other marine conditions. In addition, in the smaller populations at the southern end of the range, we expect that intrinsic factors associated with small populations are likely to influence the long-term population trajectory.

Loss of nesting habitat, primarily due to timber harvest, was the principal threat that led to listing. Current population trends may be affected by "legacy" effects of nesting habitat loss that occurred up to 20 years in the past (USFWS 2012b, p. 6). Legacy effects are related to the long lifespan of murrelets: when a breeding adult murrelet loses access to nesting habitat, it will remain present in the population, bolstering population numbers for the duration of its lifespan, but will not produce offspring. When this occurs at large scales, the age structure of the population skews toward older individuals, and dramatic reductions in population numbers can result as these older individuals die without replacing themselves. This process has been observed, for example, in a population of snail kites (*Rostrhamus sociabilis*) (Reichert et al. 2016, p. 8).

Nesting habitat loss continues today. Between 1993 and 2012, nesting habitat losses were greatest in Washington, mainly due to timber harvest on non-federal lands (Raphael et al. 2016, pp. 80-81). Nesting habitat has also been lost to wildfires; around half of all habitat loss on federal lands has resulted from wildfires in the Klamath region of Zone 4 (Raphael et al. 2016, pp. 80-81). Large populations of murrelets at sea tend to be located near large, contiguous blocks of nesting habitat, though this relationship varies across the range (Lorenz et al. 2016, pp. 12-13; Raphael et al. 2015, p. 20; Raphael et al. 2016, p. 101). Changes in the amount of nesting habitat may be more closely correlated with changes in population sizes at sea (McIver et al., in press, p. 42). In Washington there is apparently a large amount of suitable nesting habitat per individual murrelet; however, individuals are travelling long distances once at sea to reach foraging habitat (Lorenz et al. 2017, p. 318), suggesting that the amount of nesting habitat is not the only factor limiting populations.

Factors related to forage quality and availability likely also caused reductions in reproductive rates, contributing to historical and ongoing population declines (Becker et al. 2007, p. 276; Norris et al. 2007, p. 881). For example, historical murrelet diets in the Puget Sound–Georgia Basin region included mostly energy-rich fishes like northern anchovy and Pacific herring, but are now highly dependent on Pacific sand lance, a less energy-dense fish (Gutowsky et al. 2009, pp. 247, 251). Approximately six sand lance have the same energetic value as a single northern anchovy (Gutowsky et al. 2009, p. 251). Furthermore, in warm water conditions, which are becoming more frequent as a result of climate change, the energetic content of individual forage fish decreases (Piatt et al. 2020, pp. 21, 24). In addition, in the Puget Sound region, human alteration of the nearshore marine environment has a strong negative association with the at-sea population density of murrelets at sea (Raphael et al. 2016, p. 106), which implies that these human influences prevent murrelets from fully using some areas that may otherwise have provided foraging opportunities.

Factors affecting nesting habitat configuration and foraging conditions do not operate independently, but combine to affect murrelet populations. Nesting attempts are most likely to be carried out in large blocks of forested habitat close to the coast, but in years following good marine conditions, more fragmented or isolated habitat patches may be used as well (Betts et al. 2020, p. 6). In Washington, where much of the coastal nesting habitat has been replaced by urban development and industrial forests, murrelets fly extremely long distances between nesting and foraging habitats, have very large marine home ranges, and exhibit extremely low reproductive rates, as detailed above (Lorenz et al. 2017, pp. 316-319). Contiguous blocks of nesting habitat remain present on the Olympic Peninsula in closer proximity to the coast than in other parts of Washington, the adjacent OCNMS and Strait of Juan de Fuca retain relatively undisturbed nearshore habitats, and at-sea murrelet densities remain higher in these areas (i.e., Zone 1, Stratum 1 and Zone 2, Stratum 1) than in other marine areas in Washington (Zones 1 and 2) (McIver et al. 2021, pp. 11-17).

Other factors affecting survival and reproduction of murrelets include net fisheries, high predation rates in nesting habitat, and oil spills. Within the listed range, net fisheries are now expected to affect murrelets only in Washington, a reduction in the geographic range of this threat since the time of listing (McShane et al. 2004, p. ES-2). Measures have also been taken to reduce predation by corvids in nesting habitat in portions of the listed range, particularly in Northern California (Brunk et al. 2021, pp. 7-8). On the other hand, the recovery of raptors like peregrine falcons (*Falco peregrinus*) may pose increased risk to adults flying inland (Peery et al. 2006, pp. 83-84). Renewable energy presents an emerging source of mortality for murrelets, with the recent construction Skookumchuck Wind Energy Project in Western Washington, and several tidal energy projects proposed within Puget Sound.

Numerous federal actions have undergone section 7 consultation for murrelets in recent years. Most of these include effects of relatively small magnitudes, such as noise disturbance and edge effects associated with forest thinning or road work adjacent to murrelet habitat. Some are larger in scale, including the Navy's operation of airfields on Whidbey Island (in Zone 1), Treaty Tribal and all-citizens salmon net fisheries in Puget Sound (in Zone 1, but potentially affecting individuals breeding in Canada and other Zones as well), and the revision of several Bureau of Land Management Resource Management Plans for western Oregon (Zones 3 and 4). Although we anticipated that each of these actions would adversely affect individual murrelets, we concluded that none of these actions would jeopardize the species. The Service has also recently approved a number of incidental take permits for private activities associated with HCPs and Safe Harbor Agreements (SHAs) for murrelets, including permits with SHAs with timber companies Rayonier Operating Company, LLC and Sierra Pacific Land and Timber Company in western Washington; a permit with an HCP for the Skookumchuck Wind Energy Project in western Washington, and an HCP amendment incorporating a long-term conservation strategy for murrelets into the existing HCP covering forest lands managed by the Washington Department of Natural Resources. Although we anticipated that each of these projects would harm murrelets, all also include conservation actions, and each will result in a net gain of conserved nesting habitat. The Skookumchuck Wind Energy Project also includes derelict net removal from Puget Sound as an additional conservation action. The permitting of these projects underwent section 7 consultation, and in each case we concluded that the action would not jeopardize the species.

In addition to anthropogenic or climate-related threats, some populations of murrelets face intrinsic stressors related to small population sizes. At present, these stressors are likely to be most relevant to the very small population in Zone 5, and to a lesser extent in the small population in Zone 6. To the extent that separate subpopulations exist within each Zone, these intrinsic stressors will also operate at that scale, but we currently lack information about subpopulation structure within Zones. Stressors associated with small population sizes include increasing risk of extirpation due to demographic stochasticity, increasing risk of skewed sex ratios further reducing reproductive rates, and Allee effects, which occur when low population densities interfere with essential behaviors. Allee effects are discussed in greater detail below.

Demographic stochasticity refers to the random nature of survival and reproduction. This randomness has a proportionally larger influence in smaller populations. In other words, "[t]he dynamics of a small population are governed by the specific fortunes of each of its few individuals" (Caughley 1994, p. 217). In contrast, the dynamics of a large population are governed by the law of averages. Any given population has average survival and reproductive rates, but this is the result of each particular individual either surviving, or dying, and each breeding adult either breeding successfully, or not. When the population size is very small, simple bad luck on the part of a few individuals can exert a dramatic negative influence on the population trajectory, even when environmental conditions appear to be favorable, and this process increases the risk of sudden population crashes. The influence of demographic stochasticity on extirpation risk is strongest and most universal among species when a population consists of 100 individuals or fewer, i.e., at or just below the typical size of the Zone 5 murrelet population (Shaffer 1987, p. 73). Demographic stochasticity is also influential when population sizes are in the low hundreds in some circumstances, for example, in populations with monogamous mating systems or where nesting sites are limiting (Kokko et al. 1998, p. 52; Legendre et al. 1999, p. 458). Murrelets have a monogamous mating system (Sealy 1975, p. 151), and murrelets may compete for nesting sites, at least in some locations (Naslund 1993, p. 599), indicating that demographic stochasticity may affect murrelet population trajectories at population sizes between 100 and 500, the range in which Zone 6 population estimates often fall. On the other hand, murrelets' "slow" life history, with low annual fecundity and high adult survival rates, may be protective against sudden population crashes due purely to demographic stochasticity (Jeppsson and Forslund 2012, p. 714).

Skewed sex ratios are one result of demographic stochasticity that would particularly affect murrelets. Skewed sex ratios can occur for many reasons; for example, breeding males may be at greater risk than females of predation or other stressors related to inland flight, because they fly inland more often than females to provision chicks during the nestling phase (Bradley et al. 2002, p. 180; Vanderkist et al. 1999, pp. 400-401). However, even in the absence of increased risks for one sex, sex ratios can become skewed randomly in small populations, simply because more nestlings of one sex are lucky enough to survive fledging. Because murrelet reproduction requires intensive effort from both parents to produce a single chick, the maximum number of nesting pairs in a given season will be determined by whichever sex has fewer individuals present in the population. In this case, the already-small population is effectively even smaller, because not all adults have the opportunity to attempt nesting, and the influence of demographic stochasticity is larger (Engen et al. 2003, p. 2385; Møller 2003, p. 224). Skewed sex ratios in small populations have been documented during extinction and extirpation events, for example, of dusky seaside sparrows (*Ammodramus maritimus nigrescens*), heath hens (*Tympanuchus cupido cupido*), and the New Zealand mainland population of kakapos (*Strigops habroptilus*) (Donald 2007, p. 684).

Allee effects are positive associations between population density and survival or reproduction. In other words, survival or reproduction are high only when the population density is high enough. These effects occur when too few individuals are present in a population, or population density is too low, to support essential social behaviors. At small population sizes, skewed sex ratios, as discussed above, can be one source of Allee effects (Courchamp et al. 1999, pp. 405, 407; Lande 1998, p. 357). Additionally, for murrelets, there is evidence that breeding behavior is socially facilitated, and murrelets are sometimes categorized as semi-colonial in their nesting behavior. Murrelets have been observed in nesting habitat demonstrating social behaviors, such as circling and vocalizing, in groups of up to ten birds (Nelson and Peck 1995, p. 51). Spatial clustering of nest sites is documented and suggests semi-coloniality (Conroy et al. 2002, p. 131; Meyer et al. 2002, p. 103; Naslund et al. 1995, p. 12). Even in places where the clustering of nests is not marked enough to suggest semi-coloniality, usually multiple nests can be found in a contiguous forested area (Evans Mack et al. 2003, p. 6). Recent experimental work shows that playback of recorded murrelet calls within unoccupied suitable nesting habitat appears to attract murrelets to nest there during the following nesting season, indicating that murrelets select nesting habitat at least in part on the basis of the presence of conspecifics (Valente et al. 2021, pp. 50-51). Therefore, we expect that small population sizes or low population densities will interfere with breeding behavior, further reducing murrelet reproductive rates. No data are available regarding the minimum population sizes or densities needed for effective murrelet nesting behavior. However, an expert panel previously hypothesized that the threshold for Allee effects would fall between one percent of the initial population abundance and the square root of the initial abundance (Akcakaya 1997, p. 12). Calculation of the threshold would then depend on the selection of the spatial scale and timeframe of the initial conditions.

Demographic stochasticity is currently likely to be an influential factor in the population dynamics within Zone 5, and possibly within Zone 6, but effects of demographic stochasticity are likely to be negligible currently within the four larger populations within Zones 1 through 4. If Zones 1 and 2 continue to decline, they will eventually reach the point where demographic stochasticity increases the risk of extirpation. It is not clear whether Allee effects may be affecting murrelet populations within some Zones, since the appropriate spatial scale and numerical or density thresholds associated with Allee effects are not known with any certainty for murrelets. If Allee effects operate at small spatial scales, such as contiguous patches of nesting habitat, it is possible that the gap in the murrelet range at the southern end of Zone 2 and the northern end of Zone 3 has already been exacerbated by Allee effects. However, if the expert panel's hypothesized threshold of one percent of the initial population applies at large scales, it is possible that no current murrelet population is yet affected by Allee effects.

12.2.3 Survival and Recovery Needs

The Recovery Plan designated the Conservation Zones to be the functional equivalent of recovery units as defined by Service policy. Four of the six Zones are necessary to support recovery and enable long-term survival. Specifically, one of the criteria for recovery specifies that "trends in estimated population size, densities and productivity have been stable or increasing in four of the six zones over a 10-year period" (USFWS 1997, p. 113). Furthermore, the Recovery Plan described viable, well-distributed populations in each of the four northern Zones as being necessary to allow for long-term survival and eventual recovery (USFWS 1997, p. 116).

Conservation Zone 5 was not expected to contribute to survival or recovery, due to the extremely limited amounts of nesting habitat, very small population size, and resulting high risk of extirpation (USFWS 1997, p. 115). Conservation Zone 6 was also not expected to contribute to long-term survival, due to the small population size, habitat conditions, lack of federal land, and isolation from other murrelet populations, all factors that increase the risk of extirpation from stochastic or catastrophic events (USFWS 1997, p. 116). Given its slightly larger size, it was expected to contribute to recovery; however, research conducted after the publication of the Recovery Plan indicates that the Zone 6 population is a demographic sink, bolstered by immigrants from other populations but not producing enough young to contribute demographically to the larger metapopulation (Peery et al. 2006, p. 1523; Peery et al. 2010, p. 702; Vásquez-Carrillo et al. 2013, p. 177). Furthermore, current estimates indicate that one quarter of suitable nest trees were killed in 2020 wildfires, likely further reducing the reproductive capacity of this population. Therefore, we assume for this analysis that Zone 6 will not contribute to recovery.

Rangewide, the listed DPS of murrelets is not currently poised for long-term survival and recovery. Although the rangewide population does not show an increasing or decreasing trend for the 2001-2018 period, this lack of a trend is produced by a combination of consistent declines in Washington, increasing populations in Oregon and Northern California, and small populations at the southern end of the range. Long-term survival and recovery will require population stabilization or increase in Zones 1 and 2, as well as continued increase or stability in Zones 3 and 4. Throughout the listed range, all estimates of productivity indicate that reproductive rates are too low to support sustained population stability. Therefore, the DPS is not likely to recover, and the chances of long-term survival appear low, unless productivity can be increased.

Population sizes in Zones 1 through 4 are currently large enough that, if productivity can be increased, long-term survival and recovery will be possible. It is not clear what management actions may be taken to improve murrelet productivity to the point where populations will stabilize or increase. Under the NWFP, various HCPs, and other conservation efforts, some nesting habitat regrowth is expected to occur in the coming decades, to partially offset past and ongoing nesting habitat loss. However, murrelet populations will likely be unable to take advantage of additional nesting habitat if forage conditions deteriorate further. Given changing climate conditions in the marine environment, further deterioration in forage conditions are expected. Therefore, new conservation strategies will be needed to increase murrelet productivity and stabilize their populations, factors that are needed for recovery and long-term survival.

In summary, although some threats have been reduced in portions of the range, most continue unabated and new threats now strain the ability of the murrelet to successfully reproduce at replacement levels. Threats continue to contribute to murrelet population declines through adult and juvenile mortality and reduced reproduction. Therefore, given the current status of the species and background risks facing the species, murrelet populations throughout the listed range have low resilience to deleterious population-level effects and are at high risk of continuing or renewed declines. Activities that degrade the existing conditions of occupied nesting or foraging habitat, reduce adult survivorship, or reduce reproductive rates are of greatest consequence to the species.

12.2.4 Effects of the Action

12.2.4.1 Summary of Effects to Individuals

The proposed action includes numerous activities, with varying effects to murrelets. Many of the activities are expected to have insignificant or discountable effects. More substantial effects are expected from a subset of the proposed activities, all of which occur in the marine environment. Anticipated effects include behavioral disturbance, injury, and mortality. Helicopters hovering close to the surface of the water are expected to affect 9 birds at sea in Zone 1, leading to repeated avoidance diving and disruption of foraging, which creates a likelihood of injury due to reduced energy intake and increased energy expenditure. Underwater explosions are expected to affect 3 birds at sea in Zone 1 and 37 birds at sea in Zone 2, leading to hearing injury and barotrauma injuries, which range in severity from mild contusions to fatal internal bleeding. Gunnery exercises are expected to affect 16 birds at sea in Zone 2, 17 birds at sea in Zone 3, and 15 birds at sea in Zone 4, leading to hearing injury and injury or mortality associated with projectile strikes.

When breeding adults are disturbed, injured, or killed during the breeding season, we expect these effects to adults to affect eggs or chicks, in turn, with results ranging from a single missed chick meal, to nest abandonment and mortality of the egg or chick. During poor forage conditions, even one missed meal may lead to starvation of an already-malnourished chick. All helicopter hovering in Zone 1, and most of the underwater detonations, are expected to occur during the breeding season. We anticipate that 3 eggs or chicks will be affected in these ways in Zone 1 and 13 eggs or chicks will be affected in Zone 2. We do not anticipate effects to chicks resulting from Gunnery Exercises, because we expect that murrelet exposure to Gunnery Exercises will be confined to the non-breeding season, as exposure during the breeding season is extremely unlikely.

12.2.4.2 Cumulative Effects

In Zone 1, cumulative effects include alterations of the marine environment related to population growth in the Puget Sound region. Due to increased air pollution and stormwater runoff from impervious surfaces, we expect increases in the concentrations of several marine contaminants, including oil, grease, PCBs, PBDEs, and heavy metals (WDOE 2006; WDOE and King County 2011, p. 30). Oil tanker and barge traffic are also increasing in Zone 1, and in particular, increases of 7 to 14 percent over 2012 vessel traffic rates to Burnaby, British Columbia, are expected upon the completion of the Trans Mountain Pipeline Expansion Project, now projected for completion in late 2022 (Felleman 2016, pp. 27, 37-38; Etkin et al. 2015, p. 271; NEB 2019, p. 363; Trans Mountain 2021; Van Dorp et al. 2014, pp. 38, 52). Increased oil vessel traffic in the Zone 1 increases the probability of oil spills.

Shipping traffic is expected to increase throughout the range of the murrelet in the coming decades. These increases in shipping appear most likely to affect murrelets in Washington and Oregon (Zones 1 through 3, and northern portion of Zone 4), where ports are present within or near areas of relatively concentrated murrelet use, and less likely to affect murrelets in California (southern Zone 4, Zones 5 and 6), where most ports are located away from areas of concentrated murrelet use. Increased shipping results in an elevated probability of oil spills, whether from oil cargo or from the vessels' own fuel oil, as well as an increase in the number of opportunities for ships to disrupt murrelet foraging, resting, or staging activities.

Increases in vessel disturbances, marine contaminants, and oil spill risks are expected to result in a variety of effects to individuals. Effects of vessel disturbance are varied, and include temporary decreases in foraging, increases in energy expenditure, and missed feedings to chicks. At times when prey are abundant and easily accessible, these effects to individuals, especially non-breeding individuals, may be insignificant. Decreased energy intake and increased energy expenditure are likely to have more consequence during poor forage years, and to breeding individuals, given that breeding activities already likely require these individuals to operate at their maximum energy intake and output levels. Effects of contaminants include developmental abnormalities, wasting, and disruption of thyroid function, immunosuppression, and decreased reproductive success in fish-eating birds. Oil spills cause sublethal effects as well as murrelet mortality.

This combination of cumulative effects has may depress murrelet numbers, via oil spill mortality as well as incremental decreases in survival rates for birds subjected to sublethal effects. The cumulative effects are also likely to incrementally reduce reproductive success. These cumulative effects are most concentrated in Zone 1, where heavy human use of the nearshore and marine environment overlaps most heavily with murrelet populations. The cumulative effects are likely contributing to population declines in Zone 1 as well as affecting the marine distribution of murrelets within Zone 1.

12.2.5 Integration of the Status and Baseline, Effects of the Action, and Cumulative Effects

Here, we integrate the effects of the action with the status and baseline condition of the species and the cumulative effects, to determine how the action will alter the numbers, reproduction, and distribution of murrelets, and whether or how these effects in turn will change the likelihood of survival and recovery of the species. Typically, we examine the effects to numbers, reproduction, and distribution first at the scale of the action area, then at the scale of the Conservation Zone, and finally, range-wide. However, given the large action area for the proposed action, we first examine the effects to each affected Conservation Zone, followed by the range-wide analysis.

12.2.5.1 Effects to the Conservation Zones

We expect that individual murrelets will be affected in Zones 1 through 4, all of which intersect, or are situated inshore of, the areas where the Navy will conduct activities. The largest number of individuals will be affected in Zone 2, which also has the smallest murrelet population of the four Zones. Therefore, the proposed action will have the greatest proportional impact to Zone 2. For this reason, we focus particularly on our analysis of effects to the Zone 2 population. Our discussion of effects to each of the other three Zones follows our Zone 2 analysis.

12.2.5.1.1 Effects of the Action in Zone 2

As noted above, the action is expected to injure or kill 52 murrelets at sea and 13 eggs or chicks in Zone 2, for a total of 65 individuals of all life stages. Over the past five surveys (2013-2015, 2017, 2019), the estimated population size in Zone 2 has ranged from 1,271 individuals in 2013 to 3,204 individuals in 2015, with an average of 2,013 birds (McIver et al. 2021, pp. 11-17). Thus, over the 16 years of the proposed action, we expect that approximately three percent (64 divided by 2,013) of the average total population of Zone 2 will be injured or killed as a result of the proposed activities. Since this population is currently most likely to be declining, with mortality outpacing reproduction, we do not expect that the population will be resilient to these effects.

We constructed a demographic model to help us understand how these effects to individuals would affect the Zone 2 population over time. This model is described in more detail in Appendix D. The model compares a set of random future population trajectories in which the action occurs, and all murrelets assumed to be injured or killed at sea by the action are removed from the population, with an otherwise identical set of population trajectories representing the baseline, in which the action does not occur. Adjustments were made to both the baseline and action trajectories to account for the effects of recently consulted-upon actions that are not reflected in the population trend (particularly the Skookumchuck Wind Energy Project, which has only recently begun operating and has not had time to affect the population trend), and to avoid double-counting effects of the Navy's Gunnery Exercises, which to some extent are already reflected in the population trajectory. The model inputs for the action trajectory do not include effects to eggs or chicks enumerated above, but effects to eggs and chicks are reflected in the model because adults removed from the population during a given breeding season do not have any reproductive output during that (or following) seasons. In this way, we avoid double-counting effects to eggs and chicks, and we also avoid overstating the demographic effects of egg or chick mortality, which we expect to be high even without the effects of the action.

12.2.5.1.1.1 *Model Assumptions*

The results of this model, given below, must be interpreted in light of a number of the model assumptions. Some of these model assumptions have the effect of overstating effects to the population. In other cases, it is not clear whether a particular assumption would result in over- or underestimation of population-level effects. Still other assumptions have the effect of underestimating population-scale risks.

In the first category, the model incorporates the assumption that all individuals exposed to injury or mortality from the proposed action are removed from the population. In reality, some individuals, especially those exposed at the outer edges of the areas of effect, may receive relatively mild injuries that allow them to recover to some degree. We expect that murrelets, and especially breeding murrelets, operate on a tight energy budget; in other words, we expect that the energy output required for daily activities takes as much energy as they can consume via food, as is the case for related thick billed murres (*Uria lomvia*) (Elliott et al. 2014, pp. 137, 142). Therefore, any injury that impairs foraging, even to a small degree, is expected at minimum to interfere with nesting success during the season in which the injury occurs, if not increase the probability of mortality for the injured individual. However, if a murrelet is able to fully recover from injury, that individual may be able to reproduce in future seasons. Although we know that the injuries from the proposed activities will range from relatively mild injuries all the way to direct mortality, we lack finely-resolved information about the various degrees of injury that may result from the proposed activities, how easy or difficult it would be for murrelets to recover from each degree of injury, or how many murrelets might receive easily recoverable injuries versus severe or immediately fatal injuries. Therefore, we err on the side of the species by assuming that all injured murrelets are removed from the population, and acknowledge that in this way, the model may overstate effects to the population. In addition, our model of exposure itself makes a number of assumptions that may have the effect of overestimating exposure (see the Effects Section).

We calibrated demographic rates in our model to produce a near-term (20-year) rate of population change of -2.2 percent per year, as estimated from at-sea surveys (McIver et al. 2021, p. 20). However, for Zone 2, this estimate has a wide 95 percent confidence interval (-5.8 to 1.5). The fact that this confidence interval encompasses 0 and some positive numbers indicates some degree of statistical uncertainty about whether the population is actually declining, though the fact that the estimate and the bulk of the confidence interval are below 0 provide some evidence for a declining trend (McIver et al. 2021, p. 4). We note that the current every-other-year survey protocol is likely underpowered to provide high statistical confidence in trends shallower than -5 percent per year without several more years of surveys (see Bentivoglio et al. 2002, pp. 14-18). The confidence interval also indicates that the decline may be steeper than -2.2 percent per year. If the population is actually stable or increasing, or declining more much steeply than -2.2 per year, the demographic effects of the action, and our interpretation of those effects, may be altered. We attempted to account for these possibilities by examining population trajectories with the best and worst demographic rates, which resulted in rates of population change substantially different from -2.2 percent per year. Given low reproductive rates throughout the listed range of the murrelet, it is likely that if the population is stable or increasing, the increase is due to immigration, rather than reproduction. Our model does not account for immigration or emigration. We assume that all individuals present at sea within or off of Zone 2 are part of a closed population in Zone 2. If a substantial portion of murrelets present at sea are nonbreeding visitors to Zone 2, effects to these individuals would not affect the demography of the Zone 2 breeding population, and in this way our model could overestimate effects to Zone 2. On the other hand, in this case, the Zone 2 breeding population would be smaller than we currently estimate, and may be more vulnerable to the small population dynamics described above.

Finally, there are several factors not included in the model that would increase the risk to Zone 2. By not including these factors, we are likely to underestimate demographic risks to Zone 2. Our model does not account for expected effects of climate change to murrelets' marine and terrestrial habitats, nor does it account for anticipated increases in shipping or oil traffic (see Appendix B and Cumulative Effects section). These factors, individually or together, are likely to increase murrelet mortality or decrease fecundity, leading to steeper population declines. We do not have any quantitative estimate of how large these effects might be, or how they might change over time, but qualitatively, a steeper rate of decline is likely to make the small population dynamics described above occur sooner. Though our model does not include these factors, as mentioned above, we attempted to explore these possibilities by looking at the subset of our model runs with the worst demographic rates. Like other demographic models of murrelets, the model includes only females, but the exclusion of males from the model is likely to lead us to underestimate the effect of demographic stochasticity and the resulting extirpation risk (Bessa-Gomes et al. 2010, pp. 443-444; Møller 2003, p. 229). Similarly, our model does not account for Allee effects, and therefore underestimates the time until extirpation, after the population reaches small sizes (Akcakaya 1997, p. 12; Brassil 2001, p. 12).

On the whole, the demographic model likely overestimates the number and magnitude of effects to individuals, but likely underestimates the future rates of population decline in Zone 2, the baseline likelihood of reaching small population sizes, and the risk of extirpation once small population sizes are achieved. We considered these biases in interpreting model results.

Prior to constructing the model, we determined that the timeframe of the model would extend 200 years from the beginning of the proposed action, in keeping with the 200-year definition of long-term population viability given in the Recovery Plan (USFWS 1997, p. 112). However, we note that as the model extends into the future, we have decreasing confidence in our assumption that survival and reproductive rates will remain similar to current rates, and our confidence in the model results similarly is greater for years in the next few decades than for timeframes more than a century in the future. We also determined, before constructing the model, that we would examine the likelihood of achieving several benchmarks of small population size: 1,000 individuals (500 females), which is just below the lower end of the recent range of variation in Zone 2 population estimate; 500 individuals (250 females), which is the approximate size of the Zone 6 population and a population size below which small population dynamics may begin to be influential; and 100 individuals (50 females), which is the approximate size of the Zone 5 population and a size below which small population dynamics are likely to be very influential; and 0 individuals, or extirpation. Given the intrinsic biases of our demographic model explained above, our discussion below emphasizes the 500- and 100-individual benchmarks, as the chances of reaching the 0-individual benchmark (extirpation) are likely to be underestimated.

12.2.5.1.1.2 *Effects to Numbers at the Zone 2 Scale*

The best available evidence indicates that the Zone 2 population is currently declining. There is no indication that existing conservation measures have yet improved productivity, in spite of being enacted decades ago (in the case of the NWFP). Unless these conservation measures begin to improve productivity, or new, effective conservation measures can be enacted, we expect those declines to continue, regardless of the action. In our demographic model, the baseline scenario reflects this expectation. We expect that the removal of 53 birds at sea over a 16-year period will result in an approximately three percent reduction in the total population size in Zone 2, and this reduction will remain over the long term. Our demographic model results demonstrate that even as the baseline population trajectory declines, population trajectories including the effects of the action are, on average, three percent smaller than corresponding baseline population trajectories for the full 200-year model run, though with more variability in the later years of the simulation. As the simulated population sizes decline, the absolute difference between population trajectories with and without the action are also expected to decline, but the percentage difference between the two populations remains around three percent (Table 29). Qualitatively, the modeled persistence of this reduction in numbers is understandable, because we conclude that mortality outpaces reproduction in the Zone 2 population, and the population will therefore unable to replace murrelets removed from the breeding due to the proposed action.

Table 29. Median simulated Zone 2 population sizes, with and without the action.

	Median # Females		**Difference**	
Simulation Year	**Baseline**	**with Action**	**# of Females**	**as % of Baseline**
1	1,125	1,125	0	0
17	755	735	20	2.7%
50	339	331	8	2.4%
75	193	186	7	3.6%
100	110	106	4	3.6%
200	9	9	0	0

Because we expect continued population declines in Zone 2, we expect that the size of the Zone 2 population will eventually reach each of the population thresholds listed above, with or without the effects of the action. In our simulations, as expected, the likelihood of the population trajectory falling below each threshold increased over time. In the baseline simulations, population decline to the approximate size of Zone 6 (500 individuals, or 250 females) became more likely than not between year 50 and year 75 of the simulations. Population decline to the approximate size of Zone 5 (100 individuals, or 50 females) became more likely than not after year 130.

Because the action permanently reduces the population size, the action also increases the likelihood that the population will fall below a given threshold at any particular time. We calculated the increase in risk due to the action in two ways. We calculated the absolute risk increase, which is the percentage of simulation runs with the action that fell below a population threshold at a given time, minus the percentage of baseline simulation runs that fell below the same population threshold at the same time. We also calculated the relative risk increase, which is the number of simulation runs with the action that fell below a population threshold at a given time, expressed as a percentage of the number of baseline simulation runs that fell below the same population threshold at the same time, minus 100 percent.

In our simulations, the maximum absolute increase in risk, due to the action, of the Zone 2 population reaching the size of Zone 6 (500 individuals) was 2 percent, and the absolute increase in risk was greatest during the period between year 50 and year 80. The period with the greatest increases in relative risk occurred immediately after the term of the action, in years 17 through 26. During this timeframe, the baseline absolute risk of reaching the 500-individual threshold rose from 4 percent to 11 percent, and the relative risk increase caused by the action ranged from 8 percent to 14 percent. During this timeframe, the increase in absolute risk attributable to the action remained around 1 percent or lower.

The maximum absolute increase in risk, due to the action, of the Zone 2 population reaching the size of Zone 5 (100 individuals) was 2 percent, and the absolute increase in risk was greatest during the period between years 126 and 135. The period with the greatest increases in relative risk occurred during years 41 through 50. During this timeframe, the baseline absolute risk of reaching the 100-individual threshold remained below 4 percent, and the relative risk increase caused by the action ranged from 4 percent to 14 percent. During this timeframe, the increase in absolute risk attributable to the action remained less than 1 percent.

We examined the subset of model runs that had both survival and fecundity rates in the highest quartiles during the first 100 years of the simulation. These model runs were, on average, nearly stable over the 100-year timeframe. Among these model runs, the likelihood of reaching the 500-individual small population threshold remained low (less than 10 percent) during the first 100 years, though the likelihood was higher with the action than without. None of these model runs reached the 100-individual threshold during the first 100 years, with or without the action.

We examined the subset of model runs that had both survival and fecundity rates in the lowest quartiles during the first 100 years of the simulation. These model runs, on average, declined at 4.2 percent per year during the first 100 years. In these simulation runs, the baseline population was at increased risk of reaching small population thresholds, compared with the full set of simulation runs. In addition, in these model runs, the increase in risk attributable to the action was larger than in the full set of simulation runs. The maximum increase in risk of the population reaching the 500-individual threshold was 3 percent. Within this subset, between years 17 and 26, the percentage of baseline simulation runs reaching the 500-individual threshold rose from 12 to 32 percent. The relative risk increase caused by the action was between 1 and 12 percent during this period. The maximum increase in risk of the population reaching the 100-individual threshold was 4.6 percent. Between years 61 and 70, the percentage of baseline simulation runs reaching the 100-individual threshold rose from 30 to 43 percent. The relative risk increase caused by the action was between 5 and 11 percent during this period.

These two subsets of simulation runs are relevant due to the uncertainty surrounding the currently-estimated 2.2 percent annual decline in the Zone 2 population. It is statistically possible that, between 2001 and 2019, the Zone 2 population did not decline significantly and that the underlying trend was actually stable or increasing, but appears to be negative due to natural variability and sampling error. If this is the case, and the stable or increasing trend is sustained into the future, the effects of the action will still cause the Zone 2 population to be smaller than it would without the action, but the risk of reaching small population sizes, and particularly very small sizes similar to the Zone 5 population will remain low even with the action. The possibility that the Zone 2 population is increasing seems unlikely, at least with respect to the breeding population in Zone 2, given that productivity measures indicate reproductive rates far below those needed for population stability.

It is also possible that the Zone 2 population declined faster than 2.2 percent per year from 2001 and 2019, or that the 2.2 percent per year decline is fairly accurate for those years but will become steeper in the coming years or decades. Our examination of the demographically-worst model runs indicated that in this case, the population is at greater risk of reaching small population sizes over the near- to medium-term, even without the action, and that the effect of the action to increase those risks would be greater than if demographic rates consistent with a current 2.2 percent decline is maintained into the future.

12.2.5.1.1.3 Effects to Reproduction in Zone 2

Multiple lines of evidence indicate that reproduction in Zone 2 is currently too low, relative to survival rates, to allow murrelets to replace themselves during their lifetimes. Climate change trends that are currently underway, as well as cumulative effects, are likely to reduce reproductive rates further. The action is also expected to affect reproduction. The action will injure individuals in ways that will reduce their ability to nest successfully for one or more years following the injury. The action will kill individuals, removing their future reproductive potential and reducing the total reproductive output of the Zone 2 population. Finally, as described above, the action hastens the likely development of small population dynamics, which result in reduced reproduction via stochastic imbalances in sex ratios, as well as reduced social facilitation of breeding.

12.2.5.1.1.4 Effects to Distribution in Zone 2

Murrelets in Zone 2 are currently concentrated in the northern portion of the Zone, both in the marine environment and in their use of nesting habitat. At the time of the Recovery Plan (USFWS 1997, p. 126), a gap in the range within Zone 2 was noted between the northern and southern portions of the Zone. At present, the southern subpopulations within Zone 2 are sparse, and the gap in the range appears to have widened, rather than narrowed. Murrelets in Zone 2 are generally present at greater distances from shore in Zone 2 than in other portions of the listed range.

The action is not expected to result in direct alterations of murrelet distribution in Zone 2. The bulk of the injury and mortality to murrelets resulting from the action will be associated with mine countermeasure and neutralization testing activities, which will occur between 3 and 7.5 nautical miles from shore, just south of the OCNMS, in the middle of the north-south range of Zone 2, near the transition between the dense concentrations of murrelets to the north and the sparser scattering of murrelets to the south. However, mine countermeasure and neutralization testing activities will take place in these locations for a few days each year, and we expect that surviving murrelets will continue to use these marine areas as usual whenever Navy activities are not underway. Some injury and mortality are also expected from Gunnery Exercises, which occur at greater distances from shore. We expect that murrelets will rarely be present at these distances from shore during the breeding season, but will be present more often during the non-breeding season. However, we expect that on average, murrelets and Gunnery Exercises will coincide in space and time approximately once per year in Zone 2, and that murrelet distribution will be unaffected.

However, as the action reduces the population size beyond the already-occurring population decline, this reduction is likely to exacerbate the ongoing geographic contraction of the Zone 2 population. With a smaller population in Zone 2, the sparse subpopulations at the southern end of Zone 2 are likely to become even sparser or disappear, and the area of concentrated murrelet use at the northern part of Zone 2 may contract northward.

12.2.5.1.1.5 *<u>Effects to the Likelihood of Survival in Zone 2</u>*

The best available information indicates that the Zone 2 population is currently declining, and is on a trajectory to shrink, over a period of more than 100 years, to small population sizes (e.g., 500 individuals, then 100 individuals, and so on). Once the population is small enough (the precise threshold is unknown), small population dynamics and Allee effects will increase the likelihood of extirpation within Zone 2. Ongoing climate change and increasing cumulative effects are likely to speed up the process of decline.

The action will also hasten the decline to small population sizes. However, the absolute increase in risk, at any given time, is small. At demographic rates consistent with the currently-estimated 2.2 percent annual decline, at any given time, the action will lead to, at most, a 2 percent increase in absolute risk of reaching one of the small-population thresholds discussed above. Although the maximum relative risk increase is larger (up to 14 percent), the maximum increases in relative risk occur when the baseline likelihood of reaching these small population thresholds is low: 11 percent or less for the 500-individual threshold, and less than 4 percent for the 100-individual threshold.

Even considering scenarios in which the demographic rates are, or will soon become, worse than estimated – as may be the case given climate change and cumulative effects – the effect of the action on the likelihood of achieving small population sizes is relatively small. In the subset of model runs with the worst demographic rates, at any given time, the action will lead to, at most, a 4.6 percent increase in absolute risk of reaching one of the small-population thresholds discussed above. The maximum relative risk increase is smaller than in the larger set of model runs, and the maximum increases in relative risk occur when the baseline likelihood of reaching these small population thresholds is still less than even: 32 percent or less for the 500-individual threshold, and 44 percent or less for the 100-individual threshold.

As noted above, once the Zone 2 population reaches these small population sizes, and especially after it falls below the 100-individual threshold, it will be at increased risk from demographic stochasticity and Allee effects, which would likely lead to extirpation more quickly than implied by our model results.

In light of the current baseline population trend, ongoing climate change effects, and cumulative effects, we find that the action is likely to reduce numbers and reproduction to a degree that the likelihood of extirpation, at any given time in the future, will be increased in Zone 2. The action does increase the risk of reaching small population sizes, sooner than would otherwise occur. However, the increase in risk attributable to the action is relatively small, and is not substantial enough to be "appreciable" which is defined by the Services' Consultation handbook to mean "considerable" (USFWS and NMFS 1998, p. 4-36).

12.2.5.1.1.6 Effects to the Likelihood of Recovery in Zone 2

Although recovery is assessed at the scale of the listed entity, we can evaluate the likelihood that Zone 2 will achieve a condition that contributes to recovery at the larger scale. Criteria for recovery include the presence of a stable or increasing population, as well as stable or increasing productivity, over a period of at least ten years, in at least four of the six conservation zones (USFWS 1997, p. 113).

The best available information currently indicates that the Zone 2 population is most likely declining, and that productivity is currently too low to sustain population stability or increase. Therefore, in order to achieve recovery objectives, productivity must be increased in Zone 2, without simultaneous increases in mortality. Ongoing climate change trends, as well as projected cumulative effects, are more likely to lead to incremental, or periodically dramatic, decreases in productivity, and could increase mortality as well. The effects of the action to murrelet numbers and reproduction in Zone 2 involve increased mortality and decreased productivity.

The Zone 2 population is not currently positioned to contribute to the recovery of murrelets, and current trends in the baseline and projected cumulative effects are likely to make it harder, rather than easier, to increase productivity and stabilize reverse the population decline. The effects of the action will make these tasks even more difficult. Therefore, the action appreciably reduces the likelihood that Zone 2 will contribute to the recovery of the species.

12.2.5.1.2 Effects of the Action in Zone 1

As noted above, the action is expected to disrupt the essential behaviors of 8 murrelets at sea, and to injure or kill 4 murrelets at sea and 4 eggs or chicks in Zone 1, for a total of 16 individuals of all life stages. Over the past five surveys (2014-2016, 2018, 2020), the estimated population size in Zone 1 has ranged from 2,822 individuals in 2014 to 4,614 individuals in 2016, with an average of 3,742 birds (McIver et al. 2021, pp. 15-17). Thus, over the 16 years of the proposed action, we expect that 0.4 percent (16 divided by 3,742) of the average total population of Zone 1 will experience significant behavioral disruption, injury, or mortality as a result of the proposed activities. Since this population is currently declining, with mortality outpacing reproduction, we do not expect that the population will be resilient to these effects.

12.2.5.1.2.1 Effects to Numbers in Zone 1

The best available evidence indicates that the Zone 1 population is currently declining, and unless new, effective conservation measures can be enacted, we expect those declines to continue, regardless of the action. Because the number of birds affected in Zone 1 was smaller than the number of birds affected in Zone 2, and the population of Zone 1 is larger than that of Zone 2, we did not perform a demographic analysis for Zone 1, comparable to our analysis for Zone 2. Nevertheless, we can still draw conclusions about the effects of the action to numbers of murrelets in Zone 1. Because mortality is currently outpacing reproduction in Zone 1, we do not expect that the Zone 1 population will be able to replace individuals lost to the population due to the effects of the action. As was projected for Zone 2, we expect that a comparison of the population trajectory with and without the action would show a declining population in the baseline scenario, and a similarly declining, but slightly smaller, population in the action scenario. However, we expect the action to make less difference in Zone 1, as compared with Zone 2, because the action is expected to affect fewer individuals among the larger Zone 1 population. Furthermore, the majority of affected individuals are expected to experience significant disruption of normal behaviors, rather than actual injury or mortality. Although these behavioral disruptions are expected to lead to increases in energy expenditure and decreases in energy intake, thereby increasing the likelihood of injury, we expect that many murrelets experiencing significant behavioral disruption will be able to recover, survive, and potentially produce offspring in the future.

As in Zone 2, the Zone 1 population is currently on a trajectory in which it will decrease to ever-smaller population sizes, eventually reaching the 500-individual threshold, the 100-individual threshold, and extirpation. As in Zone 2, the effects of the action will hasten, rather than retard, that process. However, given the proportionally much smaller effect of the action to Zone 1, as compared with Zone 2, we expect that the increases in absolute or relative risk that can be attributed to the action will be very small.

12.2.5.1.2.2 Effects to Reproduction in Zone 1

As in Zone 2, multiple lines of evidence indicate that reproduction in Zone 1 is currently too low, relative to survival rates, to allow murrelets to replace themselves during their lifetimes. Climate change trends that are currently underway, as well as cumulative effects, are likely to reduce reproductive rates further. The action is also expected to affect reproduction. The action will disrupt foraging and resting behavior for some individuals, and injure other individuals. We expect that both of these effects will result in reduced reproductive success in one or more years following the bird's exposure to stressors associated with the proposed activities. The action will kill a smaller number of individuals, removing their future reproductive potential and reducing the total reproductive output of the Zone 1 population. Although the Zone 1 population is currently larger than the Zone 2 population, it is declining more rapidly, and without improvements in productivity, will reach small population sizes, resulting in reduced reproduction via stochastic imbalances in sex ratios, as well as reduced social facilitation of breeding. The action will slightly hasten, rather than slow, this process.

12.2.5.1.2.3 *Effects to Distribution in Zone 1*

Murrelet distribution at sea in Zone 1 during the nesting season is currently associated with the presence of large, contiguous blocks of nesting habitat in nearby terrestrial areas, as well as with lower human footprint in the marine environment. The proposed action will not alter the distribution or amount of nesting habitat. The proposed activities in Zone 1 will be carried out in locations where human activity in the marine environment is already elevated, at Navy installations or near civilian ports. Therefore, we do not expect alterations in the distribution of murrelets in Zone 1 to result directly from the proposed action.

Murrelet distribution in Zone 1 is currently patchy, with concentrations of murrelets in some areas and very low densities in other areas (for example, see Raphael et al. 2015, p. 21). As the population declines, it is not clear whether the patchiness of the murrelet distribution will increase, or whether reductions in at-sea density will occur throughout Zone 1. However, we expect that murrelet distribution in Zone 1 will continue to depend on proximity to nesting habitat and likely the availability of forage, and will continue to be influenced by the marine human footprint. Although the action will contribute slightly to population declines in Zone 1, which may result in some alteration of murrelet distribution, we do not expect that these alterations will substantially change the pattern of murrelet distribution in Zone 1.

12.2.5.1.2.4 *Effects to the Likelihood of Survival in Zone 1*

The best available information indicates that the Zone 1 population is currently declining, and is on a trajectory to shrink to small population sizes (e.g., 500 individuals, then 100 individuals, and so on), at which point small population dynamics and Allee effects will increase the likelihood of extirpation within Zone 2. We did not estimate a timeframe in which the Zone 1 population is likely to reach these small population sizes. Compared with Zone 2, which we projected will likely reach the small size of 100 individuals by around 130 years from now, the Zone 1 population is currently nearly twice as large, but also declining more than twice as fast. Ongoing climate change and increasing cumulative effects are likely to speed up the process of decline.

The action will also slightly hasten the decline to small population sizes. As noted above, once the Zone 1 population reaches these small population sizes, and especially after it falls below the 100-individual threshold, it will be at increased risk from demographic stochasticity and Allee effects, which would likely lead to extirpation relatively quickly. However, compared with the effects of the action in Zone 2, we expect that the effects of the action in Zone 1 will result in a smaller increase in the risk of reaching these small population sizes and subsequently becoming extirpated. This is because the proportion of the population affected in Zone 1 is much smaller than the proportion affected in Zone 2, and furthermore, we expect a smaller proportion of the affected individuals to be injured or killed in Zone 1, compared with affected individuals in Zone 2.

In light of the current baseline population trend, ongoing climate change effects, and cumulative effects, we find that the action is likely to reduce numbers and reproduction in Zone 1, and will contribute a small increase in risk that, at any given time, the population will reach the small population thresholds discussed above and will become extirpated. However, given that the increase in risk attributable to the action will be even smaller than the increase in risk described for Zone 2, it is not appreciable.

12.2.5.1.2.5 Effects to the Likelihood of Recovery in Zone 1

Although recovery is assessed at the scale of the listed entity, we can evaluate the likelihood that Zone 1 will achieve a condition of contributing to recovery at the larger scale. Criteria for recovery include the presence of a stable or increasing population, as well as stable or increasing productivity, over a period of at least ten years, in at least four of the six conservation zones (USFWS 1997, p. 113).

The best available information currently indicates that the Zone 1 population is declining more rapidly than populations in other Zones, and that productivity is currently too low to sustain population stability or increase. Therefore, in order to achieve recovery objectives, productivity must be increased in Zone 1, without simultaneous increases in mortality. Cumulative effects are likely to have the largest impact in Zone 1, and in addition to ongoing climate change trends, are more likely to lead to incremental, or periodically dramatic, decreases in productivity, and occasionally increases in mortality as well. The effects of the action to murrelet numbers and reproduction in Zone 1 involve some additional mortality as well as decreased productivity.

The Zone 1 population is not currently positioned to contribute to the recovery of murrelets, and current trends in the baseline and projected cumulative effects are likely to make it harder, rather than easier, to increase productivity and stabilize reverse the population decline. The effects of the action will make these tasks somewhat more difficult. Therefore, the action reduces the likelihood that Zone 1 will contribute to the recovery of the species, although we expect that this reduction in likelihood is relatively small, given the proportion of the Zone 1 population we expect will be affected by the action.

12.2.5.1.3 Effects of the Action in Zones 3 and 4

The effects of the proposed action at the Zone scale is likely to be similar in Zones 3 and 4, so we discuss these two Zones together. As noted above, the action is expected to injure or kill 17 individuals in Zone 3 and 15 individuals in Zone 4. We expect that murrelets in Zones 3 and 4 will be affected only during the non-breeding season, when murrelets are distributed farther offshore. Therefore, we do not expect effects to eggs or chicks in Zones 3 and 4. In Zone 3, the estimated population size over the past five surveys (2013-2014, 2016, 2018, 2020), has ranged from 6,813 individuals in 2016 to 8,841 individuals in 2014, with an average of 8,061 birds (McIver et al. 2021, pp. 11-17). In Zone 4, the estimated population size over the past five surveys (2012, 2013, 2015, 2017, 2019), has ranged from 4,960 individuals in 2012 to 8,743 individuals in 2015, with an average of 7,023 birds (McIver et al. 2021, pp. 11-175). Thus, over the 16 years of the proposed action, we expect that 0.2 percent (17 divided by 8,061, or 15 divided by 7,023) of the average total population of each Zone will experience injury or mortality as a result of the proposed activities.

12.2.5.1.3.1 Effects to Numbers in Zones 3 and 4

The best available evidence indicates that the number of murrelets at sea has been increasing in Zones 3 and 4. Productivity remains low in these Zones, so it is likely that at least some of these population increases result from temporary migratory movements from other Zones or from outside of the listed range. If so, this indicates a disconnect between the long-term, breeding population associated with each Zone and the birds that are present at sea. The long-term, breeding population in each Zone would presumably be a subset of the birds present at sea in that Zone during surveys, but it is impossible to estimate the size of the breeding population subset for each Zone. Given sustained low productivity rates in these Zones, it is likely that the breeding population in each Zone is declining even while the number of birds present at sea increases.

Because the action will affect birds at sea, visiting migrants in Zones 3 and 4 will be affected as well as members of the long-term breeding populations. We have no reason to believe that the effects of the action will disproportionally affect visiting migrants or long-term residents. For example, if two-thirds of the at-sea population consisted of residents and one-third consisted of visitors, then we would expect that two-thirds of the birds injured or killed would be residents and one-third of the birds injured or killed would be visitors. Because we expect that residents and visitors will be affected in proportion to their presence in that at-sea population, this means that the percentage of birds injured or killed over 16 years will still be 0.2 percent, as calculated above.

If long-term breeding populations in Zone 3 and/or Zone 4 are truly declining, then the effects of the action will contribute slightly to that decline, but this population-scale effect will be even less significant than that described for Zone 1. If the productivity data available for Zones 3 and 4 are misleading, and in fact the breeding populations are stable, the action will reduce numbers very slightly but will not alter the long-term population trend. If in fact the breeding populations are increasing, the action will temporarily retard the increase, but the effect will not be large enough to change the increase to a decline. Because there are several hypotheses that would make low productivity compatible with increasing numbers of murrelets at sea (e.g., the presence of migrants, as discussed above), we do not assume that the productivity data are wrong, but simply consider the possibility to demonstrate that in that case, the action would not result in a large effect to numbers in Zones 3 and 4.

12.2.5.1.3.2 *Effects to Reproduction in Zones 3 and 4*

Although at-sea survey data indicate that the numbers of murrelets at sea are increasing in Zones 3 and 4, productivity data indicate that reproduction in Zones 3 and 4 is currently too low, relative to survival rates, to allow murrelets to replace themselves during their lifetimes. Climate change trends that are currently underway, as well as cumulative effects, are likely to reduce reproductive rates further. The action is also expected to affect reproduction. Individuals affected by the action, if not killed, will be injured in ways that are expected to result in reduced reproductive success in one or more years following the injury. The action will also kill individuals, removing their future reproductive potential and reducing the total reproductive output of the breeding populations of Zones 3 and 4. Given the large numbers of murrelets at sea in Zones 3 and 4, we expect that it would take a very long time for these populations to decline to the small population sizes where skewed sex ratios and Allee effects would affect reproduction. However, if the long-term, breeding populations in these Zones make up only a small proportion of the birds present at sea, small breeding population sizes may come about more quickly. The action is expected to have only a minor contribution to population decline that may be occurring in these Zones.

12.2.5.1.3.3 *Effects to Distribution in Zones 3 and 4*

Murrelets in Zone 3 are currently concentrated in the central and southern portions of the Zone, both in the marine environment and in their use of nesting habitat. At the time of the Recovery Plan (USFWS 1997, p. 126), a gap in the range was noted in northern portions of Zone 3. Murrelets at sea are fairly well-distributed throughout Zone 4, with scattered areas of higher or lower density (Raphael et al. 2015, p. 21), but the distribution of nesting habitat in Zone 4 has become patchier after large fires caused significant habitat loss.

The action is not expected to result in direct alterations of murrelet distribution in either Zone. All injury and mortality in these Zones is expected to result from Gunnery Exercises, which occur far from shore. We expect that murrelets will rarely be present at these distances from shore during the breeding season, but will be present more often during the non-breeding season. However, only 5 percent of all Gunnery Exercises will occur in each Zone, and we expect that on average, approximately one murrelet will encounter Gunnery Exercises each year in each Zone. We expect that murrelet distribution will be unaffected.

12.2.5.1.3.4 *Effects to the Likelihood of Survival in Zones 3 and 4*

Given the apparently conflicting information regarding population trends and productivity in Zones 3 and 4, the likelihood of long-term survival or extirpation in these Zones is unclear. Productivity information indicates that breeding populations in these Zones are likely decreasing, which would indicate a trajectory toward extirpation. However, these Zones have typically maintained relatively large populations (around 4,000 individuals or more) throughout the last two decades, so unless the breeding populations are declining very rapidly, we would not expect small population dynamics to affect these populations in the near- or medium-term.

Instead, these populations are more likely to be at risk of sudden population decline due to large-scale environmental stochasticity or occasional catastrophes, for example, warm-water events such as the 2014-2016 marine heatwave, widespread wildfire resulting in the loss of large amounts of nesting habitat, or large-scale oil spills. Extirpations stemming from large-scale environmental stochasticity or catastrophic events may not be sensitive to initial population size, especially for species that are also vulnerable to Allee effects at low population sizes (Roth and Schreiber 2014, pp. 196-198), so even large populations may be at risk. These environmental changes are becoming more likely, given trends in climate change and projected cumulative effects. Since these risks are not necessarily sensitive to initial population size, the loss of a small number of individuals due to the proposed action will not affect the level of risk to Zones 3 and 4.

If breeding populations in Zones 3 or 4 are currently smaller than their historical sizes, and also declining as rapidly as productivity data indicate, decline to small population sizes is a possibility. As in other Zones, the slight decrease in numbers and reproduction due to the proposed action will contribute slightly to the decline to small population sizes. As in other Zones, if the Zone 3 or Zone 4 population reaches these small population sizes, and especially if it falls below the 100-individual threshold, there will be an increased risk from demographic stochasticity and Allee effects, which would likely lead to extirpation relatively quickly. However, compared with the effects of the action in Zone 1, we expect that the effects of the action in Zone 3 or 4 will result in even smaller increases in the risk of reaching these small population sizes and subsequently becoming extirpated.

In light of uncertainty regarding population trend and productivity in these Zones, ongoing climate change effects, and cumulative effects, we find that the action will reduce numbers and reproduction in Zones 3 and 4, and may contribute a small increase in risk that, at any given time, the population will reach the small population thresholds discussed above and will become extirpated. However, given that the increase in risk attributable to the action will be even smaller than the increase in risk described for Zone 1, it is not appreciable.

12.2.5.1.3.5 *Effects to the Likelihood of Recovery in Zones 3 and 4*

Although recovery is assessed at the scale of the listed entity, we can evaluate the likelihood that each of Zones 3 and 4 will achieve a condition of contributing to recovery at the larger scale. Criteria for recovery include the presence of a stable or increasing population, as well as stable or increasing productivity, over a period of at least ten years, in at least four of the six conservation zones (USFWS 1997, p. 113).

The best available information currently indicates that these two populations have been increasing during the 2001-2018 period. This is a period of greater than ten years, so the Zone 3 and Zone 4 populations do currently meet the criteria for stable or increasing populations. However, information regarding productivity indicates that it is still too low in these Zones to sustain long-term population stability or increase. Therefore, in order to achieve recovery objectives, productivity must be increased in Zones 3 and 4, without simultaneous increases in mortality. Climate change, as well as cumulative effects, will likely to lead to incremental, or periodically dramatic, decreases in productivity, and occasionally increases in mortality as well. The effects of the action to murrelet numbers and reproduction in Zones 3 and 4 involve some additional mortality as well as decreased productivity.

Because population trend information indicates that the number of birds at sea is increasing, while productivity information indicates that breeding populations should be declining, it is not clear how well positioned they are to contribute to the recovery of murrelets. Recovery criteria call for stable or increasing population numbers for a period of at least ten years, and in these Zones, the population trends measured over the period between the years 2000 and 2019, a period of more than ten years, are stable or increasing, although the numbers showed a decline (albeit not statistically significant) during the first half of that period (Miller et al. 2012, p. 775). Recovery criteria also call for stable or increasing productivity. Trends in productivity have not been measured in these Zones. There is no particular indication that productivity is declining, and it is quite possible that it is stable overall, as is the case in Zone 1 (Lorenz and Raphael 2018, p. 211); however, even if it productivity is stable, the measured rates are low to support population stability over the long term. Current trends in the baseline and projected cumulative effects are likely to make it harder, rather than easier, to increase productivity in the future. The effects of the action will make this task very slightly more difficult. Therefore, the action may reduce the likelihood that Zones 3 and 4 will contribute to the recovery of the species, but if so, we expect that any reduction in likelihood will be relatively small, and not appreciable.

12.2.5.2 Effects of the Action Rangewide

For our analysis of the effects of the action rangewide, we synthesize the effects of the action in each Zone, rather than averaging all effects over the entire rangewide population area. The action will affect four of the six conservation Zones. The four affected Zones are the same four that we rely upon for the recovery and long-term survival of the species, because the remaining two Zones have populations that are too small, and too at-risk, to be relied upon for recovery and long-term survival.

12.2.5.2.1 Effects to Numbers Rangewide

In total, over the 16 years of the action, we expect that 96 adults, subadults, or fledged juveniles will be injured, killed, or experience significant disruption of normal behaviors at sea as a result of the action, and as a result of these effects to breeding adults, 17 eggs or chicks at the nest will also be injured or killed. These effects will occur across Zones 1 through 4, but the majority of the effects will be concentrated in Zone 2. Zone 2 also currently has the smallest murrelet population of any of the four affected Zones, so the disproportionate impact on Zone 2 will be even greater.

