# APPENDIX A
# PUGET SOUND NEARSHORE HABITAT CONSERVATION CALCULATOR USER GUIDE

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# Puget Sound Nearshore Habitat Conservation Calculator
# User Guide

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## Abbreviations

| | |
|---|---|
| DSAYs | Discounted Service Acre Years |
| DSZ | Deeper Shore Zone |
| EFH | Essential Fish Habitat |
| ESA | Endangered Species Act |
| GIS | Geographic Information System |
| HAT | Highest Astronomical Tide |
| HEA | Habitat Equivalency Analysis |
| LSZ | Lower Shore Zone |
| MHHW | Mean Higher High Water |
| MLLW | Mean Lower Low Water |
| NHVM | Nearshore Habitat Values Model |
| NOAA | National Oceanographic and Atmospheric Administration |
| NRDA | Natural Resource Damage Assessments |
| OWS | Overwater Structures |
| PS | Puget Sound |
| SAV | Submerged Aquatic Vegetation |
| SSNP | Salish Sea Nearshore Programmatic |
| USACE | US Army Corps of Engineers |
| USFWS | US Fish and Wildlife Service |
| USZ | Upper Shore Zone |
| WDOE | Washing State Department of Ecology |
| WDFW | Washington Department of Fish and Wildlife |
| WDNR | Washing State Department of Natural Resources |

# Definitions

**Action Agency:** Federal Agency seeking ESA section 7 consultation with the National Marine Fisheries Service or US Fish and Wildlife Service.

**Conservation Credit (credit):** A unit of measure (e.g., a functional or areal measure or other suitable metric) representing a gain in ecological functions for Puget Sound Chinook and Hood Canal summer-run chum at a mitigation site. The measure of ecological functions is based on the resources restored, established, enhanced, or preserved. As part of Puget Sound Nearshore consultations, a credit is determined using the Conservation Calculator or other Services and Action Agency approved habitat quantification tool.

**Conservation Debit (debit):** A unit of measure (e.g., a functional or areal measure or other suitable metric) representing the loss in ecological functions at an impacted site. The measure of ecological functions is based on the resources impacted.

**Conservation Points:** Conservation Points are Discounted Service Acre Years multiplied by 100. This creates more intuitive outputs for small impacts.

**Discounted Service Acre Years (DSAYs)**: Measure of change in habitat services provided over a specific duration of time to a set of target species within the Habitat Equivalency Analysis (HEA) methodology.

**Force Majeure:** Unexpected circumstances including accidents and extreme weather that may damage structures.

**Minor Maintenance:** Minor servicing of an existing structure that does *not meaningfully prolong the life of the structure*. For minor maintenance, a structure must remain the same size and within its current footprint. Minor Maintenance activities do not have to be entered into the Conservation Calculator. Further, minor maintenance includes the repair and replacement of previously mitigated elements during the first half of their design life.

**Repair:** Partial replacement, reconstruction, or rehabilitation of a structure that meaningfully extends the life of that structure.

**Replacements:** Reconstruction of an identical or highly similar structure in the same location as the structure being replaced.

**Service Area:** The service area is the geographic area in which conservation credits and debits can be traded to ultimately offset the loss of salmonid resource functions.

**Standalone Restoration:** A standalone restoration project restores or improves habitat functions without introducing new, or temporally extending, adverse effects aside from construction-related effects. Standalone restoration projects include removal of a structure that has adverse effects but does not include any replacement.

# Introduction to the Conservation Calculator

## *What is the Puget Sound Nearshore Conservation Calculator?*

NOAA and the US Fish and Wildlife Service (USFWS), collectively "the Services," developed the Conservation Calculator as a user-accessible tool that simplifies the application of the Habitat Equivalency Analysis (HEA) and Nearshore Habitat Values Model (NHVM) (Figure 1). The goals of the Conservation Calculator are to:

- Quantify the habitat impacts relevant for Puget Sound (PS) Chinook salmon and Hood Canal summer-run chum from a proposed project and the habitat benefits from mitigation projects in terms of a common habitat metric.
- Allow the Services, Action Agencies, and project applicants to simultaneously and consistently apply both HEA and NHVM for proposed actions in the Puget Sound nearshore environment.
- Facilitate avoidance, minimization, and, where warranted or otherwise appropriate, no-net loss of nearshore habitat functions for PS Chinook salmon and Hood Canal summer-run chum by quantifying habitat impacts from proposed project actions (construction, repair, replacement, mitigation).

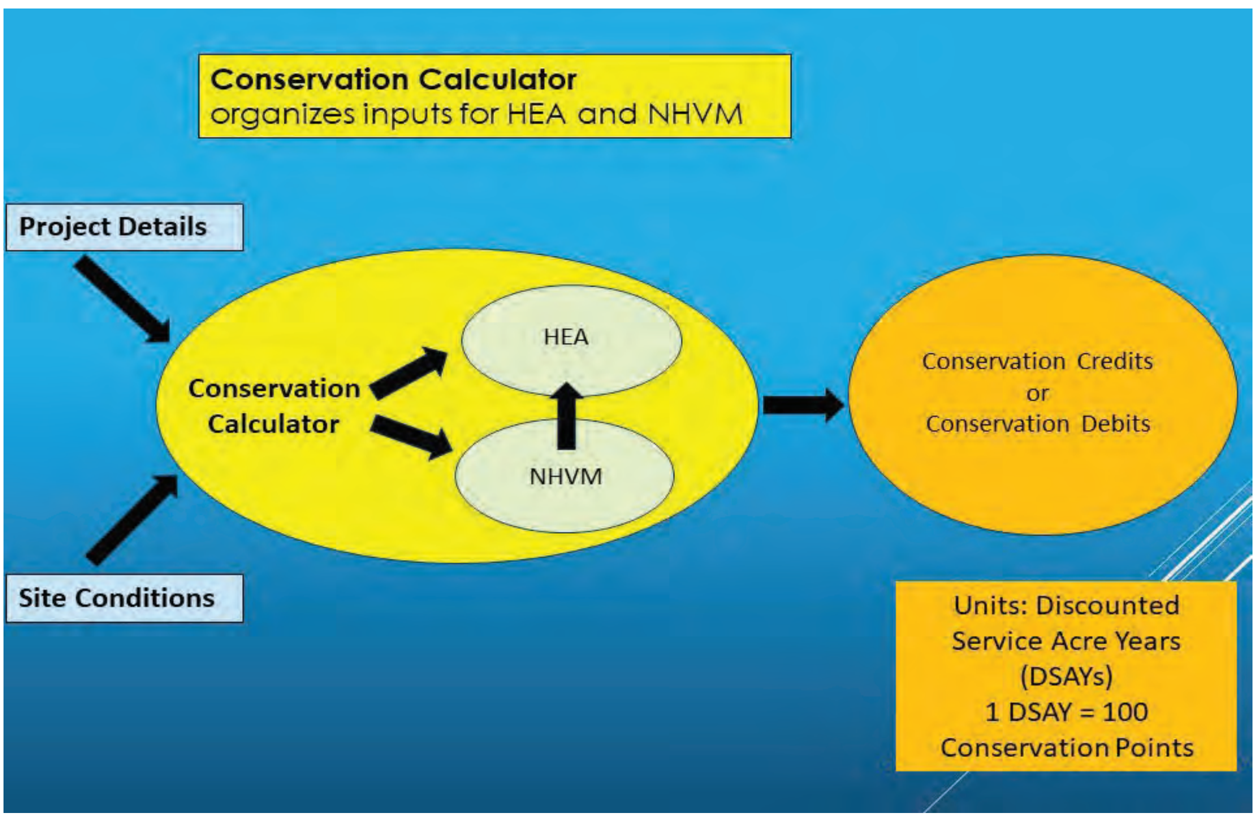



*Figure 1. The Conservation Calculator is an interface for Habitat Equivalency Analysis (HEA) and the Nearshore Habitat Values Model (NHVM).*

The Conservation Calculator is a user interface to the NHVM and HEA. It facilitates determination of **conservation debits** resulting from nearshore projects that decrease habitat function and **conservation credits** that are associated with projects that increase nearshore habitat function.

The Conservation Calculator allows the Services to assess habitat impacts and benefits in Puget Sound from several actions including:

1. Addition of new, replacement, and removal of overwater structures including piers, ramps, floats, house-boats, decks, piles, etc.
2. Removal of creosote
3. Addition of new, replacement, and removal of shoreline armoring
4. Addition of new, replacement, and removal of boat ramps, jetties, and rubble
5. Addition of new, and removal of riparian plantings
6. Addition of new submerged aquatic vegetation (SAV) plantings
7. Addition of forage fish spawning supplement/beach nourishment
8. Maintenance dredging

The Conservation Calculator is adaptable and allows the Services to make updates as new science or best available information becomes available. The Conservation Calculator also allows for expanding the types of analysis to account for the different types of nearshore development actions that could occur. (Note: These changes, if necessary, will be scheduled for predictable and regular updates. See below for specifics).

### *Habitat Equivalency Analysis*

The **Habitat Equivalency Analysis (HEA)** methodology assesses impacts (net ecological loss) and benefits (net ecological gain) to the habitat. Ecological equivalency provides the basis of HEA as a concept that uses a common medium of exchange called **Discounted Service Acre Years (DSAYs)**. DSAYs express and assign a value to functional habitat loss and gain over a certain time period. Ecological equivalency is a service-to-service approach where the ecological habitat services relevant for a species or group of species impacted by an activity are fully offset by the services gained from a conservation activity. This is further explained in Ray (2008).

The NOAA Restoration Center developed HEA in cooperation with stakeholders and it has become a common method for Natural Resource Damage Assessments (NRDAs). NOAA’s Central and North Puget Sound area offices chose the HEA methodology for its Endangered Species Act (ESA) consultations and developed the NHVM and Conservation Calculator to facilitate the use of the HEA model. Not only has HEA been successfully used in multiple NRDA proceedings, it also addresses temporal impacts of the design life of nearshore structures.

The use of HEA requires several input parameters including nearshore habitat values (Figure 2). Habitat values characterize the functions and value of a specific habitat for the target species before and after an impact/restoration. A team of NOAA biologists developed a NHVM to aid in determining these habitat values specific to juvenile PS Chinook salmon and Hood Canal summer-run chum. The NHVM’s structure and values are specific to quantifying habitat conditions for the designated critical habitat of listed PS Chinook salmon and Hood Canal summer-run chum. The NHVM accounts for a range of habitat values (low to high depending on functionality and importance to the species). The NHVM design and values were derived from scientific literature and best available information, as required by the ESA. The resulting NHVM allows for consistent determination of habitat values across the Puget Sound nearshore through consideration of site-specific conditions.

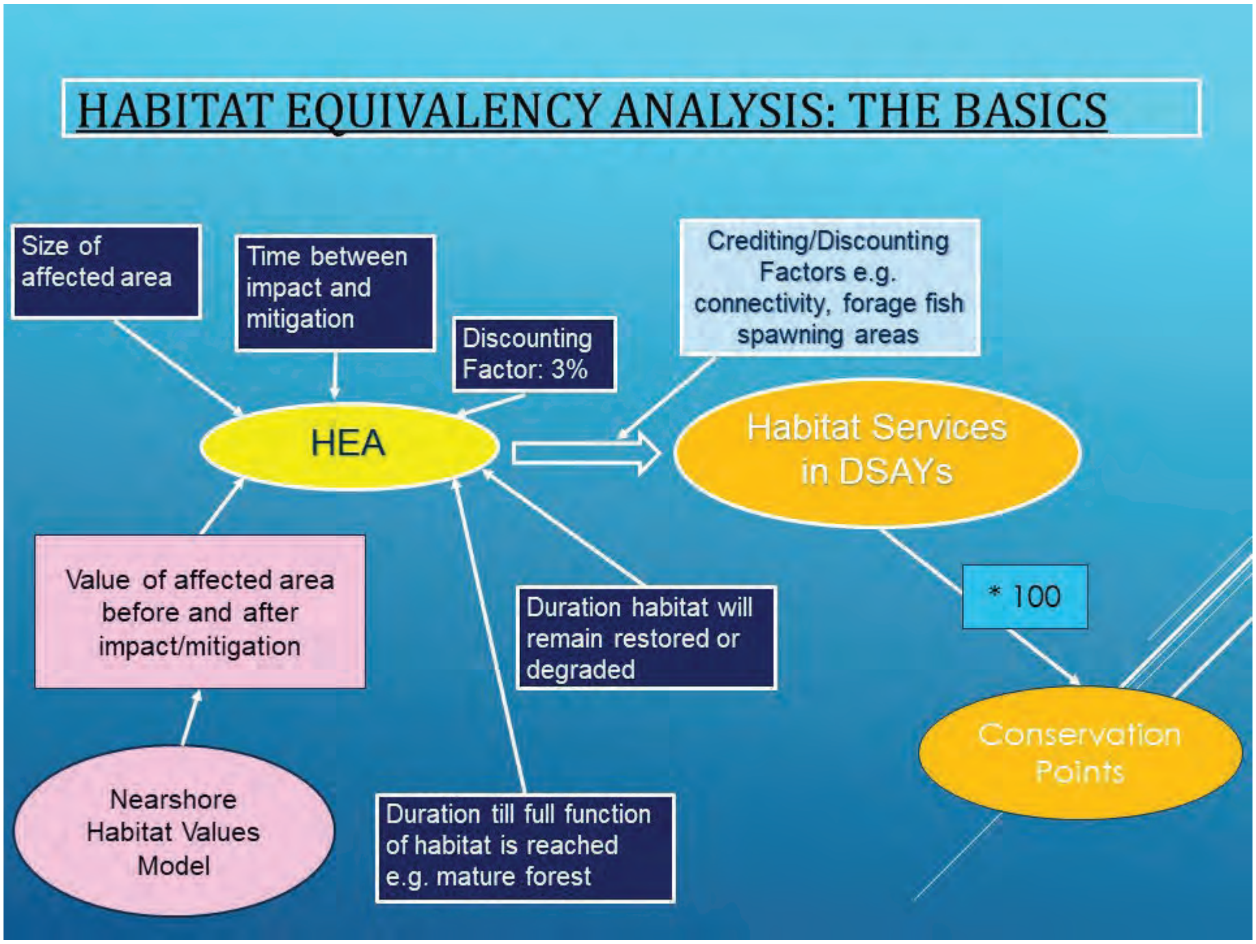



*Figure 2. Components of Habitat Equivalency Analysis (HEA). Inputs include nearshore habitat values (pink) and additional parameters (navy). Outputs (orange) include DSAYs – conservation credits or debits – and* ***Conservation Points****. Conservation points are DSAYs multiplied by 100 which allow the user to work with more intuitive outputs for small impacts.*

## *Nearshore Habitat Values Model*

The NHVM determines the habitat value, by ranking the existing conditions of physical and biological functions of salmonid critical habitat (50 CFR 226.212) for each of five elevation zones (Figure 3) in the subject habitat. The physical and biological functions for marine and estuarine critical habitat used for the NHVM include the unobstructed migratory corridor, cover and primary production, sediment quality and quantity, and water quality.

We split the marine/estuarine nearshore into five elevation zones based on their accessibility and function for the target species. The **Riparian Zone (RZ)** extends 130 feet landward from HAT. This is the area we found most relevant for supporting water quality and food provisioning for salmonids (see *Tab 3: RZ (Riparian Zone)* for more information). The **Upper Shore Zone (USZ)** extends between HAT and plus five Mean Lower Low Water (MLLW). The USZ is further split into **USZ 1** and **USZ 2** with the USZ 1 extending from HAT to Mean Higher High Water (MHHW). The duration and extent of tidal inundation in the USZ 1 is very limited and thus salmonid access, as well as sand lance and surf smelt spawning, is generally limited to the USZ 2. Based on the reduced extent, frequency, and duration of aquatic access for those species, we assigned the USZ 1 a lower maximum habitat value than the USZ 2 (Figure 4). The **Lower Shore Zone (LSZ)** extends from plus five MLLW to the deepest extent of submerged aquatic vegetation (SAV). All SAV is contained in the LSZ. The **Deeper Shore Zone (DSZ)** begins at minus 10 feet MLLW or the lowest limit of SAV growth. There is no defined limit end to the DSZ.

The five different shore zones provide different maximum habitat values for juvenile PS Chinook salmon and Hood Canal summer-run chum. Habitat values range from a minimum of 0 to a maximum of 1. The maximum habitat values that each zone can provide is based on the maximum possible contribution of habitat functions in that zone (Figure 4). For juvenile salmonids in the marine nearshore **–** a maximum habitat value of 1 **–** is an eelgrass meadow or other dense SAV providing food, cover, and an unobstructed migratory corridor. While the DSZ also provides migratory corridor function and forage (via primary production and drift in) for juvenile salmonids, it generally produces less forage than the LSZ as it does not contain SAV. While the riparian zone is not used directly by salmonids, it provides important functions for juvenile salmonids including provision of food via allochthonous[5] input including insects. Corresponding maximum habitat values are shown in Figure 4.

[5] Allochthonous: Material that has been imported from outside of the system or considered area.

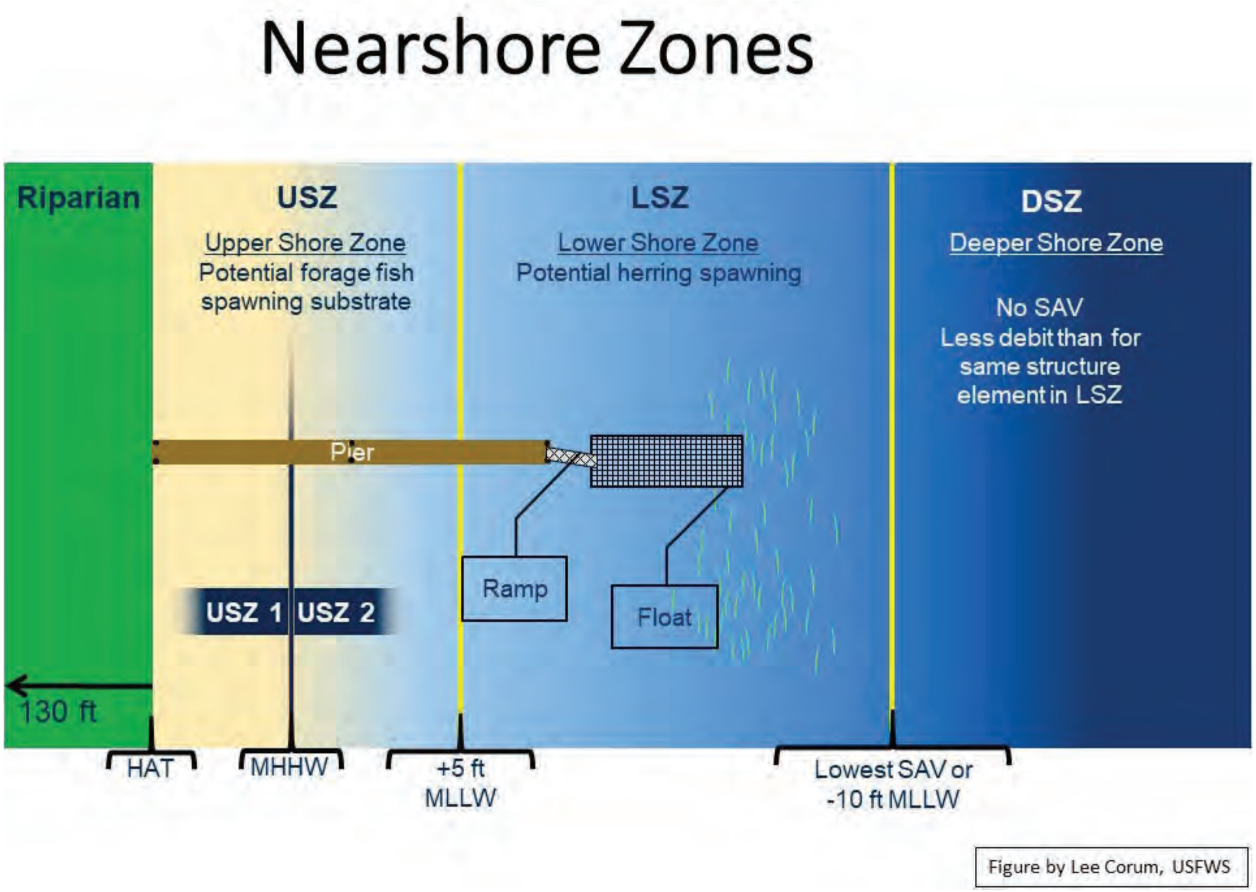



*Figure 3. The Five Puget Sound Nearshore Zones. From highest to lowest elevation they are the Riparian Zone (RZ), Upper Shore Zone 1 (USZ 1), Upper Shore Zone 2 (USZ 2), Lower Shore Zone (LSZ), and Deeper Shore Zone (DSZ). Figure by Lee Corum, USFWS.*

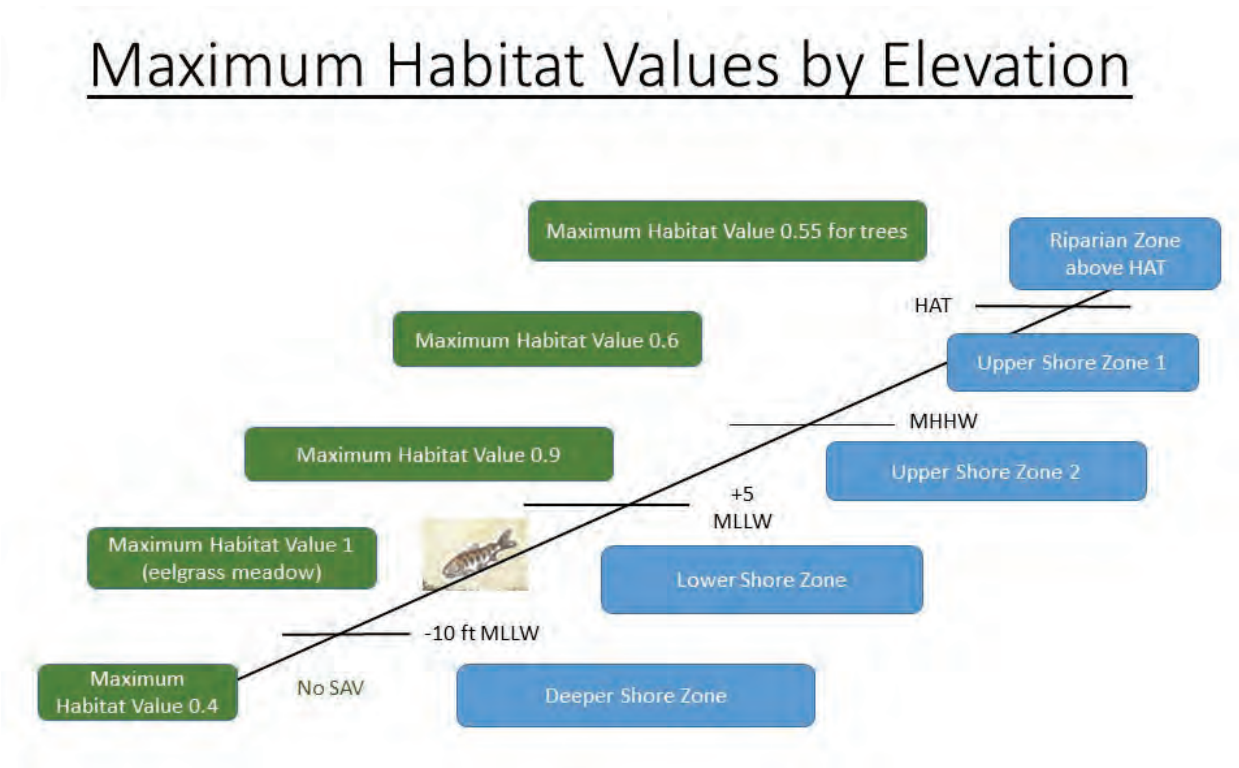



*Figure 4. Maximum Habitat Values by Elevation. The corresponding nearshore zones are shown to the right. Values for each zone range from a minimum of 0 to a maximum of 1.*

We mention these details to create an understanding about how some of the input requested for the Conservation Calculator is used. For example, to evaluate the cover and primary production in the LSZ, the NHVM uses presence and quality of SAV. For that evaluation, an assessment of the SAV condition via online resources or field surveys is needed.

## Application of this Tool

The Conservation Calculator can be used to quantify habitat impacts for projects within marine and some estuarine environments of Puget Sound, including projects within the salt wedge[6] of riverine systems. The Conservation Calculator is not appropriate for application in estuarine environments that do not fall within the shoreline descriptions outlined above (Figure 3), such as tidally influenced wetlands with backwater channels. Project elements upstream of documented salt wedges are also not suitable to be evaluated by this Conservation Calculator.

Conservation Calculator outputs are based on the evaluation of changes in physical and biological functions and their indicators relevant for PS Chinook salmon and Hood Canal summer-run chum productivity and abundance. The evaluation framework is dependent on the

[6] The salt wedge is defined as the area of intrusion of salt water into a tidal estuary in the form of a wedge along the bed of the estuary.

existing HEA model and the NHVM. The evaluation of project related changes is based on best available science, context-specific application of ecological processes, and best professional judgment. As with most other rapid assessment methods (Adamus, P, K. Verble 2020), field verification requires periodic process improvement.

If submitting a Conservation Calculator with a consultation request, ensure that the most current version of the Conservation Calculator and user guide is being utilized. The Conservation Calculator will be updated February 1st of every calendar year to incorporate applicable new science, monitoring results, additional modules, or other procedural, design, or usability improvements.

## Avoidance and Minimization

NOAA strongly encourages applicants and consultants to evaluate nearshore projects with the Conservation Calculator *prior* to engaging in ESA consultation. Consider options for reducing conservation debits before project submission by (1) reducing the size of the structure, (2) incorporating hybrid or soft armoring for bulkheads, and/or (3) evaluating possible restoration on-site or on adjacent properties. On-site offsets, such as creosote removal, riparian plantings, or structure removal, may not be enough to reduce all debits associated with high impact projects.

## Best Management Practices for Structures Evaluated with the Conservation Calculator

To reduce project impacts and associated debits, applicants should strive to minimize habitat impacts associated with their nearshore structures. Minimizing the project footprint to the greatest possible extent and avoiding areas with greater habitat value reduces the associated conservation debits.

In detail, best management practices (BMPs) to minimize impacts include:

1. Minimize the total size (area) of coverage or linear feet of the structure.
2. Reduce shoreline armoring (seawalls, bulkheads, abutments).
    a. Instead of a traditional “hard armoring” bulkheads (concrete, steel, rock), use soft-shore or hybrid armoring whenever possible. The Washington State Department of Fish and Wildlife (WDFW) Your Marine Waterfront guide is a valuable resource for minimizing your environmental impact. Soft or hybrid armoring is not entered into the Conservation Calculator, and therefore does not accrue debits!
    b. Relocate shoreline armoring as far landward as possible to reduce impacts to the USZ. Armoring landward of HAT does not incur debits for placement of armoring, but may incur small debits related to impacts to riparian vegetation.
    c. Slope rock bulkheads landward and incorporate native woody plantings.

    d. Place vulnerable structures (like homes) as far landward of the shoreline as possible to reduce dependence on shoreline modification.
3. Minimize impacts to SAV.
    a. Delineate SAV for the project area within 25 feet of proposed structures. If SAV is found within that area, then delineate the entire property and choose a location for the structures that demonstrates the greatest avoidance and minimization of vegetation.
    b. Floating structures should never "ground out" on the substrate, and stoppers/pin piles/feet should hold the structure at least 12 inches above the substrate.
    c. If SAV is present within 25 feet of the proposed float, the bottom side of the float must be elevated at least 4 feet above the substrate at low tide to reduce prop scour impacts on SAV.

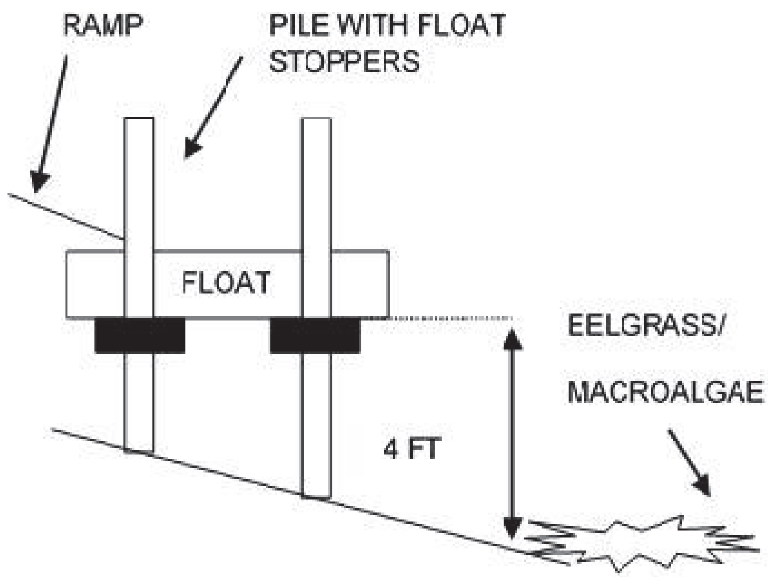



*Figure 5. Proposed Float within 25 feet of SAV. Side-view. The bottom side of the float must be elevated at least 4 feet above the substrate at low tide.*

    d. We request SAV field surveys for most replacement projects[7]. However, applicants should include a description of the SAV to the best of their ability using the following resources:
        i. Submit photographs of the LSZ taken at low tide between June 1 through October 1. An underwater camera (GoPro or equivalent) is ideal for photographing the LSZ area that is still underwater at low tide.
        ii. Washington Marine Vegetation Atlas from the Washington State Department of Natural Resources (WDNR)
        iii. Coastal Atlas mapping tool from the Washington Department of Ecology (WDOE)
        iv. Old SAV surveys and SAV surveys from adjacent areas.

[7] NMFS accepts field surveys that follow the WDFW SAV interim survey guidelines.

In the absence of a description, survey, or photos that provides reasonable certainty of a vegetation condition rating as described in this User Guide, NMFS biologists will assign an SAV score based on available data.

If the project area is located in areas with dense SAV or native eelgrass (*Zostera marina*) and avoidance and minimization of impacts cannot be achieved with on-line resources, a field survey and delineation may be required to demonstrate how the project will avoid and minimize impacts.

4. Minimize impacts to forage fish spawning substrate by avoiding spawning areas, which can be found using WDFW's Forage Fish Spawning map. If this is not possible, construct 100% grated piers and ramps over spawning habitat and minimize the number of piles in the USZ.
5. Maximize light penetration
   a. Pier surfaces and ramps should be entirely grated with at least 60% open space.
   b. Floats should be grated to the maximum extent possible. To qualify as a grated float in the Conservation Calculator, floats must have 50% effective grating with 60% or more open space (Compliant with WAC 220-660-280).
   c. Install a mooring buoy in the DSZ rather than a boat lift in the LSZ.
6. Minimize impacts from piers
   a. Minimize the width of the pier. We recommend a pier width of 4 feet for residential structures, and as narrow as possible for commercial structures (ADA compliance may impact how wide the structure must be).
   b. Piers should be a straight line rather than finger, "L," or "T" shaped.
   c. Do not construct additional structures on piers (i.e., buildings, planter boxes, slides, etc.). Solid structure areas must be entered in the Conservation Calculator as a solid pier, which has more habitat impacts than grated surfaces.
   d. Stairways should be open-frame construction and not solid structures (i.e., concrete). The width of stairway landings and steps should not exceed 4 feet for single-use and 6 feet for joint-use.

## User Requirements

Use of the Conservation Calculator requires a moderate to substantial knowledge of nearshore ecology and coastal geology, and experience with field data collection methods including determining some tidal elevation. Field data that are necessary for use of the Conservation Calculator also include SAV surveys and forage fish surveys or appropriate use of existing information. Users will need to have experience with geographic information systems (GIS) or Google Earth, aerial photo interpretation, and/or field evaluation experience, depending on project type. Users will need to be able to interpret maps related to areas valuable for the target species including maps of natal estuaries, pocket estuaries,[8] WDOE's Coastal Atlas map at https://apps.ecology.wa.gov/coastalatlas/tools/Map.aspx, and WDFW's Forage Fish Spawning

[8] Map layers are provided on NOAA web page and as hot links in the Conservation Calculator

map at https://www.arcgis.com/home/item.html?id=19b8f74e2d41470cbd80b1af8dedd6b3. In addition, the user must have access to the internet and Microsoft Excel 2007 or later. Moderate Microsoft Excel knowledge allows for further understanding of equations used within the calculator.

Resources on NOAA's PS Nearshore web page at https://www.fisheries.noaa.gov/west-coast/habitat-conservation/puget-sound-nearshore-habitat-conservation-calculator, along with this user guide, provide information regarding the use of the Conservation Calculator. Further, pre-consultation technical assistance meetings can provide a venue for applicants and consultants to get help with more complicated projects. Email PSNearshoreConservation.WCR@noaa.gov for questions. NOAA will continue to provide training and technical assistance for use of the Conservation Calculator by adding training materials and updates to the PS Nearshore web page mentioned above.