Populations in Zone 1 and 2 are currently declining, with the steepest and most statistically certain decline in Zone 1. Zones 3 and 4 are currently increasing, but other lines of evidence indicate that productivity remains low throughout the range, including in Zones 3 and 4, such that murrelets are not replacing themselves. Therefore, in general, we do not expect the populations in any of the Zones to be able to replace the loss of individuals resulting from the action. In Zones 1 and 2, we expect continuing downward population trajectories, and the action will reduce the number of individuals in the population further, especially in Zone 2. In Zone 2, in particular, we expect that the effects of the action will hasten the population’s decline to small population sizes, though we anticipate that these small population sizes will come about with or without the action. As we noted above, even with the action, we expect that it will take at least several decades for the Zone 2 population to reach a very small size (100 individuals), although intervening stressors could accelerate that timeline. In Zone 1, we expect that the action will hasten the population’s decline to a lesser degree, though we anticipate small population sizes will ultimately occur in Zone 1 as well. In Zones 3 and 4, the size and trend of breeding populations are not clear, because the at-sea populations are increasing, likely due at least in part to visiting migrants. However, in Zones 3 and 4, we expect that the action will have little influence on the population size and trend, given that the action will affect a very small proportion of the population in these zones.

In summary, given current population trends, trends in environmental conditions due to climate change, and cumulative effects, we expect continued declines in the northern populations, and we are uncertain about future trends in population sizes in Zones 3 and 4. We expect that the action will contribute to population declines in Zone 2, in particular, and to a much smaller extent, in Zone 1. We expect any effect of the action on future population trends in Zones 3 and 4 will be miniscule. The populations in Zones 5 and 6 are currently very small populations, but are unaffected by the action. We expect that over the coming decades, small population dynamics, which are likely already affecting Zone 5 and possibly Zone 6, will begin to affect Zone 2, with some contribution from the proposed action, and possibly Zone 1, with a much smaller contribution from the proposed action. Zones 3 and 4 may retain large population sizes, or as monitoring of these populations continues, we may discover that these populations have resumed their previous declines, but the action is unlikely to be influential in this process.

12.2.5.2.2 Effects to Reproduction Rangewide

Productivity data indicate that reproduction throughout the listed range is currently too low, relative to survival rates, to allow murrelets to replace themselves during their lifetimes. Climate change trends that are currently underway, as well as cumulative effects, are likely to reduce reproductive rates further. The action is also expected to affect reproduction in each of the Zones. Individuals affected by the action, if not killed, will be injured or disturbed in ways that are expected to result in reduced reproductive success in one or more years following their encounters with the action. The action will also kill individuals, removing their future reproductive potential and reducing the total reproductive output of each of the affected populations.

In Zone 2, and to a much smaller extent in Zone 1, we expect that the action will hasten the reduction in population size to sizes so small that skewed sex ratios and Allee effects will reduce reproduction further. In Zone 5, we expect that the population may already be at such a low size. In Zone 6, the population currently appears to be larger than the sizes where these small population effects are typical, but the recent loss to wildfire of more than one quarter of suitable nest trees will likely reduce reproduction over the coming decades. In Zones 3 and 4, we do not have information indicating that these effects of small population sizes are likely to occur in the foreseeable future.

In short, reproduction is impaired throughout the range. In the Zones affected by the action, reproduction will be further impaired by the effects of the action, but the demographic effects of this effect of the action will vary by Zone. Where mortality and reduced reproduction as a result of the action contribute to population trajectories toward very small population sizes, in Zone 2 and to a lesser extent in Zone 1, the direct effects to reproduction resulting from the action will later be compounded by reductions in reproduction due to small population dynamics. Effects to reproduction in Zones 3 and 4 are likely to be confined to the direct effects resulting from injuries caused by the action. Zones 5 and 6 are not affected by the action, but are already expected to experience further impaired reproduction due to the small population in Zone 5 and the loss of nesting habitat in Zone 6. Therefore, the unaffected Zones are not able to compensate in any way for the reductions in reproduction in the affected Zones.

12.2.5.2.3 Effects to Distribution Rangewide

Currently, there are two notable gaps in the distribution of murrelets within the listed range. The large gap between populations in Zone 5 and Zone 6 is a result of large-scale historical habitat loss and will be difficult to overcome. The other gap is between the southern portion of Zone 2 and the northern portion of Zone 3. Murrelets are present in the southern portion of Zone 2 and the northern portion of Zone 3, but at-sea densities are low and nesting habitat is relatively scarce in these regions.

We do not expect that the action will directly affect distribution or alter these gaps. We also do not expect any influence of the action on the gap between Zones 5 and 6. However, as the Zone 2 population reaches small population sizes as a result of baseline trends, cumulative effects, and the effects of the action, we expect that the murrelet's range in Zone 2 will contract further, increasing the gap in the range between the bulk of the populations in Zone 2 and Zone 3.

Ultimately, unless murrelet productivity increases substantially, populations in Zones 1 and 2 are, over the long term, likely to reach small population sizes such that demographic stochasticity and Allee effects lead to extirpation. The action will have some contribution to this process, with a larger contribution in Zone 2 than in Zone 1. Extirpation in Zones 1 or 2 will create much larger gaps in the range than those currently present, and substantially reduce the distribution of the species throughout the listed range.

12.2.5.2.4 Effects to the Likelihood of Survival Rangewide

Long-term survival, defined in the Recovery Plan as survival for more than 200 years, depends on the presence of stable, productive, well-distributed populations in Zones 1 through 4 (USFWS 1997, pp. 112-113). This condition will provide redundancy and increase the resiliency of the species within the listed range. Redundancy and the capacity for resilience are essential to protect the DPS from large-scale environmental fluctuations and catastrophic events, which can threaten even healthy populations. Initial population size does not necessarily protect against extirpations caused by these types of stressors, but when populations are distributed across a wide area, it is less likely that all populations will be simultaneously affected by the same stressor. Similarly, when populations are present in multiple different environmental contexts, this can protect against all populations becoming simultaneously affected by a single widespread stressor. For example, the Zone 1 population is particularly important to the resiliency of the murrelet DPS, because it is located in the Salish Sea, whereas all of the other populations are located along the California Current ecosystem. The Zone 1 population is therefore insulated from large-scale stressors affecting the other Zones, such as the marine heatwave of 2014-2016, or the delayed upwelling of 2005.

Currently, murrelet populations in Zones 1 through 4 are relatively large (more than 1,000 murrelets at sea estimated for each population), and at the Zone scale, are unlikely to be affected at present by demographic stochasticity acting on small populations. Population estimates are declining in Zones 1 and 2, and increasing in Zones 3 and 4, but productivity is low throughout the DPS. Given current population trends and low productivity, the expected degradation of marine and terrestrial habitat conditions due to climate change, and cumulative effects, we expect that murrelet populations in Zones 1 and 2 will continue to decline, eventually reaching such small population sizes that demographic stochasticity, and eventually Allee effects, will lead to extirpation. The loss of these conservation Zones will decrease redundancy and resiliency of the DPS as a whole, making its long-term persistence less likely. Although populations in Zones 3 and 4 are larger, and population estimates derived from at-sea surveys have increased in recent years, productivity in these populations remains too low to support population stability. These populations are at risk of resuming their previous declines. However, we do not at this point anticipate that these populations are in danger of reaching the small population sizes (500 individuals, 100 individuals) discussed above for Zones 1 and 2. These populations remain at risk from large-scale environmental fluctuations and catastrophic events, and their current large population sizes do not necessarily protect them from these stressors. If these become the only remaining large populations within the listed range, this will decrease the likelihood of survival of the entire listed entity, especially given that Zones 3 and 4 are adjacent to one another and will likely experience many of the same environmental stressors.

The action will contribute to declines in the northern two Zones, especially in Zone 2. However, at any given time, the absolute increase in risk that Zone 2 will reach these small population sizes remains relatively small, with a maximum increase in risk of 2 percent if demographic rates remain consistent with current declines, or less than 5 percent if demographic rates are or become somewhat worse. This increase in risk is not great enough to be called appreciable. Furthermore, even with worse demographic rates, we do not expect the Zone 2 population to reach a very small size (100 individuals) for several decades, and this remains true with or without the effects of the action.

In conclusion, the long-term survival of the murrelet DPS is at increasing risk, and the effects of the action are likely to contribute to the increase in risk, but we expect the alteration in the likelihood of survival that can be attributed to the action will be relatively small and not appreciable.

12.2.5.2.5 Effects to the Likelihood of Recovery Rangewide

Criteria for recovery include the presence of a stable or increasing population, as well as stable or increasing productivity, over a period of at least ten years, in at least four of the six conservation zones (USFWS 1997, p. 113). Zones 5 and 6 are small populations, and at high risk of further habitat loss. In most surveys, the population Zone 5 has been so small that demographic stochasticity and Allee effects appear likely (though not certain) to lead to extirpation. In Zone 6, even before the recent loss to wildfire of more than one quarter of all suitable nest trees, the population showed signs of acting as a demographic sink. These factors indicate that Zones 5 and 6 cannot be counted on to contribute to recovery, so recovery will require stable or increasing populations and productivity in the remaining populations in Zones 1 through 4. Well-distributed populations in Zones 1 through 4 are also necessary to allow for recovery (USFWS 1997, p. 116).

Population and productivity data indicate that the rangewide murrelet population is not currently poised for recovery. Zones 1 and 2 are currently declining, while Zones 3 and 4 are increasing. Productivity is too low in all Zones to support population stability, so even if productivity is currently stable, as observed in Zone 1, it must increase in order to support recovery. Because productivity data indicates low reproductive rates in Zones 3 and 4, in spite of the increasing populations in those Zones, it is likely that the population increases in those Zones are due to visiting migrants from other Zones or from outside of the listed range. Therefore, population increases in Zones 3 and 4 may be short-lived, and we consider these populations to be at high risk of resuming their previous declines. The distribution of murrelets within Zones 1 through 4 has remained relatively stable, with one exception. The gap at the southern end of Zone 2 and the northern end of Zone 3 has become wider and murrelet presence within this gap has become sparser over since the Recovery Plan was published in 1997, whereas the Recovery Plan called for increasing the distribution of murrelets in this area and not furthering the gap (USFWS 1997, p. 126).

Given current population trends and low productivity, the expected degradation of marine and terrestrial habitat conditions due to climate change, and cumulative effects, we expect that murrelet populations, especially those in Zones 1 and 2, will maintain declining population trends that are inconsistent with recovery. Productivity must increase and mortality reduced or kept stable, in order to reverse the population trend and approach recovery. The proposed action will have the opposite effect, causing additional mortality and reducing reproduction. These effects will be greatest in Zone 2, smaller in Zone 1, and minimal in Zones 3 and 4. Particularly due to the effects in Zone 2, the effects of the action will appreciably reduce the likelihood of recovery rangewide by increasing the difficulty of increasing productivity and reversing the downward population trend in Zone 2.Stable or increasing populations and productivity in Zones 1-4 are defined as necessary by the Service's recovery plan for this species.

12.3 Short-tailed Albatross

The changes proposed to the Navy's NWTT program will decrease the expected exposure of short-tailed albatross habitat to stressors associated with the action. The increasing short-tail albatross population trend will result in increasing albatross in that habitat. If the population trend continues, the short-tailed albatross population will have more than tripled by the end of the proposed action. Since the short-tail albatross population is growing and, compared to that growth, a small number of albatross will be affected, we do not anticipate the changed proposed action will appreciably alter the current population size or the increasing population trend and will therefore not appreciably reduce the likelihood of persistence or recovery of short-tailed albatross. See the 2016 Opinion for a more detailed integration and synthesis (USFWS 2016, pp. 265-266).

13 CONCLUSION

13.1 Bull Trout

After reviewing the current status of the bull trout, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is the Service's Opinion that implementation of the Navy's NWTT Activities, as proposed, is not likely to jeopardize the continued existence of the bull trout.

13.2 Marbled Murrelet

After reviewing the current status of the murrelet, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is the Service's Opinion that implementation of the Navy's NWTT Activities, as proposed, is not likely to jeopardize the continued existence of the murrelet. While we conclude that the action is reasonably expected to reduce appreciably the likelihood of recovery of the species when just recovery is evaluated, we do not conclude that the action will reduce appreciably the likelihood of both survival and recovery together. The Service intends the jeopardy standard as a joint survival and recovery concept. Except in exceptional circumstances, injury to recovery alone does not warrant a jeopardy conclusion (51 FR 19934). As an intertwined standard, we consider whether the Navy's action here would injure the murrelet's chances for survival to the point that recovery is not attainable. We expect a small reduction, not appreciable, in the likelihood of survival of the DPS as result of the proposed action. We expect that the effects of the action will make it considerably (in other words, appreciably) more difficult to achieve recovery criteria, particularly because of the impacts in Conservation Zone 2, a recovery unit essential to the recovery of the species. Taken together, however, based on the best available information at the time of this analysis, the effects of the proposed action on survival are not so outsized as to make the recovery of the species unattainable.

13.3 Short-Tailed Albatross

After reviewing the current status of the short-tailed albatross, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is the Service's Opinion that implementation of the proposed action is not likely to jeopardize the continued existence of the short-tailed albatross.

14 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. 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. Harm is defined by the Service as an act that actually kills or injures wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavior patterns, including breeding, feeding, or sheltering (50 CFR 17.3). Harass is defined by the Service in regulation as an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 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 is not considered to be a prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary and must be undertaken by the Navy for the exemption in section 7(o)(2) to apply. The Navy has a continuing duty to regulate the activity covered by this Incidental Take Statement. If the Navy 1) fails to assume and implement the terms and conditions or 2) fails to adhere to the terms and conditions of the incidental take statement, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the Navy must report the progress of the action and its impact on the species to the Service as specified in this Incidental Take Statement pursuant to the requirements of 50 CFR 402.14(i)(3).

14.1 Amount or Extent of Take

The Service anticipates take of the bull trout, the murrelet, and the short-tailed albatross as a result of the proposed action. The amount, extent, and form of take is detailed for each species below. The Service anticipates that incidental take of each species will be difficult to detect for the following reasons: the action will introduce stressors to large areas intermittently; these species are widely distributed throughout the marine environment and their distribution is likely to change frequently; when injury and mortality events occur at-sea (on, under, or over water), individuals that are killed may sink or be transported farther out to sea, where they are extremely unlikely to be recovered. In the event that injuries do not result in mortality, detection of most injuries requires capture and handling. Capture and handling is difficult or impossible to accomplish in this setting and would involve additional take of individuals, and could exacerbate the consequences of the proposed action. A further challenge specific to the take of murrelet eggs and chicks is that locating murrelet nests is difficult even when the general location of the nest is known, and nesting habitat for adult birds exposed to stressors in the marine environment is likely to encompass the entire terrestrial range of the murrelet in Washington.

However, pursuant to 50 CFR 402.14(i)(1)(i), a surrogate can be used to express the anticipated level of take in an Incidental Take Statement, provided three criteria are met: (1) measuring take impacts to a listed species is not practical; (2) a link is established between the effects of the action on the surrogate and take of the listed species; and (3) a clear standard is set for determining when the level of anticipated take based on the surrogate has been exceeded.

The Service's regulations state that significant habitat modification or degradation caused by an action that results in death or injury to a listed species by significantly impairing its essential behavior patterns constitutes take in the form of harm. Those regulations further state that an intentional or negligent act or omission that creates the likelihood of injury to a listed species by annoying it to such an extent as to significantly disrupt its normal behavioral patterns constitutes take in the form of harass. Such annoyance can be caused by actions that modify or degrade habitat conditions (e.g., excessive noise or smoke). In cases where this causal link between effects of a federal action to habitat and take of listed species is established, and the biological opinion or incidental take statement explains why it is not practical to express and monitor the level of take in terms of individuals of the listed species, the Service's regulations authorize the use of habitat as a surrogate for expressing and monitoring the anticipated level of take, provided a clear standard is established for determining when the level of anticipated take has been exceeded.

The following presents the Service's analysis and findings with respect to the three regulatory criteria for use of a surrogate in this Incidental Take Statement to express the anticipated level of take likely to be caused by the proposed action.

Bull Trout

As in the 2016 Opinion (p. 268), a coextensive surrogate based on specific project components is necessary to express the extent of take of the bull trout because it is not practical to monitor take in terms of individual bull trout due to the extremely low likelihood of finding dead or injured individuals in the marine environment, and the large geographic extent of the action area. The coextensive surrogate is the direct source of the stressors causing the taking, and a clear standard for take exceedance can be established under the monitoring requirements (below) using this surrogate. On that basis, the extent of take of the bull trout addressed in this Incidental Take Statement is categorized below by activity and described using a coextensive habitat surrogate.

Marbled Murrelet

A coextensive surrogate, based on an estimate of the number of times individuals are exposed to the stressors causing the taking, is necessary because it is not practical to monitor take in terms of individual murrelets due to the extremely low likelihood of finding dead or injured individuals in the marine environment, or dead or injured chicks and eggs in the forested environment. For the components of the proposed action expected to result in take of murrelets, the number of times the component is used is combined with information about the area affected and the murrelet population density in the affected marine areas at specific times of the year. Thus, the surrogate links the stressor (underwater or in-air pressure waves, projectile strike, or rotor wash) to the exposure of the murrelets in that geographic area of habitat. The area of habitat affected is estimated based on the spatial extent within which exposure to the stressor is expected to lead to take, and we expect that all individuals that are taken will be present within the affected area, or will be the dependent eggs or chicks of adults that are exposed in the affected area. A clear standard for take exceedance can be established under the monitoring requirements (below) using this surrogate. On that basis, the extent of take is categorized below by activity and described using a coextensive habitat surrogate.

Short-tailed Albatross

A coextensive surrogate based on specific project components is necessary to express the extent of take of the short-tailed albatross because it is not practical to monitor take of individuals due to the extremely low likelihood of finding dead or injured individuals in the marine environment. The coextensive surrogate is the direct source of the stressors causing the taking, and a clear standard for take exceedance can be established under the monitoring requirements (below) using this surrogate. On that basis, the extent of take is categorized below by activity and described using a coextensive surrogate.

14.1.1 Bull Trout

Take of sub-adult and adult bull trout in the form of harm, is reasonably certain to occur as a result of exposure to underwater pressure waves from explosions. Harm will occur within an area of habitat relative to explosive size as identified below.

E3 and E0 Detonations (unchanged from 2016 and 2018 Opinions)

- Take in the form of harm, of sub-adult and adult bull trout at the Crescent Harbor EOD Training Range site through July 21, 2036:
    - Three E3 detonations with a radius of 1,674 m over 3 events per year
    - Eighteen E0 detonations with a radius 661 m over 3 events per year

E4 Detonations

- Take in the form of harm, of sub-adult and adult bull trout three nautical miles or greater from shore, in the offshore area through July 21, 2036:
    - 216 E4 detonations with a radius of 1,829 m.

14.1.2 Marbled Murrelet

Take of the murrelet is reasonably certain to occur in the form of harm and harass as the result of exposure to helicopter hovering at-sea, exposure to underwater detonations, exposure to non-explosive projectiles, and indirect exposure of eggs and chicks due to direct take of breeding adults. Over the 16-year term of the proposed action (through July 21, 2036), 96 adult, subadult, and juvenile murrelets will be incidentally taken while at-sea. As a result of the take of adults at-sea during the breeding season, incidental take of 17 eggs or chicks will also occur, totaling 113 instances. The extent of take of the murrelet is categorized below by activity:

Helicopter Use At-Sea

- Take in the form of harass of 8.42 adult, subadult, and fledged juvenile murrelets, with resulting harm to 3.03 eggs or chicks at the nest.

The remaining 101.43 instances of incidental take are expected to be in the form of harm resulting from exposure to underwater detonations associated with Explosives Ordnance Disposal and mine countermeasure and neutralization testing activities, and to non-explosive projectiles used in Gunnery Exercises. We expect these instances of harm to affect 87.53 adult, subadult, and fledged juvenile murrelets at sea, and 13.90 eggs or chicks at the nest. Exposure will vary from year to year. Some years will have higher- or lower-than-average numbers of exposures. The Navy implements the proposed action in 7-year timeframes, and defined the maximum number of mine countermeasure and neutralization testing detonations that will occur in that timeframe. However, for activity types other than mine countermeasure and neutralization testing, only annual averages were defined. So, to calculate the 7-year maximum for activities other than mine countermeasure and neutralization testing, we multiplied the annual average by 7. As such, the take associated with mine countermeasure and neutralization testing, may be concentrated during the first 14 years of the action. For mine countermeasure and neutralization testing, we expect that the exposure over a 7-year period will not exceed the maximum 7-year usage of E4 and E7 explosives.

The following three tables (Tables 30, 31, and 32) enumerate the extent of incidental take of marbled murrelets using a coextensive habitat surrogate, evaluated on a rolling annual average basis, as well as maximum take levels per 7-year period and over the16-year term of the action (through July 21, 2036).

Table 30. Anticipated rolling average annual take of marbled murrelets.

Stressor/ Activity	**Location and Season**	**Average Annual Frequency of Use**	**Total Average Annual Area Affected**	**Average Annual Number of Murrelets Exposed At Sea & Zone**	**Average Annual Number of Eggs/ Chicks Exposed**
Helicopter rotor wash	Inland Waters, breeding season	6 instances of hovering per year	0.12 km^2	0.53 Zone 1 (harass)	0.19 (harm)
E3 underwater explosives	Crescent Harbor and Hood Canal, breeding and non-breeding season	6 per year	0.85 km^2	0.21 Zone 1 (harm)	0.06 (harm)

Stressor/ Activity	**Location and Season**	**Average Annual Frequency of Use**	**Total Average Annual Area Affected**	**Average Annual Number of Murrelets Exposed At Sea & Zone**	**Average Annual Number of Eggs/ Chicks Exposed**
E4 underwater explosives	Quinault Range Site, ≥ 3 nautical miles from shore, breeding season	13.5 per year	5.11 km^2	1.89 Zone 2 (harm)	0.68 (harm)
E7 underwater explosives	Quinault Range Site, ≥ 7.5 nautical miles from shore, breeding season	1.88 per year	10.55 km^2	0.37 Zone 2 (harm)	0.13 (harm)
Small-caliber non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	1,210 projectiles per year, fired in bursts of ≥ 5 projectiles	4.42 km^2	0.11 Zone 2 0.12 Zone 3 0.10 Zone 4 (harm)	0
Medium-caliber surface-to-surface non-explosive projectiles	Offshore Area*, < 50 nm from shore, non-breeding season	168 projectiles per year, fired in bursts of ≥ 5 projectiles	8.52 km^2	0.21 Zone 2 0.23 Zone 3 0.19 Zone 4 (harm)	0
Medium-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	78 projectiles per year, fired in bursts of ≥ 5 projectiles	1.25 km^2	0.03 Zone 2 0.03 Zone 3 0.03 Zone 4 (harm)	0
Large-caliber surface-to-surface non-	Offshore Area, < 50 nm from shore, non-	28 projectiles per year, fired in	26.42 km^2	0.64 Zone 2 0.69 Zone 3	0

Stressor/ Activity	Location and Season	Average Annual Frequency of Use	Total Average Annual Area Affected	Average Annual Number of Murrelets Exposed At Sea & Zone	Average Annual Number of Eggs/ Chicks Exposed
explosive projectiles	breeding season	bursts of ≥ 5 projectiles		0.60 Zone 4 (harm)	
Large-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	1 projectile per year	0.46 km^2	0.01 Zone 2 0.01 Zone 3 0.01 Zone 4 (harm)	0
Totals	**All areas**		**58.66 km^2**	**0.75** **Zone 1** **3.26** **Zone 2** **1.08** **Zone 3** **0.92** **Zone 4**	**0.25** **Zone 1** **0.81** **Zone 2** **0** **Zone 3** **0** **Zone 4**

Table 31. Maximum anticipated 7-year total take of marbled murrelets.

Stressor/ Activity	Location and Season	Maximum Frequency of Use over 7 Years	Maximum Area Affected over 7 Years	Maximum 7-year Number of Murrelets Exposed At Sea & Zone	Maximum 7-year Number of Eggs/ Chicks Exposed
Helicopter rotor wash	Inland Waters, breeding season	42 instances of hovering per 7-year period	6.86 km^2	3.7 Zone 1 (harass)	1.3 (harm)
E3 underwater explosives	Crescent Harbor and Hood Canal, breeding and non-breeding season	42 per 7-year period	5.97 km^2	1.5 Zone 1 (harm)	0.39 (harm)

Stressor/ Activity	Location and Season	Maximum Frequency of Use over 7 Years	Maximum Area Affected over 7 Years	Maximum 7-year Number of Murrelets Exposed At Sea & Zone	Maximum 7-year Number of Eggs/ Chicks Exposed
E4 underwater explosives	Quinault Range Site, ≥ 3 nautical miles from shore, breeding season	108 per 7-year period	40.85 km^2	15.1 Zone 2 (harm)	5.4 (harm)
E7 underwater explosives	Quinault Range Site, ≥ 7.5 nautical miles from shore, breeding season	15 per 7-year period	84.36 km^2	3.0 Zone 2 (harm)	1.1 (harm)
Small-caliber non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	8,470 projectiles per 7-year period, fired in bursts of ≥ 5 projectiles	30.97 km^2	0.75 Zone 2 0.81 Zone 3 0.70 Zone 4 (harm)	0
Medium-caliber surface-to-surface non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	1,176 projectiles per 7-year period, fired in bursts of ≥ 5 projectiles	59.61 km^2	1.45 Zone 2 1.56 Zone 3 1.34 Zone 4 (harm)	0
Medium-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	654 projectiles per 7-year period, fired in bursts of ≥ 5 projectiles	8.75 km^2	0.21 Zone 2 0.23 Zone 3 0.20 Zone 4 (harm)	0
Large-caliber surface-to-surface non-	Offshore Area, < 50 nm from shore, non-	196 projectiles per 7-year period, fired	184.9 km^2	4.50 Zone 2 4.85 Zone 3	0

Stressor/ Activity	**Location and Season**	**Maximum Frequency of Use over 7 Years**	**Maximum Area Affected over 7 Years**	**Maximum 7-year Number of Murrelets Exposed At Sea & Zone**	**Maximum 7-year Number of Eggs/ Chicks Exposed**
explosive projectiles	breeding season	in bursts of ≥ 5 projectiles		4.17 Zone 4 (harm)	
Large-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	7 projectiles per 7-year period	3.19 km^2	0.08 Zone 2 0.09 Zone 3 0.07 Zone 4 (harm)	0
Totals	**All areas**		**425.5 km^2**	**5.17** **Zone 1** **31.56** **Zone 2** **7.54** **Zone 3** **6.47** **Zone 4**	**1.72** **Zone 1** **6.50** **Zone 2** **0** **Zone 3** **0** **Zone 4**

Table 32. Maximum anticipated total take of marbled murrelets for the 16-year term of the action (through July 21, 2036).

Stressor/ Activity	**Location and Season**	**Maximum Frequency of Use over 16 Years**	**Maximum Area Affected over 16 Years**	**Maximum 16-year Number of Murrelets Exposed At Sea & Zone**	**Maximum 16-year Number of Eggs/ Chicks Exposed**
Helicopter rotor wash	Inland Waters, breeding season	96 instances of hovering	2.88 km^2	8.42 Zone 1 (harass)	3.03 (harm)
E3 underwater explosives	Crescent Harbor and Hood Canal, breeding and non-breeding season	96	13.59 km^2	3.40 Zone 1 (harm)	0.90 (harm)

Stressor/ Activity	Location and Season	Maximum Frequency of Use over 16 Years	Maximum Area Affected over 16 Years	Maximum 16-year Number of Murrelets Exposed At Sea & Zone	Maximum 16-year Number of Eggs/ Chicks Exposed
E4 underwater explosives	Quinault Range Site, ≥ 3 nautical miles from shore, breeding season	216	81.71 km^2	30.22 Zone 2 (harm)	10.88 (harm)
E7 underwater explosives	Quinault Range Site, ≥ 7.5 nautical miles from shore, breeding season	30	168.73 km^2	5.91 Zone 2 (harm)	2.13 (harm)
Small-caliber non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	19.360 projectiles, fired in bursts of ≥ 5 projectiles	70.79 km^2	1.72 Zone 2 1.85 Zone 3 1.59 Zone 4 (harm)	0
Medium-caliber surface-to-surface non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	2,688 projectiles, fired in bursts of ≥ 5 projectiles	136.25 km^2	3.31 Zone 2 3.57 Zone 3 3.06 Zone 4 (harm)	0
Medium-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	1,248 projectiles, fired in bursts of ≥ 5 projectiles	20.00 km^2	0.49 Zone 2 0.52 Zone 3 0.45 Zone 4 (harm)	0
Large-caliber surface-to-surface non-	Offshore Area, < 50 nm from shore, non-	448 projectiles, fired in	422.73 km^2	10.28 Zone 2 11.09 Zone 3	0

Stressor/ Activity	**Location and Season**	**Maximum Frequency of Use over 16 Years**	**Maximum Area Affected over 16 Years**	**Maximum 16-year Number of Murrelets Exposed At Sea & Zone**	**Maximum 16-year Number of Eggs/ Chicks Exposed**
explosive projectiles	breeding season	bursts of ≥ 5 projectiles		9.52 Zone 4 (harm)	
Large-caliber surface-to-air non-explosive projectiles	Offshore Area, < 50 nm from shore, non-breeding season	16 projectiles	7.29 km^2	0.18 Zone 2 0.20 Zone 3 0.17 Zone 4 (harm)	0
Totals	**All areas**		**938.5 km^2**	**11.82** **Zone 1** **52.10** **Zone 2** **17.22** **Zone 3** **14.79** **Zone 4**	**3.93** **Zone 1** **13.0** **Zone 2** **0** **Zone 3** **0** **Zone 4**

14.1.3 Short-tailed Albatross

Take of juvenile/immature short-tailed albatross is reasonably certain to occur in the form of harm, as a result of exposure to explosions, projectiles, fragments, and shock waves. Over the 16-year term of the action (through July 21, 2036), take will occur to 12 individuals.

E1/E2 Explosions, Projectile Strikes, Fragment Strikes, and Projectile Shock Waves from Medium-Caliber Projectiles

- Take in the form of harm of sub-adult short-tailed albatross within the offshore area through July 21, 2036:
    - 258 E1/E2 medium-caliber projectiles per year

E5 Explosions, Projectile Strikes, Fragment Strikes, Projectile Shock Waves, and Muzzle Blasts from Large-Caliber Projectiles

- Take in the form of harm of sub-adult short-tailed albatross within the offshore area through July 21, 2036:
    - 192 E5 (E3/E5) larger-caliber projectiles per year

Small-caliber Non-Explosive Projectiles – Physical Strike

- Take in the form of harm of sub-adult short-tailed albatross within the offshore area through July 21, 2036:
    - 24,200 instances (121,000 small-caliber non-explosive projectiles) per year

Medium-caliber Non-Explosive Projectiles – Physical Strike and Projectile Shock Wave

- Take in the form of harm of sub-adult short-tailed albatross within the offshore area through July 21, 2036:
    - 5,282 instances (26,410 medium-caliber non-explosive projectiles) per year

Large-caliber Non-Explosive Projectiles – Physical Strike, Projectile Shock Wave, and Muzzle Blast

- Take in the form of harm of sub-adult short-tailed albatross within the offshore area through July 21, 2036:
    - 592 instances (2,960 large-caliber non-explosive projectiles) per year

14.2 Effect of Take

14.2.1 Bull Trout

In the accompanying Opinion, the Service determined that this level of anticipated take is not likely to result in jeopardy to the bull trout.

14.2.2 Marbled Murrelet

In the accompanying Opinion, the Service determined that this level of anticipated take is not likely to result in jeopardy to the murrelet.

14.2.3 Short-tailed Albatross

In the accompanying Opinion, the Service determined that this level of anticipated take is not likely to result in jeopardy to the short-tailed albatross.

15 REASONABLE AND PRUDENT MEASURES

The Service believes the following reasonable and prudent measures (RPMs) are necessary and appropriate to minimize the impacts (i.e., the amount or extent) of incidental take of the bull trout, marbled murrelet, and the short-tailed albatross.

1. Monitor implementation of the proposed action and report the results of that monitoring program to insure that the level of take exemption provided under this Incidental Take Statement is not exceeded.

2. Minimize the number of murrelets harmed by underwater explosions during mine countermeasure and neutralization testing.

16 TERMS AND CONDITIONS

In order to be exempt from the prohibitions of section 9 of the ESA, the Navy must comply with the following terms and conditions, which implement the RPMs described above and outline required reporting/monitoring requirements. These terms and conditions are non-discretionary.

The Service is to be notified within three working days upon locating a dead, injured or sick endangered or threatened species specimen. Initial notification must be made to the nearest U.S. Fish and Wildlife Service Law Enforcement Office. Notification must include the date, time, precise location of the injured animal or carcass, and any other pertinent information. Care should be taken in handling sick or injured specimens to preserve biological materials in the best possible state for later analysis of cause of death, if that occurs. In conjunction with the care of sick or injured endangered or threatened species or preservation of biological materials from a dead animal, the finder has the responsibility to ensure that evidence associated with the specimen is not unnecessarily disturbed. Contact the U.S. Fish and Wildlife Service Law Enforcement Office at (425) 883-8122.

1. To implement RPM 1, the Navy shall submit a monitoring report, by February 15 of each year, summarizing the training and testing activities implemented in the previous calendar year. The report will be submitted to the U.S. Fish and Wildlife Service's Washington Fish and Wildlife Office. The report shall include, at a minimum, the following information for bull trout, marbled murrelet and short-tailed albatross:
    a. For each training and testing activity identified above under the *Form and Amount or Extent of Take* section:
        i. Stressor/activity name
        ii. Date and location where the stressor/activity occurred
        iii. Number and size of projectiles used, and number and size of detonations.
        iv. To track take of murrelets, dates and locations of the use of each of the stressors listed under the *Form and Amount or Extent of Take* section shall be reported in a way that the date can be identified as occurring during the breeding season (April through September) or non-breeding season (October through March), and so that the location can be identified as follows:
            1. In the inland waters, identify activities as occurring within Northwest Forest Plan Effectiveness Monitoring Zone 1, Stratum 1, 2 or 3 during the breeding season, or Washington Department of Fish and Wildlife Stratum 2, 3, or 4 during the non-breeding season;

        2. Off of Washington's outer coast, identify activities by the following distance bands: less than 4.6 nautical miles from shore, 4.6 to 7.5 nautical miles from shore, 7.5 to 12 nautical miles from shore, or 12 to 50 nautical miles from shore;
        3. Off of Oregon, north of Coos Bay, 12 to 50 nautical miles from shore during the non-breeding season;
        4. Off of Oregon, south of Coos Bay, or off of California, 12 to 50 nautical miles from shore during the non-breeding season.
    - v. To track take of murrelets, provide the breakdown of medium-caliber surface-to-surface and surface-to-air non-explosive projectiles into the specific sizes used.
    - vi. To track take of murrelets, in addition to information about the Navy's activities, the monitoring report shall include an estimate of murrelets exposed to the stressors listed under the *Form and Amount or Extent of Take* section in the previous calendar year; the rolling 7-year total (not applicable for reports due prior to February 15, 2028), and the total for the full term of the action starting on the date this opinion is finalized.
- b. If any of this information cannot be recorded and reported as specified above, the Navy shall provide the best approximation of the listed information and explain the method used in the approximation. Methods used to derive estimated information must be based on the best information that is current and available to the Navy at the time of the monitoring report.
- c. While implementing the action, the Navy shall use a spreadsheet, to be provided by the Service, to track throughout the reporting period the estimated number of murrelet exposures to stressors associated with the action that are expected to result in take. The spreadsheet will contain the exposure model that relates each stressor to the number of murrelets exposed. The use of this spreadsheet will provide the Navy with early warning if it is on track to exceed the exempted levels of take, allowing the Navy to adjust operations to reduce the number of murrelet exposures, or, if that is not possible, re-initiate consultation prior to exceeding the exempted levels of take. The Service may periodically update the spreadsheet with new information regarding the expected number of murrelets in each part of the action area.
- d. The Navy shall contact the Service if the monitoring shows that the seven-year rolling annual total or if the rate of murrelet exposures indicate that incidental take may exceed the levels exempted by this Incidental Take Statement. If updated information is available regarding murrelet population densities at sea during the Navy's training and testing operations, the Service will update the exposure model to reflect the new information and advise the Navy as to whether reinitation is warranted.

2. To implement RPM 2, during use of E4 underwater explosives for mine countermeasure and neutralization testing, the Navy shall conduct pre-test seabird surveys in those portions of the testing area that are within 4.6 nm of shore.
    a. Surveys will be conducted after targets have been placed and their locations verified but prior, and as close as possible, to the enforcement of the human safety zone (which typically corresponds to the deployment of the unmanned surface vessel tasked with locating targets). At a minimum, the Navy will survey for all seabirds in a radius of 347 m (area corresponding to expected injury to murrelets "injury zone") around targets located within 4.6 nm of shore. Surveyors will use one or more small (approximately 25-50 ft.) boats traversing the area within 347 m of each target, observing for seabirds within 100 meters of the boat. As safety allows, the boat(s) will cover the area at speeds between 5 and 10 knots. In each vessel, at least one observer who is not the boat operator shall be dedicated solely to searching for seabirds. If using a small boat is not feasible due to safety or environmental conditions, observations will be conducted using a vessel already participating in the activity. If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will also survey the injury zone for seabirds while performing their regular duties. When a seabird is observed on, or diving below, the surface, in the injury zone associated with a target, the Navy shall identify that target as not clear for detonations until observed seabirds are seen exiting the injury zone or subsequent surveys determine that the area is clear of seabirds. If seabirds are lost from view the observer will monitor the area where they were last seen for five minutes. If the seabirds are not seen after monitoring for five minutes, then the observer will resurvey the injury zone to ensure there are no seabirds in the area before the target is cleared for future detonation.
    b. An annual report, summarizing this pre-detonation seabird monitoring effort and results shall be provided along with the annual reports required in Term and Condition 1. At a minimum, the seabird monitoring report shall contain:
        i. The survey year's data sheets
        ii. A description of how and when surveys were conducted
        iii. A description of the platform(s) used for surveys
        iv. Whether surveys detected seabirds within injury zones
        v. Number and duration of curtailment events due to seabird observations in the injury zone
    c. If adjustments to the protocol are necessary to ensure human safety, the Navy shall notify the Washington Fish and Wildlife Office as soon as practicable of the adjustments to be made and the reason for the adjustments. If these adjustments can be anticipated in advance, the Navy shall discuss them with the Washington Fish and Wildlife Office before enacting a modified protocol. However, human safety is paramount, and, if necessary, adjustments to the protocol for this purpose may be made without prior notice; in that case, the Navy shall notify the Service once it is safe and practicable to do so.

3. To implement RPM 2, the Navy will coordinate with the Washington Fish and Wildlife Office to develop training materials for Navy personnel who will conduct pre-detonation seabird surveys to maximize seabird detection capabilities. The training materials shall focus on the monitoring objective, scanning techniques, and method of communication with the Littoral Combat Ship. The Navy shall submit proposed training materials to the Service's Washington Fish and Wildlife Office for approval, no less than four months before commencing the first mine countermeasure and neutralization testing activity within 4.6 nm of shore. The Service will notify Navy as soon as possible, but no later than 30 days after receipt, of any significant concerns about the materials provided. The Service will have 60 days to review and provide final approval. The Navy shall deliver the training associated with the approved training materials to Navy personnel serving as seabird observers prior to their utilization as such during mine countermeasure and neutralization testing activities.

17 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or designated critical habitat, to help implement recovery plans, or to develop information.

In order for the Service to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, we request notification of the implementation of any conservation recommendations.

The Service provides the following recommendations:

1. To refine our ongoing assessment of the effects of the Navy's activities on listed species, provide the following information along with the above annual monitoring report. The Service is available to coordinate on development of a monitoring approach that could address the following:
    a. For each activity conducted:
        i. Activity name as described in the proposed action
        ii. Number of events conducted throughout the year
        iii. Location of each event—as specific as possible (i.e., distance offshore)
        iv. Date event occurred—including beginning and end dates
        v. Time event occurred, providing as much information as possible on where specific portions of events occurred
        vi. Number of ordnances used per event

        vii. Total hours of sonar used per event

   b. For projectiles and missiles:

        i. Type of number of projectiles and missiles used per event
        ii. Firing rate, for a given event (i.e., 5 bursts per shot, number of shots per minute, etc.)
        iii. How many projectiles or missiles are fired along the same trajectory (i.e., is both the firing location and target stationary so all projectiles and missiles are fired along the same trajectory, or is the firing location moving and the target stationary, or both are moving, etc.)
        iv. Distance projectiles and missiles traveled, distance to target
        v. Accuracy of projectiles or missiles hitting the target

   c. For sonar:

        i. Type of sonar used
        ii. Duration sonar was used
        iii. Average time sonar was used per hour

2. The information used to determine effects of acoustic stressors (explosives, sonar, projectile shock wave, etc.) is based on older studies that do not account for the real-world variables that are encountered in the field. Monitoring of acoustic levels associated with technologies that the Navy utilizes during training activities could provide updated information that would benefit analytical approaches and/or development of minimization measures. Coordinate with the Service to develop an acoustic monitoring plan to provide updated information on the stressors and biological effects from the following:

   a. Underwater explosives
   b. In-air explosives
   c. Sonar SPL outputs
   d. Bow shock or projectile shockwaves

3. Minimize impacts to short-tailed albatross associated with the Navy's activities in the offshore area, by instituting a program of marine debris removal focusing on the edge of the continental shelf or around the Aleutian Islands where short-tailed albatross are most likely to occur.

4. Improve upon existing marine debris retrieval and removal processes. Retrieve debris related to detonations, weapons firing and other training activity whenever possible. Dispose debris at a secure upland location to ensure that it does not re-enter the marine environment.

5. Reduce threats to the marine forage base of the murrelet by avoiding impacts to marine shoreline, eelgrass, and other habitats where marine forage fish spawn or congregate. Offset existing and future impacts to these habitats by completing effective shoreline and marine habitat restoration projects and by conserving marine shoreline habitat areas within the range of the murrelet.

6. Develop and/or fund research programs that improve understanding of the hearing capabilities of seabirds as well as how seabirds are affected by elevated sound levels and shock waves. Involve the Service in the scoping and oversight of those efforts to ensure results are compatible with consultation needs.

7. Develop and/or fund research programs that improve understanding of the abundance, distribution, and status of marine forage fish that comprise the prey base of the murrelet.

8. Provide funding toward resuming annual Northwest Forest Plan Effectiveness Monitoring at-sea surveys of murrelets.

9. Provide funding toward monitoring murrelet productivity (e.g., surveys of juvenile ratios late in the breeding season).

10. To avoid, reduce, or minimize the take of murrelets, conduct training and testing activities that involve the following stressors beyond 50 nm from shore in the Offshore Area: E3 and E4 detonations; and small-caliber, medium-caliber, and large-caliber non-explosive projectiles

11. Conduct mine countermeasure and neutralization testing (E4 and E7 underwater detonations) farther than 12 nm from shore. When these activities must be conducted closer to shore, to the maximum extent possible, conduct E4 detonations along with E7 detonations at distances greater than 7.5 nm from shore.

12. Coordinate with the Service to develop a plan, within one year from the date of this Opinion, which utilizes adaptive management to further refine our understanding of stressors and the impacts on the bull trout, marbled murrelet, and short-tailed albatross caused by the proposed action. Such information may facilitate the identification of additional ways to further minimize the impacts of take on these species caused by the proposed action.

13. To further understanding of bull trout use of the offshore area, continue to fund acoustic telemetry studies. This could include support of the NOAA Northwest Fisheries Science Center's efforts and expanding the receiver array and number of tagged bull trout.

18 REINITIATION NOTICE

This concludes formal consultation on the action(s) outlined in the request for formal consultation. As provided in 50 CFR 402.16, reinitiation of formal consultation is required and shall be requested by the federal agency or by the Service, where discretionary federal involvement or control over the action has been retained or is authorized by law and: (a) if the amount or extent of taking specified in the incidental take statement is exceeded; (b) if new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (c) if the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the biological opinion; or (d) if a new species is listed or critical habitat designated that may be affected by the identified action.

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APPENDIX A
STATUS OF THE SPECIES: BULL TROUT

Appendix A
Status of the Species: Bull Trout

Taxonomy

The bull trout (*Salvelinus confluentus*) is a native char found in the coastal and intermountain west of North America. Dolly Varden (*Salvelinus malma)* and bull trout were previously considered a single species and were thought to have coastal and interior forms. However, Cavender (1978, entire) described morphometric, meristic and osteological characteristics of the two forms, and provided evidence of specific distinctions between the two. Despite an overlap in the geographic range of bull trout and Dolly Varden in the Puget Sound area and along the British Columbia coast, there is little evidence of introgression (Haas and McPhail 1991, p. 2191). The Columbia River Basin is considered the region of origin for the bull trout. From the Columbia, dispersal to other drainage systems was accomplished by marine migration and headwater stream capture. Behnke (2002, p. 297) postulated dispersion to drainages east of the continental divide may have occurred through the North and South Saskatchewan Rivers (Hudson Bay drainage) and the Yukon River system. Marine dispersal may have occurred from Puget Sound north to the Fraser, Skeena and Taku Rivers of British Columbia.

Species Description

Bull trout have unusually large heads and mouths for salmonids. Their body colors can vary tremendously depending on their environment, but are often brownish green with lighter (often ranging from pale yellow to crimson) colored spots running along their dorsa and flanks, with spots being absent on the dorsal fin, and light colored to white under bellies. They have white leading edges on their fins, as do other species of char. Bull trout have been measured as large as 103 centimeters (41 inches) in length, with weights as high as 14.5 kilograms (32 pounds) (Fishbase 2015, p. 1). Bull trout may be migratory, moving throughout large river systems, lakes, and even the ocean in coastal populations, or they may be resident, remaining in the same stream their entire lives (Rieman and McIntyre 1993, p. 2; Brenkman and Corbett 2005, p. 1077). Migratory bull trout are typically larger than resident bull trout (USFWS 1998, p. 31668).

Legal Status

The coterminous United States population of the bull trout was listed as threatened on November 1, 1999 (USFWS 1999, entire). The threatened bull trout generally occurs in the Klamath River Basin of south-central Oregon; the Jarbidge River in Nevada; the Willamette River Basin in Oregon; Pacific Coast drainages of Washington, including Puget Sound; major rivers in Idaho, Oregon, Washington, and Montana, within the Columbia River Basin; and the St. Mary-Belly River, east of the Continental Divide in northwestern Montana (Bond 1992, p. 4; Brewin and Brewin 1997, pp. 209-216; Cavender 1978, pp. 165-166; Leary and Allendorf 1997, pp. 715-720).

Throughout its range, the bull trout are threatened by the combined effects of habitat degradation, fragmentation, and alterations associated with dewatering, road construction and maintenance, mining, grazing, the blockage of migratory corridors by dams or other diversion structures, poor water quality, entrainment (a process by which aquatic organisms are pulled through a diversion or other device) into diversion channels, and introduced non-native species (USFWS 1999, p. 58910). Although all salmonids are likely to be affected by climate change, bull trout are especially vulnerable given that spawning and rearing are constrained by their location in upper watersheds and the requirement for cold water temperatures (Battin et al. 2007, entire; Rieman et al. 2007, entire; Porter and Nelitz. 2009, pages 4-8). Poaching and incidental mortality of bull trout during other targeted fisheries are additional threats.

Life History

The iteroparous reproductive strategy of bull trout has important repercussions for the management of this species. Bull trout require passage both upstream and downstream, not only for repeat spawning but also for foraging. Most fish ladders, however, were designed specifically for anadromous semelparous salmonids (fishes that spawn once and then die, and require only one-way passage upstream). Therefore, even dams or other barriers with fish passage facilities may be a factor in isolating bull trout populations if they do not provide a downstream passage route. Additionally, in some core areas, bull trout that migrate to marine waters must pass both upstream and downstream through areas with net fisheries at river mouths. This can increase the likelihood of mortality to bull trout during these spawning and foraging migrations.

Growth varies depending upon life-history strategy. Resident adults range from 6 to 12 inches total length, and migratory adults commonly reach 24 inches or more (Goetz 1989, p. 30; Pratt 1985, pp. 28-34). The largest verified bull trout is a 32-pound specimen caught in Lake Pend Oreille, Idaho, in 1949 (Simpson and Wallace 1982, p. 95).

Bull trout typically spawn from August through November during periods of increasing flows and decreasing water temperatures. Preferred spawning habitat consists of low-gradient stream reaches with loose, clean gravel (Fraley and Shepard 1989, p. 141). Redds are often constructed in stream reaches fed by springs or near other sources of cold groundwater (Goetz 1989, pp. 15-16; Pratt 1992, pp. 6-7; Rieman and McIntyre 1996, p. 133). Depending on water temperature, incubation is normally 100 to 145 days (Pratt 1992, p. 1). After hatching, fry remain in the substrate, and time from egg deposition to emergence may surpass 200 days. Fry normally emerge from early April through May, depending on water temperatures and increasing stream flows (Pratt 1992, p. 1; Ratliff and Howell 1992, p. 10).

Early life stages of fish, specifically the developing embryo, require the highest inter-gravel dissolved oxygen (IGDO) levels, and are the most sensitive life stage to reduced oxygen levels. The oxygen demand of embryos depends on temperature and on stage of development, with the greatest IGDO required just prior to hatching.

A literature review conducted by the Washington Department of Ecology (WDOE 2002, p. 9) indicates that adverse effects of lower oxygen concentrations on embryo survival are magnified as temperatures increase above optimal (for incubation). Normal oxygen levels seen in rivers used by bull trout during spawning ranged from 8 to 12 mg/L (in the gravel), with corresponding instream levels of 10 to 11.5 mg/L (Stewart et al. 2007, p. 10). In addition, IGDO concentrations, water velocities in the water column, and especially the intergravel flow rate, are interrelated variables that affect the survival of incubating embryos (ODEQ 1995, Ch 2 pp.

23-24). Due to a long incubation period of 220+ days, bull trout are particularly sensitive to adequate IGDO levels. An IGDO level below 8 mg/L is likely to result in mortality of eggs, embryos, and fry.

Population Dynamics

Population Structure

Bull trout exhibit both resident and migratory life history strategies. Both resident and migratory forms may be found together, and either form may produce offspring exhibiting either resident or migratory behavior (Rieman and McIntyre 1993, p. 2). Resident bull trout complete their entire life cycle in the tributary (or nearby) streams in which they spawn and rear. The resident form tends to be smaller than the migratory form at maturity and also produces fewer eggs (Goetz 1989, p. 15). Migratory bull trout spawn in tributary streams where juvenile fish rear 1 to 4 years before migrating to either a lake (adfluvial form), river (fluvial form) (Fraley and Shepard 1989, p. 138; Goetz 1989, p. 24), or saltwater (anadromous form) to rear as subadults and to live as adults (Brenkman and Corbett 2005, entire; McPhail and Baxter 1996, p. i; WDFW et al. 1997, p. 16). Bull trout normally reach sexual maturity in 4 to 7 years and may live longer than 12 years. They are iteroparous (they spawn more than once in a lifetime). Repeat- and alternate-year spawning has been reported, although repeat-spawning frequency and post-spawning mortality are not well documented (Fraley and Shepard 1989, p. 135; Leathe and Graham 1982, p. 95; Pratt 1992, p. 8; Rieman and McIntyre 1996, p. 133).

Bull trout are naturally migratory, which allows them to capitalize on temporally abundant food resources and larger downstream habitats. Resident forms may develop where barriers (either natural or manmade) occur or where foraging, migrating, or overwintering habitats for migratory fish are minimized (Brenkman and Corbett 2005, pp. 1075-1076; Goetz et al. 2004, p. 105). For example, multiple life history forms (e.g., resident and fluvial) and multiple migration patterns have been noted in the Grande Ronde River (Baxter 2002, pp. 96, 98-106). Parts of this river system have retained habitat conditions that allow free movement between spawning and rearing areas and the mainstem Snake River. Such multiple life history strategies help to maintain the stability and persistence of bull trout populations to environmental changes. Benefits to migratory bull trout include greater growth in the more productive waters of larger streams, lakes, and marine waters; greater fecundity resulting in increased reproductive potential; and dispersing the population across space and time so that spawning streams may be recolonized should local populations suffer a catastrophic loss (Frissell 1999, pp. 861-863; MBTSG 1998, p. 13; Rieman and McIntyre 1993, pp. 2-3). In the absence of the migratory bull trout life form, isolated populations cannot be replenished when disturbances make local habitats temporarily unsuitable. Therefore, the range of the species is diminished, and the potential for a greater reproductive contribution from larger size fish with higher fecundity is lost (Rieman and McIntyre 1993, p. 2).

Whitesel et al. (2004, p. 2) noted that although there are multiple resources that contribute to the subject, Spruell et al. (2003, entire) best summarized genetic information on bull trout population structure. Spruell et al. (2003, entire) analyzed 1,847 bull trout from 65 sampling locations, four located in three coastal drainages (Klamath, Queets, and Skagit Rivers), one in the Saskatchewan River drainage (Belly River), and 60 scattered throughout the Columbia River Basin. They concluded that there is a consistent pattern among genetic studies of bull trout, regardless of whether examining allozymes, mitochondrial DNA, or most recently microsatellite loci. Typically, the genetic pattern shows relatively little genetic variation within populations, but substantial divergence among populations. Microsatellite loci analysis supports the existence of at least three major genetically differentiated groups (or evolutionary lineages) of bull trout (Spruell et al. 2003, p. 17). They were characterized as:

i. "Coastal", including the Deschutes River and all of the Columbia River drainage downstream, as well as most coastal streams in Washington, Oregon, and British Columbia. A compelling case also exists that the Klamath Basin represents a unique evolutionary lineage within the coastal group.

ii. "Snake River", which also included the John Day, Umatilla, and Walla Walla rivers. Despite close proximity of the John Day and Deschutes Rivers, a striking level of divergence between bull trout in these two systems was observed.

iii. "Upper Columbia River" which includes the entire basin in Montana and northern Idaho. A tentative assignment was made by Spruell et al. (2003, p. 25) of the Saskatchewan River drainage populations (east of the continental divide), grouping them with the upper Columbia River group.

Spruell et al. (2003, p. 17) noted that within the major assemblages, populations were further subdivided, primarily at the level of major river basins. Taylor et al. (1999, entire) surveyed bull trout populations, primarily from Canada, and found a major divergence between inland and coastal populations. Costello et al. (2003, p. 328) suggested the patterns reflected the existence of two glacial refugia, consistent with the conclusions of Spruell et al. (2003, p. 26) and the biogeographic analysis of Haas and McPhail (2001, entire). Both Taylor et al. (1999, p. 1166) and Spruell et al. (2003, p. 21) concluded that the Deschutes River represented the most upstream limit of the coastal lineage in the Columbia River Basin.

More recently, the U.S. Fish and Wildlife Service (Service) identified additional genetic units within the coastal and interior lineages (Ardren et al. 2011, p. 18). Based on a recommendation in the Service's 5-year review of the species' status (USFWS 2008a, p. 45), the Service reanalyzed the 27 recovery units identified in the draft bull trout recovery plan (USFWS 2002a, p. 48) by utilizing, in part, information from previous genetic studies and new information from additional analysis (Ardren et al. 2011, entire). In this examination, the Service applied relevant factors from the joint Service and National Marine Fisheries Service Distinct Population Segment (DPS) policy (USFWS 1996, entire) and subsequently identified six draft recovery units that contain assemblages of core areas that retain genetic and ecological integrity across the range of bull trout in the coterminous United States. These six draft recovery units were used to inform designation of critical habitat for bull trout by providing a context for deciding what habitats are essential for recovery (USFWS 2010, p. 63898). The six draft recovery units identified for bull trout in the coterminous United States include: Coastal, Klamath, Mid-Columbia, Columbia Headwaters, Saint Mary, and Upper Snake. These six draft recovery units were also identified in the Service's revised recovery plan (USFWS 2015, p. vii) and designated as final recovery units.

Population Dynamics

Although bull trout are widely distributed over a large geographic area, they exhibit a patchy distribution, even in pristine habitats (Rieman and McIntyre 1993, p. 4). Increased habitat fragmentation reduces the amount of available habitat and increases isolation from other populations of the same species (Saunders et al. 1991, entire). Burkey (1989, entire) concluded that when species are isolated by fragmented habitats, low rates of population growth are typical in local populations and their probability of extinction is directly related to the degree of isolation and fragmentation. Without sufficient immigration, growth for local populations may be low and probability of extinction high (Burkey 1989, entire; Burkey 1995, entire).

Metapopulation concepts of conservation biology theory have been suggested relative to the distribution and characteristics of bull trout, although empirical evidence is relatively scant (Rieman and McIntyre 1993, p. 15; Dunham and Rieman 1999, entire; Rieman and Dunham 2000, entire). A metapopulation is an interacting network of local populations with varying frequencies of migration and gene flow among them (Meffe and Carroll 1994, pp. 189-190). For inland bull trout, metapopulation theory is likely most applicable at the watershed scale where habitat consists of discrete patches or collections of habitat capable of supporting local populations; local populations are for the most part independent and represent discrete reproductive units; and long-term, low-rate dispersal patterns among component populations influences the persistence of at least some of the local populations (Rieman and Dunham 2000, entire). Ideally, multiple local populations distributed throughout a watershed provide a mechanism for spreading risk because the simultaneous loss of all local populations is unlikely. However, habitat alteration, primarily through the construction of impoundments, dams, and water diversions has fragmented habitats, eliminated migratory corridors, and in many cases isolated bull trout in the headwaters of tributaries (Rieman and Clayton 1997, pp. 10-12; Dunham and Rieman 1999, p. 645; Spruell et al. 1999, pp. 118-120; Rieman and Dunham 2000, p. 55).

Human-induced factors as well as natural factors affecting bull trout distribution have likely limited the expression of the metapopulation concept for bull trout to patches of habitat within the overall distribution of the species (Dunham and Rieman 1999, entire). However, despite the theoretical fit, the relatively recent and brief time period during which bull trout investigations have taken place does not provide certainty as to whether a metapopulation dynamic is occurring (e.g., a balance between local extirpations and recolonizations) across the range of the bull trout or whether the persistence of bull trout in large or closely interconnected habitat patches (Dunham and Rieman 1999, entire) is simply reflective of a general deterministic trend towards extinction of the species where the larger or interconnected patches are relics of historically wider distribution (Rieman and Dunham 2000, pp. 56-57). Recent research (Whiteley et al. 2003, entire) does, however, provide genetic evidence for the presence of a metapopulation process for bull trout, at least in the Boise River Basin of Idaho.

Habitat Characteristics

Bull trout have more specific habitat requirements than most other salmonids (Rieman and McIntyre 1993, p. 4). Habitat components that influence bull trout distribution and abundance include water temperature, cover, channel form and stability, valley form, spawning and rearing substrate, and migratory corridors (Fraley and Shepard 1989, entire; Goetz 1989, pp. 23, 25; Hoelscher and Bjornn 1989, pp. 19, 25; Howell and Buchanan 1992, pp. 30, 32; Pratt 1992, entire; Rich 1996, p. 17; Rieman and McIntyre 1993, pp. 4-6; Rieman and McIntyre 1995, entire; Sedell and Everest 1991, entire; Watson and Hillman 1997, entire). Watson and Hillman (1997, pp. 247-250) concluded that watersheds must have specific physical characteristics to provide the habitat requirements necessary for bull trout to successfully spawn and rear and that these specific characteristics are not necessarily present throughout these watersheds. Because bull trout exhibit a patchy distribution, even in pristine habitats (Rieman and McIntyre 1993, pp. 4-6), bull trout should not be expected to simultaneously occupy all available habitats.

Migratory corridors link seasonal habitats for all bull trout life histories. The ability to migrate is important to the persistence of bull trout (Rieman and McIntyre 1993, p. 2). Migrations facilitate gene flow among local populations when individuals from different local populations interbreed or stray to nonnatal streams. Local populations that are extirpated by catastrophic events may also become reestablished by bull trout migrants. However, it is important to note that the genetic structuring of bull trout indicates there is limited gene flow among bull trout populations, which may encourage local adaptation within individual populations, and that reestablishment of extirpated populations may take a long time (Rieman and McIntyre 1993, p. 2; Spruell et al. 1999, entire). Migration also allows bull trout to access more abundant or larger prey, which facilitates growth and reproduction. Additional benefits of migration and its relationship to foraging are discussed below under "Diet."

Cold water temperatures play an important role in determining bull trout habitat quality, as these fish are primarily found in colder streams, and spawning habitats are generally characterized by temperatures that drop below 9 °C in the fall (Fraley and Shepard 1989, p. 137; Pratt 1992, p. 5; Rieman and McIntyre 1993, p. 2).

Thermal requirements for bull trout appear to differ at different life stages. Spawning areas are often associated with cold-water springs, groundwater infiltration, and the coldest streams in a given watershed (Pratt 1992, pp 7-8; Rieman and McIntyre 1993, p. 7). Optimum incubation temperatures for bull trout eggs range from 2 °C to 6 °C whereas optimum water temperatures for rearing range from about 6 °C to 10 °C (Buchanan and Gregory 1997, p. 4; Goetz 1989, p. 22). In Granite Creek, Idaho, Bonneau and Scarnecchia (1996, entire) observed that juvenile bull trout selected the coldest water available in a plunge pool, 8 °C to 9 °C, within a temperature gradient of 8 °C to 15 °C. In a landscape study relating bull trout distribution to maximum water temperatures, Dunham et al. (2003, p. 900) found that the probability of juvenile bull trout occurrence does not become high (i.e., greater than 0.75) until maximum temperatures decline to 11 °C to 12 °C.

Although bull trout are found primarily in cold streams, occasionally these fish are found in larger, warmer river systems throughout the Columbia River basin (Buchanan and Gregory 1997, p. 2; Fraley and Shepard 1989, pp. 133, 135; Rieman and McIntyre 1993, pp. 3-4; Rieman and McIntyre 1995, p. 287). Availability and proximity of cold water patches and food productivity can influence bull trout ability to survive in warmer rivers (Myrick 2002, pp. 6 and 13).

All life history stages of bull trout are associated with complex forms of cover, including large woody debris, undercut banks, boulders, and pools (Fraley and Shepard 1989, p. 137; Goetz 1989, p. 19; Hoelscher and Bjornn 1989, p. 38; Pratt 1992, entire; Rich 1996, pp. 4-5; Sedell and Everest 1991, entire; Sexauer and James 1997, entire; Thomas 1992, pp. 4-6; Watson and Hillman 1997, p. 238). Maintaining bull trout habitat requires natural stability of stream channels and maintenance of natural flow patterns (Rieman and McIntyre 1993, pp. 5-6). Juvenile and adult bull trout frequently inhabit side channels, stream margins, and pools with suitable cover (Sexauer and James 1997, p. 364). These areas are sensitive to activities that directly or indirectly affect stream channel stability and alter natural flow patterns. For example, altered stream flow in the fall may disrupt bull trout during the spawning period, and channel instability may decrease survival of eggs and young juveniles in the gravel from winter through spring (Fraley and Shepard 1989, p. 141; Pratt 1992, p. 6; Pratt and Huston 1993, p. 70). Pratt (1992, p. 6) indicated that increases in fine sediment reduce egg survival and emergence.

Diet

Bull trout are opportunistic feeders, with food habits primarily a function of size and life-history strategy. Fish growth depends on the quantity and quality of food that is eaten, and as fish grow their foraging strategy changes as their food changes, in quantity, size, or other characteristics (Quinn 2005, pp. 195-200). Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macrozooplankton, and small fish (Boag 1987, p. 58; Donald and Alger 1993, pp. 242-243; Goetz 1989, pp. 33-34). Subadult and adult migratory bull trout feed on various fish species (Donald and Alger 1993, pp. 241-243; Fraley and Shepard 1989, pp. 135, 138; Leathe and Graham 1982, pp. 13, 50-56). Bull trout of all sizes other than fry have been found to eat fish half their length (Beauchamp and VanTassell 2001, p. 204). In nearshore marine areas of western Washington, bull trout feed on Pacific herring (*Clupea pallasi*), Pacific sand lance (*Ammodytes hexapterus*), and surf smelt (*Hypomesus pretiosus*) (Goetz et al. 2004, p. 105; WDFW et al. 1997, p. 23).

Bull trout migration and life history strategies are closely related to their feeding and foraging strategies. Migration allows bull trout to access optimal foraging areas and exploit a wider variety of prey resources. For example, in the Skagit River system, anadromous bull trout make migrations as long as 121 miles between marine foraging areas in Puget Sound and headwater spawning grounds, foraging on salmon eggs and juvenile salmon along their migration route (WDFW et al. 1997, p. 25). Anadromous bull trout also use marine waters as migration corridors to reach seasonal habitats in non-natal watersheds to forage and possibly overwinter (Brenkman and Corbett 2005, pp. 1078-1079; Goetz et al. 2004, entire).

Status and Distribution

Distribution and Demography

The historical range of bull trout includes major river basins in the Pacific Northwest at about 41 to 60 degrees North latitude, from the southern limits in the McCloud River in northern California and the Jarbidge River in Nevada to the headwaters of the Yukon River in the Northwest Territories, Canada (Cavender 1978, pp. 165-166; Bond 1992, p. 2). To the west, the bull trout's range includes Puget Sound, various coastal rivers of British Columbia, Canada, and southeast Alaska (Bond 1992, p. 2). Bull trout occur in portions of the Columbia River and tributaries within the basin, including its headwaters in Montana and Canada. Bull trout also occur in the Klamath River basin of south-central Oregon. East of the Continental Divide, bull trout are found in the headwaters of the Saskatchewan River in Alberta and Montana and in the MacKenzie River system in Alberta and British Columbia, Canada (Cavender 1978, pp. 165-166; Brewin et al. 1997, entire).

Each of the following recovery units (below) is necessary to maintain the bull trout's distribution, as well as its genetic and phenotypic diversity, all of which are important to ensure the species' resilience to changing environmental conditions. No new local populations have been identified and no local populations have been lost since listing.

Coastal Recovery Unit

The Coastal Recovery Unit is located within western Oregon and Washington. Major geographic regions include the Olympic Peninsula, Puget Sound, and Lower Columbia River basins. The Olympic Peninsula and Puget Sound geographic regions also include their associated marine waters (Puget Sound, Hood Canal, Strait of Juan de Fuca, and Pacific Coast), which are critical in supporting the anadromous[1] life history form, unique to the Coastal Recovery Unit. The Coastal Recovery Unit is also the only unit that overlaps with the distribution of Dolly Varden (*Salvelinus malma*) (Ardren *et al*. 2011), another native char species that looks very similar to the bull trout (Haas and McPhail 1991). The two species have likely had some level of historic introgression in this part of their range (Redenbach and Taylor 2002). The Lower Columbia River major geographic region includes the lower mainstem Columbia River, an important migratory waterway essential for providing habitat and population connectivity within this region. In the Coastal Recovery Unit, there are 21 existing bull trout core areas which have been designated, including the recently reintroduced Clackamas River population, and 4 core areas have been identified that could be re-established. Core areas within the recovery unit are distributed among these three major geographic regions (Puget Sound also includes one core area that is actually part of the lower Fraser River system in British Columbia, Canada) (USFWS 2015a, p. A-1).

The current demographic status of bull trout in the Coastal Recovery Unit is variable across the unit. Populations in the Puget Sound region generally tend to have better demographic status, followed by the Olympic Peninsula, and finally the Lower Columbia River region. However, population strongholds do exist across the three regions. The Lower Skagit River and Upper Skagit River core areas in the Puget Sound region likely contain two of the most abundant bull trout populations with some of the most intact habitat within this recovery unit. The Lower Deschutes River core area in the Lower Columbia River region also contains a very abundant bull trout population and has been used as a donor stock for re-establishing the Clackamas River population (USFWS 2015a, p. A-6).

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[1] Anadromous: Life history pattern of spawning and rearing in fresh water and migrating to salt water areas to mature.

Puget Sound Region

In the Puget Sound region, bull trout populations are concentrated along the eastern side of Puget Sound with most core areas concentrated in central and northern Puget Sound.

Although the Chilliwack River core area is considered part of this region, it is technically connected to the Fraser River system and is transboundary with British Columbia making its distribution unique within the region. Most core areas support a mix of anadromous and fluvial life history forms, with at least two core areas containing a natural adfluvial life history (Chilliwack River core area [Chilliwack Lake] and Chester Morse Lake core area). Overall demographic status of core areas generally improves as you move from south Puget Sound to north Puget Sound. Although comprehensive trend data are lacking, the current condition of core areas within this region are likely stable overall, although some at depressed abundances. Two core areas (Puyallup River and Stillaguamish River) contain local populations at either very low abundances (Upper Puyallup and Mowich Rivers) or that have likely become locally extirpated (Upper Deer Creek, South Fork Canyon Creek, and Greenwater River). Connectivity among and within core areas of this region is generally intact. Most core areas in this region still have significant amounts of headwater habitat within protected and relatively pristine areas (e.g., North Cascades National Park, Mount Rainier National Park, Skagit Valley Provincial Park, Manning Provincial Park, and various wilderness or recreation areas) (USFWS 2015a, p. A-7).

Olympic Peninsula Region

In the Olympic Peninsula region, distribution of core areas is somewhat disjunct, with only one located on the west side of Hood Canal on the eastern side of the peninsula, two along the Strait of Juan de Fuca on the northern side of the peninsula, and three along the Pacific Coast on the western side of the peninsula. Most core areas support a mix of anadromous and fluvial life history forms, with at least one core area also supporting a natural adfluvial life history (Quinault River core area [Quinault Lake]). Demographic status of core areas is poorest in Hood Canal and Strait of Juan de Fuca, while core areas along the Pacific Coast of Washington likely have the best demographic status in this region. The connectivity between core areas in these disjunct regions is believed to be naturally low due to the geographic distance between them.

Internal connectivity is currently poor within the Skokomish River core area (Hood Canal) and is being restored in the Elwha River core area (Strait of Juan de Fuca). Most core areas in this region still have their headwater habitats within relatively protected areas (Olympic National Park and wilderness areas) (USFWS 2015a, p. A-7).

Lower Columbia River Region

In the Lower Columbia River region, the majority of core areas are distributed along the Cascade Crest on the Oregon side of the Columbia River. Only two of the seven core areas in this region are in Washington. Most core areas in the region historically supported a fluvial life history form, but many are now adfluvial due to reservoir construction. However, there is at least one core area supporting a natural adfluvial life history (Odell Lake) and one supporting a natural, isolated, resident life history (Klickitat River [West Fork Klickitat]). Status is highly variable across this region, with one relative stronghold (Lower Deschutes core area) existing on the Oregon side of the Columbia River. The Lower Columbia River region also contains three watersheds (North Santiam River, Upper Deschutes River, and White Salmon River) that could potentially become re-established core areas within the Coastal Recovery Unit. Although the South Santiam River has been identified as a historic core area, there remains uncertainty as to whether or not historical observations of bull trout represented a self-sustaining population. Current habitat conditions in the South Santiam River are thought to be unable to support bull trout spawning and rearing. Adult abundances within the majority of core areas in this region are relatively low, generally 300 or fewer individuals.

Most core populations in this region are not only isolated from one another due to dams or natural barriers, but they are internally fragmented as a result of manmade barriers. Local populations are often disconnected from one another or from potential foraging habitat. In the Coastal Recovery Unit, adult abundance may be lowest in the Hood River and Odell Lake core areas, which each contain fewer than 100 adults. Bull trout were reintroduced in the Middle Fork Willamette River in 1990 above Hills Creek Reservoir. Successful reproduction was first documented in 2006, and has occurred each year since (USFWS 2015a, p. A-8). Natural reproducing populations of bull trout are present in the McKenzie River basin (USFWS 2008d, pp. 65-67). Bull trout were more recently reintroduced into the Clackamas River basin in the summer of 2011 after an extensive feasibility analysis (Shively et al. 2007, Hudson et al. 2015). Bull trout from the Lower Deschutes core area are being utilized for this reintroduction effort (USFWS 2015a, p. A-8).

Klamath Recovery Unit

Bull trout in the Klamath Recovery Unit have been isolated from other bull trout populations for the past 10,000 years and are recognized as evolutionarily and genetically distinct (Minckley et al. 1986; Leary et al. 1993; Whitesel et al. 2004; USFWS 2008a; Ardren et al. 2011). As such, there is no opportunity for bull trout in another recovery unit to naturally re- colonize the Klamath Recovery Unit if it were to become extirpated. The Klamath Recovery Unit lies at the southern edge of the species range and occurs in an arid portion of the range of bull trout.

Bull trout were once widespread within the Klamath River basin (Gilbert 1897; Dambacher et al. 1992; Ziller 1992; USFWS 2002b), but habitat degradation and fragmentation, past and present land use practices, agricultural water diversions, and past fisheries management practices have greatly reduced their distribution. Bull trout abundance also has been severely reduced, and the remaining populations are highly fragmented and vulnerable to natural or manmade factors that place them at a high risk of extirpation (USFWS 2002b). The presence of nonnative brook trout (*Salvelinus fontinalis*), which compete and hybridize with bull trout, is a particular threat to bull trout persistence throughout the Klamath Recovery Unit (USFWS 2015b, pp. B-3-4).

Upper Klamath Lake Core Area

The Upper Klamath Lake core area comprises two bull trout local populations (Sun Creek and Threemile Creek). These local populations likely face an increased risk of extirpation because they are isolated and not interconnected with each other. Extirpation of other local populations in the Upper Klamath Lake core area has occurred in recent times (1970s). Populations in this core area are genetically distinct from those in the other two core areas in the Klamath Recovery Unit (USFWS 2008b), and in comparison, genetic variation within this core area is lowest. The two local populations have been isolated by habitat fragmentation and have experienced population bottlenecks. As such, currently unoccupied habitat is needed to restore connectivity between the two local populations and to establish additional populations. This unoccupied habitat includes canals, which now provide the only means of connectivity as migratory corridors. Providing full volitional connectivity for bull trout, however, also introduces the risk of invasion by brook trout, which are abundant in this core area.

Bull trout in the Upper Klamath Lake core area formerly occupied Annie Creek, Sevenmile Creek, Cherry Creek, and Fort Creek, but are now extirpated from these locations. The last remaining local populations, Sun Creek and Threemile Creek, have received focused attention. Brook trout have been removed from bull trout occupied reaches, and these reaches have been intentionally isolated to prevent brook trout reinvasion. As such, over the past few generations these populations have become stable and have increased in distribution and abundance. In 1996, the Threemile Creek population had approximately 50 fish that occupied a 1.4-km (0.9-mile) reach (USFWS 2002b). In 2012, a mark-resight population estimate was completed in Threemile Creek, which indicated an abundance of 577 (95 percent confidence interval = 475 to 679) age-1+ fish (ODFW 2012). In addition, the length of the distribution of bull trout in Threemile Creek had increased to 2.7 km (1.7 miles) by 2012 (USFWS unpublished data). Between 1989 and 2010, bull trout abundance in Sun Creek increased approximately tenfold (from approximately 133 to 1,606 age-1+ fish) and distribution increased from approximately 1.9 km (1.2 miles) to 11.2 km (7.0 miles) (Buktenica et al. 2013) (USFWS 2015b, p. B-5).

Sycan River Core Area

The Sycan River core area is comprised of one local population, Long Creek. Long Creek likely faces greater risk of extirpation because it is the only remaining local population due to extirpation of all other historic local populations. Bull trout previously occupied Calahan Creek, Coyote Creek, and the Sycan River, but are now extirpated from these locations (Light et al. 1996). This core area's local population is genetically distinct from those in the other two core areas (USFWS 2008b). This core area also is essential for recovery because bull trout in this core area exhibit both resident[2] and fluvial life histories, which are important for representing diverse life history expression in the Klamath Recovery Unit. Migratory bull trout are able to grow larger than their resident

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[2] Resident: Life history pattern of residing in tributary streams for the fish's entire life without migrating.

counterparts, resulting in greater fecundity and higher reproductive potential (Rieman and McIntyre 1993). Migratory life history forms also have been shown to be important for population persistence and resilience (Dunham et al. 2008).

The last remaining population (Long Creek) has received focused attention in an effort to ensure it is not also extirpated. In 2006, two weirs were removed from Long Creek, which increased the amount of occupied foraging, migratory, and overwintering (FMO) habitat by 3.2 km (2.0 miles). Bull trout currently occupy approximately 3.5 km (2.2 miles) of spawning/rearing habitat, including a portion of an unnamed tributary to upper Long Creek, and seasonally use 25.9 km (16.1 miles) of FMO habitat. Brook trout also inhabit Long Creek and have been the focus of periodic removal efforts. No recent statistically rigorous population estimate has been completed for Long Creek; however, the 2002 Draft Bull Trout Recovery Plan reported a population estimate of 842 individuals (USFWS 2002b). Currently unoccupied habitat is needed to establish additional local populations, although brook trout are widespread in this core area and their management will need to be considered in future recovery efforts. In 2014, the Klamath Falls Fish and Wildlife Office of the Service established an agreement with the U.S. Geological Survey to undertake a structured decision making process to assist with recovery planning of bull trout populations in the Sycan River core area (USFWS 2015b, p. B-6).

Upper Sprague River Core Area

The Upper Sprague River core area comprises five bull trout local populations, placing the core area at an intermediate risk of extinction. The five local populations include Boulder Creek, Dixon Creek, Deming Creek, Leonard Creek, and Brownsworth Creek. These local populations may face a higher risk of extirpation because not all are interconnected. Bull trout local populations in this core area are genetically distinct from those in the other two Klamath Recovery Unit core areas (USFWS 2008b). Migratory bull trout have occasionally been observed in the North Fork Sprague River (USFWS 2002b). Therefore, this core area also is essential for recovery in that bull trout here exhibit a resident life history and likely a fluvial life history, which are important for conserving diverse life history expression in the Klamath Recovery Unit as discussed above for the Sycan River core area.

The Upper Sprague River core area population of bull trout has experienced a decline from historic levels, although less is known about historic occupancy in this core area. Bull trout are reported to have historically occupied the South Fork Sprague River, but are now extirpated from this location (Buchanan et al. 1997). The remaining five populations have received focused attention. Although brown trout (*Salmo trutta*) co-occur with bull trout and exist in adjacent habitats, brook trout do not overlap with existing bull trout populations. Efforts have been made to increase connectivity of existing bull trout populations by replacing culverts that create barriers. Thus, over the past few generations, these populations have likely been stable and increased in distribution. Population abundance has been estimated recently for Boulder Creek (372 + 62 percent; Hartill and Jacobs 2007), Dixon Creek (20 + 60 percent; Hartill and Jacobs 2007), Deming Creek (1,316 + 342; Moore 2006), and Leonard Creek (363 + 37 percent;

Hartill and Jacobs 2007). No statistically rigorous population estimate has been completed for the Brownsworth Creek local population; however, the 2002 Draft Bull Trout Recovery Plan reported a population estimate of 964 individuals (USFWS 2002b). Additional local populations need to be established in currently unoccupied habitat within the Upper Sprague River core area, although brook trout are widespread in this core area and will need to be considered in future recovery efforts (USFWS 2015b, p. B-7).

Mid-Columbia Recovery Unit

The Mid-Columbia Recovery Unit (RU) comprises 24 bull trout core areas, as well as 2 historically occupied core areas and 1 research needs area. The Mid-Columbia RU is recognized as an area where bull trout have co-evolved with salmon, steelhead, lamprey, and other fish populations. Reduced fish numbers due to historic overfishing and land management changes have caused changes in nutrient abundance for resident migratory fish like the bull trout. The recovery unit is located within eastern Washington, eastern Oregon, and portions of central Idaho. Major drainages include the Methow River, Wenatchee River, Yakima River, John Day River, Umatilla River, Walla Walla River, Grande Ronde River, Imnaha River, Clearwater River, and smaller drainages along the Snake River and Columbia River (USFWS 2015c, p. C-1).

The Mid-Columbia RU can be divided into four geographic regions the Lower Mid-Columbia, which includes all core areas that flow into the Columbia River below its confluence with the 1) Snake River; 2) the Upper Mid-Columbia, which includes all core areas that flow into the Columbia River above its confluence with the Snake River; 3) the Lower Snake, which includes all core areas that flow into the Snake River between its confluence with the Columbia River and Hells Canyon Dam; and 4) the Mid-Snake, which includes all core areas in the Mid-Columbia RU that flow into the Snake River above Hells Canyon Dam. These geographic regions are composed of neighboring core areas that share similar bull trout genetic, geographic (hydrographic), and/or habitat characteristics. Conserving bull trout in geographic regions allows for the maintenance of broad representation of genetic diversity, provides neighboring core areas with potential source populations in the event of local extirpations, and provides a broad array of options among neighboring core areas to contribute recovery under uncertain environmental change USFWS 2015c, pp. C-1-2).

The current demographic status of bull trout in the Mid-Columbia Recovery Unit is highly variable at both the RU and geographic region scale. Some core areas, such as the Umatilla, Asotin, and Powder Rivers, contain populations so depressed they are likely suffering from the deleterious effects of small population size. Conversely, strongholds do exist within the recovery unit, predominantly in the Lower Snake geographic area. Populations in the Imnaha, Little Minam, Clearwater, and Wenaha Rivers are likely some of the most abundant. These populations are all completely or partially within the bounds of protected wilderness areas and have some of the most intact habitat in the recovery unit. Status in some core areas is relatively unknown, but all indications in these core areas suggest population trends are declining, particularly in the core areas of the John Day Basin (USFWS 2015c, p. C-5).

Lower Mid-Columbia Region

In the Lower Mid-Columbia Region, core areas are distributed along the western portion of the Blue Mountains in Oregon and Washington. Only one of the six core areas is located completely in Washington. Demographic status is highly variable throughout the region. Status is the poorest in the Umatilla and Middle Fork John Day Core Areas. However, the Walla Walla River core area contains nearly pristine habitats in the headwater spawning areas and supports the most abundant populations in the region. Most core areas support both a resident and fluvial life history; however, recent evidence suggests a significant decline in the resident and fluvial life history in the Umatilla River and John Day core areas respectively. Connectivity between the core areas of the Lower Mid-Columbia Region is unlikely given conditions in the connecting FMO habitats. Connection between the Umatilla, Walla Walla and Touchet core areas is uncommon but has been documented, and connectivity is possible between core areas in the John Day Basin. Connectivity between the John Day core areas and Umatilla/Walla Walla/Touchet core areas is unlikely (USFWS 2015c, pp. C-5-6).

Upper Mid-Columbia Region

In the Upper Mid-Columbia Region, core areas are distributed along the eastern side of the Cascade Mountains in Central Washington. This area contains four core areas (Yakima, Wenatchee, Entiat, and Methow), the Lake Chelan historic core area, and the Chelan River, Okanogan River, and Columbia River FMO areas. The core area populations are generally considered migratory, though they currently express both migratory (fluvial and adfluvial) and resident forms. Residents are located both above and below natural barriers (*i.e.*, Early Winters Creek above a natural falls; and Ahtanum in the Yakima likely due to long lack of connectivity from irrigation withdrawal). In terms of uniqueness and connectivity, the genetics baseline, radio-telemetry, and PIT tag studies identified unique local populations in all core areas. Movement patterns within the core areas; between the lower river, lakes, and other core areas; and between the Chelan, Okanogan, and Columbia River FMO occurs regularly for some of the Wenatchee, Entiat, and Methow core area populations. This type of connectivity has been displayed by one or more fish, typically in non-spawning movements within FMO. More recently, connectivity has been observed between the Entiat and Yakima core areas by a juvenile bull trout tagged in the Entiat moving in to the Yakima at Prosser Dam and returning at an adult size back to the Entiat. Genetics baselines identify unique populations in all four core areas (USFWS 2015c, p. C-6).

The demographic status is variable in the Upper-Mid Columbia region and ranges from good to very poor. The Service's 2008 5-year Review and Conservation Status Assessment described the Methow and Yakima Rivers at risk, with a rapidly declining trend. The Entiat River was listed at risk with a stable trend, and the Wenatchee River as having a potential risk, and with a stable trend. Currently, the Entiat River is considered to be declining rapidly due to much reduced redd counts. The Wenatchee River is able to exhibit all freshwater life histories with connectivity to Lake Wenatchee, the Wenatchee River and all its local populations, and to the Columbia River and/or other core areas in the region. In the Yakima core area some populations exhibit life history forms different from what they were historically. Migration between local populations and to and from spawning habitat is generally prevented or impeded by headwater storage dams on irrigation reservoirs, connectivity between tributaries and reservoirs, and within lower portions of spawning and rearing habitat and the mainstem Yakima River due to changed flow patterns, low instream flows, high water temperatures, and other habitat impediments. Currently, the connectivity in the Yakima Core area is truncated to the degree that not all populations are able to contribute gene flow to a functional metapopulation (USFWS 2015c, pp. C-6-7).

Lower Snake Region

Demographic status is variable within the Lower Snake Region. Although trend data are lacking, several core areas in the Grande Ronde Basin and the Imnaha core area are thought to be stable. The upper Grande Ronde Core Area is the exception where population abundance is considered depressed. Wenaha, Little Minam, and Imnaha Rivers are strongholds (as mentioned above), as are most core areas in the Clearwater River basin. Most core areas contain populations that express both a resident and fluvial life history strategy. There is potential that some bull trout in the upper Wallowa River are adfluvial. There is potential for connectivity between core areas in the Grande Ronde basin, however conditions in FMO are limiting (USFWS 2015c, p. C-7).

Middle Snake Region

In the Middle Snake Region, core areas are distributed along both sides of the Snake River above Hells Canyon Dam. The Powder River and Pine Creek basins are in Oregon and Indian Creek and Wildhorse Creek are on the Idaho side of the Snake River. Demographic status of the core areas is poorest in the Powder River Core Area where populations are highly fragmented and severely depressed. The East Pine Creek population in the Pine-Indian-Wildhorse Creeks core area is likely the most abundant within the region. Populations in both core areas primarily express a resident life history strategy; however, some evidence suggests a migratory life history still exists in the Pine-Indian-Wildhorse Creeks core area. Connectivity is severely impaired in the Middle Snake Region. Dams, diversions and temperature barriers prevent movement among populations and between core areas. Brownlee Dam isolates bull trout in Wildhorse Creek from other populations (USFWS 2015c, p. C-7).

Columbia Headwaters Recovery Unit

The Columbia Headwaters Recovery Unit (CHRU) includes western Montana, northern Idaho, and the northeastern corner of Washington. Major drainages include the Clark Fork River basin and its Flathead River contribution, the Kootenai River basin, and the Coeur d'Alene Lake basin. In this implementation plan for the CHRU we have slightly reorganized the structure from the 2002 Draft Recovery Plan, based on latest available science and fish passage improvements that have rejoined previously fragmented habitats. We now identify 35 bull trout core areas (compared to 47 in 2002) for this recovery unit. Fifteen of the 35 are referred to as "complex" core areas as they represent large interconnected habitats, each containing multiple spawning streams considered to host separate and largely genetically identifiable local populations. The 15 complex core areas contain the majority of individual bull trout and the bulk of the designated critical habitat (USFWS 2010).

However, somewhat unique to this recovery unit is the additional presence of 20 smaller core areas, each represented by a single local population. These "simple" core areas are found in remote glaciated headwater basins, often in Glacier National Park or federally-designated wilderness areas, but occasionally also in headwater valley bottoms. Many simple core areas are upstream of waterfalls or other natural barriers to fish migration. In these simple core areas bull trout have apparently persisted for thousands of years despite small populations and isolated existence. As such, simple core areas meet the criteria for core area designation and continue to be valued for their uniqueness, despite limitations of size and scope. Collectively, the 20 simple core areas contain less than 3 percent of the total bull trout core area habitat in the CHRU, but represent significant genetic and life history diversity (Meeuwig et al. 2010). Throughout this recovery unit implementation plan, we often separate our analyses to distinguish between complex and simple core areas, both in respect to threats as well as recovery actions (USFWS 2015d, pp. D-1-2).

In order to effectively manage the recovery unit implementation plan (RUIP) structure in this large and diverse landscape, the core areas have been separated into the following five natural geographic assemblages.

Upper Clark Fork Geographic Region

Starting at the Clark Fork River headwaters, the *Upper Clark Fork Geographic Region* comprises seven complex core areas, each of which occupies one or more major watersheds contributing to the Clark Fork basin (*i.e.*, Upper Clark Fork River, Rock Creek, Blackfoot River, Clearwater River and Lakes, Bitterroot River, West Fork Bitterroot River, and Middle Clark Fork River core areas) (USFWS 2015d, p. D-2).

Lower Clark Fork Geographic Region

The seven headwater core areas flow into the *Lower Clark Fork Geographic Region*, which comprises two complex core areas, Lake Pend Oreille and Priest Lake. Because of the systematic and jurisdictional complexity (three States and a Tribal entity) and the current degree of migratory fragmentation caused by five mainstem dams, the threats and recovery actions in the Lake Pend Oreille (LPO) core area are very complex and are described in three parts. LPO-A is upstream of Cabinet Gorge Dam, almost entirely in Montana, and includes the mainstem Clark Fork River upstream to the confluence of the Flathead River as well as the portions of the lower Flathead River (*e.g.*, Jocko River) on the Flathead Indian Reservation. LPO-B is the Pend Oreille lake basin proper and its tributaries, extending between Albeni Falls Dam downstream from the outlet of Lake Pend Oreille and Cabinet Gorge Dam just upstream of the lake; almost entirely in Idaho. LPO-C is the lower basin (*i.e.*, lower Pend Oreille River), downstream of Albeni Falls Dam to Boundary Dam (1 mile upstream from the Canadian border) and bisected by Box Canyon Dam; including portions of Idaho, eastern Washington, and the Kalispel Reservation (USFWS 2015d, p. D-2).

Historically, and for current purposes of bull trout recovery, migratory connectivity among these separate fragments into a single entity remains a primary objective.

Flathead Geographic Region

The *Flathead Geographic Region* includes a major portion of northwestern Montana upstream of Kerr Dam on the outlet of Flathead Lake. The complex core area of Flathead Lake is the hub of this area, but other complex core areas isolated by dams are Hungry Horse Reservoir (formerly South Fork Flathead River) and Swan Lake. Within the glaciated basins of the Flathead River headwaters are 19 simple core areas, many of which lie in Glacier National Park or the Bob Marshall and Great Bear Wilderness areas and some of which are isolated by natural barriers or other features (USFWS 2015d, p. D-2).

Kootenai Geographic Region

To the northwest of the Flathead, in an entirely separate watershed, lies the *Kootenai Geographic Region*. The Kootenai is a uniquely patterned river system that originates in southeastern British Columbia, Canada. It dips, in a horseshoe configuration, into northwest Montana and north Idaho before turning north again to re-enter British Columbia and eventually join the Columbia River headwaters in British Columbia. The *Kootenai Geographic Region* contains two complex core areas (Lake Koocanusa and the Kootenai River) bisected since the 1970's by Libby Dam, and also a single naturally isolated simple core area (Bull Lake). Bull trout in both of the complex core areas retain strong migratory connections to populations in British Columbia (USFWS 2015d, p. D-3).

Coeur d'Alene Geographic Region

Finally, the *Coeur d'Alene Geographic Region* consists of a single, large complex core area centered on Coeur d'Alene Lake. It is grouped into the CHRU for purposes of physical and ecological similarity (adfluvial bull trout life history and nonanadromous linkage) rather than due to watershed connectivity with the rest of the CHRU, as it flows into the mid-Columbia River far downstream of the Clark Fork and Kootenai systems (USFWS 2015d, p. D-3).

Upper Snake Recovery Unit

The Upper Snake Recovery Unit includes portions of central Idaho, northern Nevada, and eastern Oregon. Major drainages include the Salmon River, Malheur River, Jarbidge River, Little Lost River, Boise River, Payette River, and the Weiser River. The Upper Snake Recovery Unit contains 22 bull trout core areas within 7 geographic regions or major watersheds: Salmon River (10 core areas, 123 local populations), Boise River (2 core areas, 29 local populations), Payette River (5 core areas, 25 local populations), Little Lost River (1 core area, 10 local populations), Malheur River (2 core areas, 8 local populations), Jarbidge River (1 core area, 6 local populations), and Weiser River (1 core area, 5 local populations). The Upper Snake Recovery Unit includes a total of 206 local populations, with almost 60 percent being present in the Salmon River watershed (USFWS 2015e, p. E-1).

Three major bull trout life history expressions are present in the Upper Snake Recovery Unit, adfluvial[3], fluvial[4], and resident populations. Large areas of intact habitat exist primarily in the Salmon drainage, as this is the only drainage in the Upper Snake Recovery Unit that still flows directly into the Snake River; most other drainages no longer have direct connectivity due to irrigation uses or instream barriers. Bull trout in the Salmon basin share a genetic past with bull trout elsewhere in the Upper Snake Recovery Unit. Historically, the Upper Snake Recovery Unit is believed to have largely supported the fluvial life history form; however, many core areas are now isolated or have become fragmented watersheds, resulting in replacement of the fluvial life history with resident or adfluvial forms. The Weiser River, Squaw Creek, Pahsimeroi River, and North Fork Payette River core areas contain only resident populations of bull trout (USFWS 2015e, pp. E-1-2).

Salmon River

The Salmon River basin represents one of the few basins that are still free-flowing down to the Snake River. The core areas in the Salmon River basin do not have any major dams and a large extent (approximately 89 percent) is federally managed, with large portions of the Middle Fork Salmon River and Middle Fork Salmon River - Chamberlain core areas occurring within the Frank Church River of No Return Wilderness. Most core areas in the Salmon River basin contain large populations with many occupied stream segments. The Salmon River basin contains 10 of the 22 core areas in the Upper Snake Recovery Unit and contains the majority of the occupied habitat. Over 70 percent of occupied habitat in the Upper Snake Recovery Unit occurs in the Salmon River basin as well as 123 of the 206 local populations. Connectivity between core areas in the Salmon River basin is intact; therefore it is possible for fish in the mainstem Salmon to migrate to almost any Salmon River core area or even the Snake River.

Connectivity within Salmon River basin core areas is mostly intact except for the Pahsimeroi River and portions of the Lemhi River. The Upper Salmon River, Lake Creek, and Opal Lake core areas contain adfluvial populations of bull trout, while most of the remaining core areas contain fluvial populations; only the Pahsimeroi contains strictly resident populations. Most core areas appear to have increasing or stable trends but trends are not known in the Pahsimeroi, Lake Creek, or Opal Lake core areas. The Idaho Department of Fish and Game reported trend data from 7 of the 10 core areas. This trend data indicated that populations were stable or increasing in the Upper Salmon River, Lemhi River, Middle Salmon River-Chamberlain, Little Lost River, and the South Fork Salmon River (IDFG 2005, 2008). Trends were stable or decreasing in the Little-Lower Salmon River, Middle Fork Salmon River, and the Middle Salmon River-Panther (IDFG 2005, 2008).

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[3] Adfluvial: Life history pattern of spawning and rearing in tributary streams and migrating to lakes or reservoirs to mature.

[4] Fluvial: Life history pattern of spawning and rearing in tributary streams and migrating to larger rivers to mature.

Boise River

In the Boise River basin, two large dams are impassable barriers to upstream fish movement: Anderson Ranch Dam on the South Fork Boise River, and Arrowrock Dam on the mainstem Boise River. Fish in Anderson Ranch Reservoir have access to the South Fork Boise River upstream of the dam. Fish in Arrowrock Reservoir have access to the North Fork Boise River, Middle Fork Boise River, and lower South Fork Boise River. The Boise River basin contains 2 of the 22 core areas in the Upper Snake Recovery Unit. The core areas in the Boise River basin account for roughly 12 percent of occupied habitat in the Upper Snake Recovery Unit and contain 29 of the 206 local populations. Approximately 90 percent of both Arrowrock and Anderson Ranch core areas are federally owned; most lands are managed by the U.S. Forest Service, with some portions occurring in designated wilderness areas. Both the Arrowrock core area and the Anderson Ranch core area are isolated from other core areas. Both core areas contain fluvial bull trout that exhibit adfluvial characteristics and numerous resident populations. The Idaho Department of Fish and Game in 2014 determined that the Anderson Ranch core area had an increasing trend while trends in the Arrowrock core area is unknown (USFWS 2015e).

Payette River

The Payette River basin contains three major dams that are impassable barriers to fish: Deadwood Dam on the Deadwood River, Cascade Dam on the North Fork Payette River, and Black Canyon Reservoir on the Payette River. Only the Upper South Fork Payette River and the Middle Fork Payette River still have connectivity, the remaining core areas are isolated from each other due to dams. Both fluvial and adfluvial life history expression are still present in the Payette River basin but only resident populations are present in the Squaw Creek and North Fork Payette River core areas. The Payette River basin contains 5 of the 22 core areas and 25 of the 206 local populations in the recovery unit. Less than 9 percent of occupied habitat in the recovery unit is in this basin. Approximately 60 percent of the lands in the core areas are federally owned and the majority is managed by the U.S. Forest Service. Trend data are lacking and the current condition of the various core areas is unknown, but there is concern due to the current isolation of three (North Fork Payette River, Squaw Creek, Deadwood River) of the five core areas; the presence of only resident local populations in two (North Fork Payette River, Squaw Creek) of the five core areas; and the relatively low numbers present in the North Fork core area (USFWS 2015e, p. E-8).

Jarbidge River

The Jarbidge River core area contains two major fish barriers along the Bruneau River: the Buckaroo diversion and C. J. Strike Reservoir. Bull trout are not known to migrate down to the Snake River. There is one core area in the basin, with populations in the Jarbidge River; this watershed does not contain any barriers. Approximately 89 percent of the Jarbidge core area is federally owned. Most lands are managed by either the Forest Service or Bureau of Land Management. A large portion of the core area is within the Bruneau-Jarbidge Wilderness area. A tracking study has documented bull trout population connectivity among many of the local populations, in particular between West Fork Jarbidge River and Pine Creek. Movement between the East and West Fork Jarbidge River has also been documented; therefore, both resident and fluvial populations are present. The core area contains six local populations and 3 percent of the occupied habitat in the recovery unit. Trend data are lacking within this core area (USFWS 2015e, p. E-9).

Little Lost River

The Little Lost River basin is unique in that the watershed is within a naturally occurring hydrologic sink and has no connectivity with other drainages. A small fluvial population of bull trout may still exist, but it appears that most populations are predominantly resident populations. There is one core area in the Little Lost basin, and approximately 89 percent of it is federally owned by either the U.S. Forest Service or Bureau of Land Management. The core area contains 10 local populations and less than 3 percent of the occupied habitat in the recovery unit. The current trend condition of this core area is likely stable, with most bull trout residing in Upper Sawmill Canyon (IDFG 2014).

Malheur River

The Malheur River basin contains major dams that are impassable to fish. The largest are Warm Springs Dam, impounding Warm Springs Reservoir on the mainstem Malheur River, and Agency Valley Dam, impounding Beulah Reservoir on the North Fork Malheur River. The dams result in two core areas that are isolated from each other and from other core areas. Local populations in the two core areas are limited to habitat in the upper watersheds. The Malheur River basin contains 2 of the 22 core areas and 8 of the 206 local populations in the recovery unit. Fluvial and resident populations are present in both core areas while adfluvial populations are present in the North Fork Malheur River. This basin contains less than 3 percent of the occupied habitat in the recovery unit, and approximately 60 percent of lands in the two core areas are federally owned. Trend data indicates that populations are declining in both core areas (USFWS 2015e, p. E-9).

Weiser River

The Weiser River basin contains local populations that are limited to habitat in the upper watersheds. The Weiser River basin contains only a single core area that consists of 5 of the 206 local populations in the recovery unit. Local populations occur in only three stream complexes in the upper watershed: 1) Upper Hornet Creek, 2) East Fork Weiser River, and 3) Upper Little Weiser River. These local populations include only resident life histories. This basin contains less than 2 percent of the occupied habitat in the recovery unit, and approximately 44 percent of lands are federally owned. Trend data from the Idaho Department of Fish and Game indicate that the populations in the Weiser core area are increasing (IDFG 2014) but it is considered vulnerable because local populations are isolated and likely do not express migratory life histories (USFWS 2015e, p.E-10).

St. Mary Recovery Unit

The Saint Mary Recovery Unit is located in northwest Montana east of the Continental Divide and includes the U.S. portions of the Saint Mary River basin, from its headwaters to the international boundary with Canada at the 49th parallel. The watershed and the bull trout population are linked to downstream aquatic resources in southern Alberta, Canada; the U.S. portion includes headwater spawning and rearing (SR) habitat in the tributaries and a portion of the FMO habitat in the mainstem of the Saint Mary River and Saint Mary lakes (Mogen and Kaeding 2001).

The Saint Mary Recovery Unit comprises four core areas; only one (Saint Mary River) is a complex core area with five described local bull trout populations (Divide, Boulder, Kennedy, Otatso, and Lee Creeks). Roughly half of the linear extent of available FMO habitat in the mainstem Saint Mary system (between Saint Mary Falls at the upstream end and the downstream Canadian border) is comprised of Saint Mary and Lower Saint Mary Lakes, with the remainder in the Saint Mary River. The other three core areas (Slide Lakes, Cracker Lake, and Red Eagle Lake) are simple core areas. Slide Lakes and Cracker Lake occur upstream of seasonal or permanent barriers and are comprised of genetically isolated single local bull trout populations, wholly within Glacier National Park, Montana. In the case of Red Eagle Lake, physical isolation does not occur, but consistent with other lakes in the adjacent Columbia Headwaters Recovery Unit, there is likely some degree of spatial separation from downstream Saint Mary Lake. As noted, the extent of isolation has been identified as a research need (USFWS 2015f, p. F-1).

Bull trout in the Saint Mary River complex core area are documented to exhibit primarily the migratory fluvial life history form (Mogen and Kaeding 2005a, 2005b), but there is doubtless some occupancy (though less well documented) of Saint Mary Lakes, suggesting a partly adfluvial adaptation. Since lake trout and northern pike are both native to the Saint Mary River system (headwaters of the South Saskatchewan River drainage draining to Hudson Bay), the conventional wisdom is that these large piscivores historically outcompeted bull trout in the lacustrine environment (Donald and Alger 1993, Martinez et al. 2009), resulting in a primarily fluvial niche and existence for bull trout in this system. This is an untested hypothesis and additional research into this aspect is needed (USFWS 2015f, p. F-3).

Bull trout populations in the simple core areas of the three headwater lake systems (Slide, Cracker, and Red Eagle Lakes) are, by definition, adfluvial; there are also resident life history components in portions of the Saint Mary River system such as Lower Otatso Creek (Mogen and Kaeding 2005a), further exemplifying the overall life history diversity typical of bull trout. Mogen and Kaeding (2001) reported that bull trout continue to inhabit nearly all suitable habitats accessible to them in the Saint Mary River basin in the United States. The possible exception is portions of Divide Creek, which appears to be intermittently occupied despite a lack of permanent migratory barriers, possibly due to low population size and erratic year class production (USFWS 2015f, p. F-3).

It should be noted that bull trout are found in minor portions of two additional U.S. watersheds (Belly and Waterton rivers) that were once included in the original draft recovery plan (USFWS 2002) but are no longer considered core areas in the final recovery plan (USFWS 2015) and are not addressed in that document. In Alberta, Canada, the Saint Mary River bull trout population is considered at “high risk,” while the Belly River is rated as “at risk” (ACA 2009). In the Belly River drainage, which enters the South Saskatchewan system downstream of the Saint Mary River in Alberta, some bull trout spawning is known to occur on either side of the international boundary. These waters are in the drainage immediately west of the Saint Mary River headwaters. However, the U.S. range of this population constitutes only a minor headwater migratory SR segment of an otherwise wholly Canadian population, extending less than 1 mile (0.6 km) into backcountry waters of Glacier National Park. The Belly River population is otherwise totally dependent on management within Canadian jurisdiction, with no natural migratory connection to the Saint Mary (USFWS 2015f, p. F-3).

Current status of bull trout in the Saint Mary River core area (U.S.) is considered strong (Mogen 2013). Migratory bull trout redd counts are conducted annually in the two major SR streams, Boulder and Kennedy creeks. Boulder Creek redd counts have ranged from 33 to 66 in the past decade, with the last 4 counts all 53 or higher. Kennedy Creek redd counts are less robust, ranging from 5 to 25 over the last decade, with a 2014 count of 20 (USFWS 2015f, p. F-3).

Generally, the demographic status of the Saint Mary River core area is believed to be good, with the exception of the Divide Creek local population. In this local population, there is evidence that a combination of ongoing habitat manipulation (Smillie and Ellerbroek 1991, F-5 NPS 1992) resulting in occasional historical passage issues, combined with low and erratic recruitment (DeHaan et al. 2011) has caused concern for the continuing existence of the local population.

While less is known about the demographic status of the three simple cores where redd counts are not conducted, all three appear to be self-sustaining and fluctuating within known historical population demographic bounds. Of the three simple core areas, demographic status in Slide Lakes and Cracker Lake appear to be functioning appropriately, but the demographic status in Red Eagle Lake is less well documented and believed to be less robust (USFWS 2015f, p. F-3).

Reasons for Listing

Bull trout distribution, abundance, and habitat quality have declined rangewide (Bond 1992, pp. 2-3; Schill 1992, p. 42; Thomas 1992, entire; Ziller 1992, entire; Rieman and McIntyre 1993, p. 1; Newton and Pribyl 1994, pp. 4-5; McPhail and Baxter 1996, p. 1). Several local extirpations have been documented, beginning in the 1950s (Rode 1990, pp. 26-32; Ratliff and Howell 1992, entire; Donald and Alger 1993, entire; Goetz 1994, p. 1; Newton and Pribyl 1994, pp. 8-9; Light et al. 1996, pp. 6-7; Buchanan et al. 1997, p. 15; WDFW 1998, pp. 2-3). Bull trout were extirpated from the southernmost portion of their historic range, the McCloud River in California, around 1975 (Rode 1990, p. 32). Bull trout have been functionally extirpated (i.e., few individuals may occur there but do not constitute a viable population) in the Coeur d'Alene River basin in Idaho and in the Lake Chelan and Okanogan River basins in Washington (USFWS 1998, pp. 31651-31652).

These declines result from the combined effects of habitat degradation and fragmentation, the blockage of migratory corridors; poor water quality, angler harvest and poaching, entrainment (process by which aquatic organisms are pulled through a diversion or other device) into diversion channels and dams, and introduced nonnative species. Specific land and water management activities that depress bull trout populations and degrade habitat include the effects of dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (Beschta et al. 1987, entire; Chamberlain et al. 1991, entire; Furniss et al. 1991, entire; Meehan 1991, entire; Nehlsen et al. 1991, entire; Sedell and Everest 1991, entire; Craig and Wissmar 1993pp, 18-19; Henjum et al. 1994, pp. 5-6; McIntosh et al. 1994, entire; Wissmar et al. 1994, entire; MBTSG 1995a, p. 1; MBTSG 1995b. pp. i-ii; MBTSG 1995c, pp. i-ii; MBTSG 1995d, p. 22; MBTSG 1995e, p. i; MBTSG 1996a, p. i-ii; MBTSG 1996b, p. i; MBTSG 1996c, p. i; MBTSG 1996d, p. i; MBTSG 1996e, p. i; MBTSG 1996f, p. 11; Light et al. 1996, pp. 6-7; USDA and USDI 1995, p. 2).

Emerging Threats

Climate Change

Climate change was not addressed as a known threat when bull trout was listed. The 2015 bull trout recovery plan and RUIPs summarize the threat of climate change and acknowledges that some extant bull trout core area habitats will likely change (and may be lost) over time due to anthropogenic climate change effects, and use of best available information will ensure future conservation efforts that offer the greatest long-term benefit to sustain bull trout and their required coldwater habitats (USFWS 2015, p. vii, and pp. 17-20, USFWS 2015a-f).

Global climate change and the related warming of global climate have been well documented (IPCC 2007, entire; ISAB 2007, entire; Combes 2003, entire). Evidence of global climate change/warming includes widespread increases in average air and ocean temperatures and accelerated melting of glaciers, and rising sea level. Given the increasing certainty that climate change is occurring and is accelerating (IPCC 2007, p. 253; Battin et al. 2007, p. 6720), we can no longer assume that climate conditions in the future will resemble those in the past.

Patterns consistent with changes in climate have already been observed in the range of many species and in a wide range of environmental trends (ISAB 2007, entire; Hari et al. 2006, entire; Rieman et al. 2007, entire). In the northern hemisphere, the duration of ice cover over lakes and rivers has decreased by almost 20 days since the mid-1800's (Magnuson et al. 2000, p. 1743). The range of many species has shifted poleward and elevationally upward. For cold-water associated salmonids in mountainous regions, where their upper distribution is often limited by impassable barriers, an upward thermal shift in suitable habitat can result in a reduction in range, which in turn can lead to a population decline (Hari et al. 2006, entire).

In the Pacific Northwest, most models project warmer air temperatures and increases in winter precipitation and decreases in summer precipitation. Warmer temperatures will lead to more precipitation falling as rain rather than snow. As the seasonal amount of snow pack diminishes, the timing and volume of stream flow are likely to change and peak river flows are likely to increase in affected areas. Higher air temperatures are also likely to increase water temperatures (ISAB 2007, pp. 15-17). For example, stream gauge data from western Washington over the past 5 to 25 years indicate a marked increasing trend in water temperatures in most major rivers.

Climate change has the potential to profoundly alter the aquatic ecosystems upon which the bull trout depends via alterations in water yield, peak flows, and stream temperature, and an increase in the frequency and magnitude of catastrophic wildfires in adjacent terrestrial habitats (Bisson et al. 2003, pp 216-217).

All life stages of the bull trout rely on cold water. Increasing air temperatures are likely to impact the availability of suitable cold water habitat. For example, ground water temperature is generally correlated with mean annual air temperature, and has been shown to strongly influence the distribution of other chars. Ground water temperature is linked to bull trout selection of spawning sites, and has been shown to influence the survival of embryos and early juvenile rearing of bull trout (Baxter 1997, p. 82). Increases in air temperature are likely to be reflected in increases in both surface and groundwater temperatures.

Climate change is likely to affect the frequency and magnitude of fires, especially in warmer drier areas such as are found on the eastside of the Cascade Mountains. Bisson et al. (2003, pp. 216-217) note that the forest that naturally occurred in a particular area may or may not be the forest that will be responding to the fire regimes of an altered climate. In several studies related to the effect of large fires on bull trout populations, bull trout appear to have adapted to past fire disturbances through mechanisms such as dispersal and plasticity. However, as stated earlier, the future may well be different than the past and extreme fire events may have a dramatic effect on bull trout and other aquatic species, especially in the context of continued habitat loss, simplification and fragmentation of aquatic systems, and the introduction and expansion of exotic species (Bisson et al. 2003, pp. 218-219).

Migratory bull trout can be found in lakes, large rivers and marine waters. Effects of climate change on lakes are likely to impact migratory adfluvial bull trout that seasonally rely upon lakes for their greater availability of prey and access to tributaries. Climate-warming impacts to lakes will likely lead to longer periods of thermal stratification and coldwater fish such as adfluvial bull trout will be restricted to these bottom layers for greater periods of time. Deeper thermoclines resulting from climate change may further reduce the area of suitable temperatures in the bottom layers and intensify competition for food (Shuter and Meisner 1992. p. 11).

Bull trout require very cold water for spawning and incubation. Suitable spawning habitat is often found in accessible higher elevation tributaries and headwaters of rivers. However, impacts on hydrology associated with climate change are related to shifts in timing, magnitude and distribution of peak flows that are also likely to be most pronounced in these high elevation stream basins (Battin et al. 2007, p. 6720). The increased magnitude of winter peak flows in high elevation areas is likely to impact the location, timing, and success of spawning and incubation for the bull trout and Pacific salmon species. Although lower elevation river reaches are not expected to experience as severe an impact from alterations in stream hydrology, they are unlikely to provide suitably cold temperatures for bull trout spawning, incubation and juvenile rearing.

As climate change progresses and stream temperatures warm, thermal refugia will be critical to the persistence of many bull trout populations. Thermal refugia are important for providing bull trout with patches of suitable habitat during migration through or to make feeding forays into areas with greater than optimal temperatures.

There is still a great deal of uncertainty associated with predictions relative to the timing, location, and magnitude of future climate change. It is also likely that the intensity of effects will vary by region (ISAB 2007, p 7) although the scale of that variation may exceed that of States. For example, several studies indicate that climate change has the potential to impact ecosystems in nearly all streams throughout the State of Washington (ISAB 2007, p. 13; Battin et al. 2007, p. 6722; Rieman et al. 2007, pp. 1558-1561). In streams and rivers with temperatures approaching or at the upper limit of allowable water temperatures, there is little if any likelihood that bull trout will be able to adapt to or avoid the effects of climate change/warming. There is little doubt that climate change is and will be an important factor affecting bull trout distribution. As its distribution contracts, patch size decreases and connectivity is truncated, bull trout populations that may be currently connected may face increasing isolation, which could accelerate the rate of local extinction beyond that resulting from changes in stream temperature alone (Rieman et al. 2007, pp. 1559-1560). Due to variations in land form and geographic location across the range of the bull trout, it appears that some populations face higher risks than others. Bull trout in areas with currently degraded water temperatures and/or at the southern edge of its range may already be at risk of adverse impacts from current as well as future climate change.

The ability to assign the effects of gradual global climate change to bull trout or to a specific location on the ground is beyond our technical capabilities at this time.

Conservation

<u>*Conservation Needs*</u>

The 2015 recovery plan for bull trout established the primary strategy for recovery of bull trout in the coterminous United States: 1) conserve bull trout so that they are geographically widespread across representative habitats and demographically stable1 in six recovery units; 2) effectively manage and ameliorate the primary threats in each of six recovery units at the core area scale such that bull trout are not likely to become endangered in the foreseeable future; 3) build upon the numerous and ongoing conservation actions implemented on behalf of bull trout since their listing in 1999, and improve our understanding of how various threat factors potentially affect the species; 4) use that information to work cooperatively with our partners to design, fund, prioritize, and implement effective conservation actions in those areas that offer the greatest long-term benefit to sustain bull trout and where recovery can be achieved; and 5) apply adaptive management principles to implementing the bull trout recovery program to account for new information (USFWS 2015, p. v.).

Information presented in prior draft recovery plans published in 2002 and 2004 (USFWS 2002a, 2004) have served to identify recovery actions across the range of the species and to provide a framework for implementing numerous recovery actions by our partner agencies, local working groups, and others with an interest in bull trout conservation.

The 2015 recovery plan (USFWS 2015) integrates new information collected since the 1999 listing regarding bull trout life history, distribution, demographics, conservation successes, etc., and integrates and updates previous bull trout recovery planning efforts across the range of the single DPS listed under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*) (Act).

The Service has developed a recovery approach that: 1) focuses on the identification of and effective management of known and remaining threat factors to bull trout in each core area; 2) acknowledges that some extant bull trout core area habitats will likely change (and may be lost) over time; and 3) identifies and focuses recovery actions in those areas where success is likely to meet our goal of ensuring the certainty of conservation of genetic diversity, life history features, and broad geographical representation of remaining bull trout populations so that the protections of the Act are no longer necessary (USFWS 2015, p. 45-46).

To implement the recovery strategy, the 2015 recovery plan establishes categories of recovery actions for each of the six Recovery Units (USFWS 2015, p. 50-51):

1. Protect, restore, and maintain suitable habitat conditions for bull trout.
2. Minimize demographic threats to bull trout by restoring connectivity or populations where appropriate to promote diverse life history strategies and conserve genetic diversity.
3. Prevent and reduce negative effects of nonnative fishes and other nonnative taxa on bull trout.
4. Work with partners to conduct research and monitoring to implement and evaluate bull trout recovery activities, consistent with an adaptive management approach using feedback from implemented, site-specific recovery tasks, and considering the effects of climate change.

Bull trout recovery is based on a geographical hierarchical approach. Bull trout are listed as a single DPS within the five-state area of the coterminous United States. The single DPS is subdivided into six biologically-based recover units: 1) Coastal Recovery Unit; 2) Klamath Recovery Unit; 3) Mid-Columbia Recovery Unit; 4) Upper Snake Recovery Unit; 5) Columbia Headwaters Recovery Unit; and 6) Saint Mary Recovery Unit (USFWS 2015, p. 23). A viable recovery unit should demonstrate that the three primary principles of biodiversity have been met: representation (conserving the genetic makeup of the species); resiliency (ensuring that each population is sufficiently large to withstand stochastic events); and redundancy (ensuring a sufficient number of populations to withstand catastrophic events) (USFWS 2015, p. 33).