## Conservation Calculator as part of ESA Consultations in Puget Sound

The Conservation Calculator is a tool that can be used by agency staff, environmental consultants, non-profit and corporate staff, and project proponents. Users can download the Conservation Calculator and enter project specifications to determine credit and/or debits. Project specific Conservation Calculators are needed for most, if not all, Puget Sound nearshore projects. For example, no-net loss is required as part of the proposed action in the Salish Sea Nearshore Programmatic (SSNP) biological opinion. A tool to demonstrate no-net loss is the Conservation Calculator.

## Conservation Calculator Process Improvements

The Services will apply new science, incorporate monitoring results, and process improvements to the Conservation Calculator, NHVM, and this user guide with thorough, regular and predictable updates.

Throughout the year, we encourage users to send improvement suggestions, new and relevant science, and potential bugs to PSNearshoreConservation.WCR@noaa.gov at NOAA and questions specific to USFWS species to (annelise_hill@fws.gov).

The Services plan to post any updates to the Conservation Calculator and user guide, if needed, on February 1st of every year. We document all updates and additions in a separate document, the Change Log, also available on our web page. In the event a more critical update would need to occur sooner, the Service will make every effort to update the website and user forums. Annual updates may include adjustments to credit factors, updates to maps related to the credit factors, and changes based on new science, policies, and feedback from applicants. Changes may also include improvements to the layout of the Conservation Calculator and user guide.

When a project specific Conservation Calculator is submitted as part of an ESA consultation initiation package, NMFS requests that applicants and their agent submit the most recent version of the Conservation Calculator posted on NMFS's web page. Once a project is initiated or for a programmatic implementation NMFS confirms that the project fits the programmatic, the project Conservation Calculator version is final and will stay with the project.

After the annual February Conservation Calculator update has been posted on PS Nearshore web page, applicants whose projects have not been initiated or whose programmatic implementation has not been confirmed by NMFS to fit a programmatic may amend their project file with a new Conservation Calculator using the updated February version.

## Conservation Calculator Training

Materials from the January 2021 Conservation Calculator workshop are available on the PS Nearshore web page at https://www.fisheries.noaa.gov/west-coast/habitat-conservation/puget-sound-nearshore-habitat-conservation-calculator. We strongly encourage users to review this training before sending questions about calculator entry or requesting additional training. We plan to offer follow-up training. To receive updates regarding training, new material, updated versions of the Conservation Calculator and User Guide, sign up for our listserv on the PS Nearshore web page.

## Conservation Offsets

### *Applicant-responsible Credit Generation*

Conservation credits to offset impacts can be generated by engaging in **standalone restoration** actions. Applicant-generated conservation credits can be generated on the same site as a project causing debits or within the same service area. For example, an applicant may remove structures in the nearshore of the same service area of an impacting project to generate conservation credits to offset debits. The removal must be a standalone and separate action and cannot be integral to another project.[9] Standalone applicant-responsible credit generation includes:

- Removal of individual creosote piles not associated with a structure
- Removal of an overwater structure (either containing creosote or not)
- Removal of a portion of a structure[10]
- Removal of shoreline armoring (complete armoring, not a portion)
- Riparian plantings
- Beach nourishment

[9] Residual applicant-generated credits from a replacement project cannot be used as credits for a different debit project.

[10] Partial structure removal is limited to distinct portions that can be removed as a standalone project without increasing the environmental risk associated with the remaining portion of the structure.

### *Reporting for Applicant-responsible Credit Generation*

Creosote: After creosote removal and upland disposal, applicants must submit the disposal receipts and a picture of the dump truck on the scale to the Services. Disposal receipts need to contain actual weight of the total removed creosote.

### *Removal Credits*

The Conservation Calculator is set up to determine credit for the removal of existing structures and creosote. For existing structures, we make the average estimate that at the time of permit application, the existing structure would remain in lawful and in a structurally sound and good condition for a period of 10 years. If structures are non-functioning, deteriorated and/or falling apart, or otherwise not in good condition as required by their permit, removal credit is generally not justified.

A request for an emergency authorization usually indicates that a structure has not been maintained in a good condition. Conservation credit for the removal of structures under an emergency authorization may apply only in very limited circumstances. Projects that received an emergency authorization from the US Army Corps of Engineers (USACE) may receive credits for the removal of the old/existing structures if the structure was in good condition at the time of permit application and dated pictures can be provided.

## Preservation of Existing Habitat

Stay tuned for February 2023, we are working on this.

## Service Area

To effectively offset impacts, credits must originate from within the same **service area** as debits. Service areas may vary depending on the credit provider and extend as far upriver as the maximum extent of saltwater intrusion in river mouths. See the PS Nearshore web page for links to conservation credit provider specific service area maps.

## General Information - Applicable to All Structure Entries

### *Replacements, Repairs, Minor Maintenance*

For the purposes of SSNP, replacements, repairs, and minor maintenance are part of individual activity categories. SSNP activity categories that cover repairs and replacement and require conservation offsets are “Shoreline modifications” and “Repair or replace an existing structure.” “Repair or replace an existing structure” includes: Aids to navigation; boat houses; covered boat houses; boat garages; breakwaters; commercial; industrial and residential piers; pier, ramp, and floats; float plane hangars; floating walkways; groins and jetties; house boats; boat ramps;

wharfs, port, industrial and marina facilities; dolphins, float storage units, debris booms. “Minor maintenance of an existing structure” is a separate SSNP activity category and includes: Pile resets, replacement of rubber strips, encapsulation of flotation material, replacement of fender piles that do not contribute to the structural integrity of the structure, capping of piles, replacement of flat stops, height extension of existing piles. Minor maintenance, which does not meaningfully extend the life of a structure, does not require conservation offsets.

For filling out the calculator, **replacement** means reconstruction of an identical or highly similar structure in the same location as the structure being replaced. In general, the structure that is being replaced has to be in the environment at the point of permit application for an installation to be considered a replacement. Further, to receive removal credit for the existing structure, the existing structure has to be in good condition. For more information on removal credits, review the Removal Credits section above.

For filling out the calculator, **repair** means to conduct partial reconstruction or rehabilitation of a structure that meaningfully extends the life of that structure. Repairs have impacts on critical habitat that are similar to the impacts of replacements. Like replacements, repairs extend the duration of an impact to the nearshore into the future. Such repairs include: Resurfacing boat ramps and encasing bulkheads. Most repairs have very similar or the same environmental effects as replacements. Thus, removal credits apply to most repairs even if the existing structure is not removed for the same reasons as discussed above in Removal Credits.

“Piece by Piece” approach for replacements and repairs: Only the element to be repaired or replaced is entered into the Conservation Calculator. For example, if X square feet of a boat ramp are proposed to be replaced, only those X square feet are entered into the Conservation Calculator.

In more detail, to quantify impacts from repairs and partial replacements:

1. Enter the footprint of the existing structure element that is proposed to be repaired or replaced into Entry Block III for Removal, Removal as Part of Replacements, and Repair. Structure elements to be repaired are generally eligible for removal credit.
2. Enter the footprint of the proposed replaced/repaired structure element (which should not exceed the existing footprint) in Entry Block II for Repair and Replacement. If partial replacements and repairs include design changes or improvements, like an increase in grating, those should also be reflected in Entry Block II.
3. “No Double Offsets” when replacing structurally overlapping elements. This mostly applies to overwater structures and is discussed in more detail in the section on *Tab 4: Overwater Structures - Repair of Overwater Structures*.

When filling out the Conservation Calculator, **minor maintenance** activities do not have to be entered. Minor maintenance means carrying out minor servicing at an existing structure that we have determined at this time does not meaningfully extend the life of the structure. Maintenance activities include pile resets, capping of piles, replacement of rubber strips, replacement of float stops, encapsulation of existing floatation material, height extension of existing piles, and replacement of fender piles that do not contribute to the structural integrity of the structure.

Further, for filling out the Conservation Calculator, minor maintenance includes the repair and replacement of previously mitigated elements during the first half of their design life. This includes unexpected damages caused by a force majeure, if it occurs during the first half of the structure's design life. For these situations, the Conservation Calculator does not have to be used. If a structure or elements of a structure for which conservation offsets were previously provided must be repaired or replaced for any reason during the second half of their design life, it is considered a replacement and under SSNP conservation offsets apply.

### *Entering Length and Width*

Entering floats, boat ramps, and jetties into the Conservation Calculator generally requires input of length and width parameters, rather than simply square footage. In addition to impacts related to square footage of structure, these structures have a physical buffer with added impacts factored into the final credits/debits (based on Ono et al. 2010). To correctly determine buffers, the longer side of the structure should be entered into the length field, regardless of orientation. Exceptions for overwater structures spanning several zones are discussed in the *Tab 4: Overwater Structures* section.

### *Replacement vs. New*

If the area of a replacement structure exceeds the area of the existing structure, the difference is considered to be new/expanded structure. This determination is made by structure type and shore zone.

> Example – If a replacement jetty is reduced in width but extended into the LSZ where there previously was no jetty, all area of the jetty in the LSZ is considered new/expanded.
>
> Example – If a boat ramp is replaced with a jetty, the jetty is considered to be a new structure.

A detailed discussion and more examples of expanded overwater structures can be found in section *Tab 4: Overwater Structures*.

### *Increased Credits for Removals with Site Protection*

The time horizon for credit determination associated with structure removal and no site protection on the property (e.g., a deed restriction or conservation easement) is 10 years. However, the Conservation Calculator is set up to credit removals of structure where site protections are in place for time horizons longer than 10 years. If proposing structure removals with site protections following the USACE regulations, Components of a Mitigation Plan (4) site protections instrument 33 CFR 332.4(c) §332.7(a); specific for nearshore structures, the USACE informs on deed restrictions associated with compensatory mitigation here. Contact the Services for help determining credits. If you would like an immediate **estimate** of increased credits based on a site protection, you may use Entry Block II, on the appropriate tab. Enter the dimensions of the structure to be removed as though you were installing a replacement structure; the resulting negative credits reflect the positive credits you would receive for a 40-year (design life for overwater structures and boat ramps) easement.

For shoreline armor removal the Services credit easements for a time of up to 50 years (limit based on sea level rise). Please contact Services via PSNearshoreConservation.WCR@noaa.gov for help with determining increased credits for armor removal with easements following USACE regulation.

### *Submerged Aquatic Vegetation*

1. Submerged aquatic vegetation (SAV) density informs habitat values in the Conservation Calculator. SAV surveys provide site-specific information that is used in most tabs in the Conservation Calculator.
2. Use the WDFW Eelgrass/Macroalgae Habitat Interim Survey Guidelines to conduct SAV surveys and follow the USACE "Components of a Complete Eelgrass Delineation Report" for eelgrass delineations. If surveys are conducted outside of the SAV survey window (June 1st - October 1st), NOAA may increase the SAV rating in the Conservation Calculator to account for the likely underestimate of SAV coverage outside of the main growing season. This decision depends on additional site-specific information like site specific growth patterns, temperature regime of the area, and WDFW area habitat biologist input, as available.
3. When a survey shows that no macroalgae and only eelgrass is present, we also accept an Eelgrass Delineation Report based on the Components of a Complete Eelgrass Delineation Report developed by Dr. Deborah Shafer Nelson, U.S. Army Engineer Research and Development Center; Special Public Notice May 27, 2016.
4. SAV determinations should be based on the average SAV density in the footprint of the structure including a 25-foot buffer around the structure.
5. For the determination of the SAV category based on SAV density use Table 1, which is also displayed in the Conservation Calculator reference tab.
6. SAV category determinations for replacements: For most small size replacement projects, SAV information can be provided without a new survey by using a combination of older SAV surveys, SAV surveys from adjacent properties, pictures at extreme low tides, information from Washington State Department of Ecology's (WDOE's) Coastal Atlas map, or information from WDFW biologists.
7. Structure removals with SAV have two options:
    a. Enter the SAV category based on the average cover density as outlined above in number 4.
    b. Enter the SAV category based on the average cover density of the 25-foot buffer surrounding the structure. This option is appropriate if there is a distinct difference between SAV cover under an overwater structure and the area around it and it is likely that the SAV will reestablish after the structure removal.
8. Credit for the removal of unpermitted structures in the nearshore will be approved on a case-by-case basis.

| Delineation of Lower Shore Zone SAV Scenarios | | |
|---|---|---|
| ***VEGETATION SCENARIO*** | ***Native Eelgrass and/or Kelp occurs within 25 feet of project area*** | ***Other SAV occurs within 25 feet of project area (no native eelgrass or kelp present)*** |
| Scenario 0 | N/A | ≤ 10% |
| Scenario 1 | 1-25% Combined SAV | 11-25% |
| Scenario 2 | 26-69% Combined SAV | 26-75% |
| Scenario 3 | ≥ 70% Combined SAV | > 75% |
| **Delineation of Upper Shore Zone SAV Scenarios** | | |
| ***VEGETATION SCENARIO*** | ***Macro algae and saltmarsh vegetation (such as Salicornia sp. and Distichlis sp.)*** | |
| Scenario 0 | Less than 5% of cover | |
| Scenario 1 | Between >5% and < 30% of cover | |
| Scenario 2 | Between >30% and <60% of cover | |
| Scenario 3 | Between >30% and <60% of cover | |

*Table 1. Delineation of Lower Shore Zone and Upper Shore Zone SAV scenarios (categories). SAV is defined as rooted vascular plants and attached macroalgae. Drift algae and Ulva spp. are not included when determining cover percentage unless Ulva spp. occurs in documented herring spawning areas.*

## *Credit/Debit Factors*

For habitat conditions that are especially important for Puget Sound Chinook and Hood Canal summer-run chum, the final credits or debits are multiplied by a factor. The Conservation Calculator only applies these credit/debit factors to aspects of the project that would affect the important habitat condition. Table 2 shows how the credit/debit factors apply to certain project elements.

1. Major Estuary Zones: A map of Puget Sound Natal & Pocket Estuaries is available on the PS Nearshore web page. We are using the historical extent of PS Chinook salmon natal river deltas plus a 5-mile buffer (as the fish swims), as per the PS Chinook Salmon Recovery Plan nearshore chapter (Redman et al. June 2005). For Hood Canal summer-run chum, we are using a 1-mile buffer around natal rivers and rivers where re-introduction was successful based on the first priority level for recovery actions of the Hood Canal summer-run chum recovery plan (Brewer et al. 2005).
2. Pocket Estuary or Embayment: See the Puget Sound Natal & Pocket Estuaries map
3. Feeder Bluff: We currently use the WDOE Coastal Atlas map with coastal landforms data layer to determine the location of feeder bluffs.
4. Forage Fish Spawning: We rely on WDFW's Forage Fish Spawning map and surveys to determine presence and extent of Pacific herring, Pacific sand lance and surf smelt. If questions arise for a specific location, USACE, USFWS, or NOAA staff will clarify presence in consultation with WDFW.

5. Shoreline armoring that is located within the same drift cell and updrift of forage fish spawning habitat. Use the WDOE Coastal Atlas map to determine drift direction.

While the GIS layer for “Major Estuary Zones” and “Pocket Estuary or Embayment” is depicted as a band (this is an artifact of how the GIS layer was created), these landscape-scale credit/debit factors apply to all zones and the entire structure. In other words, if any part of a structure overlaps or is waterward of location that is mapped as either “Major Estuary Zones” and/or “Pocket Estuary or Embayment,” this credit/debit factor applies to all parts of that structure not just the parts that are located on the band shown on the GIS layer; also see Table 2.

| Nearshore Impact | Major Estuary Zone | Pocket Beach | Feeder Bluff | Sand lance or surf smelt spawning | Updrift of FF spawning within same drift cell | Herring spawning |
|---|---|---|---|---|---|---|
| Shoreline armoring | XX | XX | XX | XX | XX | In rare cases |
| Piers and ramps | XX | XX | | | | |
| Piles depending on zone | XX | XX | XX | XX | | XX |
| Floats (USZ) | XX | XX | | XX | In rare cases | |
| Floats (LSZ) | XX | XX | | | | XX |
| Floats (DZ) | XX | XX | | | | depends |
| Creosote Piles WQ benefit[1] | X | X | | x | | x |
| Boat ramps & Jetties (USZ) | XX | XX | XX | XX | In rare cases | |
| Boat ramps & Jetties (LSZ) | XX | XX | | | | XX |
| Boat ramps & Jetties (DZ) | XX | XX | | | | |
| Beach Nourishment | XX | XX | | XX | XX | |
| Riparian | XX | XX | XX | XX | | |

*Table 2. Project-specific application of credit/debit factors. Credit/debit factors for water quality benefits related to creosote removal are 40% of full credit/debit factor because we expect creosote piles to be on site only for approximately 40 years of the 100-year assumed benefit period. After that they likely have broken off and are floating through Puget Sound.*

## *Duplication of Tabs*

The *Overwater Structure, MDredging, Beach N,* and *SAV Planting* tabs can be duplicated as many times as necessary in one Conservation Calculator workbook. This can be helpful for entering multiple structures on complex projects.