Each of the six recovery units contain multiple bull trout core areas, 116 total, which are non-overlapping watershed-based polygons, and each core area includes one or more local populations. Currently there are 109 occupied core areas, which comprise 611 local populations (USFWS 2015, p. 3). There are also six core areas where bull trout historically occurred but are now extirpated, and one research needs area where bull trout were known to occur historically, but their current presence and use of the area are uncertain (USFWS 2015, p. 3). Core areas can be further described as complex or simple (USFWS 2015, p. 3-4). Complex core areas contain multiple local bull trout populations, are found in large watersheds, have multiple life history forms, and have migratory connectivity between spawning and rearing habitat and FMO habitats. Simple core areas are those that contain one bull trout local population. Simple core areas are small in scope, isolated from other core areas by natural barriers, and may contain unique genetic or life history adaptations.

A local population is a group of bull trout that spawn within a particular stream or portion of a stream system (USFWS 2015, p. 73). A local population is considered to be the smallest group of fish that is known to represent an interacting reproductive unit. For most waters where specific information is lacking, a local population may be represented by a single headwater tributary or complex of headwater tributaries. Gene flow may occur between local populations (e.g., those within a core population), but is assumed to be infrequent compared with that among individuals within a local population.

Recovery Units and Local Populations

The final recovery plan (USFWS 2015) designates six bull trout recovery units as described above. These units replace the 5 interim recovery units previously identified (USFWS 1999). The Service will address the conservation of these final recovery units in our section 7(a)(2) analysis for proposed Federal actions. The recovery plan (USFWS 2015), identified threats and factors affecting the bull trout within these units. A detailed description of recovery implementation for each recovery unit is provided in separate recovery unit implementation plans (RUIPs)(USFWS 2015a-f), which identify conservation actions and recommendations needed for each core area, forage/ migration/ overwinter areas, historical core areas, and research needs areas. Each of the following recovery units (below) is necessary to maintain the bull trout's distribution, as well as its genetic and phenotypic diversity, all of which are important to ensure the species' resilience to changing environmental conditions.

Coastal Recovery Unit

The coastal recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015a). The Coastal Recovery Unit is located within western Oregon and Washington. The Coastal Recovery Unit is divided into three regions: Puget Sound, Olympic Peninsula, and the Lower Columbia River Regions. This recovery unit contains 20 core areas comprising 84 local populations and a single potential local population in the historic Clackamas River core area where bull trout had been extirpated and were reintroduced in 2011, and identified four historically occupied core areas that could be re-established (USFWS 2015, pg. 47; USFWS 2015a, p. A-2). Core areas within Puget Sound and the Olympic Peninsula currently support the only anadromous local populations of bull trout. This recovery unit also contains ten shared FMO habitats which are outside core areas and allows for the continued natural population dynamics in which the core areas have evolved (USFWS 2015a, p. A-5). There are four core areas within the Coastal Recovery Unit that have been identified as current population strongholds: Lower Skagit, Upper Skagit, Quinault River, and Lower Deschutes River (USFWS 2015, p.79). These are the most stable and abundant bull trout populations in the recovery unit. The current condition of the bull trout in this recovery unit is attributed to the adverse effects of climate change, loss of functioning estuarine and nearshore marine habitats, development and related impacts (e.g., flood control, floodplain disconnection, bank armoring, channel straightening, loss of instream habitat complexity), agriculture (e.g., diking, water control structures, draining of wetlands, channelization, and the removal of riparian vegetation, livestock grazing), fish passage (e.g., dams, culverts, instream flows) residential development, urbanization, forest management practices (e.g., timber harvest and associated road building activities), connectivity impairment, mining, and the introduction of non-native species. Conservation measures or recovery actions implemented include relicensing of major hydropower facilities that have provided upstream and downstream fish passage or complete removal of dams, land acquisition to conserve bull trout habitat, floodplain restoration, culvert removal, riparian revegetation, levee setbacks, road removal, and projects to protect and restore important nearshore marine habitats.

Klamath Recovery Unit

The Klamath recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015b). The Klamath Recovery Unit is located in southern Oregon and northwestern California. The Klamath Recovery Unit is the most significantly imperiled recovery unit, having experienced considerable extirpation and geographic contraction of local populations and declining demographic condition, and natural re-colonization is constrained by dispersal barriers and presence of nonnative brook trout (USFWS 2015, p. 39). This recovery unit currently contains three core areas and eight local populations (USFWS 2015, p. 47; USFWS 2015b, p. B-1). Nine historic local populations of bull trout have become extirpated (USFWS 2015b, p. B-1). All three core areas have been isolated from other bull trout populations for the past 10,000 years (USFWS 2015b, p. B-3. The current condition of the bull trout in this recovery unit is attributed to the adverse effects of climate change, habitat degradation and fragmentation, past and present land use practices, agricultural water diversions, nonnative species, and past fisheries management practices. Conservation measures or recovery actions implemented include removal of nonnative fish (e.g., brook trout, brown trout, and hybrids), acquiring water rights for instream flows, replacing diversion structures, installing fish screens, constructing bypass channels, installing riparian fencing, culvert replacement, and habitat restoration.

Mid-Columbia Recovery Unit

The Mid-Columbia recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015c). The Mid-Columbia Recovery Unit is located within eastern Washington, eastern Oregon, and portions of central Idaho. The Mid-Columbia Recovery Unit is divided into four geographic regions: Lower Mid-Columbia, Upper Mid-Columbia, Lower Snake, and Mid-Snake Geographic Regions. This recovery unit contains 24 occupied core areas comprising 142 local populations, two historically occupied core areas, one research needs area, and seven FMO habitats (USFWS 2015, pg. 47; USFWS 2015c, p. C-1–4). The current condition of the bull trout in this recovery unit is attributed to the adverse effects of climate change, agricultural practices (e.g. irrigation, water withdrawals, livestock grazing), fish passage (e.g. dams, culverts), nonnative species, forest management practices, and mining. Conservation measures or recovery actions implemented include road removal, channel restoration, mine reclamation, improved grazing management, removal of fish barriers, and instream flow requirements.

Columbia Headwaters Recovery Unit

The Columbia headwaters recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015d, entire). The Columbia Headwaters Recovery Unit is located in western Montana, northern Idaho, and the northeastern corner of Washington. The Columbia Headwaters Recovery Unit is divided into five geographic regions: Upper Clark Fork, Lower Clark Fork, Flathead, Kootenai, and Coeur d'Alene Geographic Regions (USFWS 2015d, pp. D-2 – D-4). This recovery unit contains 35 bull trout core areas; 15 of which are complex core areas as they represent larger interconnected habitats and 20 simple core areas as they are isolated headwater lakes with single local populations. The 20 simple core areas are each represented by a single local population, many of which may have persisted for thousands of years despite small populations and isolated existence (USFWS 2015d, p. D-1). Fish passage improvements within the recovery unit have reconnected some previously fragmented habitats (USFWS 2015d, p. D-1), while others remain fragmented. Unlike the other recovery units in Washington, Idaho and Oregon, the Columbia Headwaters Recovery Unit does not have any anadromous fish overlap. Therefore, bull trout within the Columbia Headwaters Recovery Unit do not benefit from the recovery actions for salmon (USFWS 2015d, p. D-41). The current condition of the bull trout in this recovery unit is attributed to the adverse effects of climate change, mostly historical mining and contamination by heavy metals, expanding populations of nonnative fish predators and competitors, modified instream flows, migratory barriers (e.g., dams), habitat fragmentation, forest practices (e.g., logging, roads), agriculture practices (e.g. irrigation, livestock grazing), and residential development. Conservation measures or recovery actions implemented include habitat improvement, fish passage, and removal of nonnative species.

Upper Snake Recovery Unit

The Upper Snake recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015e, entire). The Upper Snake Recovery Unit is located in central Idaho, northern Nevada, and eastern Oregon. The Upper Snake Recovery Unit is divided into seven geographic regions: Salmon River, Boise River, Payette River, Little Lost River, Malheur River, Jarbidge River, and Weiser River. This recovery unit contains 22 core areas and 207 local populations (USFWS 2015, p. 47), with almost 60 percent being present in the Salmon River Region. The current condition of the bull trout in this recovery unit is attributed to the adverse effects of climate change, dams, mining, forest management practices, nonnative species, and agriculture (e.g., water diversions, grazing). Conservation measures or recovery actions implemented include instream habitat restoration, instream flow requirements, screening of irrigation diversions, and riparian restoration.

St. Mary Recovery Unit

The St. Mary recovery unit implementation plan describes the threats to bull trout and the site-specific management actions necessary for recovery of the species within the unit (USFWS 2015f). The Saint Mary Recovery Unit is located in Montana but is heavily linked to downstream resources in southern Alberta, Canada. Most of the Saskatchewan River watershed which the St. Mary flows into is located in Canada. The United States portion includes headwater spawning and rearing habitat and the upper reaches of FMO habitat. This recovery unit contains four core areas, and seven local populations (USFWS 2015f, p. F-1) in the U.S. Headwaters. The current condition of the bull trout in this recovery unit is attributed primarily to the outdated design and operations of the Saint Mary Diversion operated by the Bureau of Reclamation (e.g., entrainment, fish passage, instream flows), and, to a lesser extent habitat impacts from development and nonnative species.

Tribal Conservation Activities

Many Tribes throughout the range of the bull trout are participating on bull trout conservation working groups or recovery teams in their geographic areas of interest. Some tribes are also implementing projects which focus on bull trout or that address anadromous fish but benefit bull trout (e.g., habitat surveys, passage at dams and diversions, habitat improvement, and movement studies).

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APPENDIX B
STATUS OF THE SPECIES: MARBLED MURRELET

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Appendix B
Status of the Species: Marbled Murrelet

The marbled murrelet (*Brachyramphus marmoratus*) (murrelet) was listed by the U.S. Fish and Wildlife Service (Service) as a threatened species in Washington, Oregon, and California in 1992. The primary reasons for listing included extensive loss and fragmentation of the older-age forests that serve as nesting habitat for murrelets, and human-induced mortality in the marine environment from gillnets and oil spills (57 FR 45328 [Oct. 1, 1992]). Although some threats such as gillnet mortality and loss of nesting habitat on Federal lands have been reduced since the 1992 listing, the primary threats to species persistence continue (75 FR 3424 [Jan. 21, 2010]).

Life History

The murrelet is a small, fast-flying seabird in the Alcidae family that occurs along the Pacific coast of North America. Murrelets forage for small schooling fish or invertebrates in shallow, nearshore, marine waters and primarily nest in coastal older-aged coniferous forests. The murrelet lifespan is unknown, but is expected to be in the range of 10 to 20 years based on information from similar alcid species (De Santo and Nelson 1995, pp. 36-37). Murrelet nesting is asynchronous and spread over a prolonged season. In Washington, the murrelet breeding season extends from April 1 to September 23. Egg laying and incubation occur from April to early August and chick rearing occurs between late May and September, with all chicks fledging by late September (Hamer et al. 2003; USFWS 2012a).

Murrelets lay a single-egg which may be replaced if egg failure occurs early in the nesting cycle, but this is rare (Nelson 1997, p. 17). During incubation, one adult sits on the nest while the other forages at sea. Adults typically incubate for a 24-hour period, then exchange duties with their mate at dawn. Chicks hatch between May and August after 30 days of incubation. Hatchlings appear to be brooded by an adult for several days (Nelson 1997, p. 18). Once the chick attains thermoregulatory independence, both adults leave the chick alone at the nest for the remainder of the rearing period, except during feedings. Both parents feed the chick, which receives one to eight meals per day (Nelson 1997, p. 18). Most meals are delivered early in the morning while about a third of the food deliveries occur at dusk and intermittently throughout the day (Nelson and Hamer 1995, p. 62).

Murrelets and other fish-eating alcids exhibit wide variations in nestling growth rates. The nestling stage of murrelet development can vary from 27 to 40 days before fledging (De Santo and Nelson 1995, p. 45). The variations in alcid chick development are attributed to constraints on feeding ecology, such as unpredictable and patchy food distributions, and great distances between feeding and nesting sites (Øyan and Anker-Nilssen 1996, p. 830). Food limitation during nesting often results in poor growth, delayed fledging, increased mortality of chicks, and nest abandonment by adults (Øyan and Anker-Nilssen 1996, p. 836).

Murrelets are believed to be sexually mature at 2 to 4 years of age (Nelson 1997, p. 19). Adult birds may not nest every year, especially when food resources are limited. For example, in central California, the proportion of murrelets attempting to breed was more than four times higher (50 percent versus 11 percent) in a year when prey availability was apparently good than in a year when more foraging effort was required (Peery et al. 2004, p. 1095). In Oregon, there was similarly a four-fold increase in vacancy rates of previously-occupied nesting habitat following the poorest ocean conditions, as compared with the years following the best ocean conditions (Betts et al. 2020, p. 6). In 2017, none of the 61 murrelets radio-tagged in Oregon attempted nesting, likely because anomalous ocean conditions reduced prey availability (Horton et al. 2018, p. 77). At other times and places, radio-telemetry and demographic modeling indicate that the proportion of adults breeding in a given year may vary from 5 to 95 percent (Lorenz et al. 2017, p. 312; McShane et al. 2004, p. 3-5). In other words, in some years, very few murrelets attempt nesting, but in other years, almost all breeding-age adults may initiate nesting.

Murrelets in the Marine Environment

Murrelets spend most (>90 percent) of their time at sea. They generally forage in pairs on the water, but they also forage solitarily or in small groups. In addition to foraging, their activities in the marine environment include preening, social behaviors, and loafing. Following the breeding season, murrelets undergo the pre-basic molt, in which they exchange their breeding plumage for their winter plumage. They replace their flight feathers during this molt, and for a few weeks they are flightless. Therefore, they spend this entire period at sea. Their preferred marine habitat includes sheltered, nearshore waters, although they occur farther offshore in some locations and during the nonbreeding season (Huff et al. 2006, p. 19).

Breeding Season Distribution

The murrelet is widely distributed in nearshore waters along the west coast of North America. It occurs primarily within 5 km of shore (in Alaska, within 50 km), and primarily in protected waters, although its distribution varies with coastline topography, river plumes, riptides, and other physical features (Nelson 1997, p. 3). For example, along the Pacific coast of Washington, the most heavily-used area during the breeding season extends to at least 8 km from the coast, with use in some years concentrated in the outer portions of this area (Bentivoglio et al. 2002, p. 29; McIver et al., in press, pp. 34, 85; Menza et al. 2015, pp. 16, 20-21). The distribution of murrelets in marine waters during the summer breeding season is highly variable along the Pacific coast, with areas of high density occurring along the Strait of Juan de Fuca in Washington, the central Oregon coast, and northern California (Raphael et al. 2015, p. 20). Low-density areas or gaps in murrelet distribution occur in central California, and along the southern Washington coast (Raphael et al. 2015, p. 21). Murrelet marine habitat use is strongly associated with the amount and configuration of nearby terrestrial nesting habitat (Raphael et al. 2015, p. 17). In other words, they tend to be present in marine waters adjacent to areas of suitable breeding habitat. Local aggregations or "hot spots" of murrelets in nearshore marine waters are strongly associated with landscapes that support large, contiguous areas of mature and old-growth forest. In Puget Sound and along the Strait of Juan de Fuca, these "hot spots" are also strongly associated with a low human footprint in the marine environment, for example, areas natural shorelines and relatively little vessel traffic (Raphael et al. 2016a, p. 106).

Non-breeding adults and subadults are thought to occur in similar areas as breeding adults. This species does occur farther offshore during the breeding season, but in much reduced numbers

(Drew and Piatt 2020; Strachan et al. 1995, p. 247). Their offshore occurrence is probably related to current upwelling and plumes during certain times of the year that tend to concentrate their prey species. Even within the breeding season, individual murrelets may make large movements, and large average marine home ranges (505 km^2 and 708 km^2, respectively) have been reported for northern California and Washington (Hébert and Golightly 2008, p. 99; Lorenz et al. 2017, p. 318).

Non-breeding Season Distribution

Marbled murrelet marine habitat use during the non-breeding season is poorly documented, but they are present near breeding sites year-round in most areas (Nelson 1997, p. 3). Murrelets exhibit seasonal redistributions following the pre-basic molt (Peery et al. 2008a, p. 119), and can move up to 750 km from their breeding season locations (Hébert and Golightly 2008, p. 101; Adrean et al. 2018). The southern end of the range extends as far south as the Southern California Bight; but some individuals also move northward at the end of the breeding season (Hall et al. 2009, p. 5081; Peery et al. 2008a, p. 121). Generally they are more dispersed and may be found farther offshore than during the breeding season, up to approximately 50 miles from shore (Adams et al. 2014; Ballance 2015, in litt.; Drew and Piatt 2020; Pearson 2019, p. 5; Speich and Wahl 1995, p. 322).

The highest concentrations likely still occur close to shore and in protected waters, but given the limited data available regarding non-breeding season murrelet distribution or densities, a great deal of uncertainty remains (Nelson 1997, p. 3; Pearson 2019, p. 5). More information is available regarding non-breeding season murrelet density and distribution in some areas of Puget Sound. Murrelets move from the outer exposed coasts of Vancouver Island and the Straits of Juan de Fuca into the sheltered and productive waters of northern and eastern Puget Sound (Beauchamp et al. 1999, entire; Burger 1995, p. 297; Speich and Wahl 1995, p. 325). However, in central and southern Puget Sound, murrelet densities are lower during the non-breeding season than they are during the breeding season (McIver et al. 2021, pp. 11-17; Pearson and Lance 2020, p. 12). Known areas of winter concentration include and southern and eastern end of Strait of Juan de Fuca (primarily Sequim, Discovery, and Chuckanut Bays), San Juan Islands and Puget Sound, Washington (Speich and Wahl 1995, p. 314).

Foraging and Diet

Murrelets dive and swim through the water by using their wings in pursuit of their prey; their foraging and diving behavior is restricted by physiology. They usually feed in shallow, nearshore water less than 30 m (98 ft) deep, which seems to provide them with optimal foraging conditions for their generalized diet of small schooling fish and large, pelagic invertebrates: Pacific sand lance (*Ammodytes personatus*), northern anchovy (*Engraulis mordax*), Pacific herring (*Clupea harengus*), surf smelt (*Hypomesus* sp.), euphausiids, mysids, amphipods, and other species (Nelson 1997, p. 7). However, they are assumed to be capable of diving to a depth of 47 m (157 ft) based on their body size and diving depths observed for other Alcid species (Mathews and Burger 1998, p. 71).

Contemporary studies of murrelet diets in the Puget Sound–Georgia Basin region indicate that Pacific sand lance now make up the majority of the murrelet diet (Gutowsky et al. 2009, p. 251). Historically, energy-rich fishes such as herring and northern anchovy comprised the majority of the murrelet diet (Becker and Beissinger 2006, p. 470; Gutowsky et al. 2009, p. 247). This is significant because sand lance have the lowest energetic value of the fishes that murrelets commonly consume. For example, a single northern anchovy has nearly six times the energetic value of a sand lance of the same size (Gutowsky et al. 2009, p. 251), so a murrelet would have to eat six sand lance to get the equivalent energy of a single anchovy. Reductions in the abundance of energy-rich forage fish species is likely a contributing factor in the poor reproduction in murrelets (Becker and Beissinger 2006, p. 470).

The duration of dives appears to depend upon age (adults vs. juveniles), water depth, visibility, and depth and availability of prey. Dive duration has been observed ranging from 8 seconds to 115 seconds, although most dives are between 25 to 45 seconds (Day and Nigro 2000; Jodice and Collopy 1999; Thoresen 1989; Watanuki and Burger 1999). Diving bouts last over a period of 27 to 33 minutes (Nelson 1997, p. 9). They forage in deeper waters when upwelling, tidal rips, and daily activity of prey concentrate prey near the surface (Strachan et al. 1995). Murrelets are highly mobile and some make substantial changes in their foraging sites within the breeding season. For example, Becker and Beissinger (2003, p. 243) found that murrelets in California responded rapidly (within days or weeks) to small-scale variability in upwelling intensity and prey availability by shifting their foraging behavior and habitat selection within a 100-km (62-mile) area. In Washington, changes in water temperature, likely also related to prey availability, influence foraging habitat use, but the influence of upwelling is less clear (Lorenz et al. 2017, pp. 315, 318).

For more information on murrelet use of marine habitats, see literature reviews in McShane et al. 2004, USFWS 2009, and USFWS 2019.

Murrelets in the Terrestrial Environment

Murrelets are dependent upon older-age forests, or forests with an older tree component, for nesting habitat (Hamer and Nelson 1995, p. 69). Specifically, murrelets prefer high and broad platforms for landing and take-off, and surfaces which will support a nest cup (Hamer and Nelson 1995, pp. 78-79). In Washington, murrelet nests have been found in live conifers, specifically, western hemlock (*Tsuga heterophylla*), Sitka spruce (*Picea sitchensis*), Douglas-fir (*Pseudotsuga menziesii*), and western red cedar (*Thuja plicata*) (Hamer and Nelson 1995; Hamer and Meekins 1999). Most murrelets appear to nest within 37 miles of the coast, although occupied behaviors have been recorded up to 52 miles inland, and murrelet presence has been detected up to 70 miles inland in Washington (Huff et al. 2006, p. 10). Nests occur primarily in large, older-aged trees. Overall, nests have been found in trees greater than 19 inches in diameter-at-breast and greater than 98 ft tall. Nesting platforms include limbs or other branch deformities that are greater than 4 inches in diameter, and are at greater than 33 ft above the ground. Substrate such as moss or needles on the nest platform is important for protecting the egg and preventing it from falling off (Huff et al. 2006, p. 13).

Murrelets do not form the dense colonies that are typical of most other seabird species. Limited evidence suggests they may form loose colonies in some cases (Ralph et al. 1995). The reliance of murrelets on cryptic coloration to avoid detection suggests they utilize a wide spacing of nests in order to prevent predators from forming a search image (Ralph et al. 1995). Individual murrelets are suspected to have fidelity to nest sites or nesting areas, although this is has only been confirmed with marked birds in a few cases (Huff et al. 2006, p. 11). There are at least 15 records of murrelets using nest sites in the same or adjacent trees in successive years, but it is not clear if they were used by the same birds (McShane et al. 2004, p. 2-14). At the landscape scale, murrelets are probably faithful to specific watersheds for nesting (McShane et al. 2004, p. 2-14). Murrelets have been observed visiting nesting habitat during non-breeding periods in Washington, Oregon, and California which may indicate adults are maintaining fidelity and familiarity with nesting sites and/or stands (Naslund 1993; O'Donnell et al. 1995, p. 125).

Loss of nesting habitat reduces nest site availability and displaces any murrelets that may have had nesting fidelity to the logged area (Raphael et al. 2002, p. 232). Murrelets have demonstrated fidelity to nesting stands and in some areas, fidelity to individual nest trees (Burger et al. 2009, p. 217). Murrelets returning to recently logged areas may not breed for several years or until they have found suitable nesting habitat elsewhere (Raphael et al. 2002, p. 232). The potential effects of displacement due to habitat loss include nest site abandonment, delayed breeding, failure to initiate breeding in subsequent years, and failed breeding due to increased predation risk at a marginal nesting location (Divoky and Horton 1995, p. 83; Raphael et al. 2002, p. 232). Each of these outcomes has the potential to reduce the nesting success for individual breeding pairs, and could ultimately result in the reduced recruitment of juvenile birds into the local population (Raphael et al. 2002, pp. 231-233).

Detailed information regarding the life history and conservation needs of the murrelet are presented in the *Ecology and Conservation of the Marbled Murrelet* (Ralph et al. 1995), the Service's 1997 *Recovery Plan for the Marbled Murrelet* (USFWS 1997), and in subsequent 5-year status reviews (McShane et al. 2004; USFWS 2009; USFWS 2019).

Terrestrial Distribution

Murrelets are distributed along the Pacific coast of North America, with birds breeding from central California through Oregon, Washington, British Columbia, southern Alaska, westward through the Aleutian Island chain, with presumed breeding as far north as Bristol Bay (Nelson 1997, p. 2), and non-breeding distribution extending as far south as the Southern California Bight (Hall et al. 2009, p. 5081). The federally-listed murrelet population in Washington, Oregon, and California is classified by the Service as a distinct population segment (75 FR 3424). The coterminous United States population of murrelets is considered significant as the loss of this distinct population segment would result in a significant gap in the range of the taxon and the loss of unique genetic characteristics that are significant to the taxon (75 FR 3430).

The inland nesting distribution of murrelets is strongly associated with the presence of mature and old-growth conifer forests. Murrelets have been detected farther than100 km inland in Washington (70 miles).The inland distribution in the southern portion of the species range is associated with the extent of the hemlock/tanoak vegetation zone which occurs up to 16-51 km inland (10-32 miles) (Evans Mack et al. 2003, p. 4). Although murrelets are distributed throughout their historical range, the area of occupancy within their historic range appears to be reduced from historic levels. The distribution of the species also exhibits five areas of discontinuity: a segment of the border region between British Columbia, Canada and Washington; southern Puget Sound, WA; Destruction Island, WA to Tillamook Head, OR; Humboldt County, CA to Half Moon Bay, CA; and the entire southern end of the breeding range in the vicinity of Santa Cruz and Monterey Counties, CA (McShane et al. 2004, p. 3-70).

Murrelets use inland habitats primarily for nesting, including egg laying, incubation, and feeding of nestlings. In addition, murrelets have been observed in nesting habitat demonstrating social behaviors, such as circling and vocalizing, in groups of up to ten birds (Nelson and Peck 1995, p. 51). Nest sites tend to be clustered spatially, indicating that although murrelets are not colonial seabirds, they also are not strictly solitary in their nesting behavior; in other words, at least in some circumstances, they nest semi-colonially (Conroy et al. 2002, p. 131; Naslund et al. 1995, p. 12). In California and southern Oregon, murrelets occupy habitat more frequently when there is other occupied habitat within 5 km (Meyer et al. 2002, p. 103), and we assume that the same is true in Washington. Usually, multiple nests can be found in a contiguous forested area, even in places where they are not strongly clustered (Evans Mack et al. 2003, p. 6). In previously unoccupied nesting habitat in Oregon, murrelets were much more likely to display behaviors associated with occupancy in places where recordings of murrelet calls had been broadcast the previous year, compared with control sites where no recordings were played (Valente et al. 2021, p. 7). This indicates that murrelets select nesting habitat in part based on the apparent presence of conspecifics.

Distribution of Nesting Habitat

The loss of nesting habitat was a major cause of the murrelet's decline over the past century and may still be contributing as nesting habitat continues to be lost to fires, logging, insects, tree diseases, and wind storms (Miller et al. 2012, p. 778; Raphael et al. 2016b, pp. 80-81). Due mostly to historical timber harvest, only a small percentage (~11 percent) of the habitat-capable lands within the listed range of the murrelet currently contain potential nesting habitat (Raphael et al. 2016b, p. 69).

Monitoring of murrelet nesting habitat within the Northwest Forest Plan (NWFP, equivalent to Conservation Zones 1 through 5) area indicates nesting habitat declined from an estimated 2.53 million acres in 1993 to an estimated 2.23 million acres in 2012, a decline of about 12.1 percent (Raphael et al. 2016b, p. 72). Fire has been the major cause of nesting habitat loss on Federal lands, while timber harvest is the primary cause of loss on non-Federal lands (Raphael et al. 2016b, p. 79). While most (60 percent) of the potential habitat is located on Federal reserved-land allocations, a substantial amount of nesting habitat occurs on non-federal lands (34 percent) (Table 1).

In Zone 6, monitoring of nesting habitat has not been carried out in the same way as within the NWFP area. Most of the existing nesting habitat within Zone 6 is located on state and local public lands, where logging has not occurred (Halbert and Singer 2017, p. 1). During August of 2020, over 60 percent of the nesting habitat in Zone 6 burned in a large wildfire (Singer 2021, in litt.). Preliminary data indicate that this fire has resulted in substantial habitat loss, though some lost habitat features may recover over the next several years. Many trees within the burned areas survived the fire, including the “Father of the Forest” redwood where murrelet nesting has been documented repeatedly (California Department of Parks and Recreation 2020, p. 2; Halbert and Singer 2017, p. 35); however, suitable platforms likely burned even in trees that survived the fire, leading to a loss of suitability for many years as branches regrow (Singer 2020, in litt.). In a sample of 40 previously-identified potential nest trees within Big Basin State Park, 22 trees (55 percent) appeared to have survived the fire (Singer 2021, in litt.). If this sample is representative, more than one quarter (i.e. 45 percent x 60 percent) of potential murrelet nest trees in Zone 6 may have been killed by the fire, with platform structures lost from a substantial percentage of the remaining trees. Future monitoring will be necessary to refine these estimates of habitat loss.

Table 1. Estimates of higher-quality murrelet nesting habitat by State and major land ownership within the area of the NWFP – derived from 2012 data.

State	Habitat capable lands (1,000s of acres)	Habitat on Federal reserved lands (1,000s of acres)	Habitat on Federal non-reserved lands (1,000s of acres)	Habitat on non-federal lands (1,000s of acres)	Total potential nesting habitat (all lands) (1,000s of acres)	Percent of habitat capable land that is currently in habitat
WA	10,851.1	822.4	64.7	456	1,343.1	12 %
OR	6,610.4	484.5	69.2	221.1	774.8	12 %
CA	3,250.1	24.5	1.5	82.9	108.9	3 %
Totals	20,711.6	1,331.4	135.4	760	2,226.8	11 %
Percent		60 %	6 %	34 %	100 %	-

Source: (Raphael et al. 2016b, pp. 78-81).

Population Status

The 1997 *Recovery Plan for the Marbled Murrelet* (USFWS 1997) identified six Conservation Zones throughout the listed range of the species: Puget Sound (Conservation Zone 1), Western Washington Coast Range (Conservation Zone 2), Oregon Coast Range (Conservation Zone 3), Siskiyou Coast Range (Conservation Zone 4), Mendocino (Conservation Zone 5), and Santa Cruz Mountains (Conservation Zone 6) (Figure 1). Conservation Zones are the functional equivalent of recovery units as defined by Service policy (USFWS 1997, p. 115). The subpopulations in each Zone are not discrete. There is some movement of murrelets between Zones, as indicated by radio-telemetry studies (e.g., Bloxton and Raphael 2006, p. 162), but the degree to which murrelets migrate between Zones is unknown. Genetic studies also indicate that there is movement of murrelets between Zones, although Zone 6 is more isolated genetically than the other Zones (Friesen et al. 2005, pp. 611-612; Hall et al. 2009, p. 5080; Peery et al. 2008b, pp. 2757-2758; Peery et al. 2010, p. 703; Vásquez-Carrillo et al. 2014, pp. 251-252). For the purposes of consultation, the Service treats each of the Conservation Zones as separate sub-populations of the listed murrelet population.

Population Status and Trends

Population estimates for the murrelet are derived from marine surveys conducted during the nesting season as part of the NWFP effectiveness monitoring program. Surveys from 2001 to 2018 indicated that the murrelet population in Conservation Zones 1 through 5 (NWFP area) increased at a rate of 0.5 percent per year (McIver et al. 2021, p. 4). While the trend estimate across this period is slightly positive, the confidence intervals are tight around zero (95% confidence interval [CI]: -0.5 to 1.5 percent), indicating that at the scale of the NWFP area, the population is changing very little (McIver et al. 2021, p. 4) (Table 2). At the state scale, Washington exhibited a significant declining trend between 2001 and 2018 (3.9% decrease per year, while Oregon and California showed significant positive trends (OR = 2.2% increase per year; CA = 4.6% increase per year (McIver et al. 2021, p. 4) (Table 2). Zone 1 shows the greatest decline of 5.0 percent per year, while the decline in Zone 2 is smaller, 2.2 percent per year, and less statistically certain (Table 2). Zone 4 shows the greatest increase of 3.5 percent per year, while Zone 3 shows a smaller, and less statistically certain, increase of 1.5 percent per year (Table 2). There is great uncertainty regarding the trend in Zone 5 due to the infrequency of surveys in that zone and the influence of a single anomalous year in 2017 (McIver et al., in press, p. 37). No trend estimate is available for Zone 6.

While the direct causes for population declines in Washington are unknown, potential factors include the loss of nesting habitat, including cumulative and time-lag effects of habitat losses over the past 20 years (an individual murrelets potential lifespan), changes in the marine environment reducing the availability or quality of prey, increased densities of nest predators, and emigration (Miller et al. 2012, p. 778). As with nesting habitat loss, marine habitat degradation is most prevalent in the Puget Sound area, where anthropogenic activities (e.g., shipping lanes, boat traffic, shoreline development) are an important factor influencing the marine distribution and abundance of murrelets in Conservation Zone 1 (Falxa and Raphael 2016, p. 110).

The most recent population estimate for the entire Northwest Forest Plan area in 2019 was 21,200 murrelets (95 percent confidence interval [CI]: 16,400 to 26,000 birds) (McIver et. al 2021, p. 10). The largest and most stable murrelet subpopulations now occur off the Oregon and northern California coasts, while subpopulations in Washington have experienced the greatest rates of decline. Murrelet zones are now surveyed on an every other-year basis, so the last year that an extrapolated range-wide estimate for all zones combined is 2018 (Table 2).

The murrelet subpopulation in Conservation Zone 6 (central California- Santa Cruz Mountains) is outside of the NWFP area and is monitored separately by California State Parks and the U.S. Geological Survey using similar at-sea survey methods (Felis et al. 2020, p. 1). Surveys in Zone 6 indicate a small population of murrelets with no clear trends. Population estimates from 2001 to 2018 have fluctuated from a high of 699 murrelets in 2003, to a low of 174 murrelets in 2008 (Felis et al. 2020 p. 7). In 2019, surveys indicated an estimated population of 404 murrelets in Zone 6 (95% CI: 272-601) (Felis et al. 2020, p. 7) (Table 3).

Table 2. Summary of murrelet population estimates and trends (2001-2019/2020) at the scale of Conservation Zones and states.

Zone	**Year**	**Estimated number of murrelets**	**95% CI Lower**	**95% CI Upper**	**Average density (at sea) (murrelets /km$^{2)}$**	**Average annual rate of population change (%)**	**95% CI Lower**	**95% CI Upper**
1	2020	3,143	2,030	4,585	0.899	-5.0	-7.0	-2.9
2	2019	1,657	745	2,752	1.004	-2.2	-5.7	+1.5
3	2020	8,359	5,569	11,323	5.239	+1.5	+0.02	+3.1
4	2019	6,822	5,576	11,063	5.885	+3.5	+1.6	+5.5
5	2017	868	457	1,768	0.983	+7.2	-4.4	+20.3
Zones 1-5	2019	21,230	16,446	26,015	2.417	+0.5	-0.5	+1.5
Zone 6	2019	404	272	601	na	na	na	na
WA	2019	5,151	2,958	7,344	1.00	-3.9	-5.4	-2.4
OR	2019	10,339	7,070	13,607	4.99	+2.2	+0.9	+3.4
CA Zones 4 & 5	2019	5,741	3,894	7,588	3.67	+4.6	+2.7	+6.5

Sources: (McIver et al. 2021, pp. 16-20, Felis et al. 2020, p. 7).

Factors Influencing Population Trends

Population monitoring data show murrelet populations declining in Washington, but increasing in Oregon and northern California (McIver et al. 2021, p. 4). Murrelet population size and distribution is strongly and positively correlated with the amount and pattern (large contiguous patches) of suitable nesting habitat, and population trend is most strongly correlated with trend in nesting habitat, although marine factors also contribute to this trend (Raphael et al. 2016a, p. 115). From 1993 to 2012, there was a net loss of about 2 percent of potential nesting habitat from on federal lands, compared to a net loss of about 27 percent on nonfederal lands, for a total cumulative net loss of about 12.1 percent across the NWFP area (Raphael et al. 2016b, p. 72). Cumulative habitat losses since 1993 have been greatest in Washington, with most habitat loss in Washington occurring on non-Federal lands due to timber harvest (Raphael et al. 2016b, pp. 80-81) (Table 3).

Table 3. Distribution of higher-suitability murrelet nesting habitat by Conservation Zone, and summary of net habitat changes from 1993 to 2012 within the NWFP area.

Conservation Zone	**1993**	**2012**	**Change (acres)**	**Change (percent)**
Zone 1 - Puget Sound/Strait of Juan de Fuca	829,525	739,407	-90,118	-10.9 %
Zone 2 - Washington Coast	719,414	603,777	-115,638	-16.1 %
Zone 3 - Northern to central Oregon	662,767	610,583	-52,184	-7.9 %
Zone 4 - Southern Oregon - northern California	309,072	256,636	-52,436	-17 %
Zone 5 - north-central California	14,060	16,479	+2,419	+17.2 %

Source: (Raphael et al. 2016b, pp. 80-81).

The decline in murrelet populations from 2001 to 2013 is weakly correlated with the decline in nesting habitat, with the greatest declines in Washington, and the smallest declines in California, indicating that when nesting habitat decreases, murrelet abundance in adjacent marine waters may also decrease. At the scale of Conservation Zones, the strongest correlation between habitat loss and murrelet decline is in Zone 2, where murrelet habitat has declined most steeply and murrelet populations have also continued to decline. However, these relationships are not linear, and there is much unexplained variation (Raphael et al. 2016a, p. 110). While terrestrial habitat amount and configuration (i.e., fragmentation) and the terrestrial human footprint (i.e., cities, roads, development) appear to be strong factors influencing murrelet distribution in Zones 2-5; terrestrial habitat and the marine human footprint (i.e., shipping lanes, boat traffic, shoreline development) appear to be the most important factors that influence the marine distribution and abundance of murrelets in Zone 1 (Raphael et al. 2016a, p. 106).

Like other marine birds, murrelets depend for their survival on their ability to successfully forage in the marine environment. Despite this, it is apparent that the location, amount, and landscape pattern of terrestrial nesting habitat are strongest predictors of the spatial and temporal distributions of murrelets at sea during the nesting season (Raphael et al. 2015, p. 20). Outside of Zone 1, various marine habitat features (e.g., shoreline type, depth, temperature, human footprint, etc.) apparently have only a minor influence on murrelet distribution at sea. Despite this relatively weak spatial relationship, marine factors, and especially any decrease in forage species, likely play an important role in explaining the apparent population declines, but the ability to detect or model these relationships is currently limited (Raphael et al. 2015, p. 20). Over both the long and short term, there is evidence that diet quality is related to marbled murrelet abundance, the likelihood of nesting attempts, reproductive success (Becker et al. 2007, p. 276; Betts et al. 2020, pp. 6-7; Norris et al. 2007, p. 881).

The interplay between marine and terrestrial habitat conditions also influences murrelet population dynamics. A recent analysis indicates that in Oregon, over a 20-year period, nesting activity was most likely to occur following years with cool ocean temperatures (indicating good forage availability), and at sites where large blocks of mature forest were close to the coast (Betts et al. 2020, pp. 5-9). Even when ocean conditions were poor, nesting murrelets colonized new sites that were surrounded by abundant old forest, but during good ocean conditions, even sites with less old forest could be colonized (Betts et al. 2020, p. 6). This relationship has not been investigated in other parts of the range, but is consistent with observations in Washington, where murrelets occupy nesting habitat at lower rates, often fly long distances to reach foraging areas, breed at very low observed rates, and the population continues to decline (Lorenz et al. 2017, pp. 312-313, 318; McIver et al. 2021, p. 20).

Population Models

Prior to the use of survey data to estimate trend, demographic models were more heavily relied upon to generate predictions of trends and extinction probabilities for the murrelet population (Beissinger 1995; Cam et al. 2003; McShane et al. 2004; USFWS 1997). However, murrelet population models remain useful because they provide insights into the demographic parameters and environmental factors that govern population stability and future extinction risk, including stochastic factors that may alter survival, reproductive, and immigration/emigration rates.

In a report developed for the *5-year Status Review of the Marbled Murrelet in Washington, Oregon, and California* (McShane et al. 2004, pp. 3-27 to 3-60), models were used to forecast 40-year murrelet population trends. A series of female-only, multi-aged, discrete-time stochastic Leslie Matrix population models were developed for each conservation zone to forecast decadal population trends over a 40-year period with extinction probabilities beyond 40 years (to 2100). The authors incorporated available demographic parameters (Table 4) for each conservation zone to describe population trends and evaluate extinction probabilities (McShane et al. 2004, p. 3-49).

McShane et al. (2004) used mark-recapture studies conducted in British Columbia by Cam et al. (2003) and Bradley et al. (2004) to estimate annual adult survival and telemetry studies or at-sea survey data to estimate fecundity. Model outputs predicted -3.1 to -4.6 percent mean annual rates of population change (decline) per decade the first 20 years of model simulations in murrelet Conservation Zones 1 through 5 (McShane et al. 2004, p. 3-52). Simulations for all zone populations predicted declines during the 20 to 40-year forecast, with mean annual rates of -2.1 to -6.2 percent, depending on Zone and decade (McShane et al. 2004, p. 3-52). While these modeled rates of decline are similar to those observed in Washington (McIver et al. 2021, p. 20), the simulated projections at the scale of Zones 1-5 do not match the apparently increasing populations observed in Oregon and California during the 2001-2019 monitoring period. Comparable trend information is not available for Zone 6 in central California.

Table 4. Rangewide murrelet demographic parameter values based on four studies all using Leslie Matrix models.

Demographic Parameter	Beissinger 1995	Beissinger and Nur 1997*	Beissinger and Peery (2007)	McShane et al. 2004
Juvenile Ratio (Ŕ)	0.10367	0.124 or 0.131	0.089	0.02 - 0.09
Annual Fecundity	0.11848	0.124 or 0.131	0.06-0.12	-
Nest Success	-	-	0.16-0.43	0.38 - 0.54
Maturation	3	3	3	2 - 5
Estimated Adult Survivorship	85 % – 90%	85 % – 88 %	82 % - 90 %	83 % – 92 %

*In U.S. Fish and Wildlife (1997).

Reproduction

Overall fecundity is a product of the proportion of murrelets that attempt nesting and the proportion of nest attempts that succeed. Telemetry studies can be used to estimate both the proportion of murrelets attempting nesting, and the proportion of nest attempts that succeed. When telemetry estimates are not available, at-sea surveys that separately count the number of hatch-year and after-hatch-year birds can be used to estimate productivity. Telemetry estimates are typically preferred over marine counts for estimating breeding success due to fewer biases (McShane et al. 2004, p. 3-2). However, because of the challenges of conducting telemetry studies, estimating murrelet reproductive rates with an index of reproduction, referred to as the juvenile ratio (**Ŕ**),[1] continues to be important, despite some debate over use of this index (see discussion in Beissinger and Peery 2007, p. 296).

Murrelet fecundity is likely limited in part by low rates of nesting attempts in some parts of the range. Radio-telemetry monitoring Washington between 2004 and 2008 indicated only a small portion of 158 tagged adult birds actually attempted to nest (13 to 20 percent) (Lorenz et al. 2017, p. 316; Raphael and Bloxton 2009, p. 165). Studies from California and Oregon also report low rates. Two studies from central and northern California reported that an average of around 30 percent of radio-tagged murrelets attempted to nest (Hébert and Golightly 2006, p. 130; Peery et al. 2004, p. 1093). In preliminary results from a study in Oregon, only 11 out of 203 murrelets (5 percent) tagged between 2017 and 2019, attempted to nest (Adrean 2021, pers. comm.). This represents the lowest rate yet reported for the species; however, the study is not yet complete and is therefore not fully comparable to the others cited above. These low rates of nesting are not intrinsic to the species; other studies outside of the listed range reported that between 46 and 80 percent of murrelets attempted to breed each year (Barbaree et al. 2014, p. 177; Bradley et al. 2004, p. 323), and most population modeling studies suggest a range of 80 to 95 percent of adults breed each year (McShane et al. 2004, p. 3-5). The process of radio-tagging or the additional weight and drag of the radio tag itself may reduce the probability that a tagged

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[1] The juvenile ratio (**Ŕ**) for murrelets is derived from the relative abundance of hatch-year (HY; 0-1 yr-old) to after-hatch-year (AHY; 1+ yr-old) birds (Beissinger and Peery 2007, p. 297) and is calculated from marine survey data. All ratios presented here are date-corrected using the methods of Peery et al. (2007, p. 234) to account adults incubating and chicks not yet fledged at the time of the survey.

individual will attempt to breed, but studies reporting higher rates of attempted nesting used similar radio tags, so radio-telemetry methods do not account for differences between the studies conducted in the listed range and those conducted elsewhere (Peery et al. 2004, p. 1094).

Although difficult to obtain, nest success rates[2] are available from telemetry studies conducted in California (Hébert and Golightly 2006; Peery et al. 2004, p. 1094), Washington (Lorenz et al. 2017, p. 312; Lorenz et al. 2019, p. 160), and, preliminarily, in Oregon (Adrean et al. 2019, p. 2). In northwestern Washington, Lorenz and others (2017, p. 312; 2019, pp. 159-160) documented a nest success rate of 0.20 (3 chicks fledging from 15 nest starts). In central California, murrelet nest success is 0.16 (Peery et al. 2004, p. 1098) and in northern California it ranges from 0.069 to 0.243 (Hébert and Golightly 2006, p. 129). In Oregon, preliminary results from a telemetry study indicate that 3 of 7 active nests successfully fledged young, a rate of 0.43, but this success rate may not be comparable to the others reported above; for example, it is not clear whether it includes all nesting attempts (Adrean et al. 2019, p. 2).

At least one telemetry study reported overall fecundity rates, combining both the rates of nesting attempts with the rates of fledging success. In central California, the fecundity rate was estimated to be 0.027, or 2.7 female chicks produced per year for every 100 females of breeding age (Peery et al. 2004, p. 1094). In other studies, the overall fecundity rate is not known, because it is not clear how many of the radio-tagged birds were of breeding age. However, in northern California, of 102 radio-tagged birds, at least two and at most six successfully produced fledglings (Hébert and Golightly 2006, pp. 130-131), and in Washington and southern Vancouver Island, of 157 radio-tagged birds, four produced fledglings (Lorenz et al. 2017, p. 312). If we assume (as in Peery et al. 2004, p. 1094) that 93 percent of captured birds in each sample were of breeding age, and that half of all captured birds and half of all fledged chicks were female, fecundity rates from these samples would be 0.027 in Washington, and between 0.021 and 0.063 in northern California.

Unadjusted and adjusted values for estimates of murrelet juvenile ratios also suggest low reproductive rates. In northern California and Oregon, annual estimates for **Ŕ** range from 0 to 0.140, depending on the area surveyed (Strong 2014, p. 20; Strong 2015, p. 6; Strong 2016, p. 7; Strong 2017, p. 6; Strong 2018, p. 7; Strong 2019, p. 6; Strong and Falxa 2012, p. 4). In Conservation Zone 4, the annual average between 2000 and 2011 was 0.046 (Strong and Falxa 2012, p. 11). In central California, estimates of **Ŕ** range from 0 to 0.12, with an annual average of 0.048, over 21 years of survey between 1996 and 2019 (Felis et al. 2020, p. 9). An independent calculation of **Ŕ** among murrelets captured in central California between 1999 and 2003 resulted in estimates ranging from 0 to 0.111, with an average of 0.037 (Peery et al. 2007, p. 235). Estimates for **Ŕ** in the San Juan Islands in Washington tend to be higher, ranging from 0.02 to 0.12, with an average of 0.067, over 18 years of survey between 1995 and 2012 (Lorenz and Raphael 2018, pp. 206, 211). Notably, **Ŕ** in the San Juan Islands did not show any temporal trend over the 18-year period, even while the abundance of adult and subadult murrelets declined (Lorenz and Raphael 2018, pp. 210-211).

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[2] Nest success here is defined by the annual number of known hatchlings departing from the nest (fledging) divided by the number of nest starts.

Although these estimates of **Ŕ** are higher than one would expect based on fecundity rates derived from radio-telemetry studies, they are below the level thought to be necessary to maintain or increase the murrelet population. Demographic modeling, historical records, and comparisons with similar species all suggest that murrelet population stability requires juvenile ratios between 0.176 and 0.3 (Beissinger and Peery 2007, p. 302; USFWS 1997, p. B-13). Even the lower end of this range is higher than any current estimate for **Ŕ** for any of the Conservation Zones. This indicates that the murrelet reproductive rate is likely insufficient to maintain stable population numbers throughout all or portions of the species' listed range. These sustained low reproductive rates appear to be at odds with the potentially stable population size measured for Zones 1 through 5, and are especially confusing in light of apparent population increases in Oregon and California.

Integration and Summary: Murrelet Abundance, Distribution, Trend, and Reproduction

A statistically significant decline was detected in Conservation Zones 1 and 2 for the 2001-2019 period (Table 2). The overall population trend from the combined 2001-2019 population estimates (Conservation Zones 1 - 5) indicates a potentially stable population with a 0.5 percent increase per year (McIver et al. 2021, p. 4). Because the confidence intervals for this estimate are fairly tight around 0, there is not clear evidence of either or a positive or negative trend. At the state-scale, significant declines have occurred in Washington, while subpopulations in Oregon and California show a statistically meaningful increase (McIver et al. 2021, p. 4).

The current ranges of estimates for fecundity and for **Ŕ**, the juvenile to adult ratio, are below the level assumed to be necessary to maintain or increase the murrelet population. Whether derived from radio-telemetry, marine surveys or from population modeling (**Ŕ** = 0.02 to 0.13, Table 4), the available information is in general agreement that the current ratio of hatch-year birds to after-hatch year birds is insufficient to maintain stable numbers of murrelets throughout the listed range. The current estimates for **Ŕ** also appear to be well below what may have occurred prior to the murrelet population decline (Beissinger and Peery 2007, p. 298).

The reported stability of the population at the larger scale (Zones 1 through 5) and growth of subpopulations in Oregon and California appear to be at odds with the sustained low reproductive rates reported throughout the listed range. A number of factors could contribute to this discrepancy. For example, population increases could be caused by an influx of murrelets moving from the Canadian population into Oregon and California, or into Washington and displacing Washington birds to Oregon and California. The possibility of a population shift from Washington to Canada has previously been dismissed, based on nest-site fidelity and the fact that both Washington and British Columbia populations are declining simultaneously (Falxa et al. 2016, p. 30), but these arguments do not rule out the possibility that non-breeding murrelets originating in Canada may be spending time foraging in Oregon or California waters.

Another possibility is the proportion of birds present on the water during surveys, rather than inland at nest sites, may be increasing. If so, this would artificially inflate population estimates. Such a shift could be driven by low nesting rates, as were observed in Oregon in 2017 (Adrean et al. 2018, p. 2; Horton et al. 2017, p. 77); or by shifts toward earlier breeding, for which there is anecdotal evidence (for example, Havron 2012, p. 4; Pearson 2018, in litt.; Strong 2019, p. 6); or a combination of both factors. In either case, individuals that would in earlier years have been incubating an egg or flying inland to feed young, and therefore unavailable to be counted, would now be present at sea and would be observed during surveys. For the same number of birds in the population, the population estimate would increase as adults spend more of the survey period at sea.

Finally, the shift that occurred in 2015 to sampling only half of the Conservation Zones in each survey year (McIver et al. 2021, pp. 5-6) is increasing the uncertainty in how to interpret the survey results, especially in light of large-scale movements that can occur during the breeding season, sometimes involving numerous individuals (Horton et al. 2018, p. 77; Peery et al. 2008a, p. 116). Murrelets that move into or out of the zone being sampled during the breeding season could artificially inflate or deflate the population estimates. Even interannual movements among the Zones could temporarily resemble population growth, without an actual increase in the number of birds in the population (McIver et al., in press, pp. 14, 43).

Some of these factors would also affect measures of fecundity and juvenile ratios. For example, if murrelets are breeding earlier on average, then the date adjustments applied to juvenile ratios may be incorrect, possibly resulting in inflated estimates of **Ŕ**. If current estimates of **Ŕ** are biased high, this would mean that the true estimates of **Ŕ** are even lower, exacerbating, rather than explaining, the discrepancy between the apparently sustained low reproductive rates and the apparently stable or increasing subpopulations south of Washington. A shift toward later breeding could result in more adults being present at sea during surveys, and would also result in artificially low estimates of **Ŕ**. We are not aware of evidence for a widespread shift toward later breeding, but this kind of alteration in seasonal behavior may be more difficult to detect than a shift to earlier breeding. Early-fledging juveniles are conspicuous when observed at sea, whereas late-fledging juveniles are not.

Considering the best available data on abundance, distribution, population trend, and the low reproductive success of the species, the Service concludes the murrelet population within the Washington portion of its listed range currently has little or no capability to self-regulate, as indicated by the significant, annual decline in abundance the species is currently undergoing in Conservation Zones 1 and 2. Populations in Oregon and California are apparently more stable, but reproductive rates remain low in those areas, and threats associated with habitat loss and habitat fragmentation continue to occur. The Service expects the species to continue to exhibit further reductions in distribution and abundance , due largely to the expectation that the variety of environmental stressors present in the marine and terrestrial environments (discussed in the *Threats to Murrelet Survival and Recovery* section) will continue into the foreseeable future.

Threats to Murrelet Survival and Recovery

When the murrelet was listed under the Endangered Species Act in 1992, several anthropogenic threats were identified as having caused the dramatic decline in the species:

- habitat destruction and modification in the terrestrial environment from timber harvest and human development caused a severe reduction in the amount of nesting habitat
- unnaturally high levels of predation resulting from forest “edge effects” ;

- the existing regulatory mechanisms, such as land management plans (in 1992), were considered inadequate to ensure protection of the remaining nesting habitat and reestablishment of future nesting habitat; and
- manmade factors such as mortality from oil spills and entanglement in fishing nets used in gill-net fisheries.

The regulatory mechanisms implemented since 1992 that affect land management in Washington, Oregon, and California (for example, the NWFP) and new gill-netting regulations in northern California and Washington have reduced the threats to murrelets (USFWS 2004, pp. 11-12). However, additional threats were identified, and more information was compiled regarding existing threats, in the Service's 5-year reviews for the murrelet compiled in 2009 and 2019 (USFWS 2009, pp. 27-67; USFWS 2019, pp. 19-65). These stressors are related to environmental factors affecting murrelets in the marine and terrestrial environments. These stressors include:

- Habitat destruction, modification, or curtailment of the marine environmental conditions necessary to support murrelets due to:
    - elevated levels of toxic contaminants, including polychlorinated biphenyls, polybrominated diphenyl ether, polycyclic aromatic hydrocarbons, and organochlorine pesticides, in murrelet prey species;
    - the presence of microplastics in murrelet prey species;
    - changes in prey abundance and availability;
    - changes in prey quality;
    - harmful algal blooms that produce biotoxins leading to domoic acid and paralytic shellfish poisoning that have caused murrelet mortality;
    - harmful algal blooms that produce a proteinaceous foam that has fouled the feathers of other alcid species, and affected areas of murrelet marine habitat;
    - hypoxic or anoxic events in murrelet marine habitat; and
    - climate change in the Pacific Northwest.
- Manmade factors that affect the continued existence of the species include:
    - derelict fishing gear leading to mortality from entanglement;
    - disturbance in the marine environment (from exposures to lethal and sub-lethal levels of high underwater sound pressures caused by pile-driving, underwater detonations, and potential disturbance from high vessel traffic); and
    - wind energy generation, currently limited to onshore projects, leading to mortality from collisions.

Since the time of listing, some murrelet subpopulations have continued to decline due to lack of successful reproduction and recruitment, and while other subpopulations appear to be stable or increasing, productivity in these populations remains lower than the levels likely to support sustained population stability. The murrelet Recovery Implementation Team identified five major mechanisms that appear to be contributing to poor demographic performance (USFWS 2012b, pp. 10-11):

- Ongoing and historic loss of nesting habitat.
- Predation on murrelet eggs and chicks in their nests.
- Changes in marine conditions, affecting the abundance, distribution, and quality of murrelet prey species.
- Post-fledging mortality (predation, gill-nets, oil-spills).
- Cumulative and interactive effects of factors on individuals and populations.

Climate Change

In the Pacific Northwest, climate change affects both the marine and forested environments on which murrelets depend. Changes in the terrestrial environment may have a direct effect on murrelet reproduction, and also affect the structure and availability of nesting habitat. Changes in the marine environment affect murrelet food resources. Changes in either location may affect the likelihood, success, and timing of murrelet breeding in any given year.

Changes in the Physical Environment

Projected changes to the climate within the range of the murrelet include air and sea surface temperature increases, changes in precipitation seasonality, and increases in the frequency and intensity of extreme rainfall events (Mauger et al. 2015, pp. 2-1 – 2-18; Mote and Salathé 2010, p. 29; Salathé et al. 2010, pp. 72-73). Air temperature warming is already underway, and is expected to continue, with the mid-21st century projected to be approximately four to six degrees Fahrenheit (°F) (2.2 to 3.3 degrees Celsius [°C]) warmer than the late 20th century (Mauger et al. 2015, p. 2-5; USGCRP 2017, pp. 196-197). Similarly, sea surface temperatures are already rising and the warming is expected to continue, with increases between 2.2 °F (1.2 °C) and 5.4 °F (3 °C) projected for Puget Sound, the Strait of Georgia, and the Pacific Coast between the late 20th century and mid-or late-21st century (Mote and Salathé 2010, p. 16; Riche et al. 2014, p. 41; USGCRP 2017, p. 368). Summer precipitation is expected to decrease, while winter precipitation is expected to increase (Mauger et al. 2015, p. 2-7; USGCRP 2017, p. 217). In particular, heavy rainfall events are projected to occur between two and three times as frequently and to be between 19 and 40 percent more intense, on average, in the late 21st century than they were during the late 20th century (Warner et al. 2015, pp. 123-124).