To duplicate one of these tabs, right click a tab on the bottom and click "Move or Copy." Then select the tab to duplicate, check the box that reads "Create a Copy" on the bottom of the window, then press "Ok."

The Conservation Calculator does not allow for duplication of *ShorelStab* or *BoatR, Jetty* tabs. For Excel experts, the *Overwater Structures, MDredging, Beach N,* and *SAV Planting* tabs work with lookup tables in the background, and the other tabs use the NHVM in the background. If an additional *ShorelStab* or *BoatR, Jetty* tab is needed for a complex project, please use and submit an additional Conservation Calculator workbook.

**Important:** Conservation credit/debit totals from duplicated tabs will not auto-populate in the summary tab, so the user should make a note about any added tabs and their resulting credit/debit outputs in the *ProjectD* tab. During consultation, NOAA project biologists will unlock and modify the *Summary* tab as needed.

### *Hiding Tabs*

Tabs that are not in use can be hidden to make your calculator more user-friendly. Simply right click on the tab at the bottom and select “Hide.” To unhide tabs, right click on any existing tab, click “Unhide” and select the tab to unhide.

Advanced users: For visual ease only, we have hidden the NHVM and HEA calculation tabs. These tabs are the gears that build and populate the user-friendly Conservation Calculator you see. Using the “unhide” method described here will allow you to get into the Conservation Calculator mechanics if you wish to dig deeper.

## Puget Sound Conservation Calculator Tabs

The Conservation Calculator consists of different entry worksheets/tabs for different types of actions. The worksheets are:

1. Summary
2. ProjectD: For recording project specific details
3. RZ: Riparian Zone
4. Overwater structures
5. ShorelStab: Shoreline stabilization
6. InputShorel
7. MDredging: Maintenance Dredging
8. BoatR, Jetty: Boat ramps, Jetties, Rubble
9. BeachN: Beach Nourishment
10. SAV Planting
11. Ref.: References

The following sections describe different components of the Conservation Calculator and provide guidance for entering project information so that the outputs will be accurate and consistent.

### ***Tab 1: Summary***

A run-down of all impacts/benefits entered into the Conservation Calculator. This tab provides the total credits/debits consisting of the sum of all project elements.

### *Tab 2: ProjectD*

The *ProjectD* tab is intended for recording project specific details relevant for filling out the Conservation Calculator. This is also the place to document your work and reference external sources you used to derive input values. For example, if you are using pictures at low tide to support your SAV category selection, add a note referencing the pictures and your conclusions or copy and paste the pictures into the *ProjectD* tab.

### *Tab 3: RZ (Riparian Zone)*

Vegetation changes that occur within 130 feet of Highest Astronomical Tide (HAT) as part of a project are entered into the *RZ* tab. According to Brennan et al. (2009), various nearshore functions are supported by adjacent riparian habitat. They reviewed published literature, recommended buffers, and Forest Ecosystem Management Assessment Team (FEMAT) curves to evaluate each of these functions and propose different riparian buffer widths to maintain a minimum 80% effective function. NOAA considered the information provided in this review and designated the area within 40 meters above HAT as the riparian area for the Conservation Calculator. This width is focused on supporting shade, large woody debris recruitment, litter/organic matter inputs, water quality, and habitat function which we believe are the most impactful for aquatic ESA listed species in the region.

Square footage is entered in a **before and after** scenario in columns E and G. The key to entry is that the total square footage input into column E (before) must equal the total square footage in column G (after). Changes are represented in four categories (in Rows 14 through 17): Trees, Shrubs, Herbaceous Vegetation, and Impervious/Unvegetated. Entry represents the "changes" to the riparian from one habitat category to another.

Riparian categories are represented in the *RZ* tab with highest ecological value on top, descending to the lowest. Trees are on top, down to impervious surface/unvegetated on the bottom. A shift of square footage from impervious (in before column E) to trees (in after column G) would represent the most habitat benefit.

There may be locations in which woody vegetation growth extends below HAT, especially in areas with stabilized shorelines. In those locations, the area where woody vegetation is planted for mitigation may be entered in this tab, including any areas below HAT.

Riparian enhancements can be evaluated with the RZ tab/worksheet regardless of location as long as they are located within the same service area as the impact site.

Submit a planting plan, performance standards, proposed monitoring plans, and site protection if applicable with your consultation initiation package. You can find an example of a mitigation plan at: Components of a Mitigation Plan (4) site protections instrument; information on deed restrictions associated with compensatory mitigation here; and an example of a Mitigation Monitoring Report for riparian plantings can be found here.

*Overstory and Understory*

Ideally, native plantings should provide overstory *and* understory conditions. For overstory and understory arrangements, only the square footage of total area is entered into the Conservation Calculator – in other words, square footage cannot be "double counted" for two categories. Instead, enter the square footage as represented by the highest habitat value. For example, if trees are planted with native herbaceous vegetation below, enter only the square footage associated with the trees in the "After" column. Additional credit for shrubs or herbaceous vegetation under trees is not given.

*Entering Trees and Shrubs*

Enter trees and shrubs into the "After" column of the *RZ* tab as their full/mature crown size (area in square feet as seen from above), rather than the size when planted. The HEA model has time built into these categories and accounts for additional years needed for woody plants to reach their full size.

To find mature tree crown square footage, please use the Washington State University's PNW Plants website.

1) On the PNW plants website, find the "Width" of the tree on the right hand "Plant Characteristics" box
2) Divide the width in half to obtain the radius of the tree crown
3) Use the formula for area of the circle $A = \pi r^2$ where A is the area (the total crown square footage as seen from above), $\pi$ is pi (3.14159), and r is the radius obtained in #2 above.

Note: Only use plants native to the area and appropriate for the weather and salt water conditions.

## Tab 4: Overwater Structures

The Conservation Calculator allows for determining the impact of overwater structures (OWS) including simple piers, ramps, floats, and other structures that shade nearshore habitats. Entering measurements for typical piers, ramps, and floats into the calculator is straightforward, whereas entering measurements for more complex structures, like marinas and industrial structures, may require more explanation which is provided below.

*Simple Float Entry*

Enter the length and width of a simple float in the respective shore zone and grating category (solid or grated). Also see "Entering Length and Width" in the *General Information* section. Unlike piers and ramps, floats have associated buffers. In order to allow the Conservation Calculator to correctly determine the buffer area of the float, the length and width must be considered. Always enter the longer side of a float into the length field, regardless of orientation.

> Example – For a replacement 8 feet by 30 feet 50% grated float with 70% open space in the LSZ, in the Overwater Structures tab, enter 30 in cell 57E, and 8 in cell 58E.

In order to be entered as a grated float, floats *must* have at least 50% functional grating, with a minimum of 60% open space (consistent with WAC 220-660-140).

Covered boat slips are entered into the nearshore Conservation Calculator as solid floats.

*Simple Floats with Length and Width Spanning Two or More Shore Zones*

If a float extends across more than one shore zone, the width entity must be adjusted to avoid double-counting a portion of the buffer. To do this:

1) Enter the float dimensions (L and W) for the portion of the float located in the more landward shore zone. Enter these zone-specific dimensions in the yellow entry cells for length and width.
2) For the adjoining waterward zone(s), enter only the length (in that zone) into the yellow entry field, **leaving the width at 0**. Then, manually enter the area (in square feet) for the applicable nearshore zone in the pink square footage box.

Example – For replacement of a grated float spanning the LSZ and DSZ, manually enter the float DSZ area located in the DSZ in the pink entry cell E63.[11] Then enter the total DSZ length into the yellow entry cell E59, and enter 0 into E60.

Because other overwater structures, such as piers and ramps, do not have buffers, this modification is not needed for those structures extending across shore zones.

*Complex Floats*

Floats can have several "branches" or float components contributing to their overall shape. Enter T-shaped floats, L-shaped floats, comb-shaped floats, and other irregular-shaped floats into the Conservation Calculator as complex floats.

*Complex Floats with One Type of Decking*

Floats with decking that is entirely grated or entirely solid can be entered as a "complex float" following these two steps.

1) Enter the total length and the width at the widest point into the appropriate nearshore zone (LSZ, or DSZ) and grating category (solid or grated). This will allow for calculations of a simplified overall float buffer.
2) Determine the area of the complex float and manually enter the square footage directly into the appropriate pink[12] nearshore zone's cell. Letting the calculator determine the square footage for complex floats results in an overestimate of the total area, as it simply multiplies length by width.

[11] The Conservation Calculator determines a buffer for floats based on length and width. If a float spans two zones, entering length and width for all zones would result in an additional buffer area based on the width at the zone break. The above outlined entry method ensures correct buffer determination in that no buffer area is assigned to the width at the end of a zone.

[12] Most entry cells in the Conservation Calculator are yellow. This is one of the few cases where an area is manually entered into the pink float area cell.

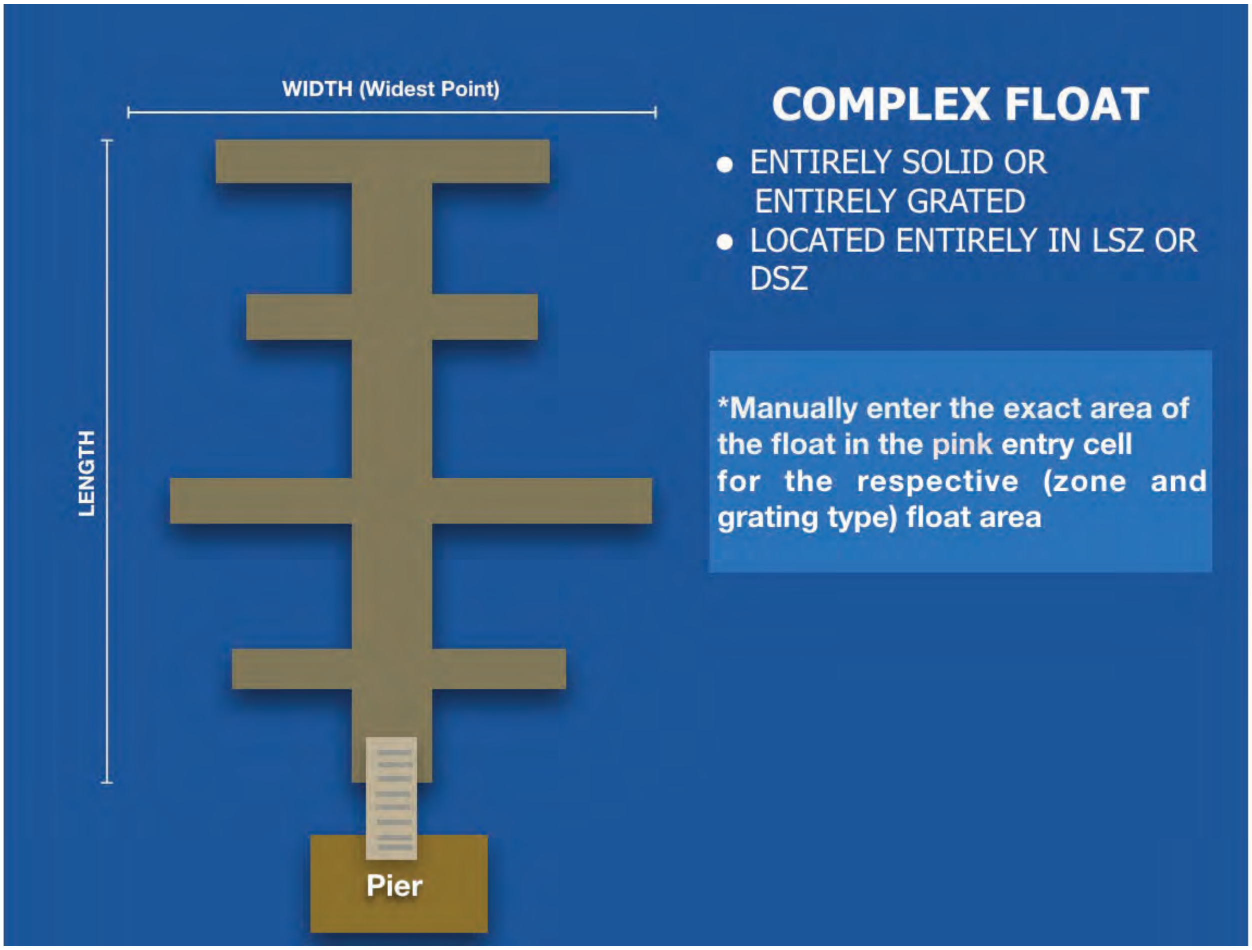



*Figure 6. Complex Float with One Type of Decking located entirely in either the LSZ or DSZ.*

### *Complex Floats with Solid Walkways and Grated Finger Slips*

Some commercial and marina floats have a combination of solid and grated floats. Since there are different float types (solid and grated) within one structure, entry must be split between the solid and grated areas of the Conservation Calculator.

When a complex float structure has a solid center "walkway" and grated fingers, enter it in the Conservation Calculator in the following way:

1) Enter the solid main walkway as a simple solid float (as outlined above under *Simple Float Entry*: Enter the longest dimension in the length entry field and the shortest in the width entry field.)

   Example – For a replacement structure in the LSZ, enter length into cell E66 and width into E67.

2) Under grated float:
   a. Enter the widest width of the entire complex float minus the center walkway as the width (the length of the longest finger floats on both sides of the center

walkway, not including the center walkway. In Figure 8: W1+W2). Leave the length at 0.

b. Manually enter the total square footage of the grated finger floats directly into the pink square foot field.

| | | | | | | |
|---|---|---|---|---|---|---|
| 54 | | | Enter average diameter of piles in DZ. | [inches] | 0 | 0.00 |
| 55 | Grated Float to be Installed | Enter the outside dimensions of replacement floats with at least 50% grating and 60% or more open space as grated floats (Compliant with WAC 220-660-140). For simplicity and as we expect floats to meet state regulations, grated floats are not split between grated and ungrated portions. For complex floats, enter the longest outside dimensions of the float. See Example Complex Float 1. | USZ Outside dimensions of replacement float. | Length [feet] | 0 | Enter length and width of floats for buffer determination. For complex floats, enter the sum of the length of each float and the widest width of the floats. See User Guide for more isntructions. **Set length and width to 0 for zones where no structure present.** |
| 56 | | | | Width [feet] | 0 | |
| 57 | | | LSZ Outside dimensions of replacement float. | Length [feet] | 0 | |
| 58 | | | | Width [feet] | 60 | |
| 59 | | | DZ Outside dimensions of replacement float. | Length [feet] | 0 | |
| 60 | | | | Width [feet] | 0 | Reference: Complex Floats |
| 61 | | The area of the float in each respective shore zone is calculated from length and width entered above. For irregularly shaped floats, user should directly enter the square footage of the float in the appropriate zone (see Notes for more information on irregularly shaped floats). BMP: Floats should not be located in the USZ and cannot ground out. | Grated Float USZ | SqFt | 0 | 0.00 |
| 62 | | | Grated Float LSZ | SqFt | 960 | -46.78 |
| 63 | | | Grated Float DZ | SqFt | 0 | 0.00 |
| 64 | Solid Float to be Installed | Solid float have higher adverse effects on the nearshore environment compared to grated floats. We highly encourage applicants to grate overwater structures as much as possible. Because of the higher impacts from solid floats compared to grated floats, resulting conservation debits are higher. Enter the length and width of the float in the appropriate shore zone (see Table 2). For complex floats, enter longest outside dimensions of float. See Example Complex Float 1 | USZ Outside dimensions of replacement float. | Length [feet] | 0 | Enter length and width of floats for buffer determination. For complex floats, enter the sum of the length of each float and the widest width of the floats. See User Guide for more isntructions. **Set length and width to 0 for zones where no structure present.** |
| 65 | | | | Width [feet] | 0 | |
| 66 | | | LSZ Outside dimensions of replacement float. | Length [feet] | 80 | |
| 67 | | | | Width [feet] | 10 | |
| 68 | | | DZ Outside dimensions of replacement float. | Length [feet] | 0 | |
| 69 | | | | Width [feet] | 0 | Reference: Complex Floats |
| 70 | | The area of a float is calculated by shore zone from the length and width entered above. For irregularly shaped floats, enter the square footage of the float in the appropriate zone (see Notes for more information on irregularly shaped floats). BMP: Floats should not be located in the USZ and cannot ground out. | Solid Float USZ | SqFt | 0 | 0.00 |
| 71 | | | Solid Float LSZ | SqFt | 800 | -72.30 |
| 72 | | | Solid Float DZ | SqFt | 0 | 0.00 |