The warming trend and trends in rainfall may be masked by naturally-occurring climate cycles, such as the El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) (Reeder et al. 2013, p. 76). These oscillations have similar effects in the Pacific Northwest, with relatively warm coastal water and warm, dry winter conditions during a “positive” warm phase, followed by cooler coastal water and cooler, wetter winter conditions during the cool “negative” phase (Moore et al. 2008, p. 1747). They differ in that one phase of the ENSO cycle typically lasts between 6 and 18 months (one to three years for a full cycle), whereas, during the 20th century, each phase of the PDO cycle lasted approximately 20 to 30 years (approximately 40 to

60 years for a full cycle) (Mantua and Hare 2002, p. 36). Some studies break the PDO into two components, one with a full cycle length between 16 and 20 years and the other with a 50 to 70 year period, with the longer component referred to as the Pacific Multidecadal Oscillation (PMO) (Steinman et al. 2015, p. 988). Another recent study has identified a 60-year cycle separate from the longer-term component of the PDO, also referring to this as the PMO (Chen et al. 2016, p. 319). An additional pattern, the North Pacific Gyre Oscillation, is associated with changes in the alongshore winds that drive upwelling, and appears to complete approximately one cycle per decade (Di Lorenzo et al. 2008, pp. 2-3).

The overall warming projections described above for the listed range of the murrelet will be superimposed over the natural climate oscillations. The climate models used to project future trends account for naturally occurring cycles (IPCC 2014, p. 56). Therefore, the projected trend combined with the existing cycles mean that temperatures during a cool phase will be less cool than they would be without climate change, and warm phases will be warmer. During the winter of 2014-2015, the climate shifted from a negative cool phase of the PDO to a positive warm phase (Peterson et al. 2016, p. 46). Additionally, one study predicts that the PMO will enter a positive warm phase around the year 2025 (Chen et al. 2016, p. 322). The phases of these long-term climate cycles in addition to the projected warming trend imply that we should expect sea surface temperatures during the period over the next couple of decades to be especially warm. However, climate change may also alter the patterns of these oscillations, for example, by shortening the cycle length of the PDO (Zhang and Delworth 2016, pp. 6007-6008). Many studies of climate effects to marine species and ecosystems use indices of these climate oscillations, rather than individual climate variables such as sea surface temperature, as their measures of the climatic state (e.g. Becker and Beissenger 2006, p. 473). Therefore, if climate factors that covary with a given oscillation become decoupled, the relationships inferred from these studies may no longer be valid in the future.

Changes in the Forest Environment

Forested habitats in the Pacific Northwest are affected by climate change mainly via changes in disturbances, including wildfire, insects, tree diseases, and drought mortality. These types of disturbances can all cause the loss of murrelet nesting habitat, though it is hoped that this loss will be offset by ingrowth as existing mid-successional forest matures. Following stand-replacing disturbances, climate conditions may not allow recruitment of the tree species that are currently present, leading to ecotype change; however, the effect of this kind of ecotype change may not directly affect murrelet habitat availability until many decades in the future.

Historical fire regimes have varied throughout the range of the murrelet. In many of the moist forests of western Washington and Oregon, the fire regime has historically been typified by large, stand-replacing fires occurring at intervals of 200 years or more (Halofsky et al. 2018a, pp. 3-4; Haugo et al. 2019, pp. 2-3; Long et al. 1998, p. 784). Parts of the murrelet range in southern Oregon and California have historically had low- and mixed-severity fires occurring every 35 years or less (Haugo et al. 2019, pp. 2-3; Perry et al. 2011, p. 707). Still other areas throughout the range historically had mixed severity fires occurring between 35 and 200 years apart (Haugo et al. 2019, pp. 2-3; Perry et al. 2011, p. 707). Within each type of historical fire regime, fire has occurred less frequently during the recent decades usually used for statistical analyses of fire behavior or projections of future fire than it did historically (Huago et al. 2019, pp. 8-9; Littell et al. 2010, p. 150).

Between 1993 and 2012, monitoring based on a database of large (1,000 acres or greater) fire perimeters detected losses associated with wildfires of 22,063 acres of Maxent-modeled high-quality murrelet nesting habitat on federal and non-federal lands in the NWFP area (Raphael et al. 2016b, pp. 80-81). Fire was the leading natural cause of habitat loss within the NWFP area, but this ranking was driven by the 20,235-acre loss to fire on federal lands in the Klamath Mountains, and fire was far less important elsewhere in the range. Within subregions overlapping the listed range of the murrelet, the proportion of area currently "highly suitable" for large fires varies from less than 1 percent in the Coast Range of Oregon and Washington to 18 percent in the Klamath Mountains (Davis et al. 2017, p. 179). The fire regime in the listed range of the murrelet has historically been sensitive to climate conditions, though less so during recent decades (Henderson et al. 1989, pp. 13-19; Littell et al. 2010, p. 140; Littell and Gwozdz 2011, pp. 130-131; Weisberg and Swanson 2003, pp. 23-25). South of the NWFP area, extreme heat and unusual lightning activity contributed to the 2020 fires that burned through much of the remaining murrelet habitat in central California, and these conditions were likely caused or exacerbated by climate change (Goss et al. 2020, p. 11; Mulkern 2020, pp. 2, 5-6; Romps et al. 2014, p. 853; Temple 2020, p. 2).

The area burned in the range of the murrelet is expected to increase in the coming decades, but there is great uncertainty about the magnitude of the increase, and it is likely to affect some areas more than others (Davis et al. 2017, pp. 179-182; Rogers et al. 2011, p. 6; Sheehan et al. 2015, p. 25). On forested lands in the Cascades, Coast Ranges, and Klamath Mountains of Washington and Oregon, the percentage of forested area highly suitable for large fires is projected to increase from the current (less than 1 percent to 18 percent, varying by ecoregion) up to between 2 and 51 percent by the late 21st century, with much of this increase projected to occur after 2050 (Davis et al. 2017, pp. 179-181). At the same time, the percentage of forested lands with low suitability for large fire is expected to decrease from the current range of 21 to 97 percent to a lower range of 4 to 85 percent, depending on ecoregion. The increase in large fire suitability is expected to have the greatest effect on the Klamath ecoregion and the smallest effect on the Coast Ranges, with Cascades ecoregions falling in between (Davis et al. 2017, pp. 181). One study has classified most of the murrelet range as having low vulnerability to fire for the 2020-2050 period, relative to all western forests, but parts of the range in southern Oregon and northern California are classified as having medium or high vulnerability (Buotte et al. 2018, pp. 5, 8). A different study found that forests west of the Cascade Crest are likely to be more vulnerable other western forests, because they will be sensitive to hotter, drier summers, but will not benefit from increased winter precipitation since soils are already saturated during winter months (Rogers et al. 2011, p. 6). Throughout the range, the annual number of days with high wildfire potential is expected to nearly double by mid-century (Martinuzzi et al. 2019, pp. 3, 6). Fire severity is also projected to increase over the 21st century (Rogers et al. 2011, p. 6).

Two recent studies have modeled future fires based on projected climate and vegetation characteristics, rather than simply using statistical projections based on past rates of wildfire. One study projected a 1.5- to 5-fold increase in forest fire in western Washington between the historical period and the 21st century (Halofsky et al. 2018b, p. 10). The baseline annual percentage of area burned was based on information about pre-European settlement fire rotation in western Washington, 0.2 to 0.3 percent of the forest land base burned per year, which is a much greater annual area burned than we have observed in the recent past. The late 21st-century annual area burned was projected to reach 0.3 to 1.5 percent of the forest land base per year, with extreme fire years burning 5 to 30 percent of the forest land base (Halofsky et al. 2018b, p. 10). The other study projected a 2- to 4-fold increase in western Washington and Oregon between the late 20th century and mid-century (Sheehan et al. 2019, p. 14). This study started with even larger baseline annual percentage of area burned, starting at 0.47 to 0.56 percent per year in the late 20th century and increasing to 1.14 to 1.99 percent per year by the mid-21st century (Sheehan et al. 2019, p. 14). In both studies, smaller increases in annual area burned were associated with a model assumption that firefighting would continue to be effective.

Insects and disease were the leading natural cause of murrelet habitat loss within most ecoregions within the NWFP area between 1993 and 2012 (Raphael et al. 2016b, p. 81). Across the NWFP area, 8,765 acres of Maxent-modeled high-quality murrelet habitat were lost to insects and disease, with the majority of these on federal lands in Washington. The USFS and WDNR have worked together since 1981 to collect and distribute aerial survey data regarding the presence of insects, disease, and other damage agents in Washington's forests (WDNR and USFS 2018). This dataset dataset indicates the identity of various insect and disease problems that have been recorded in the current murrelet habitat: Douglas-fir beetle (*Dendroctonus pseudotsugae*), "dying hemlock," fir engraver (*Scolytus ventralis*), spruce aphid (*Elatobium abietinum*), Swiss needle cast (*Phaeocryptopus gaeumannii*), and western (*Lambdina fiscellaria lugubrosa*) and phantom (*Nepytia phantasmaria*) hemlock loopers. It is likely that various root diseases have also attacked murrelet habitat, but these are generally classified as bear damage during the aerial surveys (Clark et al. 2018, p. 31). Root diseases that may be present include annosus (*Heterobasidium annosum*), armillaria (*Armillaria ostoyae*), and black stain (*Leptographium wageneri*) root diseases, as well as laminated (*Phellinus weirii*), tomentosus (*Inonotus tomentosus*), and yellow (*Parenniporia subacida*) root rots (Goheen and Willhite 2006, pp. 72-87).

Some of these pests, such as Swiss needle cast, are most typically found in younger stands, and are more likely to affect the development of murrelet habitat over the long term; whereas others, such as Douglas-fir beetle, are more likely to attack older trees (Goheen and Willhite 2006, pp. 30, 224). Swiss needle cast typically does not result in tree mortality (Maguire et al. 2011, pp. 2069-2070), but can affect mixed-species forest stands by allowing increased western hemlock growth in stands where severe Swiss needle cast affects Douglas-fir growth (Zhao et al. 2014, entire). Higher average temperatures, in particular warmer winters, and increased spring precipitation in the Oregon Coast Range have contributed to an increase in the severity and distribution of Swiss needle cast in Douglas-fir (Stone et al. 2008, pp. 171-174; Sturrock et al. 2011, p. 138; Zhao et al. 2011, p. 1,876; Lee et al. 2013, pp. 683-685; Ritóková et al. 2016, p. 2). The distribution of Swiss needle cast increased from about 131,087 ac (53,050 ha) in 1996 to about 589,840 ac (238,705 ha) of affected trees in 2015 within 31 mi (50 km) of the coast in the Oregon Coast Range (Hansen et al. 2000, p. 775; Ritóková et al. 2016, p. 5).

Drought has not historically been a major factor in most of the listed range of the murrelet, because these forests are not typically water limited, especially in Washington and northern

Oregon (Littell et al. 2010, p. 139; McKenzie et al. 2001, p. 531; Nemani et al. 2003, p. 1560). Nonetheless, every part of the listed range has been affected by multi-year drought at some point during the 1918-2014 period, varying geographically from areas with occasional mild two- to five-year droughts, to areas with moderate-severity two- or three-year droughts, to a few small areas, all in Washington, that have had at least one extreme three-year drought (Crockett and Westerling 2018, p. 345). Over the last few decades, the number of rainy summer days has decreased and the rain-free period has lengthened in much of the murrelet's listed range, especially in Oregon and Washington (Holden et al. 2018, p. 4). In the Pacific Northwest generally, drought is associated with Douglas-fir canopy declines that can be observed via satellite imagery (Bell et al. 2018a, pp. 7-10). In Western Washington, Oregon, and Southwestern British Columbia, tree mortality more than doubled (from around 0.5 percent per year to more than 1 percent per year) over the 30-year period between 1975 and 2005, likely due to increasing water stress (van Mantgem et al. 2009, pp. 522-523). Tree mortality may be caused by warm dry conditions in and of themselves (via xylem failure) or when hot, dry conditions compound the effects of insects, disease, and fire.

Some of the insects and pathogens already present in murrelet habitat, such as Douglas-fir beetles, are likely to become more prevalent and cause greater mortality in the future. Douglas-fir trees stressed by heat and drought emit ethanol, which attracts Douglas-fir beetles, and have lowered chemical defenses, which is likely to increase the endemic levels of Douglas-fir infestation and could result in higher probability of epidemic infestation (Agne et al. 2018, p. 326-327; Bentz et al. 2010, p. 605). Similarly, higher temperatures as the 21st century progresses will also increase the potential of spruce beetle (*Dendroctonus rufipennis*) outbreaks, which require mature spruce forests such as those found within the range of the murrelet (Bentz et al. 2010, p. 607). There is more uncertainty with respect to future levels of infection by Swiss needle cast, a disease that that has increased in severity over the past decade (Agne et al. 2018, p. 326). Warm, wet spring weather is thought to provide ideal conditions for Swiss needle cast infection, whereas warm, dry spring weather may inhibit the pathogen. Future spring weather will be warmer, but it is not clear whether it will be wetter, drier, or both (i.e., more variable), or perhaps current precipitation patterns will continue. Swiss needle cast effects to trees appear to be more severe during drought conditions, however. Therefore, the worst-case scenario for Swiss needle cast would be warm, wet springs followed by hot, dry summers. Swiss needle cast is also expected to spread inland and north to sites where fungal growth is currently limited by cold winter temperatures (Stone et al. 2008, p. 174; Zhao et al. 2011, p. 1,884; Lee et al. 2013, p. 688). Future climate conditions are also hypothesized to promote other diseases, such as Armillaria root disease, that could affect murrelet habitat (Agne et al. 2018, p. 326).

All climate models project increased summer warming for the Pacific Northwest, and most project decreased spring snowpack and summer precipitation, resulting in increasing demand on smaller amounts of soil water in the forest during the growing season. Forests within the murrelet range are expected to experience increasing water deficits over the 21st century (McKenzie and Littell 2017, pp. 33-34). These deficits will not be uniform, with the California and southern Oregon Coast Ranges, Klamath region, eastern Olympic Peninsula, and parts of the Cascades and northern Oregon Coast Range projected to experience much greater hydrological drought, starting sooner than in other places, while there are even projected reductions in water deficit for some other portions of the Washington Cascades and Olympic Mountains (McKenzie and Littell 2017, p. 31). Spring droughts, specifically, are projected to decrease in frequency in Washington and most of Oregon, but to increase in frequency in most of California, with some uncertainty as to the future likelihood of spring drought near the Oregon-California border (Martinuzzi et al. 2019, p. 6). The projected future warm, dry conditions, sometimes called "hotter drought" or "climate change-type drought" in the scientific literature, are expected to lead to continued increases in tree mortality. Though projections of future drought-related tree mortality in throughout the listed range of the murrelet are not available, the effects of the recent multi-year drought in the Sierra Nevada may provide some context about what to expect. Drought conditions in California during 2012 through 2015 led to an order of magnitude increase in tree mortality in Sierra Nevada forests (Young et al. 2017, p. 83). More mesic regions, including most areas of murrelet habitat, are unlikely to have near-future impacts as severe as those already seen in the Sierra Nevada. For example, redwood forests in northwestern and central California, which include areas of murrelet nesting habitat, are more resistant to drought effects than other California forests (Brodrick et al. 2019, pp. 2757-2758). However, extreme climate conditions are eventually likely to further increase drought stress and tree mortality, especially since trees in moist forests are unlikely to be well-adapted to drought stress (Allen et al. 2010, p. 669; Allen et al. 2015, pp. 19-21; Anderegg et al. 2013, p. 705; Crockett and Westerling 2018, p. 342; Prestemon and Kruger 2016, p. 262; Vose et al. 2016, p. 10).

Blowdown is another forest disturbance that has historically caused extensive stand-replacing disturbances in the Pacific Northwest. The effect of climate change on blowdown frequency, extent, and severity is unknown, and there are reasons to believe that blowdowns may become either more or less frequent or extensive. Blowdown events are often associated with extra-tropical cyclones, which are often associated with atmospheric rivers. Blowdown is influenced by wind speeds and by soil saturation. Hurricane-force winds hit the Washington coast approximately every 20 years during the 20th century (Henderson et al. 1989, p. 20). Destructive windstorms have occurred in the Pacific Northwest in 1780-1788, 1880, 1895, 1921, 1923, 1955, 1961, 1962, 1979, 1981, 1993, 1995, and 2006 (Henderson et al. 1989, p. 20; Mass and Dotson 2010, pp. 2500-2504). During the 20th century, the events in 1921, 1962, and 2006 were particularly extreme. Although there are some estimates of timber losses from these events, there are no readily available estimates of total murrelet habitat loss from particular events. In addition to habitat loss from these extreme blowdown events, a smaller amount of habitat is lost each year in "endemic" blowdown events. Wind damage may be difficult to detect via methods that rely on remotely sensed data (e.g., Raphael et al. 2016b, pp. 80-81) because much of the wind-damaged timber may be salvaged, and therefore appears to have been disturbed by harvest rather than wind. Nonetheless, between 1993 and 2012, 3,654 acres of Maxent-modeled higher suitability nesting habitat loss was detected via remote sensing and attributed to blowdown or other natural, non-fire, non-insect disturbances (Raphael et al. 2016b, pp. 80-81). Nearly all of the habitat loss in this category affected federal lands in Washington.

Because we did not locate any studies attempting to project murrelet habitat loss to blowdown into the future, we looked to studies regarding the conditions associated with blowdown: wind, rain, and landscape configuration. There are indications that average wind speeds over the Pacific Northwest have declined since 1950, and average wind speeds are projected in most climate models to decline further by the 2080s (Luce et al. 2013, pp. 1361-1362). However, it is not clear how average wind speeds might be related to blowdown, since blowdown events usually happen during extreme wind events. Extreme extra-tropical cyclones are expected to become less frequent in the Northern Hemisphere in general, and perhaps along the Pacific Northwest coastline in particular, but these predictions involve many uncertainties. Different models show local increases in storm frequency in different places (Catto et al. 2011, pp. 5344-5345). Also, how "extreme" events are categorized differs between studies, and the results vary depending on what definition of "extreme" is used (Catto et al. 2001, p. 5348; Ulbrich et al. 2009, p. 127). One recent model projects no change in the extreme ground-level winds most likely to damage nesting habitat, and an increase in the frequency of extreme high-altitude winds (Chang 2018, pp. 6531, 6539). Atmospheric rivers are expected to become wetter and probably more frequent. The frequency of atmospheric river days is expected to increase by 50 to around 500 percent over the 21st century, depending on latitude and season (Gao et al. 2015, p. 7182; Warner and Mass 2017, p. 2135), though some models project up to an 18 percent decrease in frequency for either the northern or the southern end of the listed range (Payne and Magnusdottir 2015, p. 11,184). The most extreme precipitation events are expected to be between 19 and 40 percent wetter, with the largest increases along the northern California coast (Warner et al. 2015, p. 123). If increased rain causes greater soil saturation, it is easily conceivable that blowdown would become likely at lower wind speeds than would be needed to cause blowdown in less saturated conditions, but we did not find studies addressing this relationship. Since blowdown is more likely at forest edges, increased fragmentation may lead to more blowdown for the same wind speed and amount of soil saturation. The proportion of Maxent-modeled higher suitability nesting habitat located along forest edges increased between 1993 and 2012, and now makes up the majority of habitat in the NWFP area (Raphael et al. 2016b, p. 77). Some forested areas within the range may become less fragmented over the next 30 years, as conservation plans such as the NWFP continue to allow for forest growth; other areas may become more fragmented due to harvest, development, or the forest disturbances discussed above. Thus, the amount of murrelet habitat likely to be lost to blowdown over the next 30 years is highly uncertain.

Synergistic effects between drought, disease, fire, and/or blowdown are likely to occur to some extent, and could become widespread. If large increases in mortality do occur, interactions between these agents are likely to be involved (Halofsky et al. 2018a, pp. 4-5). The large recent increase in tree mortality in the Sierra Nevada has been caused in large part due to these kinds of synergistic interactions. As noted above, range of the murrelet is unlikely to be as severely affected and severe effects are likely to happen later in time here than drier forests (where such effects are already occurring). In fact, one study rates much of the range as having low vulnerability, relative to other western forests, to drought or fire effects by 2049 (Buotte et al. 2018, p. 8). However, that study and many other studies do indicate that there is a risk of one or more of these factors acting to cause the loss of some amount of murrelet habitat over the next 30 years.

In addition to habitat loss resulting from forest disturbances at the scale of a stand or patch, habitat features may be altered as a result of climate change. For example, epiphyte cover on tree branches may change as a result of the warmer, drier summers projected for the future (Aubrey et al. 2013, p. 743). Climate-related changes in epiphyte cover will be additive or synergistic to changes in epiphyte cover resulting from the creation of forest edges through timber harvest (Van Rooyen et al. 2011, pp. 555-556). Epiphyte cover is assumed to have decreased throughout the listed range as the proportion of suitable habitat in edge condition has increased (USFWS 2019, p. 34), and as epiphyte cover decreases further, nest sites will become less available even in otherwise apparently suitable habitat.

In summary, forest disturbances, including wildfire, insect damage, disease, drought mortality, and windthrow, are likely to continue to remove murrelet nesting habitat, and many of these disturbances are likely to remove increasing amounts of habitat in the future. The effects of each type of disturbance are likely to be variable in different parts of the range, with wildfire affecting the Klamath Mountains far more than other parts of the range, and insect and disease damage largely focused in Washington. The magnitude of future increases is highly uncertain, and it is unclear whether windthrow will increase, decrease, or remain constant. Habitat not lost to disturbance may nonetheless be affected by climate change, as particular habitat features may be lost. The effects of habitat loss and the loss of habitat features will reduce the availability of nesting habitat, which will reduce the potential for murrelet reproduction.

Changes in the Marine Environment

Changes in the climate, including temperature changes, precipitation changes, and the release of carbon dioxide into the atmosphere, affect the physical properties of the marine environment, including water circulation, oxygen content, acidity, and nutrient availability. These changes, in turn, affect organisms throughout the marine food web. For top predators like the murrelet, Prey abundance, quality, and availability are all likely to be affected by climate change. Climate change is also likely to change the murrelet's level of exposure to toxic chemicals and potentially to disease agents. All of these changes are likely to alter the reproduction and survival of individual murrelets.

Marine waters within the range of the murrelet have warmed, as noted above. This warming involves not only a gradual increase in average temperatures, but also extreme marine heatwaves, which have dramatic effects on marine ecosystems. Preceding the development of El Niño conditions in 2015, a rise in sea surface temperatures in the Gulf of Alaska occurred in late 2013, likely due to a shift in wind patterns, lack of winter storms, and an increase in sea-level pressure (Bond et al. 2015, p. 3414; Leising et al. 2015, pp. 36, 38, 61). This warm water anomaly expanded southward in 2014, with further warming along the California Current in 2015, and then merged with another anomaly that developed off Baja California, becoming the highest sea surface temperature anomaly observed since 1982 when measurements began (NMFS 2016, p. 5). These anomalies became known as "the Blob" (Bond et al. 2015, p. 3414) and helped to compress the zone of cold upwelled waters to the nearshore (NMFS 2016, p. 7). During the late summer of 2019, a new marine heatwave began developing, and is currently on a trajectory to be as extreme as the 2014-2015 "Blob" (NMFS 2019).

The marine portion of the listed range of the murrelet is located along the California Current and estuary systems (including the Salish Sea) adjacent to it. The California Current is strongly influenced by upwelling, in which water rises from the deep ocean to the surface. Upwelling along the west coast leads to an influx of cold waters rich in nutrients such as nitrates, phosphates, and silicates, but that are also acidic (due to high dissolved carbon dioxide content) and low in dissolved oxygen (Johannessen et al. 2014, p. 220; Krembs 2012, p. 109; Riche et al. 2014, pp. 45-46, 48; Sutton et al. 2013, p. 7191). Changes in upwelling are likely to occur, and to influence the ecosystem components most important to murrelets. If changes in upwelling occur along the outer coast of Washington, these changes will also affect the interchange of waters through the Strait of Juan de Fuca (Babson et al. 2006, p. 30; Newton et al. 2003, p. 718). It has been hypothesized that as climate change accentuates greater warming of air over land areas than of air over the ocean, alongshore winds will intensify, which will lead to an increase in upwelling (Bakun 1990, entire). Historical records show that these winds have intensified over the past several decades (Bylhower et al. 2013, p. 2572; García-Reyes and Largier 2010, p. 6; Sydeman et al. 2014, p. 78-79; Taboada et al. 2019, p. 95; Wang et al. 2015, pp. 390-391). Projections for future changes in upwelling offer some support for this hypothesis, but are more equivocal (Foreman et al. 2011, p. 10; Moore et al. 2015, p. 5; Mote and Mantua 2002, p. 53-3; Rykaczewski et al. 2015, pp. 6426-6427; Wang et al. 2010, pp. 263, 265). Some studies indicate a trend toward a later, shorter (but in some cases, more intense) upwelling season, though at the southern end of the range the season may be lengthening (Bograd et al. 2009, pp. 2-3; Bylhower et al. 2013, p. 2572; Diffenbaugh et al. 2004, p. 30; Foreman et al. 2011, p. 8; García-Reyes and Largier 2010, p. 6). Trends and projections for the future of upwelling in the California Current may be so variable because upwelling is inherently difficult to model, or because upwelling in this region is heavily influenced by climate cycles such as the NPGO, PDO, and ENSO (Macias et al. 2012, pp. 4-5; Taboada et al. 2019, p. 95; Wang et al. 2015, p. 391).

Regardless of potential changes in the timing or intensity of upwelling, the dissolved oxygen content of the waters in the listed range is expected to decrease. The solubility of oxygen in water decreases with increasing temperature, so as the climate becomes warmer, the dissolved oxygen content of the marine environment is expected to decrease (IPCC 2014, p. 62; Mauger et al. 2015, pp. 7-3, 7-8). The oxygen content in the North Pacific Ocean has declined significantly since measurements began in 1987 (Whitney et al. 2007, p. 184), and this decline is projected to continue (Whitney et al. 2013, p. 2204). Hypoxic and anoxic events, in which the lack of dissolved oxygen creates a dead zone, have occurred in Puget Sound and along the outer coasts of Washington and Oregon (PSEMP Marine Waters Workgroup 2017, p. 22; PSEMP Marine Waters Workgroup 2016, p. 15; Oregon State University 2017, entire). These dead zones have expanded into shallower depths and areas closer to shore, and impacts are expected to increase rapidly (Chan et al. 2016, p. 4; Somero et al. 2016, p. 15). If upwelling does increase in intensity, the effect would likely be to further reduce the oxygen content of nearshore waters, but these changes are not likely to be consistent throughout the region or throughout the year. Changes in oxygen content, or in the timing of low-oxygen periods, may have important biological consequences (see below). Oxygen content also responds to biological activity. In addition to climate change-induced effects, some locations will likely experience reductions in oxygen content stemming from biological responses to eutrophication in areas that receive (and do not quickly flush) nutrient inputs from human activities (Cope and Roberts 2013, pp. 20-23; Mackas and Harrison 1997, p. 14; Roberts et al. 2014, pp. 103-104, 108; Sutton et al. 2013, p. 7191).

Similarly, acidification of waters in the listed range is expected to increase, regardless of any changes in upwelling. Acidification results when carbon dioxide in the air dissolves in surface water, and is the direct consequence of increasing carbon dioxide emissions (IPCC 2014, pp. 41, 49). Marine waters are projected to continue becoming more acidic, and ocean acidification is now expected to be irreversible at human-relevant timescales (IPCC 2014, pp. 8-9, 49; IPCC

2019, pp. 1-4, 1-7, 1-14). Both the surface and upwelled waters of North Pacific Ocean have become more acidic due to carbon dioxide emissions (Feely et al. 2008, pp. 1491-1492, Murray et al. 2015, pp. 962-963), and this trend is expected to continue (Byrne et al. 2010, p. L02601; Feely et al. 2009, pp. 40-46). These waters also contribute to acidification Conservation Zone 1 as they flow in through the Strait of Juan de Fuca (Feely et al. 2010, p. 446, Murray et al. 2015, p. 961). Any increase in upwelling intensity or changes in seasonality would respectively increase acidification or change the timing of pH changes in the murrelet range. It is unknown whether regional carbon dioxide emissions cause additional localized acidification within particular parts of the range (Newton et al. 2012, p. 36), but it is likely that other products of fossil fuel combustion, such as sulfuric acid, do contribute (Doney et al. 2007, pp. 14582-14583). Linked to reductions in dissolved oxygen (Riche et al. 2014, p. 49), acidification has important biological consequences (see below), and also responds to biological activity. For example, local areas of eutrophication are likely to experience additional acidification beyond that caused directly or indirectly by carbon dioxide emissions (Newton et al. 2012, pp. 32-33).

Sea level rise is also expected to affect the listed range of the murrelet. Sea level rise is a consequence of the melting of glaciers and ice sheets combined with the expansion of water as it warms (IPCC 2014, p. 42). At regional and local scales, numerous factors affect sea level rise, including ocean currents, wind patterns, and plate tectonics (Mauger et al. 2015, p. 4-1; Dalrymple 2012, p. 81; Petersen et al. 2015, p. 21). Sea level is rising at most coastal locations in the action area (Mauger et al. 2015, p. 4-2; Dalrymple 2012, pp. 79-81; Shaw et al. 1998, p. 37). These increases in sea level are likely to continue and may accelerate in the near future (Bromirski et al. 2011, pp. 9-10; Dalrymple 2012, pp. 71, 102; Mauger et al. 2015, pp. 4-3 – 4-5; Mote et al. 2008, p. 10; Petersen et al. 2015, pp. 21, 29, and Appendix D). However, in some places, such as Neah Bay,Washington, plate tectonics are causing upward land movement that is currently outpacing sea level rise (Dalrymple 2012, p. 80; Montillet et al. 2018, p. 1204; Mote et al. 2008, pp. 7-8; Petersen et al. 2015, pp 24-26). In other places, sea-level rise is expected to have consequences for near-shore ecosystems (see below).

Physical Changes Specific to Conservation Zone 1

Conservation Zone 1 will be affected by changes in upwelling, dissolved oxygen content, and acidification discussed above, but these effects are expected to vary, both between Conservation Zone 1 and the other Zones, and within Zone 1, based on the exchange of waters through the Strait of Juan de Fuca and water circulation patterns within Zone 1. These water circulation patterns, in and of themselves, are expected to be affected by climate change. The complexity of the physical environment within Zone 1 can make some climate change effects difficult to predict.

Changes in temperature and the seasonality of precipitation over land affect the freshwater inflows to Conservation Zone 1. Spring and summer freshwater inflows are expected to be warmer and reduced in volume, whereas winter freshwater inflows are expected to increase (Lee and Hamlet 2011, p. 110; Mauger et al. 2015, p. 3-8; Moore et al. 2015, p. 6; Mote et al. 2003, p. 56). Many watersheds draining to the Salish Sea have historically been fed by a mix of rain and snowmelt, but are expected to be increasingly dominated by rainfall, which will cause the timing of peak flows to shift from spring to winter (Elsner et al. 2010, pp. 248-249; Hamlet et al. 2001, pp. 9-11; Hamlet et al. 2013, pp. 401-404; Mauger et al. 2015, pp. 3-4 – 3-5). With winter warming and increases in heavy rainfall events, flooding has increased, and this increase is expected to continue (Hamlet and Lettenmaier 2007, pp. 25-16; Lee and Hamlet 2011, p. 113; Mauger et al. 2015, pp. 3-6 – 3-7). Increased winter freshwater inflows, in combination with melting glaciers, are expected to bring increased sediments to the mouths of rivers; however, it is uncertain whether these sediments are more likely to enter the marine waters or to be deposited in estuaries (Czuba et al. 2011, p. 2; Lee and Hamlet 2011, pp. 129-134; Mauger et al. 2015, pp. 5-7 – 5-10).

These changes in seasonal freshwater inflows are expected to alter water circulation and stratification within Conservation Zone 1, and to affect the rate and timing of exchange of waters through the Strait of Juan de Fuca between the Puget Sound and the North Pacific Ocean (Babson et al. 2006, pp. 29-30; MacCready and Banas 2016, p. 13; Mauger et al. 2015, p. 6-2, Riche et al. 2014, pp. 37-39, 44-45, 49-50). This exchange occurs in two layers, with fresh water at the surface flowing toward the ocean, and denser, saltier ocean waters flowing from the ocean at greater depths (Babson et al. 2006, p. 30). With the projected changes in timing of freshwater inflows, the rate of exchange is expected to increase during winter and decrease during summer (Mauger et al. 2015, pp. 6-2 – 6-3). The effect of changes in freshwater inflow on stratification is likely to vary by location within the action area, with greater potential for effect in, for example, southern Puget Sound than in well-mixed channels like Admiralty Inlet and Dana Passage (Newton et al. 2003, p. 721).

When hypoxic (low dissolved oxygen) events occur in the waters of Zone 2, these waters also flow into the inland waters of Conservation Zone 1, driving down the oxygen content there as well, although there is considerable variation over time, space, and depth, due to patterns of circulation and mixing within the Salish Sea (Bassin et al. 2011, Section 3.2; Johannessen et al. 2014, pp. 214-220). For example, Hood Canal is particularly susceptible to hypoxic conditions, partly because circulation of water through Hood Canal is slow (Babson et al. 2006, p. 30), whereas the vigorous tidal currents in Haro Strait allow for the mixing of oxygen-rich surface water throughout the water column (Johannessen et al. 2014, p. 216). Increased stratification, as is expected during winter with the larger freshwater inflows, can lead to hypoxic conditions in deeper waters (Mauger et al. 2015, p. 6-3; Whitney et al. 2007, p. 189). On the other hand, weaker stratification, as expected in the summer, may decrease the probability of low oxygen due to greater mixing, or increase the probability of low oxygen due to slower circulation (Newton et al. 2003, p. 725).

Primary Productivity

Changes in temperature, carbon dioxide, and nutrient levels are likely to affect primary productivity by phytoplankton, macroalgae, kelp, eelgrass, and other marine photosynthesizers (IPCC 2019, p. 5-72; Mauger et al. 2015, p. 11-5). In general, warmer temperatures, higher carbon dioxide concentrations, and higher nutrient levels lead to greater productivity (Gao and Campbell 2014, pp. 451, 454; Nagelkerken and Connell 2015, p. 13273; Newton and Van Voorhis 2002, p. 10; Roberts et al. 2014, pp. 11, 22, 108; Thom 1996, pp. 386-387), but these effects vary by species and other environmental conditions, such as sunlight levels or the ratios of different nutrients (Gao and Campbell 2014, pp. 451, 454; Krembs 2012, p. 109; Kroeker et al. 2013, p. 1889; Low-Decarie et al. 2011, p. 2530). In particular, phytoplankton species that form calcium carbonate shells, such as coccolithophores, show weaker shell formation and alter their physiology in response to acidification, and are expected to decline in abundance with continued acidification (Feely et al. 2004, pp. 365-366; IPCC 2019, p. 5-62; Kendall 2015, pp. 26-46). Due to changes in the seasonality of nutrient flows associated with upwelling and freshwater inputs, there may also be alterations in the timing, location, and species composition of bursts of primary productivity, for example, earlier phytoplankton blooms (Allen and Wolfe 2013, pp. 6, 8-9; MacCready and Banas 2016, p. 17; Mauger et al. 2015, p. 6-3). Changes in primary productivity may not occur in every season; for example, during winter, sunlight is the major limiting factor through most of Conservation Zone 1 (Newton and Van Voorhis 2002, pp. 9, 12), and it is not clear whether winter sunlight is likely to change with climate change. Models project reductions in overall annual marine net primary productivity in the world's oceans during the 21st century, trends will vary across the listed murrelet range, with decreases at the southern end of the range and increases at the northern end (IPCC 2019, pp. 5-31, 5-38). Changes in primary productivity are also likely to vary at smaller scales, even within a Conservation Zone; for example, primary productivity in Possession Sound is more sensitive to nutrient inputs than other areas within Puget Sound (Newton and Van Voorhis 2002, pp. 10-11). In sum, in addition to localized increases and decreases in productivity, we expect changes in the timing, location, and species dominance of primary producers.

Eelgrass (*Zostera marina*) is a particularly important primary producer in some parts of the range. In some areas, such as Padilla Bay in Zone 1, sea level rise is expected to lead to larger areas of suitable depth for eelgrass meadows. In such areas, eelgrass cover, biomass, and net primary production are projected to increase during the next 20 years (Kairis 2008, pp. 92-102), but these effects will depend on the current and future topography of the tidal flats in a given area. In addition, increasing dissolved carbon dioxide concentrations are associated with increased eelgrass photosynthetic rates and resistance to disease (Groner et al. 2018, p. 1807; Short and Neckles 1999, pp. 184-186; Thom 1996, pp. 385-386). However, increasing temperatures are not likely to be beneficial for eelgrass, and in combination with increased nutrients, could favor algal competitors (Short and Neckles 1999, pp. 172, 174; Thom et al. 2014, p. 4). Changes in upwelling are likely to influence eelgrass productivity and competitive interactions in small estuaries along the California Current (Hayduk et al. 2019, pp. 1128-1131). Between 1999 and 2013, eelgrass growth rates in Sequim Bay and Willapa Bay increased, but at a site in central Puget Sound, shoot density over a similar time period was too variable to detect trends (Thom et al. 2014, pp. 5-6). Taken together, these studies indicate that climate change may benefit eelgrass over the coming decades, but these benefits may be limited to specific areas, and negative effects may dominate in other areas (Thom et al. 2014, pp. 7-9).

Kelp forests also make important contributions to primary productivity in some parts of the range. Like eelgrass, bull kelp (*Nereocystis luetkeana*) responds to higher carbon dioxide concentrations with greater productivity (Thom 1996, pp. 385-386). On the other hand, kelp forests are sensitive to high temperatures (IPCC 2019, p. 5-72), and warming waters (among other factors) have reduced the range of giant kelp (*Macrocystis pyrifera* [Agardh]) (Edwards and Estes 2006, pp. 79, 85; Ling 2008, p. 892). In central and northern California, kelp forests have declined, but not along Oregon, Washington, and Vancouver Island (Krumhansl et al. 2016, p. 13787; Wernberg et al. 2019, p. 69). Along Washington's outer coast and the Strait of Juan de Fuca, bull kelp and giant kelp canopy area did not change substantially over the 20th century, though a few kelp beds have been lost (Pfister et al. 2018, pp. 1527-1528). In southern Puget Sound, bull kelp declines were observed between 2013 and 2017-2018, likely resulting from increasing temperature along with decreasing nutrient concentrations, suspended sediment, and the presence of parasites and herbivores (Berry et al. 2019, p. 43). In northern California, a severe decline in bull kelp occurred in conjunction with the marine heatwave of 2014 and 2015, though a number of other ecological factors were involved (Catton et al. 2019, entire). In central California, trends in giant kelp biomass are related to climate cycles such as the NPGO, making the effect of climate change difficult to detect (Bell et al. 2018b, p. 11). It is unclear what the future effects of climate change will be on kelp in the listed range of the murrelet.

In contrast, increases in harmful algal blooms (also known as red tides or toxic algae) have been documented over the past several decades, and these changes are at least partly due to climate change (IPCC 2019, pp. 5-85 – 5-86; Trainer et al. 2003, pp. 216, 222). Future conditions are projected to favor higher growth rates and longer bloom seasons for these species. In the case of one species, *Alexandrium catanella,* increases in the length of bloom season are projected primarily due to increases in sea surface temperature (Moore et al. 2015, pp. 7-9). As with other climate change effects discussed above, increases in the length of the toxic algae bloom season is likely to vary across the listed range. Even within Zone 1, in the eastern end of the Strait of Juan de Fuca and the inlets of southern Puget Sound, the *A. catanella* bloom season is projected to increase by 30 days per year by 2069, in contrast with Whidbey basin, where little or no change in season length is projected (Moore et al. 2015, p. 8). In another genus toxic algae, *Pseudo-nitzschia*, toxin concentrations increase with increasing acidification of the water, especially in conditions in which silicic acid (used to construct the algal cell walls) or phosphate is limiting (Brunson et al. 2018, p. 1; Tatters et al. 2012, pp. 2-3). These and many other harmful alga species also exhibit higher growth rates with higher carbon dioxide concentrations (Brandenburg et al. 2019, p. 4; Tatters et al. 2012, pp. 3-4). During and following the marine heatwave in 2015, an especially large and long-lasting outbreak of *Pseudo-nitzschia* species stretched from southern California to the Aleutian Islands and persisted from May to October, rather than the typical span of a few weeks (Du et al. 2016, pp. 2-3; National Ocean Service 2016; NOAA Climate 2015, p. 1). This harmful algal bloom produced extremely high concentrations of toxic domoic acid, including the highest ever recorded in Monterey Bay, California (NOAA Climate 2015, p. 2; Ryan et al. 2017, p. 5575). With future climate change, toxic algae blooms are likely to be more frequent than in the past, and the larger, more toxic event of 2015 may become more typical (McCabe et al. 2016, p. 10374).

Higher Trophic Levels

There are several pathways by which climate change may affect species at higher trophic levels (i.e, consumers, including murrelets and their prey). Changing physical conditions, such as increasing temperatures, hypoxia, or acidification will have direct effects on some species. Other consumers will be affected via changes in the abundance, distribution, or other characteristics of their competitors or prey species. Changes in the timing of seasonal events may lead to mismatches in the timing of consumers' life history requirements with their habitat conditions (including prey availability as well as physical conditions) (Mackas et al. 2007, p. 249). The combination of these effects is likely to cause changes in community dynamics (e.g. competitive interactions, predator-prey relationships, etc.), but the magnitude of these effects cannot be predicted with confidence (Busch et al. 2013, pp. 827- 831).

A wide variety of marine species are directly affected by ocean acidification. Like their phytoplankton counterparts, foraminiferans and other planktonic consumers that form calcium carbonate shells are less able to form and maintain their shells in acidified waters (Feely et al. 2004, pp. 356-366). Similarly, chemical changes associated with acidification interfere with shell development or maintenance in pteropods (sea snails) and marine bivalves (Busch et al. 2014, pp. 5, 8; Waldbusser et al. 2015, pp. 273-278). These effects on bivalves can be exacerbated by hypoxic conditions (Gobler et al. 2014, p. 5), or ameliorated by very high or low temperatures (Kroeker et al. 2014, pp. 4-5), so it is not clear what the effect is likely to be in a future that includes acidification, hypoxia, and elevated temperatures. Acidification affects crustaceans, for example, slowing growth and development in Pacific krill (*Euphausia pacifica*) and Dungeness crabs (*Cancer magister*) (Cooper et al. 2016, p. 4; Miller et al. 2016, pp. 118-119). Fish, including murrelet prey rockfish species (*Sebastes* spp.) and Pacific herring (*Clupea pallasii*), are also negatively affected by acidification. Depending on species, life stage, and other factors such as warming and hypoxia, these effects include embryo mortality, delayed hatching, reduced growth rates, reduced metabolic rates, altered sensory perception, and changes in behavior, among other effects (Baumann 2019, entire; Hamilton et al. 2014, entire; Nagelkerken and Munday 2016, entire; Ou et al. 2015, pp. 951, 954; Villalobos 2018, p. 18).

Climate effects are expected to alter interactions within the marine food web. When prey items decrease in abundance, their consumers are also expected to decrease, and this can also create opportunities for other species to increase. In California's Farallon Islands, the recently increasing variance of climate drivers is leading to increased variability in abundance of prey species such as euphausiids and juvenile rockfish, associated with corresponding variability in the demography of predators such as seabirds and salmon (Sydeman et al. 2013, pp. 1662, 1667-1672). In future scenarios with strong acidification effects to benthic prey in the California Current, euphausiids and several fish species are expected to decline, while other species are expected to increase (Kaplan et al. 2010, pp. 1973-1976). An investigation of the planktonic food web off of Oregon shows that sea surface temperature has contrasting effects on different types of zooplankton, and competitive interactions are much more prevalent during warm phases of ENSO or PDO than during cool phases (Francis et al. 2012, pp. 2502, 2505-2506). A food web model of Puget Sound shows that moderate or strong acidification effects to calcifying species are expected to result in reductions in fisheries yield for several species, including salmon and Pacific herring, and increased yield for others (Busch et al. 2013, pp. 827-829). Additionally, the same model shows that these ocean acidification effects are expected to cause reductions in forage fish biomass, which are in turn expected to lead to reductions in diving bird biomass (Busch et al. 2013, p. 829). While Busch and coauthors (2013, p. 831) express confidence that this model is accurate in terms of the nature of ocean acidification effects to the Puget Sound food web of the future, they are careful to note that there is a great deal of uncertainty when it comes to the magnitude of the changes. The model also illustrates that some of the effects to the food web will dampen or make up for other effects to the food web, so that changes in abundance of a given prey species will not always correspond directly to changes in the abundance of their consumers (Busch et al. 2013, pp. 827, 830).

Changes in seasonality at lower trophic levels may lead to changes in population dynamics or in interactions between species at higher trophic levels. In central and northern California, reproductive timing and success of common murres (*Uria aalge*) and Cassin's auklets

(*Ptychoramphus aleuticus*) are related to not only the strength but also the seasonal timing of upwelling, as are growth rates of *Sebastes* species (Black et al. 2011, p. 2540; Holt and Mantua 2009, pp. 296-297; Schroeder et al. 2009, p. 271). At the northern end of the California Current, Triangle Island in British Columbia, Cassin's auklet breeding success is reduced during years when the peak in copepod prey availability comes earlier than the birds' hatch date, and this mismatch is associated with warm sea surface temperatures (Bertram et al. 2009, pp. 206-207; Hipfner 2008, pp. 298-302). However, piscivorous seabirds (tufted puffins [*Fratercula cirrhata*], rhinoceros auklets [*Cerorhinca monocerata*], and common murres) breeding at the same Triangle Island site have, at least to some extent, been able to adjust their breeding dates according to ocean conditions (Bertram et al. 2001, pp. 292-293; Gjerdrum et al. 2003, p. 9379), as have Cassin's auklets breeding in the Farallon Islands of California (Abraham and Sydeman 2004, p. 240). Because of the changes in tufted puffin, rhinoceros auklet, and common murre hatch dates at Triangle Island, the breeding periods of these species have converged to substantially overlap with one another and with that of Cassin's auklet (Bertram et al. 2001, pp. 293-294), but studies have not addressed whether this overlap has consequences for competitive interactions among the four species. Note that all four of these bird species are in the family Alcidae, which also contains murrelets. All these species also breed and forage within the listed range of the murrelet.

Several studies have suggested that climate change is one of several factors allowing jellyfish to increase their ecological dominance, at the expense of forage fish (Parsons and Lalli 2002, pp. 117-118; Purcell et al. 2007, pp. 154, 163, 167-168; Richardson et al. 2009, pp. 314-216). Many (though not all) species of jellyfish increase in abundance and reproductive rate in response to ocean warming, and jellyfish are also more tolerant of hypoxic conditions than fish are (Purcell 2005, p. 472; Purcell et al. 2007, pp. 160, 163; see Suchman et al. 2012, pp. 119-120 for a Northeastern Pacific counterexample). Jellyfish may also be more tolerant of acidification than fish are (Atrill et al. 2007, p. 483; Lesniowski et al. 2015, p. 1380). In the California Current, jellyfish populations appear to be increasing, but nearshore areas are likely to be susceptible to being dominated by jellyfish, rather than forage fish (Schnedler-Meyer et al. 2016, p. 4). Jellyfish abundance in southern and central Puget Sound has increased since the 1970s (Greene et al. 2015, p. 164). Over the same time period, herring abundance has decreased in south and central Puget Sound, and surf smelt (*Hypomesus pretiosus*) abundance has also decreased in south Puget Sound, although other Puget Sound forage fish populations have been stable or increasing (Greene et al. 2015, pp. 160-162). Forage fish abundance and jellyfish abundance were negatively correlated within Puget Sound and Rosario Strait (Greene et al. 2015, p. 164). In the northern California Current, large jellyfish and forage fish have similar diet composition and likely compete for prey, in addition to the two groups' contrasting responses to climate and other anthropogenic factors (Brodeur et al. 2008, p. 654; Brodeur et al. 2014, pp. 177-179).

Many species of forage fish are expected to fare poorly in the changing climate, regardless of any competitive effects of jellyfish. North of the listed range, in the Gulf of Alaska, Anderson and Piatt (1999, pp. 119-120) documented the crash of capelin (*Mallotus villosus*), Pacific herring, and species of Irish lord (*Hemilepidotus* spp.), prickleback (Stichaeidae family), greenlings and mackerel (*Hexagrammos* and *Pleurogrammus* spp.), as well as several shrimp species, as part of a major community reorganization following a climate regime shift from a cool phase to a warm phase in the 1970s. In the northeastern Pacific Ocean, capelin, sand lance

(Ammodytidae family), and rockfish abundance are all negatively correlated with seasonal sea surface temperatures (Thayer et al. 2008, p. 1616). A model of multiple climate change effects (e.g., acidification and deoxygenation) to marine food webs in the Northeast Pacific consistently projects future declines in small pelagic fish abundance (Ainsworth et al. 2011, pp. 1219, 1224). Within Zone 1, abundance of surf smelt and Pacific herring in the Skagit River estuary are positively associated with coastal upwelling during the spring and early summer, likely because nutrient-rich upwelled water increases food availability (Reum et al. 2011, pp. 210-212). If projections of later, shorter upwelling seasons are correct (see above), the delays may lead to declines in these stocks of herring and surf smelt, as happened in 2005 (Reum et al. 2011, p. 212). Similarly, delayed upwelling in 2005 led to reduced growth rates, increased mortality, and recruitment failure of juvenile northern anchovies off of the Oregon and Washington coasts (Takahashi et al. 2012, pp. 397-403). In contrast, anchovy abundance in Zone 1 was unusually high in 2005, as it was in 2015 and 2016 following the marine heatwave, and is positively associated with sea surface temperature (Duguid et al. 2019, p. 38). In the northeastern Pacific, Chavez and coauthors (2003, pp. 217-220) have described a shift between an “anchovy regime” during the cool negative phase of the PDO and a “sardine regime” during the warm positive phase, where the two regimes are associated with contrasting physical and biological states. However, global warming may disrupt the ecological response to the naturally-occurring oscillation, or alter the pattern of the oscillation itself (Chavez et al. 2003, p. 221; Zhang and Delworth 2016, entire).

Marbled Murrelets

Murrelets are likely to experience changes in foraging and breeding ecology as the climate continues to change. Although studies are not available that directly project the effects of marine climate change on murrelets, several studies have been conducted within and outside the listed range regarding ocean conditions and murrelet behavior and fitness. Additionally, numerous studies of other alcids from Mexico to British Columbia indicate that alcids as a group are vulnerable to climate change in the northeastern Pacific.

These studies suggest that the effects of climate change will be to reduce murrelet reproductive success, and to some extent, survival, largely mediated through climate change effects to prey. In British Columbia, there is a strong negative correlation between sea surface temperature and the number of murrelets observed at inland sites displaying behaviors associated with nesting (Burger 2000, p. 728). In central California, murrelet diets vary depending on ocean conditions, and there is a trend toward greater reproductive success during cool water years, likely due to the abundant availability of prey items such as euphausiids and juvenile rockfish (Becker et al. 2007, pp. 273-274). Across the northern border of the listed range, in the Georgia Basin, much of the yearly variation in murrelet abundance from 1958 through 2000 can be explained by the proportion of fish (as opposed to euphausiids or amphipods) in the birds’ diet (Norris et al. 2007, p. 879). If climate change leads to further declines in forage fish populations (see above), those declines are likely to be reflected in murrelet populations.

The conclusion that climate change is likely to reduce murrelet breeding success via changes in prey availability is further supported by several studies of other alcid species in British Columbia and California. Common murres, Cassin’s auklets, rhinoceros auklets, and tufted puffins in

British Columbia; common murres in Oregon; pigeon guillemots (*Cepphus columba*), common murres, and Cassin's auklets in California; and even Cassin's auklets in Mexico all show altered reproductive rates, altered chick growth rates, or changes in the timing of the breeding season, depending on sea surface temperature or other climatic variables, prey abundance, prey type, or the timing of peaks in prey availability (Abraham and Sydeman 2004, pp. 239-243; Ainley et al. 1995, pp. 73-77; Albores-Barajas 2007, pp. 85-96; Bertram et al. 2001, pp. 292-301; Borstad et al. 2011, pp. 291-299; Gjerdrum et al. 2003, pp. 9378-9380; Hedd et al. 2006, pp. 266-275; Piatt et al. 2020, pp. 13-15; Sydeman et al. 2006, pp. 2-4). The abundance of Cassin's auklets and rhinoceros auklets off southern California declined by 75 and 94 percent, respectively, over a period of ocean warming between 1987 and 1998 (Hyrenbach and Veit 2003, pp. 2546, 2551). Although the details of the relationships between climate variables, prey, and demography vary between bird species and locations, the consistent demonstration of such relationships indicates that alcids as a group are sensitive to climate-related changes in prey availability, prompting some researchers to consider them indicator species for climate change (Hedd et al. 2006, p. 275; Hyrenbach and Veit 2003, p. 2551).

In addition to effects on foraging ecology and breeding success, climate change may expose adult and juvenile murrelets to health risks. These risks include poisoning, and potentially feather fouling, from harmful algal blooms, as well as from anthropogenic toxins. Climate change can also cause unexpected changes in disease exposure. Reductions in forage fish quality and availability may also lead to starvation in extreme circumstances, though in less extreme circumstances these reductions are more likely to preclude breeding, which could, counterintuitively, increase adult survival.

It is likely that murrelets will experience more frequent domoic acid poisoning, as this toxin originates from harmful algae blooms in the genus *Pseudo-nitzchia*, which are expected to become more prevalent in the listed range (see above). In central California, domoic acid poisoning was determined to be the cause of death for at least two murrelets recovered during a harmful algae bloom in 1998 (Peery et al. 2006, p. 84). During this study, which took place between 1997 and 2003, the mortality rate of radio-tagged murrelets was highest during the algae bloom (Peery et al. 2006, p. 83). Domoic acid poisoning has previously been shown to travel through the food chain to seabirds via forage fish that feed on the toxic algae (Work et al. 1993, p. 59). Other types of harmful algae, including the *Alexandrium* genus, which is also likely to become more prevalent in the listed range (see above), produce saxitoxin, a neurotoxin that causes paralytic shellfish poisoning. Consumption of sand lance contaminated with saxitoxin was implicated in the deaths of seven out of eight (87.5 percent) of Kittlitz's murrelet (*Brachyramphus brevirostrus*) chicks that were tested following nest failure at a study site in Alaska in 2011 and 2012 (Lawonn et al. 2018, pp. 11-12; Shearn-Bochsker et al. 2014). Yet another species of harmful algae produces a foam that led to plumage fouling and subsequent mortality of common murres and other seabird species off of Oregon and Washington during October of 2009, and similar events may become more frequent with climate change (Phillips et al. 2011, pp. 120, 122-124). Due to changes in the Salish Sea food web, climate change is projected to increase mercury and, to a lesser extent, polychlorinated biphenyls (PCB) levels in forage fish and top marine predators (Alava et al. 2018, pp. 4); presumably murrelets will experience a similar increase.

Climate change may also promote conditions in which alcids become exposed to novel pathogens, as occurred in Alaska during 2013, when crested auklets (*Aethia cristatella*) and thick-billed murres (*Uria lomvia*) washed ashore after dying of avian cholera (Bodenstein et al. 2015, p. 935). Murrelets in Oregon may be especially susceptible to novel diseases, because these populations lack diversity in genes related to immunity (Vásquez-Carrillo et al. 2014, p. 252).

In extreme warm-water conditions, adult murrelets may suffer starvation, as occurred with common murres during the marine heatwave of 2014-2016. High levels of adult mortality were observed among common murres from California to Alaska, and this mortality was likely caused by a combination of reductions in forage fish nutritional content and increases in competition with large piscivorous fish, a combination termed the "ectothermic vise" (Piatt et al. 2020, pp. 17-24). Counterintuitively, in the 1997-2003 study of radio tagged murrelets in California, murrelet adult survival was higher during warm-water years and lower during cold-water years, likely because they did not breed and therefore avoided the associated physiological stresses and additional predator risk (Peery et al. 2006, pp. 83-85).

Overall, the effects of climate change in marine ecosystems are likely to be complex, and will vary across the range. Alterations in the physical properties of the marine environment will affect the productivity and composition of food webs, which are likely to affect the abundance, quality, and availability of food resources for murrelets. These changes, in turn, will affect murrelet reproductive performance. In addition, toxic algae and potentially disease organisms are expected to present increasing risks to murrelet health and survival. Different types of effects can be predicted with varying levels of certainty. For example, large increases in the prevalence of harmful algal blooms have already been observed, whereas the likely future magnitude and direction of overall changes in net primary productivity remain highly uncertain. Some changes may be positive (for example, the potential for a northward shift in anchovy abundance), but on the whole climate change is expected to have a detrimental effect to murrelet foraging and health.

Summary of Climate Change Effects

In summary, murrelets are expected to experience effects of climate change in both their nesting habitat and marine foraging habitat. Natural disturbances of nesting habitat are expected to become more frequent, leading to accelerated habitat losses that may outpace ingrowth even in protected landscapes. Marine food chains are likely to be altered, and the result may be a reduction in food resources for murrelets. Even if food resources remain available, the timing and location of their availability may shift, which may alter murrelet nesting seasons or locations. In addition, health risks from harmful algal blooms, anthropogenic toxins, and perhaps pathogens are likely to increase with climate change.

Within the marine environment, effects on the murrelet food supply (amount, distribution, quality) provide the most likely mechanism for climate change impacts to murrelets. Studies in British Columbia (Norris et al. 2007) and California (Becker and Beissinger 2006) have documented long-term declines in the quality of murrelet prey, and one of these studies (Becker and Beissinger 2006, p. 475) linked variation in coastal water temperatures, murrelet prey quality during pre-breeding, and murrelet reproductive success. These studies indicate that murrelet recovery may be affected as long-term trends in ocean climate conditions affect prey resources and murrelet reproductive rates. While seabirds such as the murrelet have life-history strategies adapted to variable marine environments, ongoing and future climate change could present changes of a rapidity and scope outside the adaptive range of murrelets (USFWS 2009, p. 46).

Conservation Needs of the Species

Reestablishing an abundant supply of high quality murrelet nesting habitat is a vital conservation need given the extensive removal during the 20th century. Even following the establishment of the NWFP, habitat continued to be lost between 1993 and 2012, and the rate of loss on non-federal lands has been 10 times greater than on federal lands (Raphael et al. 2016b, pp. 80-81). If this rate of loss continues, the conservation of the murrelet may not be possible because almost half of the higher-suitability nesting habitat is on non-federal lands (Raphael et al. 2016b, p. 86). Therefore, recovery of the murrelet will be aided if areas of currently suitable nesting habitat on non-federal lands are retained until ingrowth of habitat on federal lands provides replacement nesting opportunities (USFWS 2019, p. 21).

There are also other conservation imperatives. Foremost among the conservation needs are those in the marine and terrestrial environments to increase murrelet fecundity by increasing the number of breeding adults, improving murrelet nest success (increasing nestling survival and fledging rates), and reducing anthropogenic stressors that reduce individual fitness or lead to mortality. The overall reproductive success (fecundity) of murrelets is directly influenced by nest predation rates (reducing nestling survival rates) in the terrestrial environment and an abundant supply of high quality prey in the marine environment before and during the breeding season (improving breeding rates, potential nestling survival, and fledging rates). Anthropogenic stressors affecting murrelet fitness and survival in the marine environment are associated with commercial and tribal gillnets, derelict fishing gear, oil spills, and high underwater sound pressure (energy) levels generated by pile-driving and underwater detonations (which can be lethal or reduce individual fitness). Anthropogenic activities, such as coastline modification and nutrient inputs in runoff, also affect prey availability and harmful algal blooms, which in turn affect murrelet fitness.

Further research regarding marine threats, general life history, and murrelet population trends in the coastal redwood zone may illuminate additional conservation needs that are currently unknown (USFWS 2019, p. 66).

Recovery Plan

The Marbled Murrelet Recovery Plan outlines the conservation strategy with both short- and long-term objectives. The Plan places special emphasis on the terrestrial environment for habitat-based recovery actions due to nesting occurring in inland forests.

In the short-term, specific actions identified as necessary to stabilize the populations include protecting occupied habitat and minimizing the loss of unoccupied but suitable habitat (USFWS 1997, p. 119). Specific actions include maintaining large blocks of suitable habitat, maintaining and enhancing buffer habitat, decreasing risks of nesting habitat loss due to fire and windthrow, reducing predation, and minimizing disturbance. The designation of critical habitat also contributes towards the initial objective of stabilizing the population size through the maintenance and protection of occupied habitat and minimizing the loss of unoccupied but suitable habitat.

Long-term conservation needs identified in the Plan include:

- increasing productivity (abundance, the ratio of juveniles to adults, and nest success) and population size;
- increasing the amount (stand size and number of stands), quality, and distribution of suitable nesting habitat;
- protecting and improving the quality of the marine environment; and
- reducing or eliminating threats to survivorship by reducing predation in the terrestrial environment and anthropogenic sources of mortality at sea.

General criteria for murrelet recovery (delisting) were established at the inception of the Plan and they have not been met (USFWS 2019, p. 65). More specific delisting criteria are expected in the future to address population, demographic, and habitat based recovery criteria (USFWS 1997, p. 114-115). The general criteria include:

- documenting stable or increasing population trends in population size, density, and productivity in four of the six Conservation Zones for a 10-year period and
- implementing management and monitoring strategies in the marine and terrestrial environments to ensure protection of murrelets for at least 50 years.

Thus, increasing murrelet reproductive success and reducing the frequency, magnitude, or duration of any anthropogenic stressor that directly or indirectly affects murrelet fitness or survival in the marine and terrestrial environments are the priority conservation needs of the species. The Service estimates recovery of the murrelet will require at least 50 years (USFWS 1997).

Survival and Recovery Role of Each Conservation Zone

The six Conservation Zones, defined in the Recovery Plan as equivalent to Recovery Units, vary not only in their population status, as described above, but also in their intended function with respect to the long-term survival and recovery of the murrelet.

Conservation Zones 1 extends inland 50 miles from the marine watersof Puget Sound and most waters of the Strait of Juan de Fuca south of the U.S.-Canadian border. The terrestrial portion of Zone 1 includes the north Cascade Mountains and the northern and eastern sections of the Olympic Peninsula. Nesting habitat in the Cascades is largely separated from high-quality marine foraging habitat by both urban development on land and highly altered coastal marine environments, leading to long commutes between nesting and foraging habitat (Lorenz et al.

2017, p. 314; Raphael et al. 2016a, p. 106; USFWS 1997, p. 125). In contrast, large blocks of nesting habitat remain near the coast along the Strait of Juan de Fuca, where there is a lower human footprint (Raphael et al. 2016b, p. 72; van Dorp and Merrick 2017, p. 5). This combination of large blocks of habitat close to foraging habitat is likely more conducive to successful production of young than conditions other portions of Zone 1. Zone 1 is unique among the six Zones in that the marine environment is not a part of the California Current ecosystem, but is part of a complex system of estuaries, fjords, and straits. This means that the Zone 1 population is subject to a different set of environmental influences than the populations in the other five zones. For example, in 2005, delayed upwelling led to widespread nesting failure of seabirds, including murrelets, along the northern California Current, while above-average productivity was observed in Zone 1 (Lorenz and Raphael 2018, pp. 208-209; Peterson et al. 2006, pp. 64, 71; Ronconi and Burger 2008, p. 252; Sydeman et al. 2006, p. 3). This example illustrates the importance of Zone 1 in bolstering the rangewide resilience of murrelets. Zone 1 is one of the four Zones where increased productivity and stable or increasing population size are needed to provide redundancy and resilience that will enable recovery and long-term survival.

Conservation Zone 2 also extends inland 50 miles from marine waters. Conservation Zone 2 includes marine waters within 1.2 miles (2 km) off the Pacific Ocean shoreline, with the northern terminus immediately south of the U.S.-Canadian border near Cape Flattery along the midpoint of the Olympic Peninsula, and extending to the southern border of Washington (the Columbia River) (USFWS 1997, pg. 126). Although Zone 2 was defined to include only the nearshore waters, murrelets in this area are regularly found up to 8 km from shore, sometimes at higher densities than in the nearshore environment, even during the breeding season (Bentivoglio et al. 2002, p. 29; McIver et al. in press, pp. 34, 85). Zone 2 includes the rich waters of the Olympic Coast National Marine Sanctuary, which are adjacent to areas of the Olympic Peninsula that retain large blocks of nesting habitat (Raphael et al. 2016b, p. 72). Like the northern Olympic Peninsula in Zone 1, parts of the western Olympic Peninsula appear to provide one of the few remaining strongholds for murrelets in Washington. The southern portion of Zone 2 previously hosted a small but consistent subpopulation of nesting murrelets, and is now only sparsely used for nesting inland or foraging at sea. This reduction in murrelet population density in the southern portion of Zone 2 represents a widening of a gap in distribution that was described in the Recovery Plan (USFWS 1997, p. 126). This gap is likely a partial barrier to gene flow (USFWS 1997, p. 145). The eventual long-term survival and recovery of listed murrelets depends on the maintenance of a viable murrelet populations that are well distributed throughout Zone 2, along with the other three Zones where increased productivity and stable or increasing population size are needed for survival and recovery.

Conservation Zone 3 extends 35 miles inland, and includes marine waters within 1.2 miles of the Pacific Ocean shoreline between the northern border of Oregon (the Columbia River) and North Bend, Oregon (USFWS 1997, pp. 126-127). The terrestrial portion of Zone 3 historically experienced large-scale wildfires and timber harvest, which together likely led to a loss of nesting habitat that caused a dramatic decline in the murrelet population in this Zone (USFWS 1997, p. 117). In the northern portion of Zone 3, this lack of nesting habitat persists, and the at-sea population density of murrelets is relatively low, extending the gap in the southern portion Zone 2 (USFWS 1997, p. 145; McIver et al. 2021, pp. 11-17). Additionally, murrelet populations in Oregon are expected to be more susceptible to novel pathogens, due to low genetic diversity coding for important immune system peptides (Vásquez-Carrillo et al. 2014, p. 252). However, in Zone 3 as a whole, at-sea population density is high, and is trending upward, though the reason for the population increase is not well understood. The murrelet population of Zone 3 is one of the two largest among the Conservation Zones. The eventual long-term survival and recovery of listed murrelets depends on the maintenance of a viable murrelet populations that is well distributed throughout Zone 3, along with the other three Zones where increased productivity and stable or increasing population size are needed for survival and recovery.

Conservation Zone 4 extends 35 miles inland, and includes marine waters within 1.2 miles of the Pacific Ocean shoreline between North Bend, Oregon and the southern end of Humboldt County, California (USFWS 1997, p. 127). Since 1993, this Zone has experienced the majority of all nesting habitat losses on federal lands within the listed range, nearly all due to large wildfires (Raphael et al. 2016b, p. 75). Much of the nesting habitat within this Zone is located within National and California State Parks, and recreation likely reduces murrelet productivity in these areas, particularly via accidental food subsidies to corvid nest predators at picnic sites and camping areas (USFWS 1997, p. 128). Over the last decade, Redwood National and State Parks have made efforts to reduce this supplemental feeding of corvids, with some success in reducing corvid density at recreation sites, but it would be difficult to detect any population-scale benefit of these efforts (Brunk et al. 2021, pp. 7-8; McIver et al., in press, p. 43). The murrelet population of Zone 4 is one of the two largest among the Conservation Zones, and is increasing, though the reason for the population increase is not well understood. The eventual long-term survival and recovery of listed murrelets depends on the maintenance of a viable murrelet populations that is well distributed throughout Zone 4, along with the other three Zones where increased productivity and stable or increasing population size are needed for survival and recovery.

Conservation Zone 5 extends 25 miles inland, and includes marine waters within 1.2 miles of the Pacific Ocean shoreline between the southern end of Humboldt County, California, and the mouth of San Francisco Bay (USFWS 1997, p. 129). Very little nesting habitat remains in this Zone, mostly in California State Parks and on private lands, though some nesting habitat ingrowth was observed between 1993 and 2012 (Raphael et al. 2016b, p. 75; USFWS 1997, p. 129). Murrelet population estimates in Zone 5 have been correspondingly low, with population estimates of less than 100 individuals in most survey years (McIver et al. 2021, pp. 11-17). The most recent survey, in 2017, resulted in a much higher estimate of 872 individuals, but multiple lines of evidence indicate that this increase was likely the result of unusual migratory patterns from other Zones during the breeding season (Adrean et al. 2018, p. 2; McIver et al., in press, pp. 43-44; Strong 2018, pp. 6-7). However, surveys in Zone 5 are now conducted only once every four years, making the status and trend of this population more difficult to discern. Given the small size of the population during most survey years, and the limited availability of nesting habitat, the ability of this population to survive over the coming decades is questionable, and Zone 5 cannot be counted on to contribute toward long-term survival or recovery of the DPS (USFWS 1997, pp. 129). In the best-case scenario, if nesting habitat ingrowth in this Zone can stimulate the restoration of a larger population in Zone 5 over the long term, this would likely improve connectivity between Zones 4 and 6, provide redundancy, and increase resiliency for the DPS as a whole.

Conservation Zone 6 extends 15 miles inland, and includes marine waters within 1.2 miles of the Pacific Ocean shoreline between the mouth of San Francisco Bay and Point Sur, in Monterey County, California (USFWS 1997, pp. 129-130). Zone 6 is unique among the Zones in that it is not within the NWFP area and is not included in NWFP effectiveness monitoring. Federal land is lacking in Zone 6, and all nesting habitat is located within State or County Parks or on private lands (McShane et al. 2004, p. 4-14). Murrelet population estimates for Zone 6 have averaged around 500 individuals for the period from 1999 through 2019, with a range between 174 and 699 birds across the years (Felis et al. 2020, p. 7). The Zone 6 population is genetically differentiated from the other Zones, likely as a result of the wide gap in the range between the Zone 6 population and the populations to the north (Hall et al. 2009, p. 5078; Peery et al. 2010, p. 703). When the Recovery Plan was written in 1997, it was anticipated that the Zone 6 population would persist long enough to contribute to recovery, but could not be relied upon to contribute to the long-term survival of the species (USFWS 1997, p. 116). Subsequent research has demonstrated that the population in Zone 6 is a demographic sink, with a shrinking breeding population bolstered by the presence of mainly non-breeding individuals originating from other Zones (Peery et al. 2006, p. 1523; Peery et al. 2010, p. 702; Vásquez-Carrillo et al. 2013, p. 177). Demographic effects of large-scale nesting habitat loss and degradation during the 2020 wildfires have not yet manifested, but are expected to be negative. Therefore, it remains unlikely that this population will contribute to recovery. The presence of a murrelet population in Zone 6 is necessary to ensure the future distribution of murrelets throughout their current and historical within the DPS, but it is not clear that this will be possible over the long term, given the vulnerability of this population to stochastic or catastrophic events (USFWS 1997, p. 116).

The Recovery Plan identified lands that will be essential for the recovery of the murrelet, including1) any suitable habitat in a Late Successional Reserve (LSR) in Forest Ecosystem Management Assessment Team (FEMAT) Zone 1 (not to be confused with Conservation Zone 1), as well as LSR in FEMAT Zone 2 in Washington, 2) all suitable habitat located in the Olympic Adaptive Management Area, 3) large areas of suitable nesting habitat outside of LSRs on Federal lands, such as habitat located in the Olympic National Park, 4) suitable habitat on State lands within 40 miles of the coast in Washington, or within 25 miles of the coast in Oregon and California, 5) habitat within 25 miles of the coast on county park land in San Mateo and Santa Cruz Counties, California, 6) suitable nesting habitat on Humboldt Redwood Company (formerly Pacific Lumber Company) lands in Humboldt County, California, and 5) habitat within occupied murrelet sites on private lands (USFWS 1997, pp. 131-133).

Marine habitat is also essential for the recovery of the murrelet. Key recovery needs in the marine environment include protecting the quality of the marine environment and reducing adult and juvenile mortality at sea (USFWS 1997, pp. 134-136). Marine areas identified as essential for murrelet foraging and loafing include 1) all waters of Puget Sound and the Strait of Juan de Fuca, and waters within 1.2 miles of shore 2) along the Pacific Coast from Cape Flattery to Willapa Bay in Washington, 3) along the Pacific Coast from Newport Bay to Coos Bay in Oregon, 4) along the Pacific Coast from the Oregon-California border south to Cape Mendocino in northern California, and 5) along the Pacific Coast in central California from San Pedro Point south to the mouth of the Pajaro River.

Summary

At the range-wide scale, annual estimates of murrelet populations have fluctuated, with no conclusive evidence of a positive or negative trend since 2001(+0.5 percent per year, 95% CI: -0.5 to +1.5%) (McIver et al. 2021, p. 4). The most recent extrapolated population estimate for the entire NWFP area was 21,200 murrelets (95 percent CI: 16,400 to 26,000 birds) in 2019 (McIver et al. 2021, p. 3). The largest and most stable murrelet subpopulations now occur off the Oregon and northern California coasts, while subpopulations in Washington have steadily declined since 2001 (-3.9 percent per year; 95% CI: -5.4 to -2.4%) (McIver et al. 2021, p. 4).

Monitoring of murrelet nesting habitat within the NWFP area indicates nesting habitat declined from an estimated 2.53 million acres in 1993 to an estimated 2.23 million acres in 2012, a decline of about 12.1 percent (Raphael et al. 2016b, p. 72). Murrelet population size is strongly and positively correlated with amount of nesting habitat, suggesting that conservation of remaining nesting habitat and restoration of currently unsuitable habitat is key to murrelet recovery (Raphael et al. 2011, p. iii). Given likely future increases in forest disturbances that can cause habitat loss, conservation of remaining nesting habitat is especially important.

The species decline has been largely caused by extensive removal of late-successional and old growth coastal forest which serves as nesting habitat for murrelets. Additional factors in its decline include high nest-site predation rates and human-induced mortality in the marine environment from disturbance, gillnets, and oil spills. In addition, murrelet reproductive success is strongly correlated with the abundance of marine prey species. Overfishing and oceanographic variation from climate events and long-term climate change have likely altered both the quality and quantity of murrelet prey species (USFWS 2009, p. 67).

Although some threats have been reduced (e.g., habitat loss on Federal lands), some threats continue and new threats now strain the ability of the murrelet to successfully reproduce. Threats continue to contribute to murrelet population declines through adult and juvenile mortality and reduced reproduction. Therefore, given the current status of the species and background risks facing the species, it is reasonable to assume that murrelet populations in Conservation Zones 1 and 2 and throughout the listed range have low resilience to deleterious population-level effects and are at high risk of continuing or renewed declines. Activities that degrade the existing conditions of occupied nesting habitat or reduce adult survivorship or nest success of murrelets will be of greatest consequence to the species. Actions resulting in the loss of occupied nesting habitat, mortality to breeding adults, eggs, or nestlings will reduce productivity, contribute to continued population declines, and prolong population recovery within the listed range of the species in the coterminous United States.

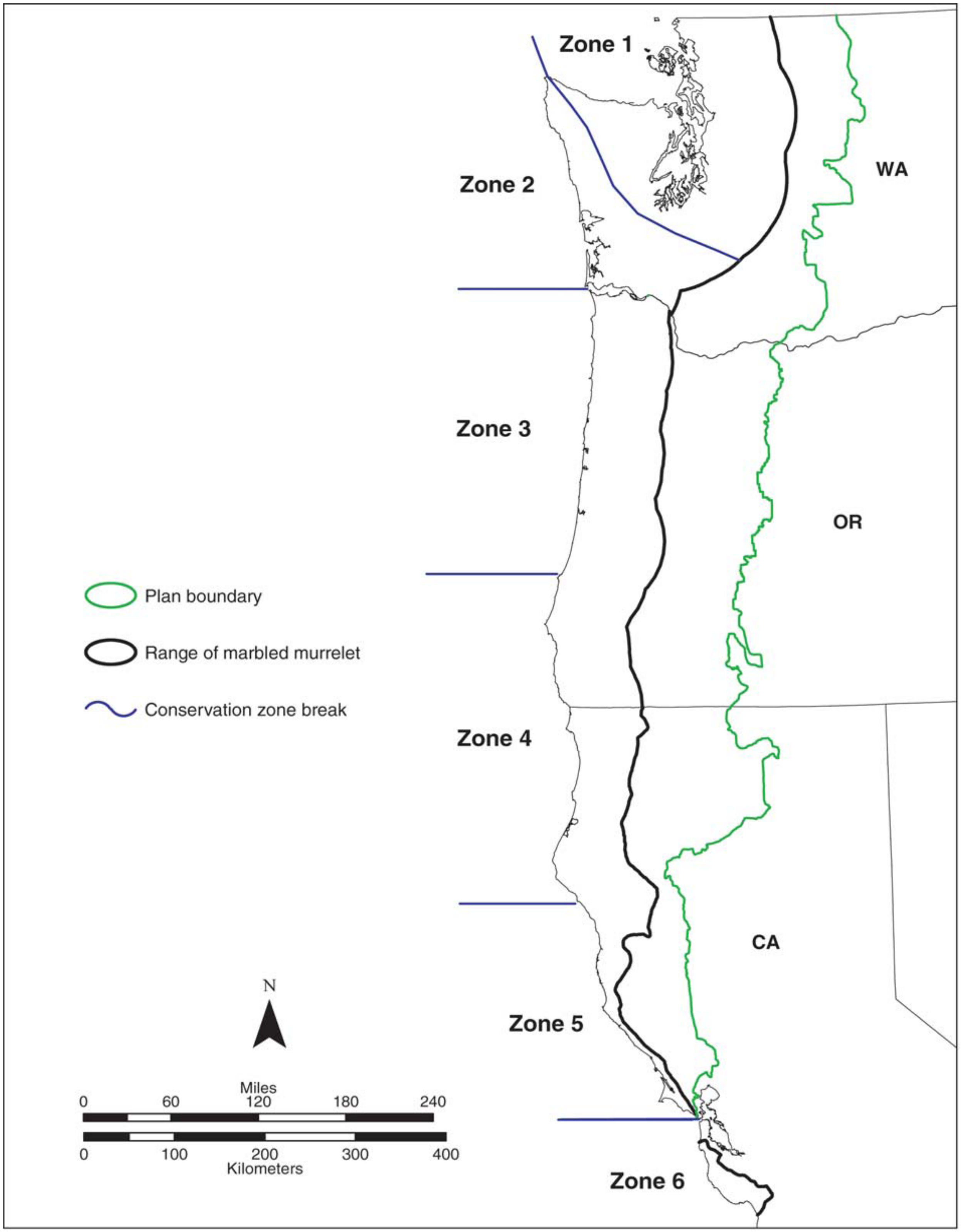

Figure 1. The six geographic areas identified as Conservation Zones in the recovery plan for the marbled murrelet (USFWS 1997). Note: "Plan boundary" refers to the NWFP. Figure adapted from Huff et al. (2006, p. 6).

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APPENDIX C
STATUS OF THE SPECIES: SHORT-TAILED ALBATROSS

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Appendix C
Status of the Species: Short-Tailed Albatross

Species Description

The short-tailed albatross is a large pelagic seabird with long, narrow wings adapted for soaring just above the water's surface. It is within the family Diomedeidae, in the order of tubenosed marine birds (Procellariiformes). Short-tailed albatross had been previously assigned to the genus *Diomedea*, but, following the results of genetic studies by Nunn et al. (1996), the family Diomedeidae was arranged in four genera. The genus *Phoebastria* (North Pacific albatrosses) now includes the short-tailed albatross, the Laysan albatross (*P. immutabilis*), the black-footed albatross (*P. nigripes*), and the waved albatross (*P. irrorata*) (American Ornithologists' Union 1998). The largest albatross species in the North Pacific, the short-tailed albatross has a body length from 84 to 94 cm (2.8 to 3.1 ft) and a wingspan from 213 to 229 cm (7 to 7.5 ft). Shortly after fledging, juveniles develop a distinctive pink color on their bills; the tips of which become progressively blue as the birds age. Sub-adults have dark blackish-brown feathers, but as the birds mature their body feathers become white and adults develop a yellow-gold crown and nape. Short-tailed albatross are the only North Pacific albatross that have a completely white back when mature (USFWS 2008, p. 1).

Status and Distribution

The short-tailed albatross was federally listed as endangered throughout its range, including the United States, on July 31, 2000 (65 FR 147:46643). At the time of listing, designation of critical habitat was determined to be not prudent (65 FR 147:46651). Historically, the short-tailed albatross was probably the most abundant albatross in the North Pacific, with 14 known breeding colonies (Olson and Hearty 2003). However, from the late 1800's, millions were hunted for feathers, oil, and fertilizer (USFWS 2008, p. 3), and by 1949 the species was thought to be extinct. The species began to recover during the 1950's and currently occurs throughout the North Pacific Ocean.

Short-tailed albatross occurrence in the North Pacific Ocean is often focused on areas along the edge of the continental shelf and other hotspots where the contour of the seafloor creates upwelling (Piatt et al. 2006, Suryan et al. 2006, p. 383). Juveniles appear to disperse more broadly throughout the range than adults, and the majority of short-tailed albatross off the west coast of the United States are immature (McDermond and Morgan 1993).

Current Population Status

In 2019 the total, range-wide population of short-tailed albatross was estimated to be 7,365 birds. Short-tailed albatross breed on remote islands in the North Pacific Ocean (Figure 1). While only 25 percent of breeding-age adults return to colonies to breed each year, a large majority of the short-tailed albatross population is still tied to one island. Torishima, a Japanese island that is an active volcano, is estimated to contain 80 to 85 percent of the existing breeding population in two main breeding colonies: Hatsunezaki and Tsubamezaki. Torishima Island hosted 1,011 short-tailed albatross breeding pairs during the 2018-2019 breeding season. Currently the

Torishima Island population of short-tailed albatross is growing at a rate of 8.9 percent per year (USFWS 2020, pp. 4, 6).

The breeding colony in the Senkaku (or Diaoyutai) Islands is in disputed ownership among China, Japan, and Taiwan, and is politically difficult to access. The majority of short-tailed albatross that do not breed on Torishima Island breed in the Senkaku Islands. The estimated population of short-tailed albatross at the Senkaku Islands during the 2018-2019 breeding season was 190 breeding pairs, with the same growth rate as Torishima Island (USFWS 2020, p.4).

In hope of re-establishing a colony 10 chicks were translocated to a former colony site on Mukojima, a non-volcanic island, south of Torishima in the Ogasawara Islands in 2008. All chicks in this group survived to fledging. From 2009 through 2012, an additional 15 chicks per year have been moved to Mukojima and reared to fledging. All but one of the 70 chicks fledged successfully. The first successful fledging on Mukojima Island occurred in the 2015-2016 breeding season. The Ogasawara Islands (Mukojima, Nakoudojima, and Yomejima) are estimated to currently have 2 to 3 nesting pairs of short-tailed albatross and could possibly reach 50 nesting pairs by 2028 (USFWS 2020, p. 4).

In the Northwestern Hawaiian Islands, one pair was breeding at Midway Atoll (having fledged a chick in 2011, 2012, and 2014) and another suspected female-female pair has been attempting to breed at Kure Atoll since 2010. The hatching in 2011 marked the first confirmed hatching of a short-tailed albatross outside of the islands surrounding Japan in recorded history. In 2016, a pair of short-tailed albatross formed on Sand Island, Midway Atoll, and in 2017 they stole a black-footed albatross (*P. nigripes*) and raised the chick to fledging. The following year the pair successfully raised their own chick to fledging. There is an additional short-tailed albatross pair (which is suspected to be a female-female pair) that has been attempting to breed on Kure Atoll, Hawaii, since 2010.

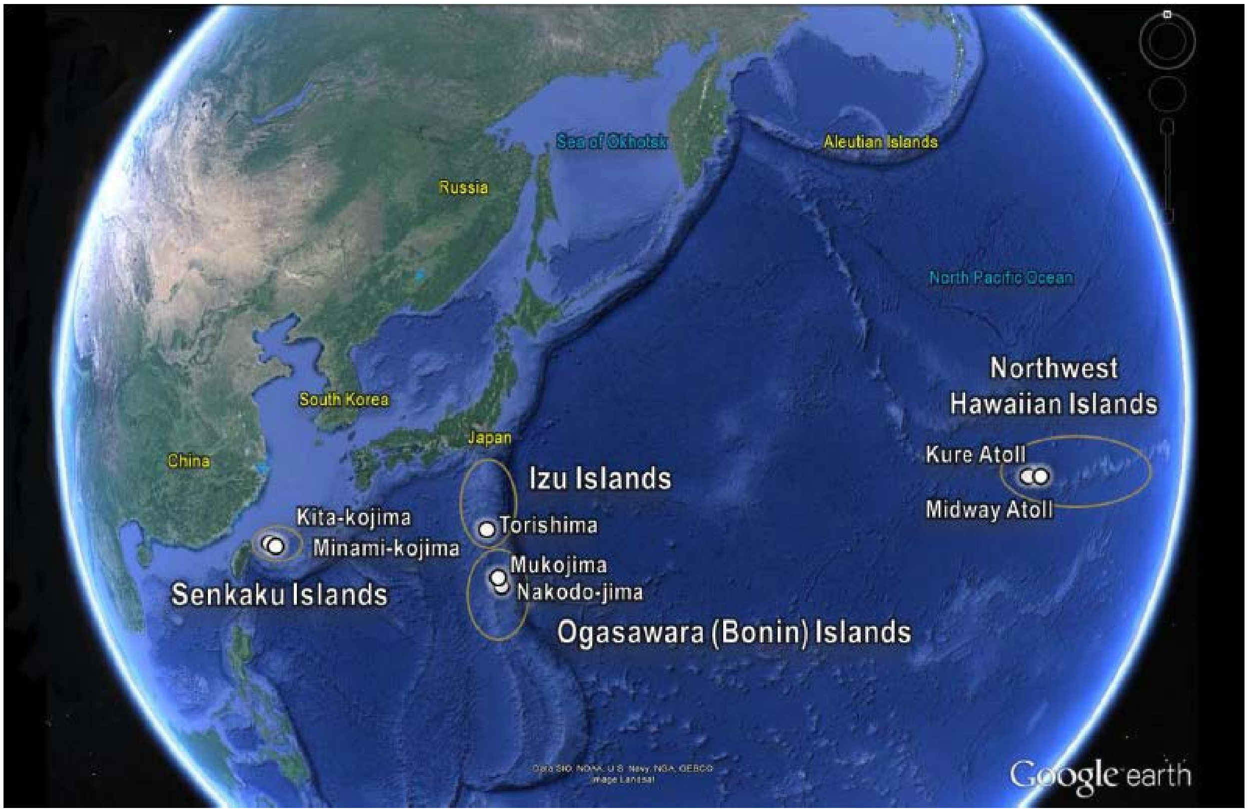

Figure 1. Short-tailed albatross breeding locations in the North Pacific. Specific islands and their island groups are indicated.

Distribution

Juveniles and younger sub-adult birds (up to two years old) have a wider range than adults and can be found in the Sea of Okhotsk, a broader region of the Bering Sea, and the west coast of North America (O'Connor 2013; USFWS 2020, pp. 9-12); Figure 2). Sub-adult birds also travel greater daily distances (mean = 191 km/day [119 mi/day] in first year of flight, 181 km/day [112 mi/day] in second year of flight; (O'Connor 2013)) than adults (133 km/day [83 mi/day]; (Suryan et al. 2007, p. 453)). Northwest Science Center observers on fishing vessels documented 207 short-tailed albatross sightings off the coasts of Washington, Oregon, and California from 2002 to 2019 (USFWS 2020, p. 11). The majority of these sightings are near the continental shelf/slope break with the largest concentration just south of San Francisco Bay and significant concentrations off of Cape Flattery and Aberdeen, Washington, Coos Bay, Oregon, and the mouth of the Columbia River (USFWS 2020, p. 11; Figure 3).

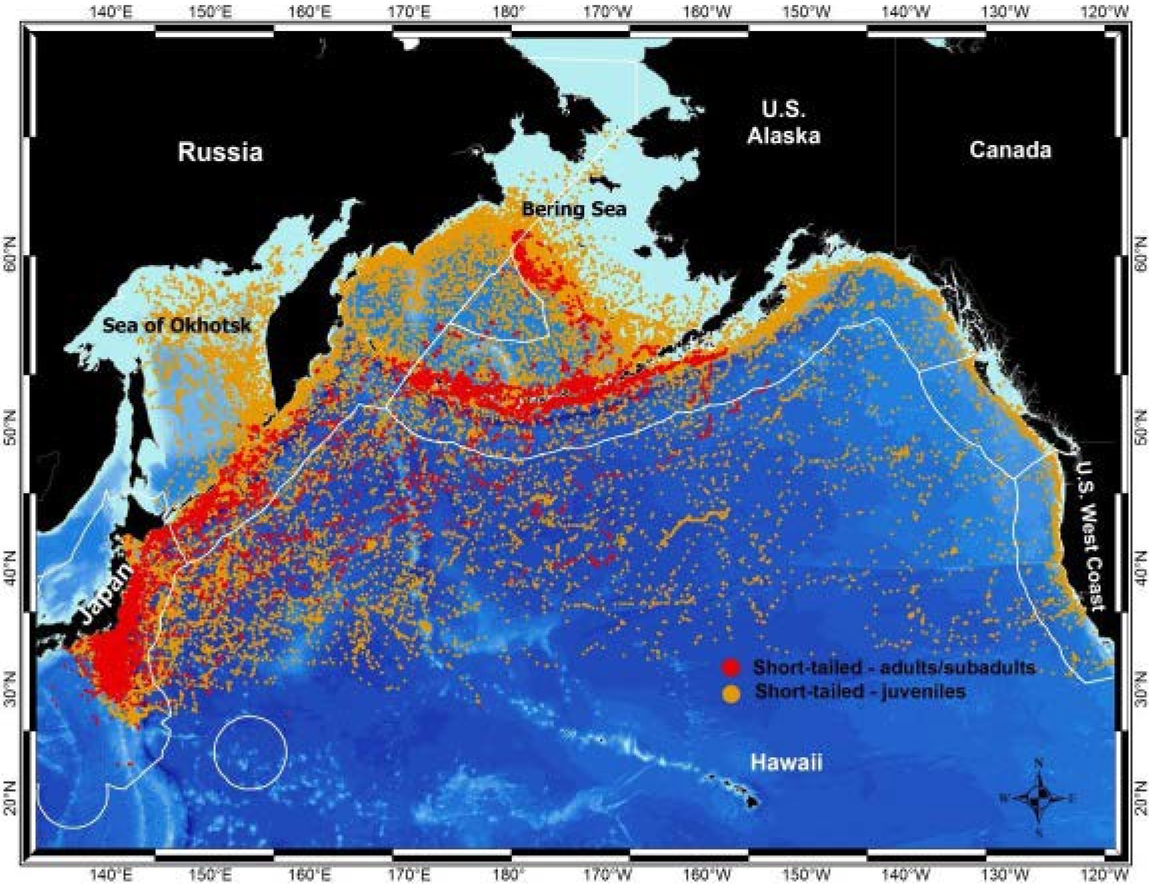

Figure 2. Locations of 99 short-tailed albatrosses tracked between 2002 - 2012, showing adult and juvenile distributions in the North Pacific (Deguchi et al. 2014; Suryan et al. 2006; 2007; 2008a; Suryan and Fischer 2010). White lines represent the exclusive economic zones of countries within the range of short-tailed albatrosses (USFWS 2014, p. 5).

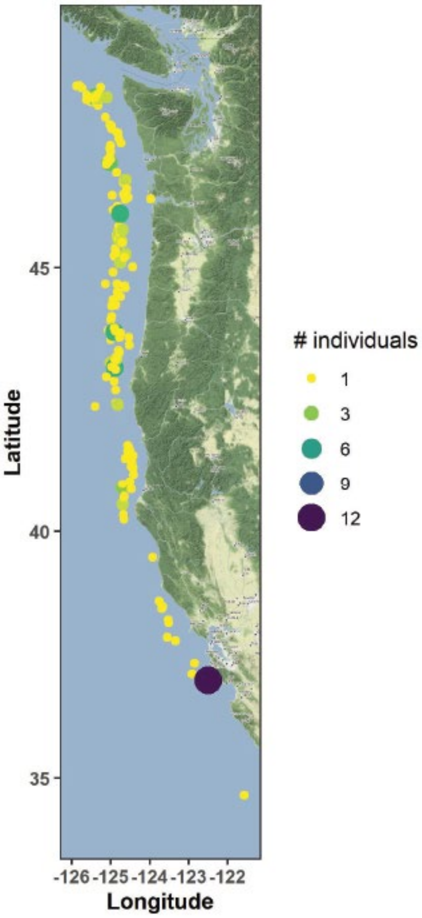

Figure 3. Short-tailed albatross sightings by fisheries observers along the west coasts of Washington, Oregon, and California from 2002 to 2019,

Post-fledging juvenile birds ranged widely throughout the North Pacific rim, and some individuals also spent time in the oceanic waters between Hawaii and Alaska (Deguchi et al. 2014). Although the highest concentrations of short-tailed albatross are found in the Aleutian Islands and Bering Sea (primarily outer shelf) regions of Alaska, sub-adults appear to be distributed along the west coast of the U.S. more than has been previously reported (Guy et al. 2013, p. 231; USFWS 2020, p. 10).

Life History

The short-tailed albatross is a colonial, annual breeding species; each breeding cycle lasts about eight months. Birds may breed at five years of age, but first year of breeding is more commonly at age six (Hasegawa, H., Toho University, pers. comm. in USFWS 2008, p. 10, USFWS 2014). Short-tailed albatross are monogamous and highly philopatric to nesting areas (they return to the same breeding site year after year). However, young birds may occasionally disperse from their natal colonies to attempt to breed elsewhere, as evidenced by the appearance of adult birds on Midway Atoll that were banded as chicks on Torishima (Richardson 1994 in USFWS 2015, p. A-10).

Birds arrive at breeding colonies in October, but as many as 25 percent of breeding age adults may not return to the colony in a given year (Hasegawa, H., Toho University, pers. comm. in USFWS 2008, p. 10). A single egg is laid in late October to late November, and is not replaced if destroyed (Austin Jr 1949). Bi-parental incubation lasts 64 to 65 days. Parents alternate foraging trips that may last 2 to 3 weeks while taking turns incubating. When one bird is foraging, the other stays on the nest without eating or drinking for up to 24 days (Sato, F., Yamashina Institute, pers. comm. in USFWS 2015, p. A-11).

Hatching occurs from late December through January (Hasegawa and DeGange 1982). For the first few days after hatching the chick is fed on stomach oil that is very rich in calories and Vitamin A. This oil also provides a source of water once metabolized, which is important when chicks may be left for several days in high temperatures on dry islands. Soon after hatching, the chicks are fed more solid food, such as squid and flying fish eggs. During the first few weeks after hatching, one adult broods the chick and the other forages at sea. Later, when the chick can regulate its body temperature, both parents leave their chick, while they forage simultaneously. During the brood-rearing period, most foraging bouts are along the eastern coastal waters of Honshu Island, Japan (Suryan et al. 2008a).

Chicks begin to fledge in late May into June (Austin Jr 1949). By late May or early June, the chicks are almost fully grown, and the adults begin abandoning the colony site (Hasegawa and DeGange 1982; Suryan et al. 2008b). The chicks fledge soon after the adults leave the colony. By mid-July, the breeding colony is empty (Austin Jr 1949). Non-breeders and failed breeders disperse earlier from the breeding colony, during late winter through spring (Hasegawa and DeGange 1982, p. 808). In summer (the nonbreeding season), short-tailed albatross disperse widely throughout the temperate and subarctic North Pacific Ocean (Suryan et al. 2007; Sanger 1972 in USFWS 2008, p. 3).

Juvenile (less than 1 year old) short-tailed albatrosses travel much more broadly throughout the North Pacific than adult birds. Seasons of overlap in telemetry tracking of non-breeding adult and juvenile/sub-adult short-tailed albatrosses (those individuals not having to return to the breeding colony to tend eggs or chicks) included summer and early fall (May-September).

During summer and early fall, juvenile short-tailed albatrosses traveled extensively in the Sea of Okhotsk, Russia, and western Bering Sea where few adults ventured. Juvenile short-tailed albatrosses also traveled to the west coast of North America and more extensively throughout the North Pacific transition zone between Hawaii and Alaska. From multi-year tracking studies of juvenile to sub-adult birds, we see that distribution patterns and habitat use of sub-adult birds become similar to adults by age three (Suryan et al. 2013, p. 9).

Foraging Ecology and Diet

Based on necropsies performed by fisheries observers, squid (especially *Gonatus onyx*) is the primary food of short-tailed albatross, accounting for 98.5 percent of the stomach contents of the necropsied birds (Walker et al. 2015, p. 171). Short-tailed albatross predominantly feed by seizing prey from the surface (Hasegawa and DeGange 1982, p. 811; Piatt et al. 2006, p. 394; Prince and Morgan 1987 in USFWS 2014, p. 14). The squid found in the short-tailed albatross stomachs inhabit depths ranging from 200 m to 1,750 m, but their bodies can become positively buoyant after death allowing albatross to scavenge them from the sea surface (Seibel et al. 2000, p.520; Walker et al. 2015, p. 171). In an analysis of historic and current distribution of North Pacific albatrosses, Kuletz et al. (2014, p. 290) speculated that the increase in albatrosses (including short-tailed albatross) and changes in their distribution over the last decade was due to possible increases in squid biomass in the Bering Sea/Aleutian Islands region. Overall, the much higher abundance of albatrosses in the Aleutians compared to the Bering Sea mirrored the relative density of squid, which is estimated to be approximately seven times higher in the Aleutians (Ormseth 2014, p. 1906).

At-sea observations during feeding indicate that short-tailed albatross diet also includes shrimp, fish (including bonitos [*Sarda sp.*], flying fishes [Exocoetidae] and sardines [Clupeidae]), flying fish eggs, and other crustaceans (Hasegawa and DeGange 1982, p. 811; Tickell 1975 and 2000 in USFWS 2008, p. 14). This species has also been reported to scavenge discarded marine mammals and blubber from whaling vessels, and they readily scavenge fisheries offal (Hasegawa and DeGange 1982, p. 811). Short- tailed albatross forage diurnally and possibly nocturnally (Hasegawa and DeGange 1982, p. 811), either singly or in groups (occasionally in the 100's, (Piatt et al. 2006, p. 391)).

Threats

Habitat Alteration and Loss

Habitat destruction from volcanic eruption continues to pose a significant threat to short-tailed albatross at the primary breeding colony on Torishima (USFWS 2020, p. 15). The main colony site, Tsubamezaki, is on a sparsely vegetated steep slope of loose volcanic soil that is subject to severe erosion, particularly during monsoon rains. A landslide at Tsubamezaki buried up to 10 chicks in February 2010 (Yamashina Institute for Ornithology, unpublished data in USFWS 2014, p. 15). Future eruptions or landslides could result in a significant loss to the primary nesting area and the population as a whole. The population on Torishima has expanded to include breeding sites at Hatsunezaki, approximately 1.2 miles (2 km) from Tsubamezaki. The Hatsunezaki site is on more stable ground compared with Tsubamezaki. While the proportion of albatross breeding a Hatsunezaki has increased, a substantial portion of the short-tailed albatross population breeds at Tsubamezaki, and erosion or landslides at that site would cause significant loss to the population (USFWS 2020, p. 15).

Global Changes

Climate change impacts to short-tailed albatrosses could include changes to nesting habitat or changes to prey abundance or distribution. Fortunately, the nesting habitats on Torishima, the Ogasawara Islands, and the Senkaku Islands are high enough above sea level (above 20 m [70 ft]) to avoid inundation by projected sea level rise. Models for the Northwestern Hawaiian Islands indicate nesting habitat used by short-tailed albatrosses on low-lying Midway and Kure Atolls is likely to be lost by the end of the century due to sea level rise and increased storm frequency and intensity (Storlazzi et al. 2013, pp. 28-29).

Sea-ice retreat in the Arctic (see "Habitat or Ecosystem Conditions") may potentially open new foraging habitat or provide a new migration corridor between the Pacific and Atlantic Oceans. A juvenile short-tailed albatross was recently sighted in the Arctic (Chukchi Sea) and evidence from other species (e.g., northern gannet (*Morus bassanus*), ancient murrelet (*Synthliboramphus antiquus*)) indicates some bird species might use ice free portions of the Arctic as a migration or population dispersion route (Gall et al. 2013, p. 56).

Increasing sea surface temperatures may affect short-tailed albatross survival. While no study has looked directly at the effect of sea surface temperature changes to short-tailed albatross, modeling suggests that increases in late winter sea surface temperature decrease the survival of long-lived black-browed albatross (*Thalassarche melanophris*; USFWS 2020, p. 17). Climate change may cause shifts in prey availability or breeding schedules, leading to mismatches in food resources needed for various life stages. Predator-prey mismatches may occur increasingly in marine ecosystems in the future, especially in northern latitudes (USFWS 2020, pp. 17-18).

Commercial Fishing

Sightings of short-tailed albatross in the West Coast National Marine Fisheries Service (NMFS) observer programs are relatively common compared to some other fisheries. For example, in Hawaiian longline fisheries, 100 percent observer coverage has yielded 16 sightings between 2000 and 2010; one in 2000, two in 2004, three in 2007, three in 2008, three in 2009, and four in 2010. Considerably lower observer coverage in the West Coast NMFS Observer Program has yielded 95 short-tailed albatross sightings between 2001 and 2011; four in 2001, 14 in 2002, five in 2003, five in 2004, five in 2005, four in 2006, three in 2007, two in 2008, 16 in 2009, 18 in 2010, and 19 in July 2011. The higher rate of sightings along the west coast compared to Hawaii is consistent with the species' primary use of continental shelf margins when not nesting.

Since 2010, six short-tailed albatross mortalities associated with commercial fisheries have been reported, three in the Alaskan cod fishery one in the Pacific Coast groundfish fishery, and one during bycatch mitigation research in Japan (Table 1).

Table 1. Known short-tailed albatross mortalities associated with North Pacific and west coast fishing activities since 1983

Date	Fishery	Observer program	Bird age	Location	Source
7/15/1983	Net	No	4 months	Bering Sea	USFWS (2008)
10/1/1987	Halibut	No	6 months	Gulf of Alaska	USFWS (2008)
8/28/1995	IFQ sablefish	Yes	1 year	Aleutian Islands	USFWS (2008)
10/8/1995	IFQ sablefish	Yes	3 years	Bering Sea	USFWS (2008)
9/27/1996	Hook-and-line	Yes	5 years	Bering Sea	USFWS (2008)
4/23/1998	Russian salmon drift net	n/a	Hatch-year	Bering Sea, Russia	USFWS (2008)
9/21/1998	Pacific cod hook-and-line	Yes	8 years	Bering Sea	USFWS (2008)

Date	Fishery	Observer program	Bird age	Location	Source
9/28/1998	Pacific cod hook-and-line	Yes	Sub-adult	Bering Sea	USFWS (2008)
7/11/2002	Russian (unknown)	n/a	3 months	Sea of Okhotsk, Russia	YIO (2011)
8/29/2003	Russian demersal longline	n/a	3 years	Bering Sea, Russia	YIO (2011)
8/31/2006	Russian (unknown)	n/a	1 year	Kuril Islands, Russia	YIO (2011)
8/27/2010	Cod freezer longline	Yes	7 years	Bering Sea/Aleutian Islands	NOAA (2010)
9/14/2010	Cod freezer longline	Yes	3 years	Bering Sea/Aleutian Islands	NOAA (2010)
4/11/2011	Sablefish demersal longline	Yes	1 year	Pacific Ocean/Oregon	USFWS (2012)
10/25/2011	Cod freezer longline	Yes	1 year	Bering Sea	NOAA (2011)
5/24/2013	Longline, seabird bycatch mitigation research	No	1 year	Pacific Ocean, Japan	YIO pers. comm. (2014)
9/7/2014	Greenland turbot hook-and-line	Yes	5 years	Bering Sea	USFWS 2020
9/7/2014	Greenland turbot hook-and-line	Yes	Sub-adult	Bering Sea	USFWS 2020
12/16/2014	Pacific cod hook-and-line	Yes	< 1 year	Bering Sea	USFWS 2020

(Data from USFWS unpublished data and Ozaki, K., Yamashina Institute, pers. comm. in USFWS 2014, p. 17)

Domestic and international efforts have been ongoing to minimize fisheries impacts on short-tailed albatross. Threats have been reduced in some areas through the establishment or improvement of regulations to minimize seabird bycatch, including within the U.S. Pacific Coast groundfish fishery and in the longline tuna fishery in Japan (USFWS 2012 and Fisheries Agency of Japan 2009 in USFWS 2014, p. 22). Even with regulatory measures to minimize impacts on short-tailed albatross (including required use of long-line deterrent devices [streamers or tori lines] and implementation of observer programs), bycatch and other injury and mortality associated with fisheries in the North Pacific remain a concern, and the magnitude of the ongoing impacts is uncertain.

Commercial fishing in Russia

Russian longline cod fisheries implemented experimental use of streamers in 2004 - 2008 (Artukhin et al. 2013). The frequency of reported seabird attacks was 5 to 9 times lower on boats with paired streamers, and total catch of fish was 4 to 12 percent higher. The study recommended wide application of streamer line in the Far Eastern Seas of Russia. Although consistent funding has been a problem, the World Wildlife Fund has continued to work with Russian partners to educate the Russian commercial fishing communities about the benefits of using streamer lines and promote their use to reduce seabird bycatch and improve fishing success (World Wildlife Fund 2014). Paired streamers to reduce sea bird bycatch have been deployed on all Pacific Halibut Longline Fisheries Association (LFA) vessels (including 30 to 40 vessels in the Russian Far East), and scientific observers were deployed on LFA vessels in 2015 and 2016 to collect data on bycatch and the performance of paired streamer lines (USFWS 2020, p. 24).

Commercial Fishing in Japan

Japan developed a National Plan of Action for seabird conservation and management (Fisheries Agency of Japan 2004 and 2009 in USFWS 2014, p. 19). In areas where short-tailed albatrosses occur (north of 23 °N latitude), vessels must employ two of the following measures, one of which must be from the first four listed, and streamer lines are obligatory within 32 km (20 mi) of Torishima in October through May: side setting with a bird curtain and weighted branch lines, night setting with minimum deck lighting, streamer (tori) lines, weighted branch lines, blue-dyed bait, deep setting line shooter, and/or management of offal discharge. Japan has also implemented an observer program on their longline and purse seine fisheries to observe bycatch of non-target species, including seabirds (Uosaki et al. 2014). The only observed seabirds incidentally caught north of the 23°N latitude were a black-footed albatross in 2012 and an unidentified petrel in 2013 (Uosaki et al. 2014). However, only a small percentage of deployed hooks are observed.

Japanese fishermen pioneered the use of streamer (tori) lines to deter seabirds, and researchers have continued to assess their use. Researchers have continued to examine methods to improve the effectiveness of streamer lines, Yokota et al. (2011 in USFWS 2014) and Sato et al. (2012) assessed types and lengths of streamers for their effectiveness and found that lighter lines with shorter streamers are as effective as those with long streamers, although the shorter lines are thought to be safer and less likely to tangle. Sato et al. (2013) further examined the use of paired versus single streamer lines and determined that paired lines were more effective than single lines in reducing bait attacks and seabird mortality. The continuing research by Japan has been an important contribution to minimizing longline fisheries bycatch of short-tailed albatrosses.

In 2016, the Japanese Fisheries Agency revised their National Plan of Action applying to tuna and other longline fisheries. The Plan requires fishers to employ methods to reduce bycatch in areas where short-tailed albatross occur and requires streamer lines within 20 miles of Torishima Island during the albatross breeding season (USFWS 2020, pp. 24-25).

Driftnet Fishing in the North Pacific

United Nations General Assembly Resolutions 44/225, 45/197, and 46/215 (United Nations 1989; 1990; 1991) called for a global driftnet moratorium on the high seas by June 30, 1992, and the resolution has been re-adopted biennially. NMFS and the State Department work to implement the moratorium for the United States. According to NMFS (2016, entire) however, high seas driftnet fishing continues to occur in the North Pacific Ocean targeting species of squid and occuring toward the end of the fishing season. Both the target and timing of that fishing increase the threat of short-tailed albatross becoming entangled in nets (USFWS 2020, pp. 25-26). Non-compliance with the driftnet moratorium continues to pose a risk of mortality to short-tailed albatross.

Canadian fishing Operations

Off Canada's west coast, deployment of seabird avoidance gear became mandatory for all hook-and-line groundfish fisheries between 2002 and 2005. Most bycatch monitoring in these fisheries is done by on-board Electronic Monitoring Systems (EMS), with about 10 percent of that data audited after each fishing trip. Although there have been no reported takes of short-tailed albatross bycatch in these groundfish fisheries, an examination of imagery collected by EMS between 2006 and 2012 identified 79 albatross; a third of which were identified only as "albatross species". Based on the proportions of sets audited, an estimated 120 albatross of various species are predicted to have been caught each year in groundfish fisheries. Given the high proportion of albatrosses that are not identified to species and the fact that more than a third of all birds detected during the audits were listed as "unidentified bird", it is likely that small numbers of short-tailed albatrosses are killed each year in Canadian west coast fisheries (Committee on the Status of Endangered Wildlife in Canada and Burger 2013, USFWS 2020, p. 27).

Contaminants

Radiation

Approximately 80 percent of the radiation released from the Fukushima Daiichi Nuclear Plant, which was damaged by a March 11, 2011 earthquake and tsunami, was believed to have entered the Pacific Ocean (Tanabe and Subramanian 2011; Steinhauser et al. 2013 and 2014 in USFWS 2014, p. 23). The area east of the plant is a primary feeding area for nesting short-tailed albatrosses. Although recent analysis has shown no detectable levels of radiation in short-tailed albatross, the impact of these continuing releases on short-tailed albatrosses or their food resources is unknown (USFWS 2014, p. 23).

Organochlorines, pesticides and metals

Albatross and other birds may be exposed to organochlorine contaminants such as polychlorinated biphenyls (PCBs) and pesticides, and to toxic metals (e.g., mercury, lead) via atmospheric and oceanic transport. Vo et al. (2011, entire) examined mercury and methylmercury in tissues of black-footed albatross. They compared the levels of mercury and methylmercury in museum specimens (n = 25) from a 120-year collection period (1880 - 2002). They found no temporal trend in mercury concentrations, but measured significantly higher concentrations of methylmercury through time. Finkelstein et al. (2007 in USFWS 2014, p. 23) found mercury concentrations in blackfooted albatross were associated with decreased immune response. Similar effects would be expected for short-tailed albatross. High concentrations of lead at Midway Atoll are a concern. Taylor et al. (2009, p. 25) described neurological impacts of lead-based paints on Laysan albatross chicks. Since then, the Service has initiated removal and remediation of lead-based paint and contaminated soils on Sand Island (USFWS 2010, entire).

Although only one pair has successfully nested on Midway at Eastern Island, this remediation will reduce exposure to any offspring or future nesting birds on Sand Island. The degree to which any of these or other toxins impact short-tailed albatross remains uncertain and further research is needed to examine the prevalence of these contaminants in short-tailed albatrosses and their impact on the population.

Plastics

Plastics have been found in most, if not all, species of albatross. Donnelly-Greenan et al. (2018, entire) necropsied six short-tailed albatross recovered from fishing gear or opportunistically salvage; the researchers found ingested plastic in four of the six birds. Ingestion of plastics may cause starvation, suppressed appetite and reduced growth, depressed weight at fledging, decreased fat deposition, increased assimilation of toxins including polychlorinated biphenyls and organochlorides, and obstruction in the gut (Auman et al. 1997). Lavers and Bond (in USFWS 2014, p. 25) have examined the role of plastic as a vector for trace metals in Laysan albatrosses. Lavers et al. (2014) studied sub-lethal effects of plastic ingestion in flesh- footed shearwaters (*Puffinus carneipes*) and found birds with high levels of ingested plastic exhibited reduced body condition and increased contaminant load ($p < 0.05$). Tanaka et al. (2013, entire) analyzed polybrominated diphenyl ethers in the abdominal adipose of short-tailed shearwaters (*Puffinus tenuirostris*). Some of the birds were found to contain higher-brominated constituents (BDE 209 and BDE 183), which were not present in their pelagic fish prey. These same birds were found to contain plastics in their stomach. Plastic ingestion is therefore not only a direct dietary risk but may contribute to chronic accumulation of contaminants that adhere to and are absorbed by plastics. Ingested plastics generally do not pass in the intestines of seabirds, most adults have the ability to regurgitate at least some plastic (Laist 1987, p. 321). If seabirds do not regurgitate it, plastic can remain in their stomachs for up to two years (Ryan and Jackson 1987, p. 218). Adult albatross can regurgitate plastics when feeding chicks (Blight and Burger 1997, p. 323; Laist 1987, p. 321; Pettit et al. 1981, p. 840). Plastics ingested by adults anywhere in the North Pacific have the potential to harm chicks due to the length of time plastics can persist within birds and the potential for adults to regurgitate plastics when feeding chicks.

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APPENDIX D

DEMOGRAPHIC MODEL OF MARBLED MURRELET

CONSERVATION ZONE 2

Appendix D

Demographic Model of Marbled Murrelet Conservation Zone 2

Matrix models have been widely used to investigate marbled murrelet (*Brachyramphus marmoratus*) (murrelet) demography, including examinations of the potential demographic effects of various human activities. In this appendix, we describe a matrix model we constructed to illustrate the demographic effects of Navy activities associated with the Northwest Training and Testing (NWTT) program on the murrelet population within Conservation Zone 2, off of the Washington Coast. The results of this model are one of many elements that inform our determination as to whether or not the proposed action will jeopardize the listed distinct population segment of murrelets, or in other words, whether the proposed action will appreciably diminish the likelihood of survival and recovery rangewide.

In particular, the effects to Conservation Zone 2 must be evaluated within the context of its role among all of the Conservation Zones, which are equivalent to recovery units (USFWS 1997, p. 115). Zone 2 is one of the four Zones which currently maintain large enough population sizes and areas of nesting habitat that they may be able to contribute to the recovery and long-term survival of the species (USFWS 1997, p. 116). The two southernmost Zones, Zone 5 and Zone 6, currently have small populations and limited areas of nesting habitat, and cannot be counted on to contribute to recovery and long-term survival. The population size of Zone 5 has been in the vicinity of 100 birds during most surveys (McIver et al. 2020, pp. 10-15). The population size of Zone 6 has fluctuated, but is often estimated to include around 500 individuals (Felis et al. 2020, p. 7).

Note that this model is not statistical. The field of statistics concerns the interpretation of empirically collected data, and is most often used to characterize uncertainty regarding how well a sample represents the whole set from which the sample is drawn. In other words, statistical analysis is a way to make inferences in the face of uncertainty, when that uncertainty comes from sampling error. In contrast, the model described here is not intended or used to analyze empirical data, and the primary sources of uncertainty affecting the model are 1) uncertainty about some details of murrelet life history; 2) uncertainty regarding how the environment is changing, and 3) uncertainty about how those changes will influence murrelet populations. Some of the information used to inform our parameter choice is subject to uncertainty based in part on sampling error, as well as potential bias in measurement techniques, but this uncertainty is less influential in the model than the three sources of uncertainty listed above. Variation in the model outputs stems in large part from variability that we deliberately included in the model to represent environmental variability and a range of conditions, rather than the "statistical noise" that results from empirical sampling.

In the sections below, we review previous matrix models of murrelet demography, as well as empirical and theoretical estimates of murrelet demographic rates. We outline our model structure and methods used to parameterize and run the model. We present the model results, and offer some considerations for the interpretation of these results.

Literature Review

Previous Matrix Models

We reviewed all of the previous matrix models of murrelet demography that we could locate (Table 1). These models fell into two major categories: deterministic models, in which demographic rates (survival and fecundity) are fixed and result in the same proportion of individuals surviving and the same per-individual productivity in every model year, and stochastic models, which incorporate some element of randomness into demographic changes. Stochastic models incorporated environmental stochasticity, in which survival rates and fecundity rates varied between model years, or demographic stochasticity, in which even for years the same survival or fecundity rate, there was a random element, analogous to repeatedly flipping a coin weighted by the demographic rate, to determine how many modeled individuals would survive or reproduce successfully. Many stochastic models incorporated both demographic and environmental stochasticity.

All previous murrelet matrix models were female-based models. These models are predicated on the assumption that the population includes equal numbers of males and females at all life stages, and that a breeding adult male mate is available for each breeding adult female murrelet.