*Figure 7. Example Entries for a Replacement Complex Float with two types of decking (solid walkway and grated finger slips) in the LSZ.*

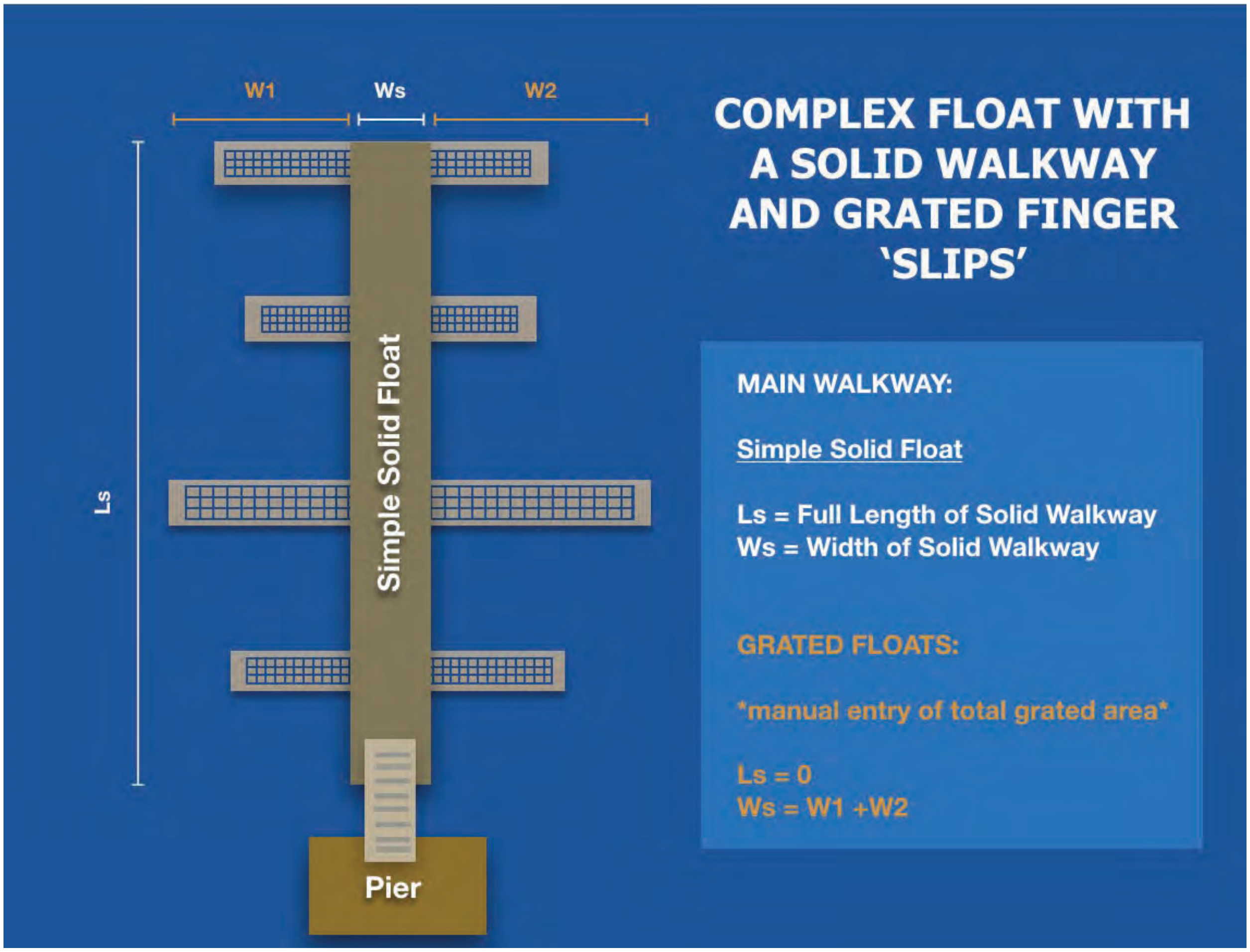



*Figure 8. Complex Float with Two Types of Decking (grated finger slips and a solid walkway) located entirely in either the LSZ or DSZ.*

### *Complex Floats Spanning Two or Three Shore Zones*

When complex floats extend across several nearshore zones (Figure 9), the float area as well as length and width entries for buffer calculations must be zone specific. To enter complex floats in more than one shore zone:

1) Enter the length of the complex float portion that exists in the most landward shore zone in the yellow entry field for its corresponding shore zone. Length, in this case, represents a portion of the longest dimension as it spans all shore zones.
2) Enter the maximum width of all the floats together (finger floats and walkways) in the yellow entry field for width in the most landward shore zone.
3) Manually enter the area of the float portion located in the most landward shore zone in the pink field for square footage.
4) For the adjoining waterward zone(s), enter the total zone-specific length into the yellow entry field for float length. Manually enter a width of 0.

5) Manually enter the area of the float located in each waterward shore zone into the respective pink field for square footage.

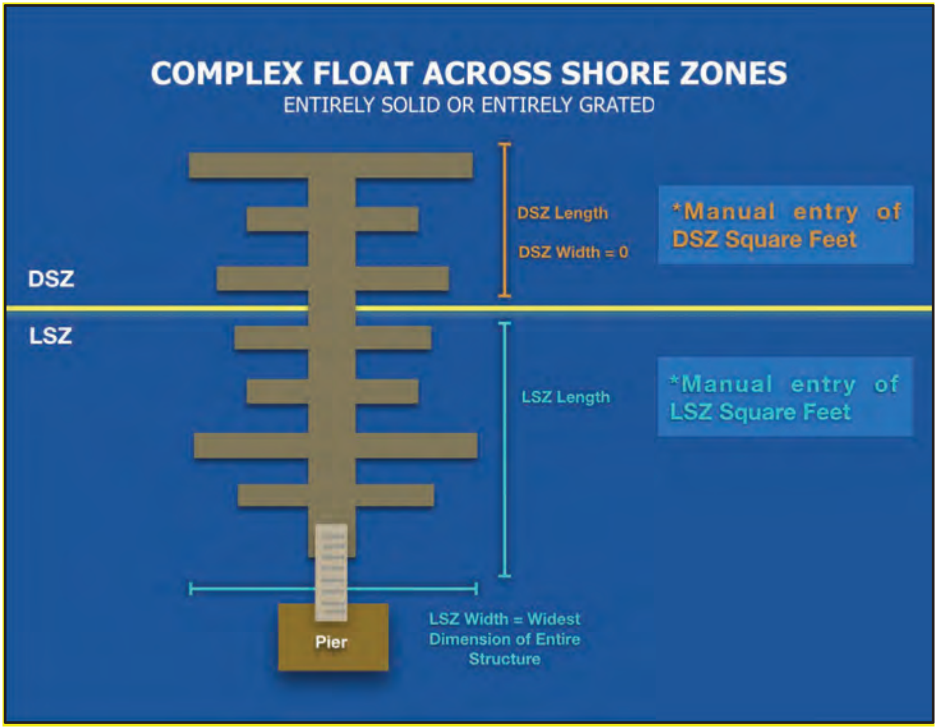



*Figure 9. Complex Float with One Type of Decking (either solid or grated) extending across Multiple Shore Zone.*

Conservation Calculator entry instructions for complex floats spanning two or three shore zones with both grated and solid decking combine the approaches outlined above under *Complex Floats with Solid Walkways and Grated Finger Slips* and *Complex Floats Spanning Two or Three Shore Zones*:

1) Enter the length of the complex float portion in the most landward shore zone in the yellow entry field for length of a *solid* float in its corresponding shore zone. Length, in this case, represents a portion of the longest dimension as it spans all shore zones, LSZ L1 in Figure 9.
2) Enter the longest width of *all* the floats together (e.g., finger floats and walkways) in the yellow entry field for width of a *grated* float in the most landward shore zone, LSZ W1 in Figure 8.
3) Enter 0 for the remaining landward shore zone dimension fields (solid float width and grated float length).
4) Manually enter the square footage of the solid portion of the float in the landward shore zone in the pink field for solid square footage. And manually enter the square footage of

the grated potion of the float in the landward shore zone in the pink field for grated square footage.

5) For the adjoining waterward zone(s), enter the total zone-specific length into the yellow entry field for the length of a **solid float,** DSZ L2 in Figure 9. Manually enter a width of 0. Manually enter the square footage for both the grated square footage and solid square footage for the total waterward shore zone of the float.

*Replacement vs. New Overwater Structures*[13]

If the area of a replacement overwater structure is larger than the area of the removed structure (Figure 10), the difference is entered in the Conservation Calculator as new or expanded overwater coverage. The area entry for an expanded structure will be split between Entry Blocks I: New/Expanded area and Entry Blocks II: Replacement area and must be entered in the respective nearshore zone.

The Conservation Calculator determines impacts/benefits based on the affected area *in each shore zone*. Thus, the determination of what is new or expanded coverage is zone specific. Exception for legacy structures[14]: Replacing floats in the USZ with same size floats in the LSZ can be entered as a replacement.

Finally, to enter a structure element as a replacement, it must be a "like structure." Like structures are those that would be entered into the same structure category in the Conservation Calculator. For example, piers and ramps are like structures. Grated floats, solid floats and pontoons are also like structures.

Typically, the same square footage in the same zone can be minimally realigned. However, large shifts in location that cause increased habitat impacts should be entered as new structures (e.g., if a replacement pier is shifted 50 feet from its original location, and/or to an area with more SAV).

[13] The "New" category also applies to the ShorelStab and BoatR, Jetty tabs (Tabs 5 and 7 respectively). In these tabs it is a drop down "yes" or "no" selection, rather than a separate entry block.

[14] New and replacement floats are usually not placed in the USZ where the water depth is insufficient to prevent the structure from grounding out on the substrate during normal low tides.

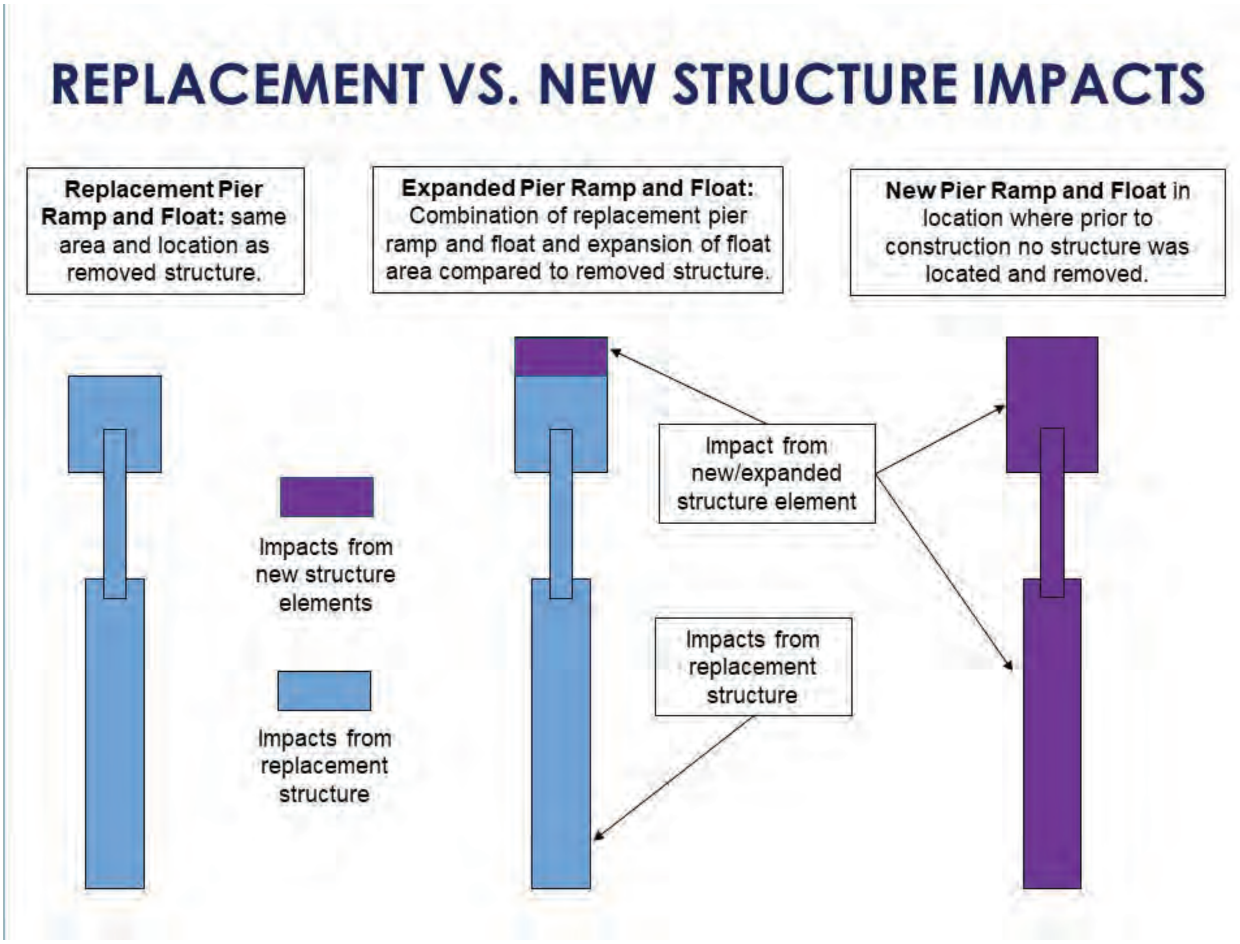



*Figure 10. Replacement versus New/Expanded Structure Impacts*

Entering Replacements with Expansions in the Calculator:

1. Confirm the total square footage of each like structure category to be removed within each shore zone. This is your maximum replacement square footage. Enter this in Entry Block III.
2. Enter the total square footage for the replacement structures in Entry Block II. This area must not exceed the values for zone specific areas entered above in Entry Block III.
3. Excess "replacement" structure square footage exceeding the removal square footage within a shore zone are considered expansions and must be placed in Entry Block I: New/Expanded. Enter zone specific expansions for each like structure in Entry Block I. Exception for legacy structures[15]: Replacing floats in the USZ with same size floats in the LSZ can be entered as a replacement.

[15] New and replacement floats are usually not placed in the USZ where water depth is insufficient to prevent the structure from grounding out on substrate during normal low flow or low tide conditions.

We refer to area and square footage in the above section to focus on the concept of what is considered new/expanded area. To enter the area in the Conservation Calculator, this requires in most cases determining relevant length and width. As discussed with floats spanning different shore zones, the entry of float dimensions for expanded floats also has to consider the buffer area and is explained below.

Example – When removing a 5x10 foot solid float from the LSZ and installing a 5x5 foot grated float with a 3x3 foot solid mooring buoy in the LSZ:

- The total removal square footage in LSZ = 50 square feet. Enter 50 square feet solid float into the Removal Entry Block, along with the dimensions of the float.
- The total replacement square footage in LSZ = 25 + 9 = 34 square feet, which is less than the original 50 square feet. Therefore, both of these structures are entered in the Replace Entry Block. Even though the new float is grated (not solid) and the mooring buoy is "new," the square footage from Remove is applied to the Replace section because the float and mooring buoy are "like structures." The Replace structures are still entered as a 5x5 grated float and a 3x3 solid float in different entry boxes.

### *Length and Width Entry for Expanded Floats*

This section covers directions for entering the length and width for replacement floats with expansions accounting for the buffer around floats. Enter the replaced square footage of the float with the actual Length and Width in Entry Block II. Enter the New/Expansion portion in Entry Block I, manually enter the square footage in the pink entry filed, enter the expanded length, and leave the width entry field at 0. This allows for the buffer area to match the new dimensions.

Example – For a 30x8 foot grated float being replaced with a 40x8 foot grated float in the LSZ (50% grated, > 60% open space) (Figure 11):

- In the Entry Block for replacement structures, enter 30 and 8 as the length and width respectively.
- In the Entry Block for new structures, enter 10 for the length of the "new/expanded" float, leave the width as 0, and enter 80 square feet as the area of the new/expanded float.