Most previous murrelet matrix models were stage-structured modeled, with at least three life stages: juvenile, subadult, and adult. Several more recent models included two subadult stages (Beissinger and Peery 2007, p. 301; Peery et al. 2006a, p. 1521; Peery and Henry 2010, p. 2415; Peery and Jones 2019, p. 7), and one included five subadult stages (Chambers and WEST 2019, pp. 4-5). Two models (Peery and Henry 2010, p. 2415; Peery and Jones 2019, p. 7) divided the adult population into two stages: breeding adults (presumed to have access to a nest site) and non-breeding adults (presumed to lack access to a nest site). One model was age- rather than stage-structured, allowing for a 25-year lifespan. The authors (McShane et al. 2004, pp. 3-34 – 3-36) made this choice to avoid a small bias associated with models that use a single adult stage. The use of a single adult stage technically implies that the maximum lifespan is infinite, and models built this way may underestimate population decline, though this underestimate is small unless the actual decline is steep (McShane et al. 2004, p. 3-30). Over a reasonable range of survival rates, this assumption leads to survivorship curves in which between 0.1 percent and 2.2 percent of individuals would survive at least to age 40 (Burger 2002, pp. 17-18).

Table 1. Previous matrix models of murrelet population dynamics

Authors	Year	# stages	Deterministic or stochastic	Demographic stochasticity	Environmental stochasticity
Peery and Jones	2019	5	Stochastic	n/a	Beta distribution
Chambers and WEST	2019	7	Stochastic	Binomial distribution	Beta & uniform distributions
Peery and Henry	2010	5	Both	Binomial distribution	Beta distribution
Beissinger and Peery	2007	4	Deterministic	n/a	n/a
Peery and others	2006a	4	Deterministic	n/a	n/a
McShane and others	2004	25	Stochastic	Binomial distribution	Uniform distribution
Beissinger	2002	3	Deterministic	n/a	n/a
Boulanger and others	1999	3	Deterministic	n/a	n/a
Akcakaya	1997	3	Stochastic	Binomial distribution	Lognormal distribution
Beissinger and Nur	1997	3	Deterministic	n/a	n/a
Beissinger	1995	3	Deterministic	n/a	n/a

A few previous models included components representing dispersal, or immigration and emigration. Akcakaya (1997, pp. 5, 14) modeled three populations making up the Conservation Zone 4 metapopulation: Southern Oregon, Northern Humboldt County, and the "Bioregion" including the area to be covered under a draft Habitat Conservation Plan. Juveniles and, to a lesser extent, subadults were modeled as dispersing between these populations. Peery and Jones (2019, pp. 11-13) modeled dispersal of subadults within Washington between a sub-population nesting in habitat managed by Washington Department of Natural Resources (WDNR) and a sub-population nesting in habitat on other lands. Peery and others (2006a, pp. 1521-1523) used a matrix model as one component of a study investigating immigration and emigration, but did not include these processes in the matrix model itself.

Several previous models investigated the effect of particular human actions on murrelet populations. Two of these models (Akcakaya 1997, p. 15; Peery and Jones 2019, pp. 3, 10-11) examined the effects of timber harvest and forest conservation, and therefore these models incorporated an assumption that nesting habitat may be a major limiting factor determining the environmental carrying capacity of murrelets. Peery and Henry (2010, pp. 2415, 2417) also assumed that nesting habitat was a limiting factor, but investigated the effect of corvid control, rather than timber harvest. Other models examined effects of collisions with wind turbines adjacent to nesting habitat (Chambers and WEST 2019, pp. 10-11), gill net fisheries and oil spills (McShane et al. 2004, pp. 3-57 – 3-58), or the loss of ten percent of the population for unspecified reasons (Beissinger 2002, pp. 8-10). These latter models do not incorporate any assumptions about the carrying capacity of nesting habitat, but instead model exponential population growth.

Demographic Parameters

We combed through previous demographic models, empirical field studies of murrelets, and comparative taxonomic analyses to determine the bounds of reasonable ranges for each demographic parameter relevant to the matrix model: survival, breeding propensity, nesting success, overall fecundity, and juvenile ratios. Note that overall fecundity is a product of breeding propensity and nesting success, if nesting success is defined as success in producing a fledgling that survives its first flight to the ocean. Some studies measure nesting success by examining the fecal ring at the nest, which offers no indication as to the chick's survival during its first flight, so fecundity estimates based on these nest success rates are likely to be overestimates. Juvenile ratios are the ratios of juveniles to older birds (subadults and adults) at sea, and can be converted to fecundity rates if the relative proportions of subadults and adults are known.

In some cases, particular study methods are known to produce biased estimates of demographic rates. In particular, we expect that the stress of capture and the burden of carrying a radio tag reduces the survival and breeding success of murrelets that are monitored via radiotelemetry. For example, Peery and others (2006b, pp. 79-82) estimated survival rates in the Central California murrelet population using two methods simultaneously: radiotelemetry and mark-recapture. Survival rate estimates based mark-recapture data ranged from 0.85 to 0.91, in line with other estimates of murrelet survival rates, whereas those based on radiotelemetry data were dramatically lower, between 0.44 and 0.60 (Peery et al. 2006b, p. 83). In a study using larger satellite tags, three of the seven tagged murrelets were found dead after less than one month, and the remaining four also appeared to have ceased movement, indicating either death or loss of the transmitter (Northrup et al. 2018, p. 51). When results from radiotelemetry studies provided substantially lower estimates of key demographic rates than those generated using other methods, we did not necessarily consider these estimates to be part of the reasonable range.

We also combed through the existing literature to determine reasonable ways to represent variability in demographic parameters. In general, previous modeling studies offered the best source of information about how demographic variability would best be modeled. Though most empirical studies offered some measure of variability (for example, standard errors or 95 percent confidence intervals), these measures generally represented sampling error or statistical uncertainty about the data from one given sampling season, rather than variability from year to year stemming from environmental fluctuations. In a demographic model, variability in demographic parameters represents environmental variability rather than sampling error.

Survival

Early models of murrelet demography used survival rates derived from allometric or comparative life history studies. These methods yield adult survival rates between 0.714 and 0.957, though most of the modeling studies used rates between 0.83 and 0.9 (Beissinger 1995, pp. 389-390; Beissinger and Nur 1997, p. B-12; Beissinger and Peery 2007, p. 300; DeSanto and Nelson 1995, pp. 36-37). Using similar methods, Beissinger and Nur (1997, p. B-12) estimated that juvenile survival was likely to be 70.1 percent of the adult survival rate, and second-year (subadult) survival was likely to be 88.8 percent of the adult survival rate. For example, if the adult survival rate is 0.85, according to this scheme, juvenile survival rate would be 0.596 (0.85*0.701) and subadult survival rate would be 0.755 (0.85*0.888). Lank and others (2003, p. 38) discuss juvenile and subadult survival rates of 71 and 88 percent of the adult rate, roughly in line with the 70.1 and 88.8 percent figures cited above.

More recently, empirical estimates of survival rates have been generated from data gathered using mark-recapture and radiotelemetry methods, as noted above, and by examining the age distribution of museum specimens. The after-hatch-year (adult and subadult) survival rate derived from museum specimens was 0.84 (Beissinger and Peery 2007, pp. 299-300). Mark-recapture methods were used in Central California and in British Columbia, yielding after-hatch-year survival rates between 0.829 and 0.929, depending on sex and capture method (Cam et al. 2003, pp. 1121-1122; Peery et al. 2006a, p. 1522; Peery et al. 2006b, p. 83). Survival rates derived from museum specimens and mark-recapture data are consistent with one another and with the information derived from allometric studies.

As noted above, survival rates estimated from radiotelemetry data are substantially lower. Radiotelemetry based estimates of after-hatch-year survival are available from Central and Northern California, and range from 0.531 to 0.733 for annual survival rates (Hébert and Golightly 2006, p. 87; Peery et al. 2006b, 83). Radiotelemetry studies from Oregon and Southeastern Alaska did not report annual survival rates, but provided information about mortality over shorter periods of time. In Oregon, a minimum of zero and maximum of four out of nine birds (44 percent) survived over a 25-day period (depending on the fate assumed for birds when the transmitter indicated a lack of movement, and no remains could be found), which extrapolates to an annual survival rate between 0 and 0.000006 (calculated as 0.44 ^ [365/25]) (Northrup et al. 2018, p. 51). In Alaska, seven or eight of nine (78 or 89 percent) remained alive after an average of 67 days of tracking, which extrapolates to an annual survival rate between 0.25 and 0.53 (calculated as 0.78 ^ [365/25] or calculated as 0.89 ^ [365/25]) (Whitworth et al. 2000, p. 454). Because radiotelemetry methods may increase mortality of murrelets wearing radio tags, we do not consider these estimates to be reliable.

One study in British Columbia used radiotelemetry to measure juvenile survival rates, with a 0.86 survival rate over an 80-day period, which extrapolates to 0.51 over a 1-year period (Parker et al. 2003, p. 209). This estimate of juvenile survival does not account for periods of likely higher mortality risk during and immediately after fledging, or during the bird’s first winter, but as discussed above, the attachment of the radio tag may also increase mortality, so it is not clear whether this estimate may be biased high or low. Because the study did not report adult or subadult survival rates, this estimate cannot be used to test the assumption that juvenile survival is 70.1 percent of adult survival.

After evaluating these sources of information regarding murrelet survival rates, we concluded that reasonable estimates of average annual adult survival would fall roughly between 0.82 and 0.93. We assume that juvenile survival is 71 percent of adult survival, following the lead of Beissinger and Nur (1997, p. B-12) and most previous modeling studies, since this remains the best available information regarding juvenile survival rates. It is not clear whether subadult survival rate should be modeled separately from adult survival rates, since empirical studies generally cannot distinguish between subadults and adults, and report after-hatch-year survival rates instead. If a separate subadult survival rate is needed, the estimate of subadult survival rate as 88.8 percent of adult survival rate (Beissinger and Nur 1997, p. B-12) remains the best available information.

As noted above, previous modeling studies offered the most relevant representations of variability regarding survival rates. Previous models have incorporated survival rates with coefficients of variation (CV) between 0.03 and 0.2. Akcakaya (1997, p. 12) modeled the effect of environmental stochasticity on survival rates in three scenarios. Survival rates were drawn at each time step from lognormal random distributions with CVs of 0.03, 0.07, and 0.10, respectively, for scenarios with low, medium, and high levels of variation. Peery and Jones (2019, p. 26) found that drawing survival rates from a beta distribution with variance of 0.004, equivalent to a CV of approximately 0.07, provided good biological realism in comparison with at-sea survey data, with relatively few years in which survival rates were below 0.75. Peery and Henry (2010, p. 2418) drew survival rates from a beta distribution with variance of 0.01, equivalent to a CV of approximately 0.11, which resulted in occasional years with high mortality, as might be expected to result from a major oil spill. Based on life-history theory, Chambers and WEST (2019, p. 7) modeled survival rates drawn from beta distributions with CVs of 0.1 (for older subadults and subadults), 0.15 (for second-year subadults), and 0.2 (for juveniles). Based on these previous modeling studies, we considered the reasonable range of CVs for survival rates to fall between 0.03 and 0.2.

Breeding Propensity

Breeding propensity is an important component of fecundity, but is not always measured or modeled separately. A variety of techniques have been used to measure or model breeding propensity, including comparative life history, examination of brood patch development on captured birds, tests of blood hormone levels, and radiotelemetry. It is not clear which of these methods might offer the best estimate of breeding propensity, as each method has drawbacks.

Similar to survival, early estimates of breeding propensity were derived from information about breeding propensity in related species. For example, Beissinger and Nur (1997, p. B-11) used information about other alcids as the basis for the assumption that approximately 90% of adult murrelets would nest in any given year. Other modeling studies explicitly modeling or discussing breeding propensity have typically maintained this assumption, but with variations. Chambers and WEST (2019, p. 5) followed the example of McShane and others (2004, pp. 3-5, 3-40) in assuming that 90% of adults would attempt nesting, except in El Niño years, when only 50% would make a nesting attempt. Peery and Jones (2019, pp. 9-11, 18) assumed that adults in possession of a nest site would breed in 9 of every 10 years (i.e., 90% would breed in a given year), but only 60% of adults were in possession of a nest site at the start of the model period, and this percentage varied based on timber harvest and habitat ingrowth.

Empirical estimates have typically indicated lower breeding propensity rates. However, researchers cannot distinguish between subadult and adult murrelets in the field, so empirical estimates would ideally be corrected to account for the presence of subadults in the sample. However, the age or stage distribution of murrelets is rarely known or even estimated for any given population, and the average age at first breeding is not known with any certainty, so it is not clear how best to make this kind of correction.

Radiotelemetry studies offer the clearest evidence that an individual murrelet is actually attempting nesting. Non-breeding murrelets do fly inland during the breeding season (Peery et al. 2004a, p. 349), but a pattern in which an individual bird is consistently present inland on one day and at sea the next (sometimes called an "on-off" pattern) indicates incubation, and consistent daily inland flights are likely to be associated with chick provisioning (Bradley et al. 2004, p. 322; Peery et al. 2004a, p. 349). Radiotelemetry can be used to locate nest sites, as well, providing even clearer evidence of a nesting attempt (for example, Lorenz et al. 2017, p. 307).

However, like survival rates, breeding propensity rates measured via radiotelemetry are likely to be biased low, given that the radio tag imposes a literal burden on the tagged bird, potentially contributing to a decision not to breed in a given year (Peery et al. 2004b, p. 1094). Eggs may be laid or damaged during capture and handling (Bloxton and Raphael 2009, p. 3; McFarlane Tranquilla 2003a, p. 62). If, after losing an egg due to capture, a bird subsequently did not engage in the inland flight behavior indicative of nesting, that bird would be incorrectly classified as a non-breeder. In at least one case, a bird successfully fledged a chick after laying its first egg during capture, but it appears likely that only a minority of failed breeders lay replacement eggs for second attempts (McFarlane Tranquilla et al. 2003b, pp. 77-78). Therefore, even if the radio tag itself does not interfere with breeding propensity, the process of capture is likely to reduce the likelihood of nesting at least for individuals whose eggs are affected by the capture process. Furthermore, handling and especially the attachment of tag may lead to the abandonment of eggs or chicks, as has been demonstrated for the closely-related rhinoceros auklet (*Cerorhinca monocerata*) (Sun et al. 2020, p. 178).

In radiotelemetry studies, the proportion of tracked birds initiating nests varied widely, between 13.1 percent in Washington, and 65 percent in Desolation Sound, British Columbia (Lorenz et al. 2017, p. 313; Bradley et al. 2004, p. 323). Note that both the high and low values come from multi-year studies, and therefore do not represent anomalous conditions in a single year. Radiotelemetry studies showing breeding propensities across the full range between these values have been conducted in Central and Northern California, several different locations in British

Columbia, and Southeast Alaska (Barbaree et al. 2014, p. 177; Hébert and Golightly 2008, p. 101; Lougheed et al. 2002a, p. 323; Manley et al. 2001, p. 16; McFarlane Tranquilla et al. 2003a, p. 112; McFarlane Tranquilla et al. 2005, p. 360; Peery et al. 2004b, p. 1093). All of these breeding propensity estimates are substantially lower than the 90 percent estimate based on related species, as would be expected based on the potential for radiotelemetry methods to interfere with breeding attempts. However, the large range in estimates among studies using the same methods indicates that there is likely to be substantial variation in breeding propensity from one place to another.

Several studies have drawn blood from captured murrelets and tested it for the presence of vitellogenin, a protein involved in the production of egg yolk, which is present at elevated levels in the plasma of female murrelets while they are producing eggs (Vanderkist et al. 2000, p. 759). This test applies only to female murrelets, and murrelets cannot generally be distinguished by sex on the basis of external features, so in most cases blood tests for vitellogenin are done in conjunction with DNA testing for sex chromosomes. Also, vitellogenin levels are expected to remain elevated only during the 2-week period of egg production, so breeding females captured after laying eggs would not register as egg-producers based on these blood tests alone, nor would females captured before they initiate egg production (McFarlane Tranquilla et al. 2003c, p. 514; Vanderkist et al. 2000, p. 763). It is not clear whether all egg-producing females actually initiate nesting attempts by laying eggs in a nest (McFarlane Tranquilla et al. 2003c, p. 514). Most studies testing vitellogenin in murrelets have taken place in British Columbia, where between 38 percent and 55 percent of females caught in dip-nets showed elevated vitellogenin levels (Lougheed et al. 2002a, pp. 322-323, McFarlane Tranquilla et al. 2003c, p. 514, McFarlane Tranquilla et al. 2003d, p. 375; Vanderkist et al. 2000, p. 763). In Central California, 60 percent of females captured for a radio-telemetry study had elevated vitellogenin levels at the time of capture in April or May (Peery et al. 2004a, p. 351; Peery et al. 2004b, p. 1093). As might be expected, the proportion of females with elevated vitellogenin levels changed over the course of the breeding season, with no females captured later than July 6 in British Columbia showing signs of ongoing egg development (Lougheed et al. 2002a, p. 323; McFarlane Tranquilla et al. 2003c, p. 516).

Several studies have also examined captured murrelets for the development of a brood patch. Both male and female murrelets develop brood patches to aid in the incubation of eggs, and the presence of a well-developed brood patch may indicate that the bird is involved in a nesting attempt, but it is not clear that brood patches are reliable indicators of nesting status (McFarlane Tranquilla et al. 2003a, entire). In Southeastern Alaska, between 83.5 and 100 percent of captured after-hatch-year murrelets had fully or partially developed brood patches (Barbaree et al. 2014, p. 177; Whitworth et al. 2000, p. 454). In British Columbia, between 37 and 52.3 percent of captured after-hatch-year birds had fully developed brood patches (McFarlane Tranquilla et al. 2003a, pp. 111-112; McFarlane Tranquilla et al. 2005, p. 360). Another study in British Columbia noted that "almost all" captured after-hatch-year birds had brood patches, but did not offer a percentage (Lougheed et al. 2002b, p. 316). In Central California, half of a sample of radio-tagged murrelets had brood patches at the time of capture (Peery et al. 2004a, p. 351; Peery et al. 2004b, p. 1093).

In some cases, multiple methods have been used to estimates breeding propensity within the same group of birds, and the various potential indicators of breeding propensity do not seem to be closely related to one another. In British Columbia, McFarlane Tranquilla and others (2003a, p. 112) found that when brood patches were assessed at the time of radio-tagging, there was no relationship between brood patch at time of capture and eventual incubation behavior. Similarly, in a group of female murrelets that were assessed for plasma vitellogenin and then radio-tagged, only half of the egg-producing females went on to display incubation behavior, and only 60 percent of those that displayed incubation behavior were producing eggs at the time of capture, indicating little if any association between the two potential indicators of breeding propensity (McFarlane Tranquilla 2001, p. 111). In Northern California, 79 percent of birds that initiated nesting had brood patches at the time of capture, but only 43 percent of birds with brood patches made detectable nesting attempts (Hébert and Golightly 2006, p. 130). Similarly, in Central California, 89 percent of birds that initiated incubation had brood patches, but only half of the birds with brood patches at capture went on to display incubation behavior (Peery et al. 2004a, p. 351; Peery et al. 2004b, p. 1093). In the Central California study, vitellogenin levels were more closely related to a female's eventual incubation behavior: all females displaying incubation behavior had elevated vitellogenin levels at capture, and 71 percent of females with elevated vitellogenin levels at capture went on to display incubation behavior (Peery et al. 2004a, p. 351; Peery et al. 2004b, p. 1093). However, based on these comparisons, it does not appear that any of these indicators, on its own, allows for a good estimate of the true breeding propensity within a given murrelet population.

Some additional information about breeding propensity comes from studies that were not designed to measure or estimate this value, but nonetheless provide some relevant information. For example, a study investigating the prevalence of "replacement breeding" used a combination of radiotelemetry and vitellogenin measurement to classify 78.7 percent of monitored birds as incubators, failed nesters, replacement nesters, with the other 21.3 percent consisting of non-nesters (McFarlane Tranquilla et al. 2003b, p. 78). Another report describes nest monitoring of a single Northern California pair over a period of ten years, during which they made nesting attempts seven times (Golightly and Schneider 2011, p. 4). This equates to a breeding propensity rate of 70 percent for this one pair. Strong and others (1995, p. 352) note that breeding propensity is likely to be lower, perhaps less than 50 percent, in areas where major nesting habitat losses have occurred within the last 20 years or so.

In summary, empirically-derived breeding propensity estimates range from 13 to 100 percent. Values at the low end of this range (lower than 37 percent) come only from radiotelemetry-based estimates, which are probably biased low. Values at the high end of the range (higher than 65 percent) come only from examination of brood patches, which may not reliably indicate that a bird is involved in an actual nesting attempt. Therefore, considerable uncertainty remains regarding the reasonable range for murrelet breeding propensity, and it seems possible that for a particular population, the breeding propensity could be any fraction larger than 13.1 percent, the lowest empirically-derived value, up to around 90 percent, the value originally derived from comparative life history information and sometimes still used in modeling studies.

Nesting success

Nesting success is the other important component of fecundity. Empirical estimates of nesting success are based on nest monitoring, on evidence gathered at the nest, often by tree-climbing, or on the duration of inland flight patterns associated with incubation and chick rearing, as observed via radiotelemetry. Some of these estimates are given as separate estimates of hatching success and fledging success, and some estimates extend only to the middle of the chick-rearing period. Other studies have estimated nesting success on the basis of total fecundity estimates (for example, those derived from the ratio of juveniles to older birds at sea) divided by estimates of breeding propensity.

Early studies reporting nest success estimates were based on monitoring of opportunistically-located nests. Nelson and Hamer (1995a, pp. 90-91) gathered nest monitoring data from across the species range; among nests where the ultimate fate could be documented, 28 percent successfully fledged chicks. Beissinger and Peery (2003, p. 5) examined the subset of these nests from Oregon and Washington and found that the nest success rate was 43 percent. Boulanger and others (1999, p. 56) combined the information from Nelson and Hamer with information from additional nests in British Columbia to arrive at a nest success rate of 34 percent. At one nest site in Northern California, which was monitored over ten years, two of seven nesting attempts (29 percent) resulted in successful fledging (Golightly and Schneider 2011, p. 4). In Alaska, Naslund and others (1995, p. 15) found that every one of the nests with known fates had failed.

Several studies monitored nests found via radiotelemetry or a combination of radiotelemetry and opportunistic nest locations. In these studies, fledging success is generally determined when fledging is actually observed, sometimes by video monitoring, or when the tree is climbed after the nesting season to find a complete fecal ring and down. Using these methods, between 25 and 33 percent of nests with known fates in British Columbia were found to be successful (Hull et al. 2001, p. 1039; Manley 1999, p. 34). In a Northern California radiotelemetry study, 40 percent of nests that were initiated achieved hatching success (excluding those subjected to experimental disturbance), and of the chicks with known fates (not excluding those subjected to experimental disturbance), 42 percent fledged (Hébert and Golightly 2006, pp. 127-128). Combining the success rates for each of these stages yields an overall nest success rate of 17 percent (0.40 * 0.42 = 0.17). Two studies also included some “successful” nests where success was determined when radiotelementry information indicated that parents visited the nest to feed the chick for the full nestling phase (Lorenz et al. 2019, p. 159; Peery et al. 2004b, p. 1092). In Washington and British Columbia, 20 percent of the nests of radio-tagged birds successfully fledged young, and when the fates of other, opportunistically monitored nests were also considered, the nest success rate was 25 percent (Lorenz et al. 2019, pp. 160-162). In Central California, only 16 percent of nests successfully fledged young (Peery et al. 2004b, p. 1093). Another study in British Columbia estimated a nesting success rate of 48 percent (Bradley et al. 2004, pp. 323-324). This estimate was based on a combination of radiotelemetry observations used to determine nesting success through the incubation and “mid-chick” (at least ten days after the estimated hatch date) phases, and tree climbing to determine fledging success at a subset of radio-monitored sites.

Some radiotelemetry studies defined nests as “successful” when they observed regular adult inland flights consistent with chick provisioning for at least 20 days, or at least several times during the “mid-chick” period as defined above. This method is likely to result in an overestimate of nest success, given that the nestling phase lasts between 27 and 40 days (Nelson and Hamer 1995b, p. 60), and may fail at any point prior to fledging. Nonetheless, in Southeastern Alaska, nest success measured in this way was only 20 percent (Barbaree et al. 2014, p. 177). Other studies in British Columbia that used this method reported success rates to the “mid-chick” period ranging from 59 to 69 percent (Bradley et al. 2004, p. 324; Zharikov et al. 2006, p. 114). One British Columbia study reported incubation success, which, if used as a measure of nest success, would be an even greater overestimate. Among birds that initiated incubation, 59 continued exhibiting the “on-off” pattern for the full 30-day incubation period, and were presumed to have successfully completed incubation (McFarlane Tranquilla et al. 2003, p. 78).

In short, nest success estimates derived from empirical data vary widely, from 16 to 48 percent. Success to the mid-chick phase, which does not account for mortality in the later part of nestling development, varies even more widely, from 20 to 69 percent. The high and low values for nest success and success to the mid-chick phase all come from multi-year studies, and therefore do not reflect anomalous conditions in a single year. However, all include small sample sizes, and may be subject to large sampling errors. All of these studies are radiotelemetry studies, which may reduce the nest success of tagged birds. Some include information from haphazard or non-representative samples of nests, and it is not clear how this method might influence the results.

A few studies have addressed correlations between nest success and features of the nest location. For example, success to the mid-chick stage, as measured by radiotelemetry observations of inland flights, varied depending on whether the site was accessible or inaccessible to researchers, for example, due to extremely steep slopes or lack of nearby helicopter landing areas. Inaccessible nests had higher success to the mid-chick phase, 76 percent, compared with 57 percent at nests located in more accessible areas (Bradley et al. 2004, p. 324). Nests located close to the tree trunk are reported to have a nest success rate of 71 percent (Manley and Nelson 1999, p. 40). Nests located in the interior of forest stands are reported to have a nest success rate of 55 or 62 percent, whereas nests located near the edge have a success rate of only 38 percent (Manley and Nelson 1999, p. 40; McShane et al. 2004, p. 4-89). In their demographic model of the effects to murrelet populations of WDNR forest management alternatives, Peery and Jones (2019, p. 19) included differential nest success rates (from 38 to 55 percent) for nest locations in varying edge and interior forest conditions. Many nest failures are apparently due to predation (Golightly and Schneider 2011, p. 4; McShane et al. 2004, p. 2-16; Peery et al. 2004b, pp. 1093-1094), leading Peery and Henry (2010, pp. 2416-2418) to model varying nest success depending on corvid predation and the effectiveness of corvid control. Nest success is likely to vary for a variety of other reasons, as well, including prey availability, falls from the nest, inviable eggs, and possibly exposure to algal toxins (Lorenz et al. 2019, pp. 160, 162; Peery and Henry 2010, p. 2418; Shearn-Bochsler et al. 2014, pp. 934-935). To account for year-to-year variation in nest success, Chambers and WEST (2019, p. 7) drew yearly nest success rates from a beta distribution with a coefficient of variation of 0.25.

In summary, empirically-derived estimates of average nest success range from 16 to 48 percent, with some higher estimates for nests with particular features, such as those located in interior forest, or close to the tree trunk. All estimates are based on radiotelemetry information, nests that were accessible enough to be found and monitored without radiotelemetry, or some combination of the two, and therefore may be biased low. For these reasons, considerable uncertainty remains regarding the reasonable range for murrelet nest success, and it seems possible that for a particular population, the breeding propensity could be any fraction larger than 16 percent, the lowest empirically-derived value, up to around 50 percent, a value slightly larger than the highest empirical estimate (which is likely to be biased low).

Juvenile Ratios

Strictly speaking, fecundity refers to the number of offspring produced per adult. For murrelets, which produce at most one fledgling per pair per year, annual fecundity is more or less equivalent to the breeding propensity rate multiplied by the nest success rate, divided by two. This method of calculating fecundity does not account for mortality that occurs after fledging; for example, a juvenile's first flight from the nest to the sea may be perilous, and little is known about the success rates of these flights, though some authors have ventured rough estimates of 90 to 95 percent (Beissinger and Nur 1997, p. B-11; Burger 2002, p. 14). For murrelets, some researchers have defined fecundity as the number female offspring reaching the ocean per female adult, which would need to account for success during the first flight (Beissinger and Nur 1997, p. B-11, Burger 2002, p. 13). Observations of the ratio of juveniles to older birds at sea offers an index of fecundity that does account for these factors, but there are difficulties involved in estimating fecundity based on the juvenile ratio, as well.

Four factors create difficulties in the use of juvenile ratios as measures of productivity: the murrelet's asynchronous breeding chronology; the difficulty of distinguishing different age or stage classes; the potential for differential habitat usage by different age classes; and migratory movements during the late breeding season. Various methods and mathematical corrections have been proposed to account for these difficulties, but none of the methods have been adopted universally, and some of the corrections may introduce other errors into the estimate.

Murrelets breed asynchronously, and sometimes make second a nesting attempt if a first attempt fails. Fledging may occur over a large portion of the breeding season. For example, a review of radiotelemetry and nest site data from Oregon, Washington, and British Columbia indicates that the earliest fledging dates are in mid- to late June, and the latest fledging dates are in mid- to late September; and more recent information indicates that fledging sometimes occurs even earlier (Havron 2012, p. 4; Pearson 2018, in litt.; USFWS 2012, p. 2). Ideally, at-sea surveys used to estimate juvenile ratios should occur after all juveniles have fledged, but in practice most at-sea surveys occur earlier, during the peak of the breeding season. Therefore, many studies have combined observed juvenile ratios with information about the distribution of fledging dates to arrive at a corrected juvenile ratio that accounts for nestlings that have yet to fledge.

During the peak of the breeding season, as juveniles begin to fledge, they are easily distinguished from after-hatch-year birds by their black and white plumage, which contrasts with the brown alternate (breeding) plumage of the older birds (Strong 1998, p. 6). As the breeding season continues and more juveniles fledge, after-hatch-year birds progress through the phases of prebasic molt. Birds in late stages of prebasic molt are difficult to distinguish from juveniles, though observers with good training are able to differentiate the two, at a short enough distance and in good light conditions (Strong 1998, pp. 6, 8). When observers are inexperienced, or cannot get a good view of the bird, it is not possible to make the distinction (Havron 2012, pp. 4-5; Strong 1998, p. 8). Conducting productivity surveys late in the breeding season is useful, because more juveniles have fledged and less correction of the observed data are needed, but the difficulty of distinguishing juveniles from older birds can lead to many observations of birds of "unknown" stage. Adults and subadults cannot be distinguished in at-sea surveys, if at all (Beissinger and Peery 2007, p. 298), so juvenile ratios are typically expressed as the ratio of juveniles to all after-hatch-year birds. However, to estimate fecundity from a juvenile ratio, the proportion of the population consisting of subadults must be accounted for. Because these ratios cannot be measured in practice, they must be estimated based on demographic models or life history theory (for example, Peery 2004b, p. 1094).

In some areas, it appears that juvenile murrelets differ from older birds in their patterns of marine habitat use (Andersen and Beissinger 1995, pp. 80-81; Beissinger 1995, pp. 387-389; Kuletz and Piatt 1999, pp. 257, 260; Wong et al. 2008, pp. 311-312), though in other areas they do not differ substantially (Kuletz and Kendall 1998, p. 457; Lougheed et al. 2002b, p. 316; Peery et al. 2007, p. 230). Also, in some areas, both juveniles and older birds migrate during the late breeding season away from the waters used earlier in the breeding season, but the timing of juvenile and adult movements may differ (Kuletz and Kendall 1998, pp. 450-453; Lougheed et al. 2002b, p. 315; Wong et al. 2008, p. 311). Therefore, Kuletz and Kendall (1998, p. 456) proposed that a better productivity index would be expressed as the ratio between numbers of juveniles observed during late-season surveys and numbers of adults observed earlier in the breeding season, during incubation. However, this method does not appear to be widely used.

Uncorrected (or "raw") juvenile ratios have been reported, with single-season ratios ranging from 0, if no juveniles were observed on surveys, to 0.4 (Beissinger 2002, p. 6; Beissinger and Nur 1997, p. B-26; Peery et al. 2007, p. 235; Strong 2014, p. 22; Strong 2016, p. 14; Wong et al. 2008, p. 311). Raw, single-season juvenile ratios in between these values have been reported throughout the listed range and beyond, for Central California (Beissinger and Nur 1997, p. B-26; Beissinger and Peery 2003, p. 19; Beissinger and Peery 2007, p. 300; Peery et al. 2004b, p. 1094); Southern Oregon and Northern California (Beissinger 1995, pp. 388-389; Beissinger 2002, p. 6; Beissinger and Nur 1997, p. B-26; Beissinger and Peery 2003, p. 19; Strong 2014, p. 22; Strong 2016, p. 14; Strong 2018, p. 22); Northern and Central Oregon (Beissinger 1995, pp. 388-389; Beissinger and Nur 1997, p. B-26; Beissinger and Peery 2003, p. 19; Strong 2014, p.

22; Strong 2015, p. 6; Strong 2017, p. 6; Strong 2019, p. 6); the inland waters and straits of Washington (Beissinger and Peery 2003, p. 19; Hamer and Brennan 1994, p. 11; Lorenz and Raphael 2018, p. 206); British Columbia (Beissinger 1995, pp. 388-389; Lougheed et al. 2002b, p. 314; Ronconi and Burger 2008, p. 255; Wong et al. 2008, p. 311); and Alaska (Andersen and Beissinger 1995, p. 80; Kuletz 2005, p. 108; Kuletz and Kendall 1998, p. 252). For a single season, the maximum reported raw juvenile ratio within a single season is 0.23, observed in Northern California and Southern Oregon (Zone 4) in 2006 (Strong 2016, p. 14).

The failure to detect any juveniles on surveys could result from widespread nesting failure, or from some combination of predominantly late nesting in a given year with surveys being conducted too early in the season; for example, some of the surveys in which no juveniles were observed were conducted only between July 13 and July 31, whereas a large proportion of juveniles would be expected to fledge after July 31 (Strong 2014, p. 22; Strong 2016, p. 14; USFWS 2012, p. 2). Even when single-season raw juvenile ratios can be surveyed at the optimal time, after nearly all juveniles have fledged but before post-breeding season migratory movements are underway or plumage changes make juveniles difficult to distinguish, substantial variation would be expected from year to year, reflecting years with good and poor conditions for nesting (e.g., Ronconi and Burger 2008, p. 255; Strong 2018, pp. 8-11; Strong 2019, pp. 8-9). Over the long-term, multi-year average juvenile ratios provide a better indication of whether the population is likely to exhibit sustained growth, stability, or decline. Raw juvenile ratios, averaged over multiple years, vary from 0.015 in Conservation Zone 5, California (Strong 2018, p. 22) to 0.056 in Washington’s inland waters (Beissinger and Peery 2003, p. 19), with values in between reported for Central California (Beissinger and Peery 2003, p. 19; Peery et al. 2007, p. 235); Northern California and Southern Oregon (Beissinger and Peery 2003, p. 19; Strong 2014, p. 22; Strong 2018, p. 22); Central and Northern Oregon (Beissinger and Peery 2003, p. 19; Strong 2014, p. 22); Washington’s inland waters (Lorenz and Raphael 2018, p. 206); and British Columbia (Lougheed et al. 2002b, p. 314).

Corrected ratios are more reliable indicators of productivity, since they depend less on the exact timing of the surveys, and are designed to represent the ratio between the number of chicks successfully fledged over the whole breeding season and the number of older birds in the population. However, it is essential to use a correction factor appropriate to the place and time of the surveys; for example, fledging may occur during a different date range in different places, and post-breeding migratory movements affect the ratio differently in different places. Corrected ratios are more comparable to one another, as compared with raw ratios, as long as we assume that researchers offering corrected ratios have made the most appropriate corrections for their respective locations. Corrected ratios for a single season and location range from 0 to 0.45 (Beissinger 2002, p. 6; Beissinger 2002, p. 6; Felis et al. 2020, p. 9; Kuletz 2005, p. 121; Peery et al. 2007, p. 234). Corrected, single-season juvenile ratios in between these values have been reported throughout the listed range and beyond, for Central California (Beissinger and Nur 1997, p. B-26; Felis et al. 2020, p. 9; Peery et al. 2007, p. 234); Southern Oregon and Northern California (Beissinger 1995, pp. 388-389; Beissinger 2002, p. 6; Beissinger and Nur 1997, p. B-26; Strong 2014, p. 20; Strong and Falxa 2012, p. 11); Northern and Central Oregon (Beissinger

1995, pp. 388-389; Beissinger and Nur 1997, p. B-26; Strong 2014, p. 20; Strong 2015, p. 6); the inland waters and straits of Washington (Lorenz and Raphael 2018, p. 206); British Columbia (Beissinger 1995, pp. 388-389; Ronconi and Burger 2008, p. 255); and Alaska (Kuletz and Kendall 1998, p. 254). For a single season, the maximum corrected ratio reported within the listed range is 0.157, observed in Northern Oregon in 1995 (Beissinger and Nur 1997, p. B-26).

Corrected ratios of 0, similar to raw ratios of 0, could result from widespread nesting failure, or from surveys being done early enough that few or no juveniles had fledged yet. When surveys are done too early, these scenarios cannot be distinguished from one another. When surveys are conducted at the appropriate time and the correction factor is accurate, variation in corrected ratios would be expected due to a combination of varying environmental conditions, demographic stochasticity (especially in small populations), and sampling error. A multi-year average of corrected juvenile ratios provides the best indication of whether a population is likely to exhibit sustained growth, stability, or decline. Direct measurement of demographic rates may provide information that is equally good, or even better, but measuring these rates is difficult and costly, and therefore large-scale, long-term demographic datasets are not available (see previous sections). Multi-year averages of corrected juvenile ratios range from 0.032 in Central California (Peery et al. 2007, p. 234) to 0.13 in Desolation Sound, British Columbia (Lougheed et al. 2002b, p. 314), with values in between reported for Central California (Beissinger and Peery 2003, p. 19; Felis et al. 2020, p. 9; Peery et al. 2006a, p. 1522; Peery et al. 2007, p. 234); Northern California and Southern Oregon (Beissinger and Peery 2003, p. 19; Strong and Falxa 2012, p. 11); Central and Northern Oregon (Beissinger and Peery 2003, p. 19); and Washington's inland waters (Beissinger and Peery 2003, p. 19; Lorenz and Raphael 2018, p. 206). In the listed range, the maximum reported multi-year average corrected ratio is 0.089, observed in Washington's inland waters between 1993 and 2002 (Beissinger and Peery 2003, p. 19). More recent research showed that, during the years from 1995-2012, the corrected juvenile ratio in the San Juan Islands showed no significant directional trend, and averaged 0.07 (Lorenz and Raphael 2018, p. 206).

In comparison, the historical ratio of juveniles to after-hatch-year birds, among museum specimens collected during and immediately after the breeding season, was 0.297 (Beissinger and Peery 2007, p. 300). Assuming a survival rate of 0.84 for subadults and adults, this juvenile ratio would be associated with a stable or slightly increasing population (Beissinger and Peery 2007, p. 299). With higher subadult and adult survival rates, between 0.85 and 0.9, the juvenile ratios needed to sustain population stability range from 0.176 to 0.279 (Beissinger and Nur 1997, p. B-13). While a number of single-season estimates of the juvenile ratio are greater than 0.176, no murrelet population has exhibited long-term juvenile ratios in this range.

Juvenile ratios have not been reported for Washington's outer coast, though they have been measured elsewhere in Washington, as well as to the north and south in British Columbia and Oregon. We assume that juvenile ratios along the outer coast of Washington are most likely to be within the span of those observed in the rest of the listed range, or in British Columbia. That is, we expect that the average juvenile ratio is likely to be approximately 0.13 or less, in keeping with corrected ratios recorded for British Columbia and the listed range. We assume that average juvenile ratios greater than 0.16 would be quite unrealistic, as this would be higher than the highest single-year value observed within the listed range.

Fecundity

Some researchers have estimated fecundity using observed juvenile ratios as a starting point. Fecundity can be estimated from a corrected juvenile ratio by accounting for the proportion of the population consisting of subadults. Because subadults are not distinguishable from adults during at-sea surveys, the proportion of subadults is must be calculated using a population age structure derived from demographic modeling. Beissinger (1995, p. 389) used juvenile ratios from Oregon and British Columbia to arrive at a rough fecundity estimate of 0.05 to 0.1 fledged juveniles per breeding-aged adult, without providing an explicit correction for subadults. Peery and others (2004b, p. 1094) used date-corrected juvenile ratios, and corrected for subadults using a demographic model, to arrive at fecundity estimates between 0.029 and 0.055 fledged juveniles per breeding-aged adult in Central California.

Fecundity may also be estimated by multiplying together breeding propensity and nest success. In Central California, this procedure generated an estimate of 0.027 juveniles per breeding-aged adult (Peery et al. 2004b, p. 1094). In British Columbia, this method generated estimates between 0.19 and 0.23 juveniles per breeding-aged adult (Bradley et al. 2004, p. 324), among the highest fecundity estimates reported for the species. However, these estimates were based on breeding propensities between 0.8 and 0.95, whereas the observed breeding propensity in this study was conservatively estimated at 0.65 (Bradley et al. 2004, p. 323). Beissinger (1995, p. 389) noted that when fecundity is estimated by multiplying breeding propensity and nest success, an additional correction must be made to account for juvenile survival between leaving the nest and arriving at sea. The first flight from the nest to the sea is apparently hazardous, given that grounded juveniles are sometimes found, but the rate at which they succeed in reaching the ocean is unknown. Therefore, most models do not explicitly include this component of fecundity.

Previous studies offer little information regarding year-to-year variation in fecundity over the long term. Long-term studies of juvenile ratios indicate that there is likely substantial year-to-year variation, but the variability of these studies represents a combination of actual variability in fecundity, variability in other factors, such as the timing of breeding, and sampling error. Furthermore, only two of these studies report CVs or standard deviations, which can be divided by the mean to compute the CV; others report standard errors instead. In a 3-year data set from British Columbia, the CV of the corrected juvenile ratio was 0.7 (Lougheed et al. 2002b, p. 314). In an 18-year data set from the San Juan Islands, a CV of 0.29 can be computed from the mean (0.07) and standard deviation (0.02) of the corrected juvenile ratio (Lorenz and Raphael 2018, p. 206). Because the variability measured in these studies combines multiple sources of variation, it does not seem appropriate to adopt the coefficients of variation derived from these studies directly into our modeling exercise.

As with survival rates, the most relevant representations of variability regarding fecundity would come from previous modeling studies. We found only one modeling study that incorporated a coefficient of variation for specifically for fecundity. Akcakaya (1997, p. 12) modeled the effect of environmental stochasticity on fecundity in three scenarios. Survival rates were drawn at each time step from lognormal random distributions with CVs of 0.2, 0.35, and 0.3, respectively, for scenarios with low, medium, and high levels of variation. Other studies modeled variability in only one component of fecundity, or used different methods to model each component of fecundity. Peery and Jones (2019, p. 27) used a low mean nest success rate for nests in forest edge and a high mean nest success rate for nests in interior forest; the variance of nest success was 0.016 in both cases, which translates to CVs of 0.33 for forest edge and 0.23 for interior forest. Two studies modeled breeding propensity as being fixed at 0.9 in most years, and 0.5 in El Niño years; in both of these studies, a given year had a 12 percent chance of being an El Niño year (Chambers and WEST 2019, p. 5; McShane et al. 2004, p. 3-40). This distribution of breeding propensities has a CV of approximately 0.15. Chambers and WEST (2019, p. 7) also modeled nest success drawn from a beta distribution with a CV of 0.125. Based on these previous modeling studies, the range of CVs for fecundity in previous models is 0.2 to 0.5, but it is plausible that variation in fecundity would fall outside of this range, given the variety of methods used to model fecundity, and the fact that a very long-term study of juvenile ratios (Lorenz and Raphael 2018, p. 206) had a level of variability consistent with the lower side of this range, even though it incorporated additional sources of variation beyond simply variation in fecundity.

Modeling Methods

We performed all steps of the model requiring calculation in R version 3.6.1 (R Core Team 2019).

Model Structure

Consistent with most other murrelet demographic models (see Table 1), our model was stage-structured. We used four stages: juvenile, first-year subadult, second-year subadult, and adult, similar to models by Beissinger and Peery (2007, p. 299) and Peery and others (2006a, p. 1521). Models by Peery and Henry (2010, p. 2415) and Peery and Jones (2019, p. 7) use a similar stage structure, but further subdivide adults into breeding and non-breeding classes, as their models were used to investigate the effects of management actions occurring within nesting habitat, which was not the purpose of our model. Like all previous demographic models of murrelets, we used a female-only model, and assumed that the number of females and males would remain equal at all times. Our model, like other murrelet models, represented time in discrete, one-year steps, updating at the end of each breeding season.

We incorporated both demographic stochasticity and environmental stochasticity into our model. As in previous murrelet models incorporating demographic stochasticity, we used binomial functions to model the random processes of survival and reproduction (Akcakaya 1997, p. 13; Chambers and WEST 2019, p. 9; McShane et al. 2004, pp. 3-36 - 3-37; Peery and Henry 2010, p.

2418). Binomial functions specify the number of times a "success" occurs in a series of N trials; for a familiar example, think of the number of "heads" in a series of N coin flips. In our model, "successes" represented survival or production of female offspring, where N is the number of individuals in a stage or breeding-age adults, respectively. We represented environmental stochasticity by selecting survival and reproductive rates from beta functions, a method used in recent murrelet models (Chambers and WEST 2019, pp. 8-9; Peery and Henry 2010, p. 2418; Peery and Jones 2019, p. 27). Beta functions are useful for this application because they vary continuously between 0 and 1, which covers the potential range of survival rates and number of offspring per female adult. We represented the number of female offspring per female adult as simply half of the total number of offspring per female adult, in keeping with the assumption that the sex ratio would remain equal at all times.

<u>Parameterization</u>

We parameterized the demographic model with the goal of simulating a 2.2% annual rate of decline over a 20-year period, as observed in Conservation Zone 2 between 2001 and 2019 (McIver et al. 2020, p. 18). First, we made some simplifying assumptions. Second, we used a version of the model that incorporated only demographic stochasticity, but not environmental stochasticity, to select mean survival and reproductive rates that resulted in a 2.2% annual rate of decline. Below, we refer to this version of the model as the reduced model. Lastly, we used the version of the model including both demographic and environmental stochasticity to select coefficients of variation. We did not have a specific target level of variation. The variability observed so far during population monitoring (McIver et al. 2020, pp. 10-15, 21) includes effects of sampling and measurement errors, which we did not intend to include in our model, so it is possible that the appropriate level of variability would be less than what has been observed over the 19-year survey period. At the same time, it is possible that the observed variation does not encompass the full range of environmental variation that would be observed over a longer time period, in which case the appropriate level of variability would be greater than what has been observed so far. We evaluated coefficients of variation by examining the plausibility of the shapes of the resulting beta distributions, and erred on the side of including a wider range of variation in order to examine the consequences of the action over a broader suite of population trajectories. For each parameter set, we also evaluated model output for realistic values of the juvenile ratio.

We simplified our parameterization by assuming that, in any given year, the survival of juveniles over their first year was a fixed percentage of adult survival, in the range of values used in most previous murrelet models (see the literature review of survival rates, above). Since empirical studies do not distinguish subadult and adult survival rates, we assumed that all after-hatch-year age classes would have identical survival rates. We further assumed that survival and reproduction varied independently from one another. This assumption may not be strictly realistic, since lower survival rates have been observed in years with higher rates of breeding, possibly because breeding involves increased risks of predation associated with inland flights (Peery et al. 2006b, p. 84). Correlations between these variables are likely to be complex, and we lack information to fully describe these correlations, so for simplicity, we modeled the two parameters independently.

We began our selection of parameter means by running the reduced model with survival rates, breeding propensity, and nest success set to their highest plausible values and then by running the reduced model with parameters set to their lowest plausible values. The highest plausible values were after-hatch-year survival rate of 0.93, juvenile survival 71 percent of adult survival (or 0.66), breeding propensity of 0.95, and nest success of 0.55, resulting in approximately 0.26 female offspring per female adult. Without environmental stochasticity, these parameters resulted in population growth of 5.6 percent annually over 20 annual time steps, and a juvenile ratio of 0.19. The lowest plausible values were after-hatch-year survival rate of 0.83, juvenile survival of 0.51 (rather than a percent of adult survival), breeding propensity of 0.13, and nest success of 0.16, resulting in approximately 0.01 female offspring per female adult. Without environmental stochasticity, these parameters resulted in population decline of 17.3 percent annually over 20 annual time steps, and a juvenile ratio of 0.012. We then tested various combinations of parameters in between the highest and lowest plausible values, settling at this stage on after-hatch-year survival rate of 0.9, juvenile survival 71 percent of adult survival (or 0.693), breeding propensity of 0.675, and nest success of 0.42, resulting in approximately 0.14 female offspring per female adult. Without environmental stochasticity, these parameters resulted in population decline of 2.2 percent annually, as desired, and a juvenile ratio of 0.123. Although this juvenile ratio is higher than any multi-year average observed in the listed range, it is similar to a multi-year average observed in neighboring British Columbia, so we considered it to be acceptably realistic.

We next began our selection of coefficients of variation by using the full model that included both demographic and environmental stochasticity. We varied the adult survival rate and the number of offspring per female adult (i.e., twice the number of female offspring per female adult). Because we assumed that there would always be equal numbers of males and females, the maximum average number of female offspring per female adult would be 0.5, but the beta function we used to represent fecundity has a range from 0 to 1, so we varied the average total number of offspring per female adult and divided by 2 to calculate the average number of female offspring per female adult. We did not vary juvenile survival separately, but set it to 71 percent of the adult value, so that it varied along with the adult survival rate. To test a given combination of coefficients of variation, for each of the two parameters, we first used the mean and coefficient of variation to calculate the variance, using the formula:

$$\mathit{variance} = (\mathit{mean} * \mathit{coefficient\ of\ variation})^2$$

We then derived alpha and beta parameters that shape the beta distribution, using the formulae:

$$\mathit{alpha} = [(1 - \mathit{mean})/\mathit{variance} - 1/\mathit{mean}] * \mathit{mean}^2$$

$$\mathit{beta} = \mathit{alpha} * (1/\mathit{mean} - 1)$$

We examined the shape of the resulting beta distributions, and continued testing a set of covariances only if the beta distribution shapes made sense biologically.

As with the setting of parameter means, we began by testing the highest coefficients of variation found in our literature review. The highest plausible coefficients of variation were 0.27 for adult survival, and 0.5 for fecundity. We combined these with the survival and fecundity rates selected above, 0.9 and 0.28, respectively. The resulting beta probability density function for the survival rate was U-shaped, showing high probabilities of both very low and very high survival in a given year. This is not biologically plausible for murrelets; if annual survival rates were frequently very low, the species would disappear very quickly, given that even in the best circumstances it takes individuals many years to replace themselves. The resulting beta probability density function for fecundity showed the highest probabilities near 0, but also became somewhat U-shaped, with increasing probabilities near 1. This is also not biologically plausible, since we expect years would almost never occur in which all adults attempted reproduction and all were successful. We concluded that the high end of the "plausible" range of variation from the literature was too variable for our model.

We then tested the lowest coefficients of variation found in our literature review. The lowest plausible coefficients of variation were 0.02 for adult survival, and 0.2 for fecundity. We combined these with the survival and fecundity rates selected above, 0.9 and 0.28, respectively. The resulting beta probability density function for the survival rate showed a tight peak around the mean of 0.9. This seems much more biologically plausible than the U-shape seen in the earlier version, but more variation would seem more realistic, since we expect there would be at least occasional years with reduced survival rates. The resulting beta probability density function for fecundity showed the highest probability at lower values, gradually decreasing to 1; this is a biologically plausible shape for the probability density function of fecundity, although it overemphasized the likelihood of implausibly high fecundity values (i.e., values indicating that nearly all adults successfully fledged offspring).

Therefore, we tested coefficients of variation that fell between the high and low levels from the literature for survival, but found that lower coefficients of variation for fecundity increased the plausibility of the resulting probability distribution. We also found that adding variation accelerated the annual rate of population decline, so we revisited the means of each parameter to re-center the rate of population change at -2.2 percent annually. Our final selection of parameter estimates included breeding propensity of 0.74 and nest success of 0.42, resulting in an overall fecundity (i.e., chicks of both sexes per female adult) of 0.3108, with a CV of 0.085; and a subadult and adult survival rate of 0.9, with a CV of 0.1. These parameters resulted in biologically reasonable probability densities for survival and fecundity, the appropriate -2.2 percent mean rate of annual change, and a slightly wider range of population trends than captured in the 95 percent confidence interval for the empirical population trends. Simulated juvenile ratios ranged between 0 and 0.42, but were most often around 0.12 to 0.13; the range is appropriate, and the means are plausibly realistic, but do tend toward the high end of observed long-term average juvenile ratios.

Baseline Model

After selecting the parameters, we designed a version of the model to represent a future scenario in which the Navy's proposed action is not carried out. We made two adjustments to the model to represent this future. First, we incorporated the anticipated effects of the Skookumchuck Wind Energy Project, which was recently constructed in the terrestrial portion of Zone 2 and is expected to injure and kill murrelets each year for 30 years. Although the wind turbines are located in the terrestrial portion of Zone 2, it is not clear whether the murrelets likely to be injured or killed should be considered members of the population in Zone 1 or in Zone 2 (see USFWS 2019, pp. 152, 163-165). For the present analysis, we made the conservative assumption that all would spend time at sea in Zone 2, and therefore be counted as members of the Zone 2 population. Because this represents a new mortality source, the mortality from this source is additional to the mortality that has contributed to the 2.2 percent annual decline from 2001 to 2019. Second, we removed the estimated effects of the Navy's ongoing Gunnery Exercises. Although our 2016 opinion addressing NWTT was the first time we estimated the effect of these exercises to murrelets, these activities have been ongoing in much the same way since 2001 or before. Therefore, the effect of murrelet injury and mortality related to the Gunnery Exercises is already reflected in the 2.2 percent annual decline. To represent a future without any of the effects of the proposed action, we must remove the effects of ongoing Gunnery Exercises from our model.

Effects of Skookumchuck Wind Energy Project

We anticipated that the Skookumchuck Wind Energy Project would kill 77 adults, subadults, or fledglings, between 2020 and 2049, via collisions with turbines or meteorological towers. On average, this amounts to approximately 2.6 individuals killed each year. In keeping with our assumption that the Zone 2 murrelet population would always maintain an equal sex ratio, we also assumed that on average, half of the individuals killed each year would be female. There is some indication that, during the chick provisioning period, breeding males fly inland more often than breeding females (Bradley et al. 2002, p. 180; Vanderkist et al. 1999, pp. 400-401); therefore, the assumption of equal risk may overestimate effects to females. However, our female-only model does not allow us to examine risks to the population from unbalanced sex ratios, which may be exacerbated by male-biased risks at wind farms (See *Single Sex Model* section in the discussion below).

We modeled the effect of the Skookumchuck Wind Energy Project to the Zone 2 population by removing adult females each year prior to calculating reproduction. Although we anticipated that some of the individuals killed would be subadults and adults, we expected that most flights through wind farm would be undertaken by breeding adults. Therefore, in the present model, we assigned all mortalities associated with the wind farm to the adult age class. In the model, we represented these mortalities using a Poisson random variable, with a mean of 1.28. In each of the first 30 years of each model run, we drew a number from the Poisson distribution and subtracted this number of individuals from the adult population, prior to determining the year's reproductive output.

Effects of Ongoing Gunnery Exercises

We used information provided by the Navy to estimate the number of individuals that are affected each year by the level of Gunnery Exercises that have been ongoing over the last decades. Naval personnel indicated that annual reports made between 2016 and 2019 would provide a good indication of the annual number of Gunnery Exercises carried out within 50 nautical miles of shore, during the October through March timeframe, since 2001 (Kunz 2020a, in litt.). We do not expect murrelets to be exposed to Gunnery Exercises conducted during April through September, or farther than 50 nautical miles from shore (see USFWS 2016, Appendix A). Only the 2016 annual report provided explicit information regarding the number of exercises carried out within 50 nautical miles of shore; it indicated all Gunnery Exercises were conducted farther from shore during that year (Navy 2017, p. 2). The reports for 2017 through 2019 did not explicitly enumerate exercises conducted within 50 nautical miles of shore, but rather listed the exercises according to designated training locations. We assumed that exercises occurred within 50 nautical miles of shore if they were conducted in a training locations (W-237 C, W-237 E) that included areas within 50 nautical miles of shore (Kunz 2020b, in litt.), even if those locations also included areas farther than 50 nautical miles from shore.

The monitoring report listed the number of non-explosive projectiles used in the small-, medium-, and large-caliber categories, but did not distinguish between surface-to-air and surface-to-surface projectiles, or between different sizes within the medium-caliber category. We assumed that the proportions of surface-to-surface and surface-to-air projectiles in each caliber category was equal to the proportions included in the 2016 proposed action: all small-caliber projectiles were surface-to-surface; 78 percent of medium caliber projectiles were surface-to-surface and 22 percent were surface-to-air; and 97 percent of large caliber projectiles were surface-to-surface and 3 percent were surface-to-air. We also assumed that the proportions of different sizes of medium caliber projectiles were the same as those described in recent clarifications of the proposed action (Kunz 2020c, in litt.): 40 percent 20 mm projectiles, 53 percent 35 mm projectiles, and 7 percent 40 caliber projectiles. After distributing the reported projectiles into these categories, we applied the same exposure analysis used to predict future murrelet exposure to stressors associated with Gunnery Exercises (see Effects of the Action in attached opinion). This resulted in an average estimate of 0.76 murrelets associated with Zone 2 injured or killed each year, or around three individuals injured or killed every four years, by stressors resulting from ongoing Gunnery Exercises.

The demographic effects of this loss of individuals is already reflected in the decreasing population trend observed for Zone 2. To remove the influence of Gunnery Exercises from the future population trajectory, we replaced these individuals in our simulated populations. We adjusted the total expected level of effect in a given year according to the simulated population size in that year. In other words, the quantification of 0.76 individuals affected per year was based on a population estimate of 2,013 individuals. Because our exposure model depends linearly on the density of murrelets, we would expect a population half this size to have half the population density and half the expected number of murrelets affected by the Gunnery Exercise activities. We assumed that all age classes could be affected, in proportion to their representation in the whole population. After calculating the expected number of murrelets affected in each age class, adjusting for both the size of the population and proportion of the population in each age class, we used a Poisson random variable for each age class to determine how many birds would be affected by the ongoing level of Gunnery Exercise-related stressors in a given year. After using the binomial distribution to simulate survival in a given age class and a given simulated future year, we drew from the Poisson distribution to determine how many individuals to add back into the population. However, if the number drawn from the Poisson distribution for an age class was greater than the number simulated to have died in that age class, we added back the number simulated to have died, instead, because survival rates cannot be greater than 1. We performed this step prior to determining the reproductive output of the adult age class.

Modeling Effects of the Action

In our exposure analysis (see Effects of the Action in the attached opinion), we estimated that within Zone 2, 52.1 adults, subadults and fledged juveniles would be injured or killed at sea as a result of the action. Across the 16-year term of the action, this equates to an average of 3.26 individuals injured or killed each year (52.1 divided by 16). We assumed that half of the injured or killed murrelets would be female, or an average of 1.63 females killed per year. We assumed that these were distributed among the age classes in proportion to their representation in the population. We did not have a "fledged juvenile" age class, and furthermore we made the simplifying assumption that injuries and mortalities would occur during the incubation or nestling phase of the breeding season, so these effects were distributed among adults and first- and second-year adults. As described for ongoing Gunnery Exercise effects, above, we adjusted the expected number of females injured or killed in a given simulation year based on the simulated population size for that year. We used a Poisson random variable to determine how many individuals in each age class would be affected by injury or mortality resulting from the action, and subtracted the resulting numbers of individuals from the population. We performed this step prior to determining the reproductive output of the adult age class.

Aside from subtracting the number of individuals simulated to be affected by the action in each time step, the version of the model that included the effects of the action was identical to the baseline version. Both included the same adjustments for effects of Skookumchuck Wind Energy Project and the same adjustments for the ongoing Gunnery Exercise effects.

Running the Model

For each of 10,000 model runs, we generated a sequence of 200 annual survival rates, drawing from the survival rate beta distribution described above, and 200 annual fecundity rates, drawing from the fecundity rate beta distribution described above. For each model run, we also generated a sequence of 30 expected values representing the effects of the Skookumchuck Wind Energy Project. We generated a starting population size for each model run by drawing from a uniform distribution between 600 and 1,650 females, the approximate range of population sizes estimated from at-sea surveys between 2001 and 2018 (McIver et al. 2020, pp. 10-15). For each model run, the steps described in this paragraph were identical for both the baseline scenario and the action scenario; in other words, for each simulation run, the baseline and action scenarios had identical starting population sizes, identical sequences of demographic parameters (representing environmental variation), and identical effects resulting from Skookumchuck.

The remaining steps depended in part on the simulated population size at each step, which diverged for the baseline and action scenarios. Therefore, the remaining steps were performed separately for the two scenarios. For each scenario, in each year, we used binomial random variables, with the expected proportions set by the survival rate previously drawn for that year, and the sample sizes set by the number of females in each age class at the end of the previous year, to simulate survival since the previous year, which, for non-adults, was equivalent to transition from one life stage to the next. We used population-adjusted Poisson random variables for each scenario to represent the ongoing effects of Gunnery Exercises, and added the resulting numbers of individuals back into each age class to remove these effects from each scenario, as described above. For the action scenario only, for years 2 through 17, we used population-adjusted Poisson random variables to determine the number of birds to remove from each age class due to action-related effects, and subtracted the resulting number of birds. We then simulated reproduction for scenario in each year using a binomial random variable, with the expected number of fledged juveniles set by the fecundity rate previously drawn for that year, divided in half to represent only female fecundity, and the sample size set by the number of surviving adults. We repeated these steps 200 times for each of the 10,000 simulations, to give us 10,000 sets of two scenarios, which in each case were subjected to identical environmental variation but different demographic stochasticity.

Model Outputs

The basic model output consists of 10,000 sequences of 200 population vectors for each scenario (baseline and action scenarios), with each vector containing entries for the number of female juveniles, first-year subadults, second-year subadults, and adults at each of the 200 annual time steps. We also preserved the model inputs described in the first paragraph under Running the Model (the sequences of survival rates, fecundity rates, and effects of Skookumchuck Wind Energy Project), as well as the number of individuals simulated to have been removed from the population, in each of years 2 through 17, as a result of the proposed action.

For each of the two scenarios, we calculated the proportions of simulated populations falling below particular small population size thresholds during each time step. The small population size thresholds we selected were 1) 500 females, which is outside of the range of population estimates based on at-sea surveys in 2001 through 2018; 2) 250 females, a population approximately the same size as the Zone 6 population, and the largest population size at which small population dynamics might begin to have a noticeable effect on the population trajectory; 3) 50 females, a population approximately the same size as the Zone 5 population, and a population size at which small population dynamics would likely be a major factor affecting the population trajectory; and 4) 0 females, or in other words, extirpation. We calculated the proportions of simulated populations in each scenario falling below these sizes for every year, but we also selected several particular years as "check-in" points, for comparison with other recent murrelet demographic models (e.g., Peery and Jones 2019, entire, and Chambers and WEST 2019, entire). The check-in years were at year 17, immediately following the term of the action, and years 50, 75, 100, and 200. At each year, and at each check-in year, for which at least 1 percent of the baseline population runs met or fell below a particular small population threshold, we calculated the change in absolute and relative risk, attributable to the action, of achieving or falling below that threshold.

We also calculated logarithmic rates of population decline (see Miller et al. 2012, p. 774) for each simulation run for the time periods from the beginning of the simulations until each of these check-in years. In addition, we calculated the difference in population numbers between the action and baseline scenarios for each pair of simulated populations, both in absolute numbers and as a proportion of the average baseline population size.

To make sure that we repeated the simulation enough times, we graphed the mean of each of these results (population size at each check-in year, proportion of the population below each small population threshold during each check-in year, population trends, and difference between scenarios) as a function of the number of simulations. When the number of simulations is small, the estimated value of each of these outputs varies dramatically as new simulations are added to the average, but as the number of simulations increases, the mean stabilizes. We did not use particular numerical thresholds to judge whether the means were stable enough, but rather we considered that we had performed enough simulations if the graphs showed that the mean visibly stabilized when the number of simulations included was smaller than 10,000.

To explore questions about what would happen if the rate of population change in Zone 2 changes, we examined two subsets of the simulations. One subset consisted of the simulation runs that had both average survival and average fecundity rates in the highest quartile among all of the simulations, during the first 100 years of the simulations. The other subset consisted of the simulation runs that had both average survival and average fecundity rates in the lowest quartile among all of the simulations, during the first 100 years of the simulations. Because these subsets were much smaller than the full set of 10,000 simulations, the results were not as stable as the results from the full set of simulations and should not be regarded as precise. However, these results can provide some idea of how the results would vary in different environmental conditions.

Results

Population Numbers and Rates of Decline

As expected, in both the baseline and action scenarios, simulated populations declined, on average, over time. In both scenarios, the median population size was close to 0 by the 200 year mark. Median population numbers and rates of decline are shown in Tables 2 and 3, respectively. Mean population trajectories, along with 20 randomly-selected population trajectories from individual simulations, are shown in Figures 1, 2, and 3. Note that the example trajectories in these figures come from the same simulation runs for both scenarios, and therefore represent the same series of expected demographic rates (representing the same environmental variation) for each scenario. Differences are due to the effects of the action and to demographic stochasticity.

Table 2. Median female population sizes for the two scenarios for the six check-in years

Simulation Year	1	17	50	75	100	200
Baseline Scenario	1,125	754.5	339	193	110	9
Action Scenario	1,125	733.5	331	186	106	9

Table 3. Median rates of population change for the two scenarios for the six check-in years

Simulation Years	1-17	1-50	1-75	1-100	1-200
Baseline Scenario	-2.147	-2.340	-2.303	-2.271	-2.345
Action Scenario	-2.358	-2.380	-2.345	-2.291	-2.359

Figure 1. Mean and example population trajectories over time, baseline scenario

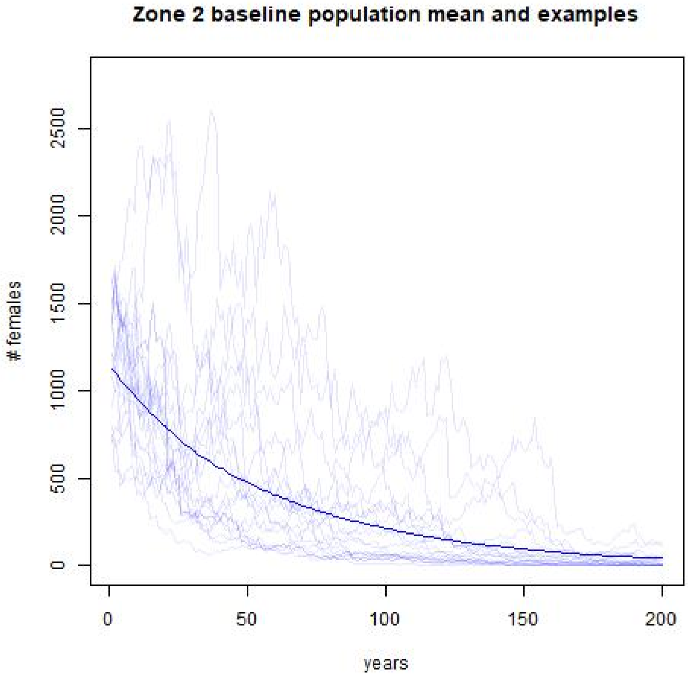


Figure 2. Mean and example population trajectories over time, action scenario

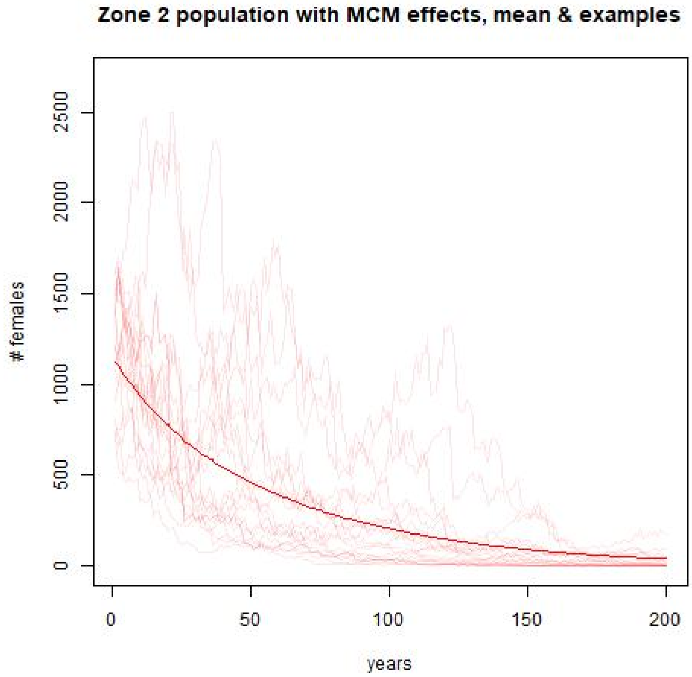
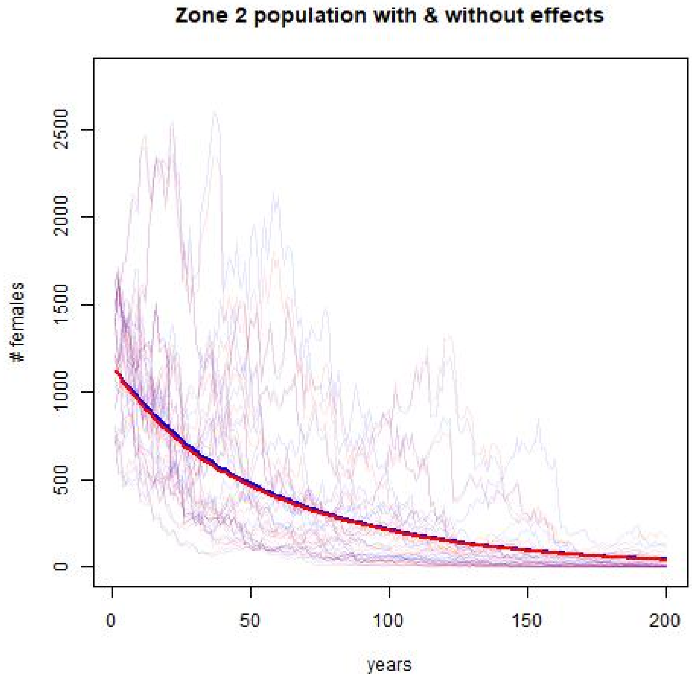


Figure 3. Mean and example population trajectories, blue: baseline, red: action scenario



Differences between Scenarios

The average simulated baseline population was larger than the average simulated population subjected to the effects of the action throughout the 200-year simulation period. This difference increased initially, during the term of the action. Subsequently, the difference in the number of females in the population declined over time, as both populations became smaller (Figure 4). However, the average percentage difference between the baseline and action scenarios was fairly stable, and following an initial change during the term of the action, remained between -2.6 and -3.6 through year 200 (Figure 5). The percentage difference did appear more variable during the later years of the simulation period.

Figure 4. Average change in number of females (action scenario – baseline), with examples

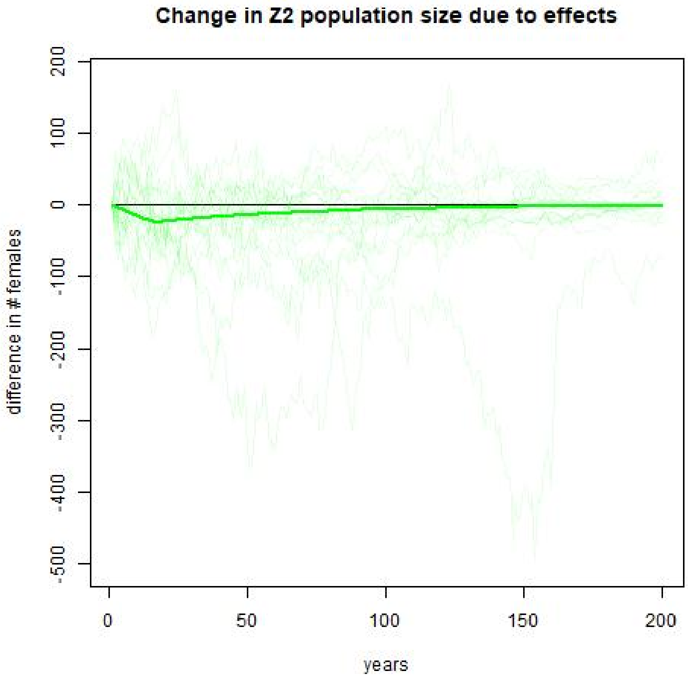
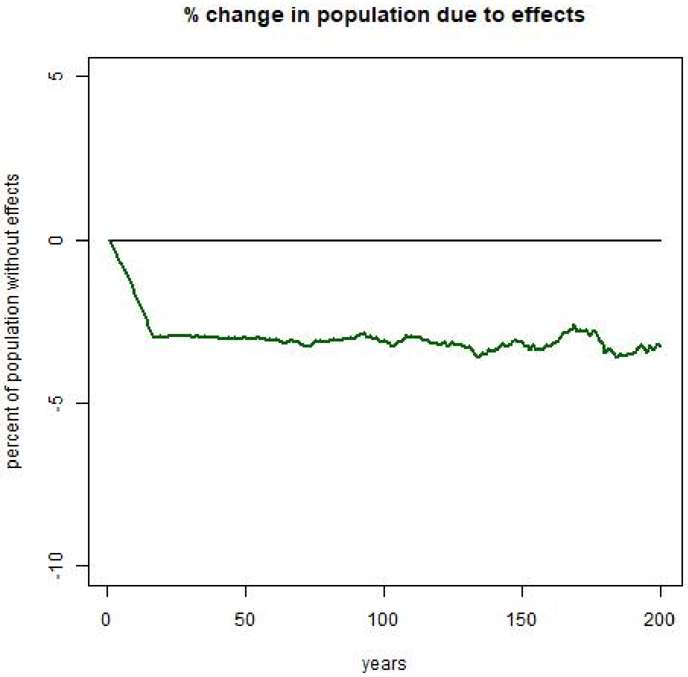


Figure 5. Percentage change in number of females (action scenario – baseline)/mean baseline

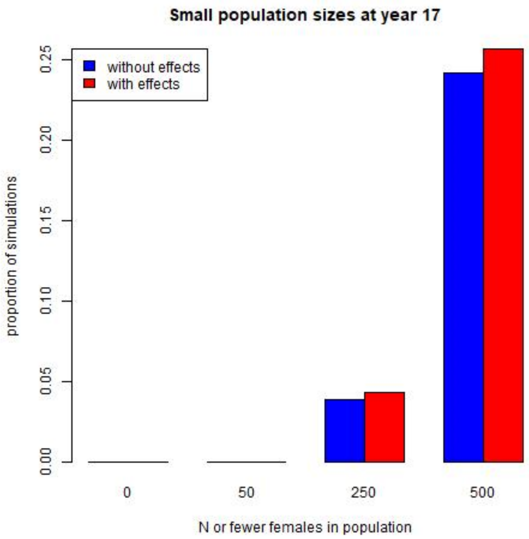


Likelihood of Small Population Sizes

As the simulated populations continued to decline over time, the proportion of simulated population trajectories at or below each of the small population thresholds increased, with or without the action. For the baseline scenario, achieving or falling below the 500-female population threshold became more likely than not at year 34, 250 or fewer females became more likely than not at year 64, and 50 or fewer females became more likely than not at year 135. For the action scenario, achieving or falling below the 500-female population threshold became more likely than not at year 32, 250 or fewer females became more likely than not at year 63, and 50 or fewer females became more likely than not at year 133. During the 200 years, the proportion of simulations reaching extirpation remained below 50 percent for both the baseline and action scenarios.

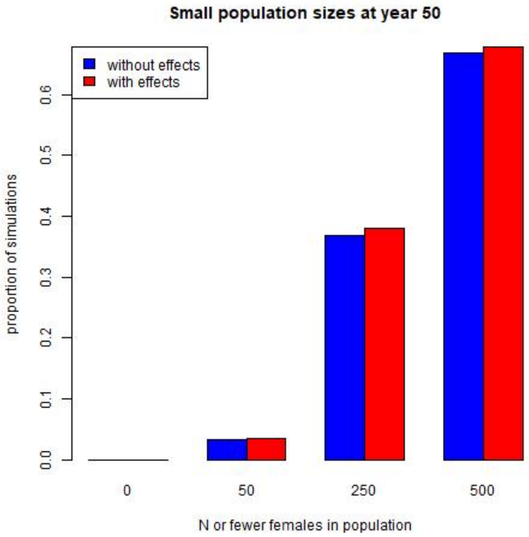
At year 17, the proportions of simulations reaching or falling below the 500-female threshold were near 25 percent for both scenarios, and for the 250-female threshold, the proportions were less than 5 percent for both scenarios. In each case, the proportions were higher for the action scenario than for the baseline scenario (Figure 6, Tables 4-7). None of the trajectories fell below the 50-female threshold for either scenario during year 17.

Figure 6. Proportions of simulations at or below small population thresholds, year 17



At year 50, around two-thirds of the simulated populations were smaller than the 500-female threshold, nearly 40 percent of simulated populations were at or smaller than the 250-female threshold, and fewer than 4 percent of simulated populations were at or smaller than the 50-female threshold. In each case, the proportions were higher for the action scenario than for the baseline scenario (Figure 7, Tables 4-7). Fewer than 1 percent of the trajectories reached extirpation by year 50.

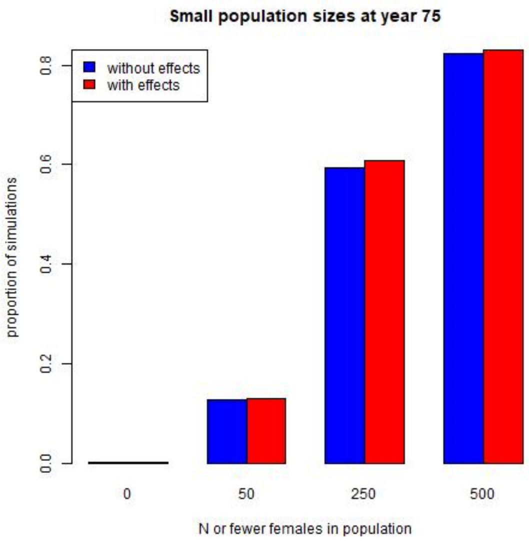
Figure 7. Proportions of simulations at or below small population thresholds, year 50



At year 75, around 80 percent of simulated populations were smaller than the 500-female threshold, most simulated populations were at or smaller than the 250-female threshold, and more than 10 percent of simulated populations were at or smaller than the 50-female threshold.

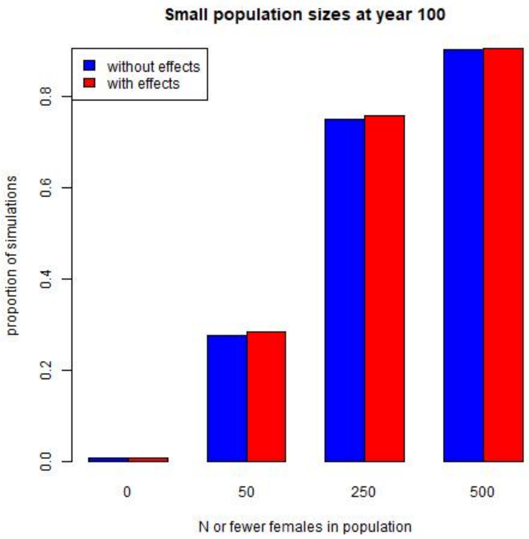
In each case, the proportions were higher for the action scenario than for the baseline scenario (Figure 8, Tables 4-7). Fewer than 1 percent of population trajectories reached extirpation by year 75.

Figure 8. Proportions of simulations at or below small population thresholds, year 75



At year 100, around 90 percent of simulated populations were smaller than the 500-female threshold, around 75 percent of simulated populations were at or smaller than the 250-female threshold, and nearly 30 percent of simulated populations were at or smaller than the 50-female threshold. In each case, the proportions were higher for the action scenario than for the baseline scenario (Figure 9). Fewer than 1 percent of population trajectories reached extirpation by year 100.

Figure 9. Proportions of simulations at or below small population thresholds, year 100



At year 200, nearly all simulated populations were smaller than both the 500-female threshold and the 250-female threshold, and more than 80 percent of simulated populations were at or smaller than the 50-female threshold. Approximately 25 percent of the population trajectories had reached extirpation. Although the proportions of action scenario populations reaching each threshold were higher than the corresponding proportions of baseline scenarios, these differences were very small (less than one percent difference) for all except for the 50-female threshold (Figure 10).

Figure 10. Proportions of simulations at or below small population thresholds, year 200

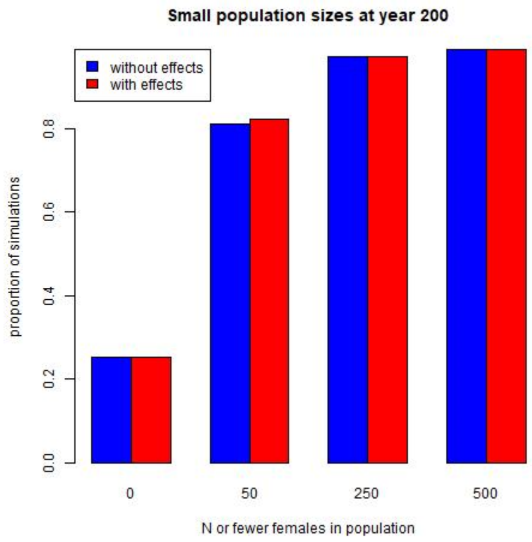


Table 4. Action-related increases in relative and absolute risk of reaching 500-female threshold

Simulation Year	17	50	75	100	200
% of Baseline Runs	24.14	66.80	82.25	90.16	99.00
% of Action Runs	25.70	67.90	83.20	90.60	99.04
Relative risk increase %	6.46	1.65	1.16	0.49	0.04
Absolute risk increase %	1.56	1.10	0.95	0.44	0.04

Table 5. Action-related increases in relative and absolute risk of reaching 250-female threshold

Simulation Year	17	50	75	100	200
% of Baseline Runs	3.88	36.81	59.35	74.82	97.10
% of Action Runs	4.31	37.92	60.77	75.80	97.22
Relative risk increase %	11.08	3.02	2.39	1.31	0.12
Absolute risk increase %	0.43	1.11	1.42	0.98	0.12

Table 6. Action-related increases in relative and absolute risk of reaching 50-female threshold

Simulation Year	17	50	75	100	200
% of Baseline Runs	0	3.28	12.68	27.49	81.06
% of Action Runs	0	3.40	13.06	28.40	82.14
Relative risk increase %	n/a	3.66	3.00	3.31	1.33
Absolute risk increase %	n/a	0.12	0.38	0.91	1.08

Table 7. Action-related increases in relative and absolute risk of reaching 0-female threshold

Simulation Year	17	50	75	100	200
% of Baseline Runs	0	< 1.00	< 1.00	< 1.00	25.21
% of Action Runs	0	< 1.00	< 1.00	< 1.00	25.30
Relative risk increase %	n/a	n/a	n/a	n/a	0.36
Absolute risk increase %	n/a	n/a	n/a	n/a	0.09

The greatest increases in relative and absolute risk of reaching small population thresholds, attributable to the action, generally did not occur during one of the selected check-in years, but rather between check-in years. The maximum relative risk increase in a given threshold generally occurred while the proportions of simulated populations reaching that threshold were relatively low, whereas the maximum absolute risk increase for a given threshold generally occurred when somewhere near half of the populations had reached that threshold.

For the 500-female threshold, the maximum increase in relative risk occurred during and after the action; from years 11-20, the relative risk of reaching this threshold was between 4.14 and 7.40 percent higher with the action than without. During years 11-20, the proportion of baseline simulations reaching this threshold rose from 14.02 to 29.49 percent, and the proportion of action simulations reaching this threshold rose from 14.60 to 30.81 percent. The maximum increase in absolute risk occurred shortly after the term of the action, during years 21-30, during which time the proportion of populations reaching this threshold was between 1.55 and 1.86 percent higher with the action than without the action. During years 21-30, the proportion of baseline simulations reaching this threshold rose from 31.40 to 46.18 percent, and the proportion of action simulations reaching this threshold rose from 32.95 to 47.79 percent.

For the 250-female threshold, the maximum increase in relative risk occurred immediately after the action; from years 17-26, the relative risk of reaching this threshold was between 7.81 and 14.38 percent higher with the action than without. During years 17-26, the proportion of baseline simulations reaching this threshold rose from 3.88 to 11.49 percent, and the proportion of action simulations reaching this threshold rose from 4.31 to 12.47 percent. The maximum increase in absolute risk occurred several decades later, during years 61-70, during which time the proportion of populations reaching this threshold was between 1.14 and 1.93 percent higher with the action than without the action. During years 61-70, the proportion of baseline simulations reaching this threshold rose from 47.44 to 55.35 percent, and the proportion of action simulations reaching this threshold rose from 48.92 to 57.08 percent.

For the 50-female threshold, the maximum increase in relative risk occurred in year 41; from years 41-50, the relative risk of reaching this threshold was between 3.66 and 13.89 percent higher with the action than without. During years 41-50, the proportion of baseline simulations reaching this threshold rose from 1.44 to 3.28 percent, and the proportion of action simulations reaching this threshold rose from 1.64 to 3.40 percent. The maximum increase in absolute risk occurred many decades later, during year 130. During years 126 through 135, the proportion of populations reaching this threshold was between 1.23 and 2.03 percent higher with the action than without the action. During years 126-135, the proportion of baseline simulations reaching this threshold rose from 44.57 to 50.45 percent, and the proportion of action simulations reaching this threshold rose from 45.90 to 51.82 percent.

The proportion of simulated populations reaching extirpation did not reach or exceed 1 percent until year 108 in the baseline scenario and year 106 in the action scenario. The maximum increase in relative risk of extirpation occurred in year 130; from years 126-135, the relative risk of reaching this threshold was between 2.72 and 4.32 percent higher with the action than without. During years 126-135, the proportion of baseline simulations reaching extirpation rose from 2.87 to 4.16 percent, and the proportion of action simulations reaching extirpation rose from 3.08 to 4.37 percent. The maximum increase in absolute risk occurred several decades later, during year 169. During years 161 through 170, the proportion of populations reaching extirpation was between 0.04 and 0.41 percent higher with the action than without the action. During years 161-170, the proportion of baseline simulations reaching this threshold rose from 10.48 to 13.33 percent, and the proportion of action simulations reaching this threshold rose from 10.52 to 13.64 percent.

Convergence to Stable Results

Most results reached stable means or proportions, as visually evaluated using graphs, after 5,000 or fewer simulation runs. Rates of population change over the full 200-year period provided a slight exception; although mean trends were fairly stable after approximately 5,000 repetitions, some slight fluctuation in the means continued through the full 10,000-repetition series. Therefore, the 200-year trend estimates should be interpreted as having lower precision than other results presented above.

Subset of Simulation Runs with Best Demographic Rates

Among the 625 simulation runs for which both survival and fecundity rates fell within the best quartiles for the first 100 years, the average adult survival rate during this time was 0.91 and the average total fecundity rate was 0.33. These demographic rates can be compared with the averages across all simulations and years, which were 0.91 for survival and 0.31 for fecundity.

Population Numbers and Rates of Change

In this subset of simulation runs, populations declined much more slowly during the first 100 years, as compared with the full set of simulation runs. Median rates of population change are negative, but close to 0, or in other words, the median population is nearly stable. After the first 100 years, the demographic rates of this subset reverted to their means, and the median population became very small by the 200-year mark, though not as close to extirpation as the median population at 200 years in the full set of simulation runs. Median population numbers and rates of decline are shown in Tables 8 and 9, respectively.

Table 8. Median female population sizes for the two scenarios: runs with best demographic rates

Simulation Year	1	17	50	75	100	200
Baseline Scenario	1,128	1,028	827	734	639	63
Action Scenario	1,128	996	789	702	599	57

Table 9. Median population change rates for the two scenarios: runs with best demographic rates

Simulation Years	1-17	1-50	1-75	1-100	1-200
Baseline Scenario	-0.39	-0.64	-0.53	-0.53	-1.41
Action Scenario	-0.64	-0.70	-0.57	-0.58	-1.37

Differences between Scenarios

The average simulated baseline population was larger than the average simulated population subjected to the effects of the action throughout the 200-year simulation period for this subset of simulation runs, as for the full set of simulation runs. The average difference in the number of females in the populations remained fairly stable, between 30 and 40 females, between years 17 (the end of the term of the action) and year 100 (the end of the period in which these simulated populations had elevated demographic rates) (see Table 8). The difference in the number of females declined thereafter as both populations became smaller. The average percentage difference between the baseline and action scenarios was slightly larger than that for the full set of scenarios during years 17 through 100, remaining between -2.8 and -4.3, and became more variable thereafter, ranging from -1.5 to -4.5.

Likelihood of Small Population Sizes

In this subset of simulation runs, no simulated population fell beneath the 50-female threshold during the first 100 years. The proportion of simulated populations meeting or falling beneath the 250-female threshold remained below 10 percent for the action scenario, and below 7 percent for the baseline scenario, during the first 100 years. The proportion of simulated populations meeting or falling beneath the 500-female threshold reached a maximum of 36 percent for the action scenario, and remained below 35 percent for the baseline scenario, during the first 100 years. By 200 years, the proportions of simulated baseline populations falling beneath each threshold were 93.12 percent for the 500-female threshold, 83.84 percent for the 250-female threshold, 42.24 for the 50-female threshold, and 2.08 extirpated. For simulated action populations the proportions were 93.76 percent for the 500-female threshold, 84.32 percent for the 250-female threshold, 45.28 for the 50-female threshold, and 2.08 extirpated.

For the 500-female threshold, the maximum increase in relative risk occurred during and after the action; from years 11-20, the relative risk of reaching this threshold was between 8.11 and 28.13 percent higher with the action than without. During years 11-20, the proportion of baseline simulations reaching this threshold rose from 5.12 to 11.84 percent, and the proportion of action simulations reaching this threshold rose from 6.56 to 12.80 percent. The maximum increase in absolute risk occurred in year 107; during years 101 to 110, the proportion of populations reaching this threshold was between 1.28 and 4.64 percent higher with the action than without the action. During years 101-110, the proportion of baseline simulations reaching this threshold rose from 33.28 to 47.68 percent, and the proportion of action simulations reaching this threshold rose from 37.44 to 51.68 percent.

For the 250-female threshold, the maximum increase in relative risk occurred around 10 years after end of the term of the action; from years 26-35, the relative risk of reaching this threshold was between 11.54 and 69.23 percent higher with the action than without. During years 26-35, the proportion of baseline simulations reaching this threshold rose from 1.92 to 4.16 percent, and the proportion of action simulations reaching this threshold rose from 2.24 to 4.64 percent. Another period of elevated relative risk occurred between years 56 and 85; in all but four of these years, the relative risk increase exceeded 20 percent, but never rose higher than 41 percent. During years 56-85, the proportion of baseline simulations reaching this threshold fluctuated between 5.76 and 6.88 percent, and the proportion of action simulations reaching this threshold fluctuated between 7.04 to 9.28 percent. The maximum increase in absolute risk was 2.88 percent, occurring in years 137 and 165, but there was no single period during which the increase in risk was particularly elevated. Instead, the increase in absolute risk was greater than 2 percent during year 53, several years around year 70, and sporadically after year 100.

For the 50-female threshold, the maximum increase in relative risk occurred in year 130; from years 129-138, the relative risk of reaching this threshold was between 8.70 and 128.57 percent higher with the action than without. During years 129-138, the proportion of baseline simulations reaching this threshold rose from 1.12 to 3.84 percent, and the proportion of action simulations reaching this threshold rose from 2.08 to 5.12 percent. The maximum increase in absolute risk occurred many decades later, during years 188 and 194. During years 186 through 195, the proportion of populations reaching this threshold was between 0.80 and 3.64 percent higher with the action than without the action. During years 186 through 195, the proportion of baseline simulations reaching this threshold rose from 32.48 to 39.68 percent, and the proportion of action simulations reaching this threshold rose from 34.56 to 42.08 percent.

The proportion of simulated populations reaching extirpation did not reach or exceed 1 percent until year 185 in the baseline scenario and year 187 in the action scenario. The relative risk of extirpation was increased with the action only in four of the last 5 years of the simulation (and no earlier years); from years 196-200, the relative risk of reaching extirpation was between 0 and 18.18 percent higher with the action than without. During years 196-200, the proportion of baseline simulations reaching extirpation rose from 1.76 to 2.08 percent, and the proportion of action simulations reaching extirpation rose from 1.92 to 2.08 percent. The maximum increase in absolute risk of extirpation also occurred during year 197, and was 0.32 percent.

Convergence to Stable Results

Because this subset only included 625 simulation runs, the results did not stabilize as fully as the results for the full set of simulation runs. However, the largest fluctuations in means or proportions occurred while the number of simulations was less than 500. Therefore, we regard the results from this subset of simulation runs as acceptable to indicate the approximate pattern of population responses to the action, for populations exhibiting the better demographic rates present in this subset, but as much less precise than the results for the full set of simulation runs.

Subset of Simulation Runs with Worst Demographic Rates

Among the 637 simulation runs for which both survival and fecundity rates fell within the worst quartiles for the first 100 years, the average adult survival rate during this time was 0.889 and the average total fecundity rate was 0.290. These demographic rates can be compared with the averages across all simulations and years, which were 0.91 for survival and 0.31 for fecundity.

Population Numbers and Rates of Change

In this subset of simulation runs, populations declined much more rapidly during the first 100 years, as compared with the full set of simulation runs. Median rates of population decline are steeper than those currently estimated for Zone 2, but are less severe than declines currently estimated for Zone 2, and are similar to those estimated for all of Washington (see McIver et al. 2020, p. 18). After the first 100 years, the demographic rates of this subset reverted to their means, but populations were so small at the 100 year mark that the majority reached extirpation by year 200. Median population numbers and rates of decline are shown in Tables 10 and 11, respectively.

Table 10. Median female population sizes: runs with worst demographic rates

Simulation Year	1	17	50	75	100	200
Baseline Scenario	1,105	555	122	44	16	0
Action Scenario	1,105	533	118	41	15	0

Table 11. Median population change rates: runs with worst demographic rates

Simulation Years	1-17	1-50	1-75	1-100	1-200
Baseline Scenario	-4.11	-4.44	-4.27	-4.20	-11.13
Action Scenario	-4.19	-4.51	-4.31	-4.23	-10.66

Differences between Scenarios

The average simulated baseline population was larger than the average simulated population subjected to the effects of the action throughout the 200-year simulation period. As in the full set of simulations, this difference increased initially, during the term of the action, and declined over time, as both populations became smaller. However, the average percentage difference between the baseline and action scenarios was fairly stable, similar to but more variable than the full set of simulations; following an initial change during the term of the action, remained between -2.6 and -4.6 through year 85 (Figure 5). The percentage difference was much more variable after year 85; between year 86 and 200, the percentage difference between the two scenarios ranged from -10.4 to 0.4 (the positive number indicating a year in which, on average populations in the action scenario were slightly larger than populations in the baseline scenario).

Likelihood of Small Population Sizes

In this subset of simulation runs, most simulated populations fell beneath the 50-female threshold by year 73 for the baseline scenario and year 72 for the action scenario. Most simulated populations fell beneath the 250-female threshold by year 34 for the both scenarios. The majority of simulated populations fell beneath the 500-female threshold by year 19 for both scenarios. By 200 years, all simulated populations in this subset, from both scenarios, were smaller than 250 females. In the baseline scenario, 98.74 percent of simulations had 50 or fewer females, and 66.09 percent were extirpated. In the action scenario, 99.22 percent of simulations had 50 or fewer females, and 64.67 percent were extirpated.

For the 500-female threshold, the maximum increase in relative risk occurred in year 2, when 1.26 percent of baseline populations and 1.41 percent of action populations were smaller than or equal to 500 females, a 12.5 percent increase in relative risk. However, this increase in risk was not sustained; during some of the early years of the term of the action, higher percentages of the baseline populations than the action populations fell below the 500-female threshold. Immediately after the term of the action, from years 17-26, the relative risk of reaching this threshold was between 0.58 and 4.78 percent higher with the action than without. During years 17-26, the proportion of baseline simulations reaching this threshold rose from 42.70 to 69.07 percent, and the proportion of action simulations reaching this threshold rose from 44.74 to 69.70 percent. The maximum increase in absolute risk occurred in years 27 and 28; during years 21 to 30, the proportion of populations reaching this threshold was between 0.63 and 2.20 percent higher with the action than without the action. During years 21-30, the proportion of baseline simulations reaching the 500-female threshold rose from 56.36 to 76.61 percent, and the proportion of action simulations reaching this threshold rose from 57.61 to 78.65 percent.

For the 250-female threshold, the maximum increase in relative risk occurred immediately after end of the term of the action; from years 17-26, the relative risk of reaching this threshold was between 1.27 and 11.76 percent higher with the action than without. During years 17-26, the proportion of baseline simulations reaching this threshold rose from 12.40 to 31.71 percent, and the proportion of action simulations reaching this threshold rose from 12.56 to 32.34 percent. The maximum increase in absolute risk was 2.98 percent, occurring in year 43. During years 41-50, the proportion of populations reaching this threshold was between 0.78 and 2.98 percent higher with the action than without the action. During years 41-50, the proportion of baseline simulations reaching the 250-female threshold rose from 64.99 to 81.16 percent, and the proportion of action simulations reaching this threshold rose from 67.03 to 81.95 percent.

For the 50-female threshold, the maximum increase in relative risk occurred in year 49, when 14.76 percent of simulated baseline populations and 16.64 percent of action populations were at or below this threshold, a 12.77 percent increase in relative risk. However, this increase in risk was not sustained, and the relative proportions of simulated populations varied between the two scenarios during the years before and after year 49. A more sustained increase in relative risk occurred between years 61 to 70, during which time the relative risk of reaching this threshold was between 5.24 and 10.47 percent higher with the action than without. During years 61-70, the proportion of baseline simulations reaching the 50-female threshold rose from 29.98 to 43.49 percent, and the proportion of action simulations reaching this threshold rose from 31.55 to 48.04 percent. The maximum increase in absolute risk, 4.55 percent, occurred during year 70.

The maximum increase in relative risk of extirpation occurred in year 90; from years 90-99, the relative risk of reaching this threshold was between 8 and 77.78 percent higher with the action than without. During years 90-99, the proportion of baseline simulations reaching this threshold rose from 1.41 to 3.92 percent, and the proportion of action simulations reaching this threshold rose from 2.41 to 4.55 percent. The maximum increase in absolute risk of extirpation was 3.45 percent, occurring in year 139. During years 131-140, the proportion of populations reaching extirpation was between 2.35 and 3.45 percent higher with the action than without the action. During years 131-140, the proportion of baseline simulations reaching extirpation rose from 18.37 to 25.43 percent, and the proportion of action simulations reaching this threshold rose from 21.35 to 27.79 percent.

Convergence to Stable Results

Because this subset only included 637 simulation runs, the results did not stabilize as fully as the results for the full set of simulation runs. However, the largest fluctuations in means or proportions occurred while the number of simulations was less than 500. Therefore, we regard the results from this subset of simulation runs as acceptable to indicate the approximate pattern of population responses to the action, for populations exhibiting the lower demographic rates present in this subset, but as much less precise than the results for the full set of simulation runs.

Discussion

The results of this demographic model show that, with demographic rates consistent with a declining population, the effects of the proposed action lead to a permanent reduction in the Zone 2 population size, and some increase in the likelihood, at any given time, that the population will be reduced to small population sizes, or even extirpation. In the long run, with demographic rates consistent with the current -2.2 percent annual trend in population change, we expect that the population will reach small population sizes with or without the effects of the action.

Our examination of subsets of the simulations leads us to conclude that the qualitative results in the previous paragraph are not highly sensitive to the particular demographic rates used, or the rate of population decline. The subset of populations in which the demographic rates were better, which on average were nearly stable and showed only a very slow decline, still showed persistent reductions in the population size with the action, and increases in the risk of reaching small population sizes. The subset of populations in which the demographic rates were worse, and the rate of population decline steeper, also showed qualitatively similar results. These subsets differed in that few of the simulated populations with better demographic rates declined to the very small population sizes during the period where their demographic rates remained elevated, whereas all of the simulated populations with worse demographic rates declined to the larger two population thresholds, and many declined to the smaller thresholds, during the first 100 years of the simulation.

Model Structure

Our model structure is similar to most demographic models for murrelets that have previously been published or used. Understanding some aspects of the model structure allows us to better interpret the model results.

Stages

We used a model with a limited number of stages, for simplicity. As noted above, the use of a single adult stage is equivalent to assuming that the lifespan could be infinite, and models built this way may underestimate population decline, though this underestimate is small unless the actual decline is steep (McShane et al. 2004, p. 3-30). Given our assumed adult survival rate of 0.91, around 2 percent of murrelets would be expected to survive at least to age 40 (Burger 2002, pp. 17-18), which may not be realistic for murrelets. Murrelet longevity is not well-understood, but the average lifespan is expected to be around 10 years (USFWS 1997, p. 22).

We do not expect that our model underestimates the rate of demographic decline, because we parameterized our model to be consistent with a particular rate of decline estimated from empirical surveys. However, a more complex, but more realistic age-structured model, like that of McShane and others (2004, pp. 3-27 – 3-61) may have required higher demographic rates to arrive at the same rate of decline. The demographic rates needed to match a -2.2 percent annual rate of change were already higher than many of the empirically-derived demographic rate estimates for the species, as noted below in Model Parameters. The need to increase these rates still further to more realistically arrive at the same rate of decline implies that the empirically-derived rate of population change may underestimate the actual rate of decline in the breeding population.

Single-sex Model

Like other demographic models of murrelets, our model includes only females. However, males are equally important for murrelet reproduction. The exclusion of males from the model is likely to lead us to underestimate the effect of demographic stochasticity and the resulting extirpation risk (Bessa-Gomes et al. 2010, pp 443-444; Møller 2003, p. 229). Skewed sex ratios are a significant result of demographic stochasticity at small population sizes, and effectively make the population smaller than it already is, because not all adults have the opportunity to attempt nesting (Engen et al. 2003, p. 2385; Møller 2003, p. 224). Skewed sex ratios in small populations have been documented during extinction and extirpation events, for example, of dusky seaside sparrows (*Ammodramus maritimus nigrescens*), heath hens (*Tympanuchus cupido cupido*), and the New Zealand mainland population of kakapos (*Strigops habroptilus*) (Donald 2007, p. 684).

Stochasticity

Our model does account for demographic stochasticity arising from random variability in the number of individuals surviving and successfully fledging offspring. Our model does not account for other sources of demographic stochasticity, for example, variation in individual fitness caused by differences in ability (e.g., in flight speed, foraging efficiency, or immunity to disease) among individual birds. Furthermore, as mentioned above, our model does not account for demographic stochasticity related to skewed sex rations. Our model likely underrepresents the effects of demographic stochasticity at the smallest population sizes (i.e., populations with 50 females or fewer). Therefore, the time between reaching a size of 50 females and extirpation may be smaller than shown in model results.

Our model also accounts for environmental stochasticity, but we lack the knowledge to make connections between particular environmental parameters and murrelet demographic parameters. Given the lack of specific information regarding murrelet responses to particular elements of environmental variability, we have great uncertainty regarding the appropriate level of variability that should be included in our model. Our model may overestimate or underestimate the effects of environmental variation, or miss important aspects of how environmental variation affects murrelets. Furthermore, patterns of environmental variation and their effects on murrelet demographic rates may change in the future, as climate change alters marine and terrestrial conditions.

Immigration and Emigration

Our model does not account for immigration or emigration. We assume that all individuals present at sea within or off of Zone 2 are part of a closed population in Zone 2. In reality, we know that the Zone 2 population is not entirely closed; at minimum, there is some blurring between the Zone 1 and Zone 2 populations. In a radiotelemetry study, all murrelets captured and in Zone 2 waters were later observed foraging in Zone 1 waters, and some murrelets captured in Zone 1 waters were later observed foraging in Zone 2 waters (Lorenz et al. 2017, p. 312). One murrelet nesting in the interior of the Olympic Peninsula foraged in the waters of both Zones (Lorenz et al. 2017, p. 313). At minimum, the mixing of Zone 1 and Zone 2 birds at sea could obscure the true population size and trend within Zone 2. If there are two, mostly closed breeding populations, the geographic extent of the two populations may differ from the geographic extent of the Conservation Zones, again obscuring the actual population sizes and trends. If the Zone 1 and Zone 2 populations are more realistically one large, continuous, population, the population size and trend would be best represented by the estimate for Washington: 5,551 individuals as of 2018, declining at 3.9 percent per year (McIver et al. 2020, pp. 16, 18). This difference in size and trend would affect the timing of the onset of small population dynamics, but given the increased rate of decline, relative to that estimated for Zone 2 alone, over time, we would expect the population to decline to the small population sizes discussed above.

It is also possible that at least during some years, non-breeding visitors are present in the waters of Zone 2. If a substantial portion of murrelets present at sea are nonbreeding visitors to Zone 2, effects to these individuals would not affect the demography of the Zone 2 breeding population, and in this way our model could overestimate effects to Zone 2. On the other hand, in this case, the Zone 2 breeding population would be smaller than we currently estimate, and may be more vulnerable demographic stochasticity and other aspects of small population dynamics like Allee effects (see below).

Furthermore, a steady stream of immigrants, most likely from Canada, where populations are declining (Bertram et al. 2015, pp. 9, 11-12), could disguise a steeper decline in the breeding population. In this case, too, the Zone 2 breeding population would be more vulnerable to small population dynamics.

It is also possible, in theory, that the Zone 2 population decline is caused by emigration, rather than mortality. In this case, mortality of individuals that would later have emigrated to populations in other Zones within the listed range would result in effects to those other Zones, in addition to any effects that caused by proposed activities occurring within those Zones. We judge this possibility to be unlikely, however, because if we accounted for this type of immigration, the survival and fecundity parameters we would need to match a -2.2 percent growth rate for Zone 2 would need to be better than those we estimated, which already may have been unrealistically good (see below).

No Density Dependence

Our model does not include density dependence. Previous murrelet models have only included density dependence when investigating the effects of terrestrial management actions, such as forest harvest or conservation, or actions to reduce nest predator density, on murrelet populations (Akcakaya 1997, entire; Peery and Henry 2010, entire; Peery and Jones 2019, entire). Murrelet populations are most likely to be sensitive to these actions when their populations are controlled by the capacity of terrestrial nesting habitat to support reproduction. We assume that this is and has been the case in some places and times. For example, the large gap in murrelet distribution between the small murrelet populations in Mendocino County, California (Zone 5) and Central California (Zone 6) most likely came about due to extensive timber harvest in the past (USFWS 1997, p. 15). However, in Washington, it appears that the amount of nesting habitat is not the only limiting factor for murrelet populations, given the large amount of apparently suitable nesting habitat per individual marbled murrelet, and long at-sea commuting distances to reach foraging habitat (Lorenz et al. 2017, p. 318). Furthermore, if the population decline in Washington were a result of the population being above the current carrying capacity, we would expect to see increases in the ratio of juveniles to older birds as the population declines (Lorenz and Raphael 2018, PAGE). Instead, juvenile ratios have remained stable and low (Lorenz and Raphael 2018, PAGE). Therefore, it seems most likely that the current population declines in Washington are not controlled simply by a carrying capacity, and that our model is reasonably realistic in not including one.

Allee effects are another form of density dependence not included in our model. Allee effects are positive associations between population density and survival or reproduction. These effects occur when too few individuals are present in a population, or population density is too low, to support essential social behaviors. At small population sizes, skewed sex ratios, as discussed above, can be one source of Allee effects (Courchamp et al. 1999, pp. 405, 407; Lande 1998, p. 357). Additionally, for murrelets, there is evidence that breeding behavior is socially facilitated, and murrelets are sometimes categorized as semi-colonial in their nesting behavior. Murrelets have been observed in nesting habitat demonstrating social behaviors, such as circling and vocalizing, in groups of up to ten birds (Nelson and Peck 1995, p. 51). Spatial clustering of nest sites is documented and suggests semi-coloniality (Conroy et al. 2002, p. 131; Meyer et al. 2002, p. 103; Naslund et al. 1995, p. 12). Even in places where the clustering of nests is not marked enough to suggest semi-coloniality, usually multiple nests can be found in a contiguous forested area (Evans Mack et al. 2003, p. 6). Recent experimental work shows that playback of recorded murrelet calls within unoccupied suitable nesting habitat appears to attract murrelets to nest there during the following nesting season, indicating that murrelets select nesting habitat at least in part on the basis of the presence of conspecifics (Valente et al. 2021, pp. 7-9). Therefore, we expect that small population sizes or low population densities would interfere with breeding behavior, further reducing marbled murrelet reproductive rates. No data are available regarding the minimum population sizes or densities needed for effective murrelet nesting behavior. However, an expert panel previously hypothesized that the threshold for Allee effects would fall between one percent of the initial population abundance and the square root of the initial abundance (Akcakaya 1997, p. 12). Calculation of the threshold would then depend on the selection of the spatial scale and timeframe of the initial conditions. Because we do expect that Allee effects will influence murrelet population dynamics at some small population size, our model likely underestimates the likelihood or speed of extirpation, once that unknown small population size is achieved.

Model Parameters

As noted above, the demographic parameters needed to reproduce the current best estimate of a -2.2 percent annual population trend in Zone 2 are toward the high end of those observed in the entire murrelet range, and are higher than most of those observed within the listed range. This leads us to the hypothesis that the current estimate of a -2.2 percent annual rate of population change may underestimate the decline of the population of murrelets breeding in Zone 2, perhaps due to the presence at sea of murrelets from other parts of the listed rage (e.g., Zone 1) or from outside of the listed range.

Survival

Our model incorporated an adult survival rate of 0.91. As noted in the literature review above, a reasonable range of adult survival rates for murrelets, derived from the literature, would be between 0.82 and 0.93. Within the listed range, the highest empirical estimates of adult survival rate were made within the Central California population, and were at or near 0.91 (Beissinger and Peery 2007, p. 199; Peery et al. 2006b, p. 83). We note that in one case, the estimate of a 0.912 survival rate applied only to females, but males had a lower survival rate of 0.846 (Peery et al. 2006b). Therefore, we regard 0.91 as an optimistic estimate of survival rates in Zone 2, though within the range that is realistically possible.

Fecundity

Our model incorporated a fecundity rate (annual number of female offspring fledged per female adult) of 0.155. There are multiple combinations of breeding propensity and nest success that could lead to this fecundity rate. We arrived at this fecundity rate by combining a breeding propensity of 0.74 with a nesting success rate of 0.42. Alternatively, a fecundity rate of 0.155 would be possible with a breeding propensity of 0.95 and a nesting success rate of 0.326. Since 0.95 was the highest breeding propensity rate we considered plausible, this means that 0.326 is the lowest nesting success rate that could result in fecundity of 0.155. The same fecundity could result from a nesting success rate of 0.48 and breeding propensity of 0.69. Since 0.48 was the highest empirically-derived nesting success rate we found in the literature, 0.69 would be the lowest breeding propensity plausibly compatible with a 0.155 fecundity rate.

In contrast, empirical data specifically from Washington indicate that breeding propensity and nesting success are outside of these ranges. A radiotelemetry study including birds from both Zone 1 and Zone 2 indicated that breeding propensity was somewhere between 0.13 and 0.2, and the nest success rate was 0.2 (Lorenz et al. 2017, pp. 312, 316). These estimates are outside of the range that would be compatible with a 0.155 fecundity rate. These estimates could be biased low due to the use of radiotelemetry, but the bias would need to be dramatic to bring them into the range compatible with our fecundity rate. Empirical breeding propensity estimates made within the listed range may be as high as 0.68, if brood patches are counted as evidence of a nesting attempt (Peery et al. 2004b, p. 1093). This estimate is compatible with a 0.155 fecundity rate. Also, one study did provide a nest success rate of 0.43 for Washington and Oregon combined, similar to the 0.42 nest success rate we used (Hamer and Nelson 1995a, p. 90). Combining these two estimates, however, would result in fecundity of 0.146, rather than 0.155. Therefore, although our fecundity estimate of 0.155 is realistically plausible for the species, it is at the high end of what we might expect within the listed range, or potentially higher.

Overall trend

The annual population trend estimate for Zone 2, of -2.2 percent per year, is subject to considerable uncertainty. The 95 percent confidence interval includes rates as low as -5.8 percent per year and as high as 1.5 percent per year (McIver et al. 2020, p. 18). Although we regard the -2.2 percent per year estimate as the best available information, the demographic rates associated with a -2.2 percent annual trend are higher than we would typically expect based on empirical data collected within the listed range. In light of the demographic information available for murrelets in Washington and generally within the listed range, rates of population change in the lower portion of the 95 percent confidence interval are more plausible than rates of population change in the higher portion of the 95 percent confidence interval. In this case, the timing with which the Zone 2 population reaches the small population sizes discussed above is likely to be faster than that described for our full set of simulation runs, and perhaps more similar to that described for the subset of simulation runs with the worst demographic rates.

Other Factors to Consider Qualitatively

Climate change

Climate change is occurring within Zone 2, and is likely already affecting murrelet populations in Zone 2 to some extent (see Status of the Species Appendix X). In general, the effects of climate change are expected to be negative, and may include reductions in the availability of nesting habitat and prey, as well as reduced nutritional content of prey, increased toxicity of prey, and exposure to novel pathogens. We expect that climate change will result in reductions in fecundity and, at least occasionally, adult survival. However, we currently have no way of estimating the magnitude of these reductions. Climate change is likely to increase the variability of environmental conditions, but we are unable to quantify the ways in which this increased variability will be likely to alter murrelet vital rates. We did not attempt to include climate change in our model. However, because the most likely effects of climate change include reductions in survival and reproduction, the subset of simulation runs with the worst demographic rates may represent, to some degree, the future effects of climate change on the Zone 2 murrelet population.

Human population growth

Human population growth in the terrestrial areas of Zone 2 are likely to result in the addition of increased anthropogenic toxins to the marine environment, as well as increased shipping traffic, including the shipping of oil, within and near Zone 2 (see attached Opinion, Cumulative Effects section). Although these issues are not likely to be as acute in Zone 2 as in other parts of the range, they also may contribute to reduced reproduction and occasional increases in mortality (e.g., from oil spills) within Zone 2. Although we have no way to quantify the demographic effects associated with increased human population growth in this region, we again refer to the subset of simulation runs with the worst demographic rates as a potential representation of the combined effects of climate change and human population growth.

Conclusion

Although the various factors discussed above alter some aspects of the interpretation of the model, none of those factors undermine the basic conclusion that the Zone 2 population is now declining and, as long as that remains true, will eventually approach small population sizes such that small population dynamics will become the determining factor in the fate of the population. The effects of the proposed action will lead to a long-term reduction in the population size, and increase in the likelihood, at any given time, that the population will be reduced to small population sizes.

Given the factors discussed above, we make the following recommendations regarding the interpretation of the model results. First, model results for time periods in the distant future are less reliable than model results for the near- to medium-term (i.e., the next few decades), because we were unable to incorporate information regarding the future effects of climate change and human population growth into our model. Second, model results for the likelihood of extirpation are likely to be unreliable, given that our model does not include the effects of skewed sex ratios or other aspects of reproduction that would lead to Allee effects at small population sizes. We recommend, instead, focusing on the likelihood of reaching the small population thresholds that precede extirpation. Finally, although the Zone 2 population trend of -2.2 percent, which we used to set the demographic rates for this model, represents the best currently-available estimate of population change, the associated demographic rates appear rather optimistic in light of empirical information collected within the listed range, especially in light of the likely effects of climate change and human population growth. Therefore, the subset of simulation runs with the worst demographic rates may be informative regarding the potential future trajectory of the Zone 2 population. However, because this was a fairly small subset of all simulation runs, results associated with this subset should be regarded more as ballpark estimates rather than as precise predictions.

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