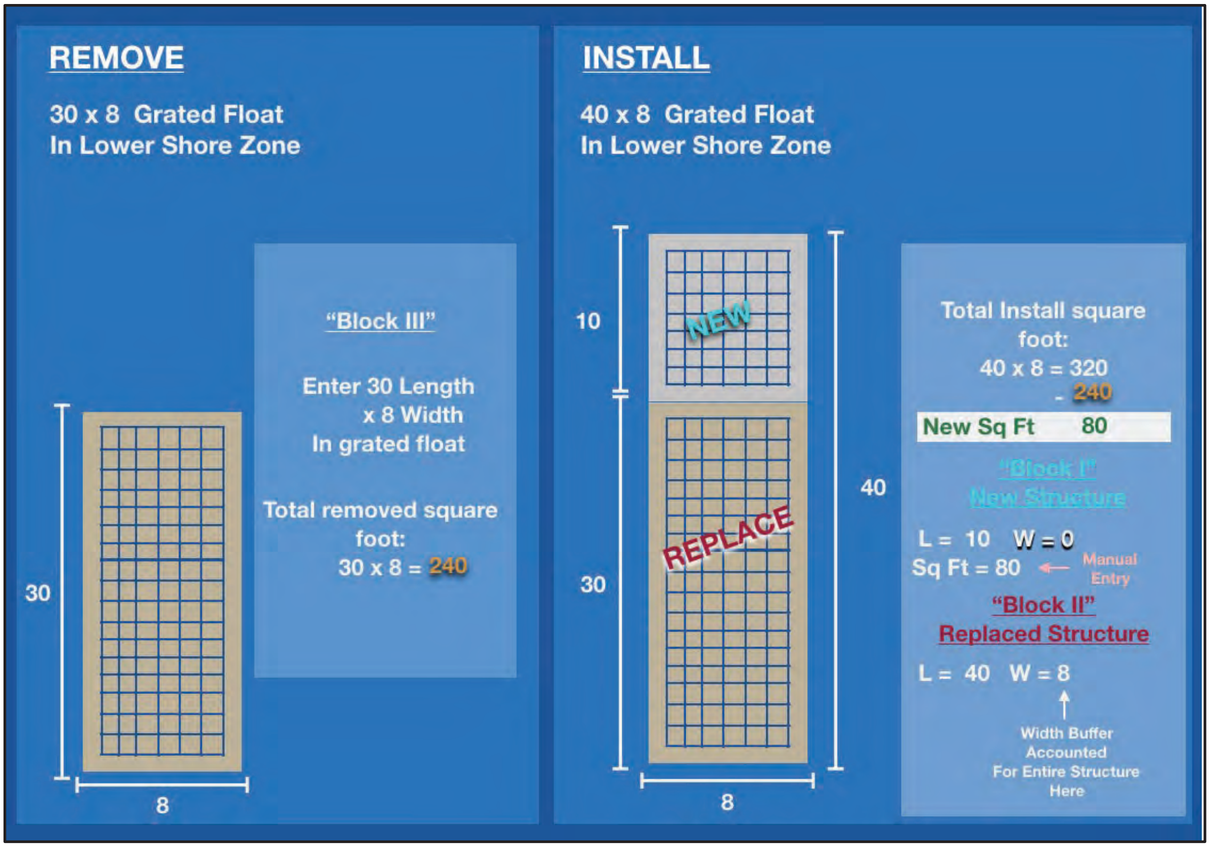



*Figure 11. Example Visualization of Replacement Float: Above example where a 30x8 foot grated float is being replaced with a 40x8 foot grated float in the LSZ.*

### *Mooring Buoys*

In general, when a mooring buoy reduces or prevents ongoing adverse impacts, mooring buoys do not need to be entered into the Conservation Calculator. This applies in situations where vessels are currently moored in areas where they have adverse impacts and would without the placement of a new mooring buoy likely continue to be moored and have negative impacts. This includes situations where mooring buoys would re-direct vessel moorage away from areas where vessels ground out, or where vessels impact dense SAV (SAV score 2 or more), or areas with any kelp, or any eelgrass. In such cases, the applicant should provide information and evidence of ongoing adverse effects and their reduction based on the placement of the mooring buoys.

Otherwise, mooring buoys act similar to and should be entered into the Conservation Calculator as simple solid floats. Enter the length and width into the yellow entry fields.

The situations where mooring buoys should not be entered into the Conservation Calculator are limited to scenarios where the benefits from indirect effects of the mooring buoys[16] that are

[16] Benefits from placement of mooring buoys include redirecting shading associated with vessels away from areas with SAV to areas with less or no SAV and redirecting vessels from areas where they ground out and create sediment disturbance.

otherwise not considered in the Conservation Calculator outweigh the adverse effects from the placement of the mooring buoy. While adverse effects from boats on critical habitat need to be addressed in ESA Section 7 consultations, the Conservation Calculator currently does not assign debits from boats. If, however, unregulated adverse effects from boats exist, are ongoing, and would be reduced by the placement of mooring buoys, the mooring buoys do not have to be entered in the Conservation Calculator as impacting structures.

### *Large Solid Decks/Piers*

Generally, elevated decks and piers have a smaller impact than floats because side lighting reduces the amount of shading. However, the wider a deck is, the less effective the side lighting compared to a long and narrow deck (e.g., a pier). In wide decks, much of the center of the deck is not affected by side lighting because light does not reach under the center of a wide deck (Figure 12).

To account for the dark center on wide decks, enter the deck area within 20 feet from the edge as a pier, and enter the remaining center deck area more than 20 feet from the edge as a float; enter the float area directly into the pink entry cell for solid floats.

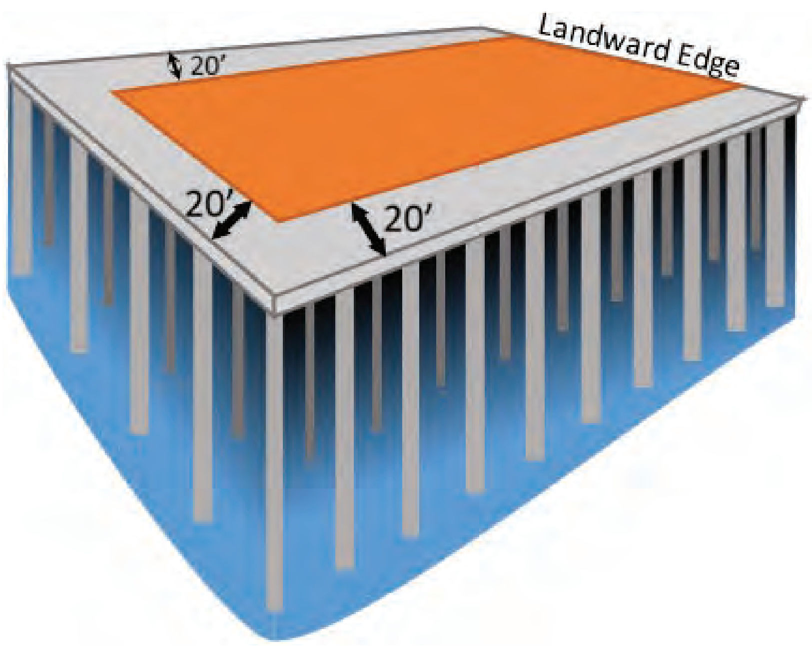



*Figure 12. Conservation Calculator Entry for Large Solid Decks. Use two different entry zones: The orange center represents the area of a solid deck that is entered as a float due to the lack of light penetration from the sides. The 20 feet gray area is entered as an elevated solid deck. The blue gradient shows how the lighting dims towards the center underneath a large deck.*

### *Houseboats and other 3-dimensional Overwater Structures*

Three-dimensional structures, including net sheds and houseboats, create a larger shadow than flat decks. To account for the larger shadow, add half of the square footage of the largest shade producing vertical wall to the area of solid overwater coverage derived from the horizontal coverage (Figure 13).

1) Enter the length and width (Y and X in Figure 13 below) into the yellow entry fields for solid floats.
2) Manually enter the total shade producing area into the pink area entry field for the applicable nearshore zone. The total shade producing area is = X*Y + ½(A*B).

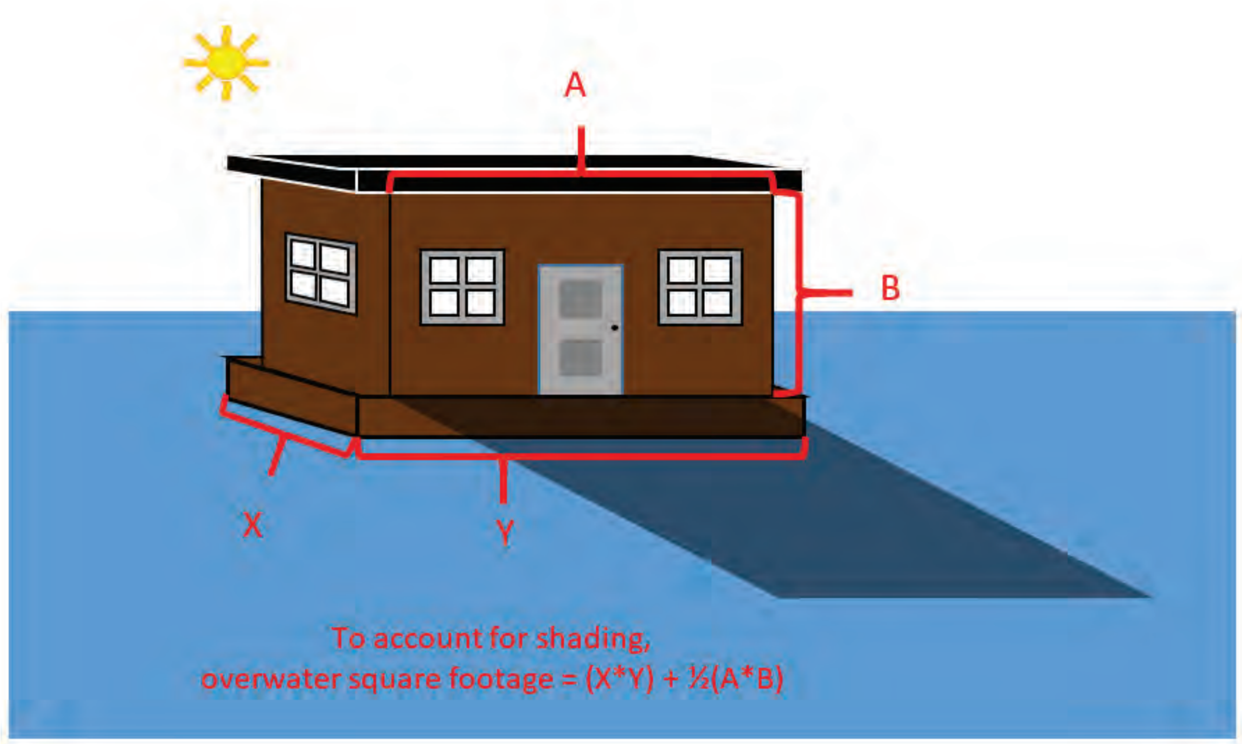



*Figure 13. Houseboats: Three-dimensional Overwater Structure. For Conservation Calculator entry, include height for determining the total shade producing area.*

### *Boat Lifts*

Boat lifts are generally entered as solid or grated piers. If the boatlift is covered, the covered area between the pontoons should be entered as a solid pier. Uncovered boat lifts are entered as grated piers. Dimensions of boats (even if stored in the lift) are not entered into the Conservation Calculator. Piles associated with boat lifts are entered as piles.

Pontoons integrated within lifts that are permanently in contact with water should be entered as a complex float (see complex float entry above). Enter the longest length and width of both pontoons as the dimensions, then manually enter the pontoon area (Figure 14).

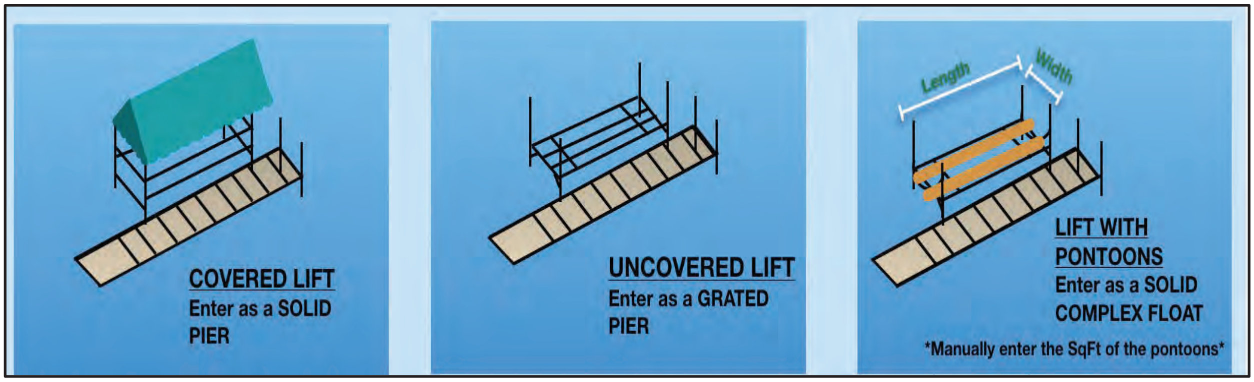



*Figure 14. Different Types of Boat Lifts.*

*Repair and Replacement of Overwater Structures*

**Overlapping Structural Elements:** Overwater structures contain overlapping structural elements like float tubes and decking. As debits/credits are based on area impacts, only the element with the largest area should be entered. Use the examples below to inform entries for similar situations.

1. Repairs to the float structural components, such as the frame and stringers: Enter 100% of the float square footage into the calculator (solid or grated surface as applicable) to determine impacts. If float tubes are replaced at same time, no extra entry is required for float tubes (no double offsets).
2. If decking on a float is proposed to be replaced, enter the area of the decking unless:
   a. Within the last 20 years (first half the design life of a float) the frame and stringers, or decking have been replaced and conservation debits were provided for the entire float or proposed to be replaced element.
   b. Solid well-functioning decking is being replaced with grated decking (see below).

To quantify impacts from repairs and partial replacements with the Conservation Calculator:

1. Enter the footprint as determined using the principles above into Overwater Entry Block for Removal, Removal as Part of Replacements, and Repair.
2. Enter the footprint of the replaced/repaired structure element (proposed), in the Overwater Entry Block for Repair and Replacement. If the footprint of the replaced/repaired structure exceeds the footprint of the existing structure, you need to enter the expanded footprint as new/expanded area, see Figure 10.

*Replacement of Well-Functioning Solid Decking with Fully Grated Decking for the Purpose of Reducing Shading*

If well-functioning solid decking on overwater structures (floats, ramps, and piers) is proposed to be replaced with fully grated decking (defined as a minimum of 60% open space, in compliance with WAC 220-660-140), the decking replacement does not have to be entered into the calculator[17] if **all** of the following conditions are met:

1. The solid decking being replaced is in well-functioning condition with a remaining functional life of more than 10 years (dated photos of existing decking must be provided). If any solid surface on the overwater structure is proposed to be replaced with new solid surface decking, then this condition would not be met. In that case, both grated and solid replacement decking must be entered into the Conservation Calculator because the replacement of the solid surface with solid surface suggests the decking was approaching the end of its design life and needed to be replaced. For example, if solid decking is proposed to be replaced with decking that has a "grated, solid, grated" pattern and all decking is replaced, then all decking replacement and removal would be entered in the Conservation Calculator.
2. Decking replacement aims at reducing adverse effects from shading rather than extending the life of the structure.[18] For example, state and local agencies or tribal entities often ask applicants to replace well-functioning solid decking with grated decking to reduce impacts. Such replacements would not have to be entered into the Conservation Calculator if conditions 1 and 3 in this description are also met.
3. No other structural replacements on the subject structure beyond decking are proposed. If other replacements or upgrades (like the replacement of piles, frame, stringers or float tubes) are proposed at time of the decking replacement, or within the 10 years following the decking replacement, all elements must be entered into the Conservation Calculator. The rationale is that the replacement of other elements at the same time or within 10 years suggests that at the time of decking replacement condition (#1 above) was not met and the structure, including the solid decking, had no more than 10 years of remaining functional life left.

   If an applicant proposes to replace additional components of an overwater structure within ten years of replacing the solid decking, the evaluation of the later-proposed project would likely need to consider the long-term impacts of the previously replaced decking. In other words, the completed solid decking replacement will have to be entered into the Conservation Calculator along with the proposed project at the time of the later replacement.

   <u>Example</u> – If the upgrade of an old float includes the replacement of solid decking with grated decking along with replacement of a float tube, then <u>all elements</u> are entered in the Conservation Calculator.

[17] If the decking replacement is not entered into the Conservation Calculator, there will be no removal credit for the removal of the solid decking. This is based on the fact that the removal credit for solid decking with a remaining life of approximately 30 years is about equal to the placement of the same area of grated decking with a design life of 40 years.

[18] Expected remaining life is more than 10 years.

Example – If a float that had its solid decking replaced under this provision and proposes nine years later to replace a different float element, the replaced decking has to be considered retroactively as the replacement of other float elements suggests that the float including the solid decking had less than 10 years of remaining life left and did not meet the conditions above.

*Piles*

This section outlines specifics regarding entering different types of piles into the *Overwater Structures* tab of the Conservation Calculator.

1) Structural piles excluding batter piles, or fender piles: (a) entering the number of piles to be placed, replaced or repaired and (b) entering the diameter of piles.
2) Multiple pile sizes: If different pile sizes are being installed, enter the average diameter of all the piles. A quick-use calculator provided in the *Overwater Structures* tab at row 129 allows for easy determination of the average pile diameter for each nearshore zone.
3) Batter piles and fender piles: Enter installation of new but not replacement piles. This is a simplified approach to account for the frequent replacement of non-structural piles intended to be hit by vessels.
4) Creosote removal: Residential creosote piles usually weigh ½ a ton or less rather than the 1 ton for industrial-sized, 70-ft-long piles. Use the tonnage estimator provided in the *Overwater Structures* tab at row 154 to determine the weight of creosote treated wood piles for known length and average diameter. Long wood piles often vary in diameter between top and bottom. Use average pile diameter for weight estimation.
5) Monitoring/Reporting of Creosote removal: After creosote removal and upland disposal, applicants must submit the disposal receipts and a picture of the dump truck on the scale to the Services. Disposal receipts need to contain actual weight of the total removed creosote. **Estimated credit calculations may require adjustment if the estimated creosote removal weight is greater or less than the actual disposed quantity.** The Services may use the average difference between estimated and actual creosote removal quantities over a year as an adjustment factor for the following year. In other words, if year one estimates were on average 8% higher than actual disposal quantities, then all estimated creosote removal quantities may be automatically discounted by 8% in year two.
6) Pile Repair: Pile repair (including adding sleeves/jackets) extends the life of a pile just like a replacement. Thus, enter the numbers of repaired piles including their increased diameter (example below) along with replaced piles. Removal credit applies to repairing piles.

    Example – Pile jacketing increases the diameter of piles. Enter the average pile diameter for partially jacketed piles and the number of to-be-repaired piles in Entry Block II: Repair and Replace of Overwater Structure Elements. Also enter the number of to-be-repaired piles and existing diameter of the old piles in Entry Block III: Removal. In terms of effects to habitat, repairs and replacements are similar and thus treated the same in the Conservation Calculator.

If creosote piles are repaired, enter only the weight of creosote treated wood that is proposed to be removed in the entry cell for ”tons of creosote to be removed.” If strut repair is proposed, usually the bottom section of the creosote pile remains in place.

#### *Crediting/Debiting Factors for OWS*

As described in the *General Information Applicable to Most Tabs: Credit/Debit Factors* section below, effects to habitat features that are especially important to Puget Sound Chinook and Hood Canal summer-run chum are multiplied by a factor. This gives more weight to the impact/credit of a proposed action on these especially important habitats. New in Conservation Calculator V 1.4, crediting/debiting factors can be entered in the ProjectD tab. They are applicable to the entire project. If a project consists of different locations that required application of different credit/debit factors, please fill out one Conservation Calculator per project location. We found that in Conservation Calculator V 1.3, applicants rarely used the separate entry blocks for credit/debit factors that allows for the installation of a new structure and the removal of an existing structure to be at different locations.

Floats in the DSZ in herring spawning and holding areas may have a herring factor applied depending on site conditions. The application of the herring spawning & holding factor to OWS in the DSZ is based on the consulting biologist’s and WDFW’s assessment of impacts related to the proximity of structure to holding and spawning areas, the size, type, and configuration of the proposed structure, and frequency and duration of use of the affected area.

### Tab 5: ShorelStab (Shoreline Stabilization)

#### *Hard Armoring*

Shoreline armoring results in reducing the available nearshore habitat landward of hard armoring. Hard armoring cuts off access to the shallow nearshore area that is preferred early marine rearing habitat for juvenile PS Chinook salmon. This is called **intertidal encroachment** and is depicted in Figure 15. Intertidal encroachment encompasses the area between the toe of armoring and the HAT. Critical habitat for PS Chinook salmon is listed under the ESA up to the HAT (50 CFR 226.212). Hard shoreline armoring can also reduce the habitat quality waterward of the hard armoring via adverse effects. Such adverse effects include reducing wrack and large wood accumulation (and thus food availability for juvenile salmonids, also known as habitat provision), changing the wave regime (wave reflection), coarsening substrate, and lowering the beach profile (Figure 16) (Dethier et al. 2016a; Dethier et al. 2016b; Heerhartz et al. 2014; Heerhartz et al. 2016; Prosser et al. 2018). The Conservation Calculator evaluates these impacts to intertidal critical habitat for ESA-listed PS Chinook and Hood Canal summer-run chum salmon via an area based functional assessment. It evaluates the respective functional loss for the area of the intertidal encroachment and for a standard area waterward of armoring. Most functional loss occurs via intertidal encroachment.

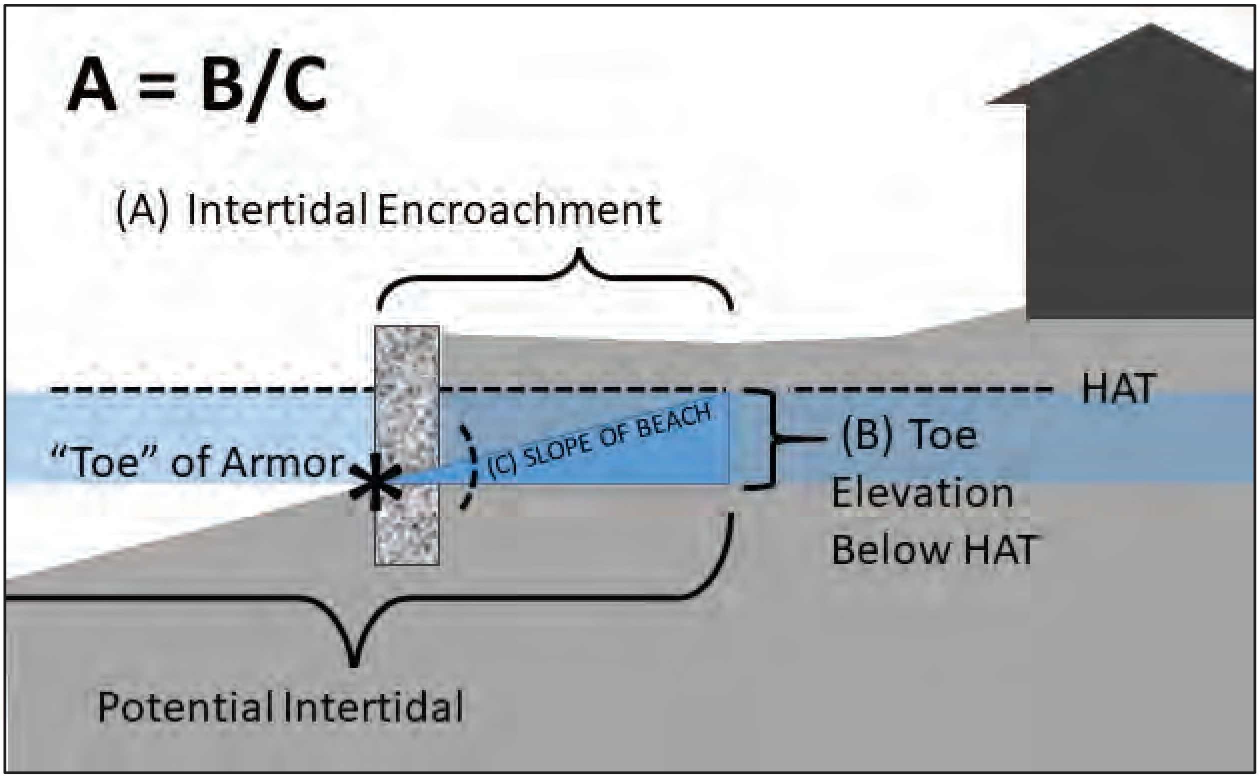



*Figure 15. Intertidal encroachment. Figure designed by Paul Cereghino.*

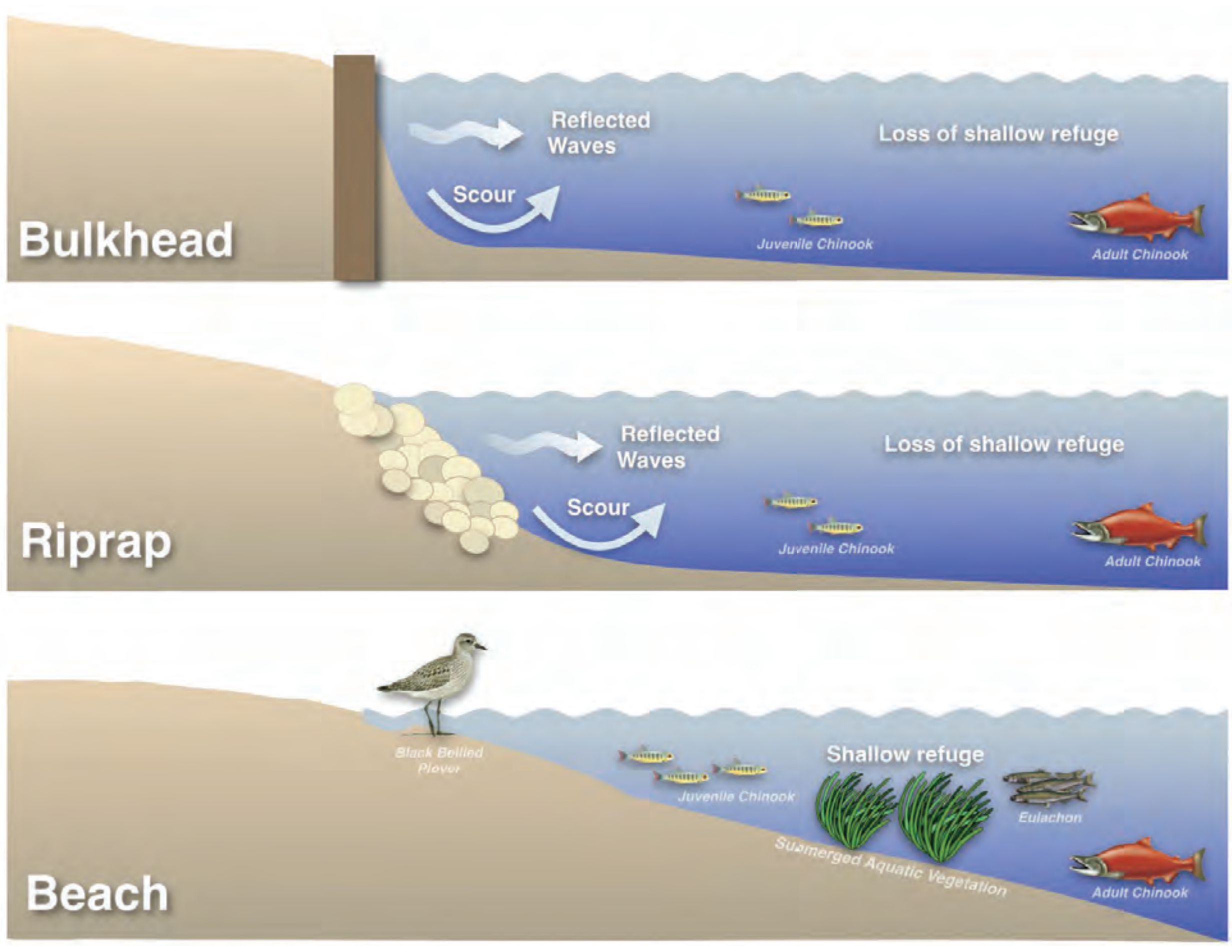



*Figure 16. Effects of hard shoreline armoring adapted from Prosser et al. 2018.*

The Conservation Calculator determines the area of intertidal encroachment considering the following three factors:

1. The length of the armoring
2. The location (elevation) of the toe of armoring relative to Mean Higher High Water (MHHW), and
3. The distance between MHHW and HAT

For the first factor, the length of armoring paralleling the shoreline should be taken from design plans. At times, armoring may wrap into the upland or encircle features jutting out into the intertidal. For such situations, the length relevant for the Conservation Calculator is the length parallel to the shoreline, only. Also see Figure 17.


*Figure 17: Determination of Length of Shoreline Stabilization.*

The second factor, the elevation of the toe of armoring[19] relative to MHHW, also is taken from design drawings. If no survey information is available to determine the elevation of the toe of hard armoring, follow the instructions in the *Toe of Armoring Relative to MHHW* section on page 50 of this guide for approximating the toe elevation.

To determine the third piece of information necessary for calculating the affected area, the distance between MHHW and HAT, NOAA recently developed an approach. We document this approach in more detail in Cereghino et al. (2022) (NOAA White Paper *in draft*). In short, NOAA developed tidal contour lines for the entire Puget Sound region outlining MHHW and HAT. We used NOAA tidal datum model outputs and a USGS high-resolution topobathymetric digital elevation model. These tidal contour lines provide site-specific elevations. Tidal contour lines are currently available on NOAAs GIS server at https://noaa.maps.arcgis.com/home/item.html?id=69c1c16ba7c8473d890e9eaed9fc6d4f#visualize.

We used the horizontal distance between MHHW and HAT based on typical beach slopes rather than measuring site specific distances in the GIS layer. Reasons include that determining site-

[19] The toe of a bulkhead, for the purpose of Conservation Calculator entry, is where the sand or other beach substrate naturally meets the bulkhead, not at the deepest portion of the bulkhead (extending below the beach grade).

specific horizontal distances between MHHW and HAT is subject to errors related to limited resolution (1 meter for recent USGS CoNED 2020 data), low confidence of the method at beaches with steep slopes, inter-annual beach profile variability, and finally that site-specific distances between MHHW and HAT cannot consistently be determined at hydromodified sites.[20] To reduce errors, NOAA developed typical average beach slope values for unarmored beaches stratified by marine basin and beach type (Table 3). NOAA used the beach types described by MacLennan et al. 2017. In the Conservation Calculator, these typical beach slopes can then be used in combination with the site-specific elevations for MHHW and HAT taken from the GIS contour elevation lines to derive the site-specific distance between MHHW and HAT. NOAA documents the results from this approach in *Tab 5: InputShorel* of the Conservation Calculator.

The formulaic expression of the horizontal distance between MHHW and HAT is:

> *Horizontal Distance (HAT - MHHW) = (site specific: HAT elevation – MHHW elevation)/typical slope (taken from Table 3).*

The horizontal distance between MHHW and HAT is determined by a Service biologist on the *InputShorel* tab. The result will be entered by a Service biologist in cell C32 of the *ShorelStab* tab.

| **Typical Stratified Beach Slopes** | | | | |
|---|---|---|---|---|
| **Basin/Service Area** | **Beach Type** | **Slope (rise over run)** | **Percent Slope *** | **Degrees **** |
| Hood Canal | Accretion | 0.142 | 14.2 | 8.1 |
| Hood Canal | Feeder Bluff | 0.28 | 28 | 15.6 |
| Hood Canal | FB Exceptional | 0.17 | 17 | 9.7 |
| Hood Canal | Transport | 0.287 | 28.7 | 16 |
| North Puget Sound | Accretion | 0.191 | 19.1 | 10.8 |
| North Puget Sound | Feeder Bluff | 0.177 | 17.7 | 10 |
| North Puget Sound | FB Exceptional | 0.176 | 17.6 | 10 |

[20] Hydromodified sites are sites where the beach profile has been altered by structures, for example existing bulkheads.

| Typical Stratified Beach Slopes | | | | |
|---|---|---|---|---|
| North Puget Sound | Transport | 0.799 | 79.9 | 38.6 |
| South Central Puget Sound | Accretion | 0.134 | 13.4 | 7.6 |
| South Central Puget Sound | Feeder Bluff | 0.316 | 31.6 | 17.5 |
| South Central Puget Sound | FB Exceptional | 0.26 | 26 | 14.6 |
| South Central Puget Sound | Transport | 0.295 | 29.5 | 16.4 |
| Strait of Juan de Fuca | Accretion | 0.126 | 12.6 | 7.18 |
| Strait of Juan de Fuca | Feeder Bluff | 0.177 | 17.7 | 10.04 |
| Strait of Juan de Fuca | FB Exceptional | 0.12 | 12 | 6.8 |
| Strait of Juan de Fuca | Transport | 0.24 | 24 | |
| Whidbey | Accretion | 0.143 | 14.3 | |
| Whidbey | Feeder Bluff | 0.243 | 24.3 | |
| Whidbey | FB Exceptional | 0.241 | 24.1 | |
| Whidbey | Transport | 0.262 | 26.2 | |

*Table 3. Typical stratified beach slopes by marine basin, beach type, and their slopes. This table is included in the ProjectD tab of the Conservation Calculator.*

* *What is a 25% slope?* A 25 % slope is simply a ratio of 25:100. In other words, the ground rises 2.5 inches every 10 inches of horizontal distance.

** *How does percent slope relate to degrees?* A 100% slope corresponds to 45 degrees. Convert the slope percentage to a ratio (slope (rise over run)) and look up the ratio in a tangent table

Additional notes:

- See NOAA's Nearshore Conservation Calculator webpage for basin/service areas map.
- For sites at hydromodified locations, use adjacent beach types.
- For sites with "no appreciable drift," err on the side of the species and use the lowest slope value for that basin. Such sites (unless misclassified) often do not need armoring; instead, consider a hybrid approach.

### *Soft and Hybrid Bank Stabilization*

Placement of soft or hybrid bank stabilization currently does not incur debits as it mostly allows aquatic access across the elements of stabilization. Replacing hard armoring with soft or hybrid approaches can result in conservation credits. Soft and hybrid armoring are defined below.

Soft Shoreline Treatments - Soft shore approaches allow for the following functions:

- Connectivity between terrestrial and aquatic habitats
- Natural fine sediment transport or accretion rates (i.e., does not coarsen the substrate)
- Does not inhibit sediment transport from upslope sources
- Retains native vegetation
- Supports forage fish spawning
- Does not increase erosion on the project beach or on adjacent properties
- Does not cause lowering of beach elevation
- Allows for woody debris and wrack to accumulate

Criteria for soft shore approaches:

1. No, or minimal, use of artificial structural elements
2. Incorporate beach nourishment (sand and small gravel)
3. Incorporate riparian plantings or allow for recruitment of native vegetation, including overhanging vegetation
4. Incorporate or allow for large wood recruitment, including allowances for small toe erosion protection where necessary, but where the wood does not act as a berm or a crib.
5. Large wood may be chained as part of the design.
6. Boulders may be incorporated into the design, but must not be used as a primary slope stabilizing element.
7. Degradable fabric and support filters may be used but must be designed and constructed to prevent surface exposure of the material through time.
8. Cannot not resemble a wall in any respect

Hybrid Shoreline Treatments – Hybrid shore approaches allow for the following functions:

- The hybrid method itself does not inhibit sediment transport from upslope sources (e.g., an adjacent road that is not part of the project may inhibit sediment transport that would not reflect on the hybrid technique).
- Retains native vegetation
- Supports forage fish spawning
- Does not increase erosion on the project beach or on adjacent properties
- Minimizes lowering of beach elevation
- Allows for woody debris and wrack to accumulate

Criteria for hybrid approaches:

1. Contains artificial structure that allows for some biological processes to occur (such as forage fish spawning), but inhibits some ecological processes from fully occurring (such

as suppressing some sediment transport, supply or accretion, but not fully ceasing the process as with hardened approaches).

2. Exposed rock, if used, must be discontinuously placed on the beach (i.e., not act as a berm or scour sediments)
3. For any individual project, a hybrid approach may not contain more than 30% of exposed rock as measured against the length of the project beach.
4. Buried rock may be used below grade where necessary to stabilize the toe of the slope, but must not form a wall or resemble rip rap, and must be covered with sand/small gravel mixes in such a way to minimize net erosion through time.
5. Incorporate beach nourishment (sand and small gravel) as needed to minimize lowering of beach grade and net erosion.

*Repair of Shoreline Armoring*

If shoreline armoring is repaired in place, treat it the same as a replacement:

1. Fill in the metrics for replacement armoring in Entry Block I: Armoring to be Installed
2. Click "yes" for replacement
3. Fill in the metrics of the armoring to be repaired in Entry Block II

If a shoreline armoring repair does not remove the old structure but places a replacement structure waterward of the existing armoring or encases the existing structure with material to extend the life of the structure, proceed as explained above. However, reflect the new impact footprint in the slope distance in Entry Block I: Armoring to be Installed.

Repairs involving creosote: When repairing structures that contain creosote, creosote removal credit applies only to removed quantities of creosote.

*COMMON QUESTION: When does removal of an existing bulkhead (BH) generate credit?*

1. As with all structures that are proposed to be removed, removal credit[21] is tied to the structure being in good condition. For a bulkhead, that means the area landward of the structure is cut off from tides and aquatic access, preventing natural processes from occurring and aquatic use of that habitat.
2. Creosote bulkhead remnants that no longer function as a bulkhead anymore should be entered into the Conservation Calculator as creosote removal only. See Figure 17 and Figure 18 for examples of non-functioning bulkheads that would not be considered in good condition.
3. Concrete bulkhead remnants that no longer function as a bulkhead should be entered into the Conservation Calculator as rubble removal only.

[21] For a standard remaining life of 10 years.

*Figure 18. Removal Credits for Old Creosote Bulkhead: Removal credit applied for creosote, not for remnants of bulkhead. Picture by and with permission from Doris Small, WDFW.*

*Figure 19. Removal Credits for Non-Functional Shoreline Armoring that is not in good condition. Removal credits do not apply for horizontal pile stabilizer as there is no functioning bulkhead effect (like sediment retention behind the bulkhead or elimination of water exchange).*

### *Site Conditions Landward of Hard Armoring*

This section assesses the value of the riparian habitat rendered inaccessible to fish via armoring. The inputs in cells C5-C7 are used to determine the area weighted habitat value of the riparian habitat after installation (new or replacement) of armoring. If just one habitat type is present, it is sufficient to enter a 1 into the respective row. If there is a 50% split of the area between two habitat types, enter a 1 into each row for respective habitat types. For more complicated scenarios, enter respective Square Foot for each habitat type.

For armor installation, the conditions described need to match the *after* conditions in the *RZ* tab (column G) if any changes in the RZ are proposed. Evaluate habitat improvement/degradation through actions like tree or shrub plantings separately in the RZ spreadsheet/tab.

For standalone shoreline armor removal projects, describe the before RZ conditions in cells C5 through C6. Armor removal is also entered in the *RZ* tab as a change from before = armored to after = unarmored in Row 21. The reverse is also true.

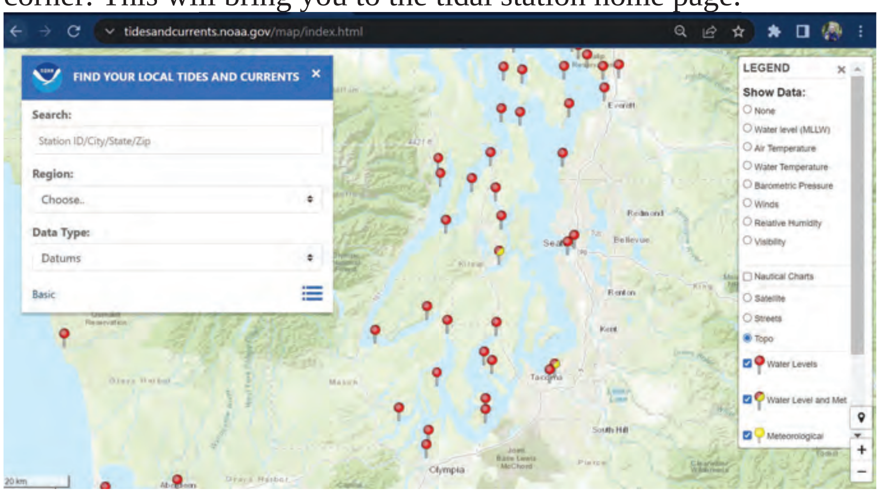
### *Toe of Armoring Relative to MHHW*

This entry is needed for *Tab 5: ShorelStab* cells C15, C16 and C30, C31.

Toe of Bulkheads: The toe of a bulkhead, for the purpose of entry into the Conservation Calculator, is where the sand or other beach substrate naturally meets the bulkhead, at grade. Often, we receive bulkhead replacement project packages where MHHW is not known or shown on a cross section of a bulkhead.

The following steps can be taken for bulkheads where the elevation of MHHW is not known or documented at the site:

1) <u>If a beach survey is available</u>: Use the beach survey to determine whether the toe of the armoring is located above or below MHHW (cell C 15 and C 30). Then determine how much the toe of armoring is located above or below MHHW (cell C 16 and C 31). If the distance between the toe of armoring and MHHW water varies along the armoring, calculate a length weighted average and document your determination in *Tab 2: ProjectD.*
2) <u>If no beach survey is available</u>: We realize that surveys are costly and noticed that many armor replacement projects do not provide information on the toe elevation. This is our currently best draft approach to determining the toe elevation absent a survey. We appreciate your feedback and improvement suggestions.
    a. **Locate the nearest tidal station to the project** in the NOAA Tides and Currents map. On the search bar, click "Advanced," and under "Data Type" select "Datums." On the map, click on the red location marker that is closest (by water) to your project site. The maker symbolizes a NOAA tidal station with tide predictions and datums. An information box for that station will open. In the information box click on the "Station Home" drop down menu in the upper right corner. This will bring you to the tidal station home page.




    b. On the top of the station home page, click on the "Tides/Water Levels" drop down and click on "Datums." This will open a page showing tidal data for this station. **Record the MHHW value** shown towards the top of the elevations list. All data values are relative to the Mean Lower Low Water (MLLW). Note - some tidal stations do not have a "Datums" page. If this is the case, go back to the station map and locate the next closest tidal station.
    c. From there, go back up to the "Tides/Water Levels" drop down menu at the top of the page and click on "NOAA Tide Predictions." This will open a page showing tidal predictions for the station. Using the chart's date options, **locate days when a high tide (either the high tide or higher high tide) is near (within 0.1 foot) of the MHHW value** recorded in step b.
        i. You can click the blue button "plot calendar" on the bottom right to show an entire month of high and low tides.

ii. Hover your mouse over a high tide that is within 0.1 ft of the Datum MHHW value to find out the exact time that high tide will occur. Take a screenshot or your result.
iii. Alternatively, you can find a high tide within 0.1 ft of MHHW using the Data Listing below the graphic.

d. At the bulkhead site, take clear photographs within 10 minutes of the high tide time as determined above (where high tide is within 0.1 ft of the MHHW for the closest NOAA Tide Predictions station). Photos need to show:
    i. The water in relation to the bulkhead as viewed from multiple angles and along the entire existing bulkhead, at multiple photo locations.
    ii. Have a date, time, and GPS stamp. (Free smartphone apps can create this stamp see "Timestamp Camera Enterprise" for iPhone or android)
    iii. Include an object for scale reference (such as a 5-gallon bucket).
    iv. <u>For armoring above MHHW:</u> Lay out a tape measure from the water line landward to the bulkhead toe to determine the distance between the toe of armoring and the water. Take photos of the tape measure documenting this distance. If the distance between the toe and the waterline varies across the length, take several pictures and develop an area weighted average distance. Enter that distance in cell C 31 and/or C 16 depending on whether this is a replacement or new installation.
    v. <u>For armoring below MHHW:</u> Hold a tape measure showing the vertical distance between the toe of armoring and the water level. If you can't find or see the toe of the armoring (this can be challenging with rip-rap) use a marker to mark where the water level was at high tide and take a picture with a date stamp showing the mark at high tide. At low tide, take a second picture identifying the vertical distance between the MHHW line and the toe of the bulkhead. Use the appropriate slope from Table 3 (*Tab 6: InputShorel*) to determine the horizontal distance between the toe of the armoring and MHHW. Enter that distance in cell C 31 and/or C 16 depending on whether this is a replacement or new installation.

e. We would greatly appreciate it if you can take the time and submit distance determinations from two separate days. We are still in the test phase for this method and are trying to evaluate possible variability between different dates.

f. Submit these photos in an email along with the NWS and WCRO project number and which tidal reference station was used to the NOAA project biologist or PSNearshoreConservation.WCR@noaa.gov.

g. A NOAA biologist will review the submitted information and will update the Nearshore Conservation Calculator (if applicable). The biologist may also request additional information.

## Tab 6: InputShorel

This tab supports entries for *Tab 5: ShorelStab*. It is designed to determine the horizontal distances between MHHW and HAT and between MHHW and the toe of armoring in feet.

To determine the **horizontal distance between MHHW and HAT**:

1. Open NOAA's Beach Slope Reference Line GIS layer located at: https://noaa.maps.arcgis.com/home/item.html?id=69c1c16ba7c8473d890e9eaed9fc6d4f#visualize.
2. Locate your project site and click on the reference line to open the information box.
3. Copy the MHHW and HAT elevations in feet from the information box (see Figure 20) into cells B6 and C6 in the *InputShorel* tab.
4. Using the Marine Basin Name and Shoretype_Beach from the information box, go to the the *InputShorel* tab to find the appropriate slope value in column K of the Typical Stratified Beach Slopes table.
5. In the *InputShorel* tab, enter or link the slope value from column K into cell D6. You can either type the slope value directly into cell D6, or link cell D6 to the applicable beach slope cell. For example, for a Hood Canal Accretion beach type you would enter "=K3" into cell D6.
6. Sea level rise was determined for three distinct areas: The Strait of Juan de Fuca, North Puget Sound, and a combination of South Central Puget Sound, Whidbey, and Hood Canal marine basins. No additional entries are needed for inclusion of sea level rise.

The site appropriate horizontal distance between MHHW and HAT in feet is calculated and displayed in E6 on the *InputShorel* tab. It is automatically copied into cells C17 and C32 in the *ShorelStab* tab.

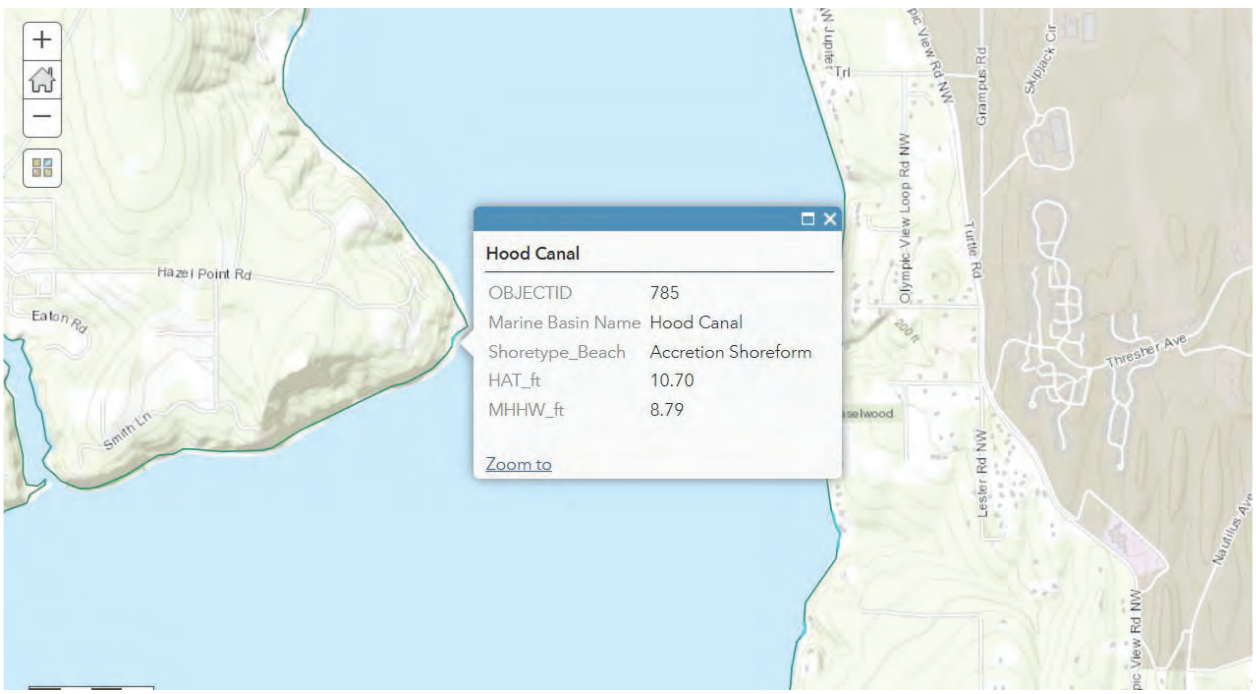



*Figure 20. Beach Slope Reference Line Information Box.*

**Horizontal Distances between MHHW and the toe of armoring** can be determined in row 15 for installation and row 19 for removal of armoring. The only needed entry is the vertical distance between MHHW and the toe of the bulkhead which is to be entered in the yellow entry cells B/C10 and B/C14. Site-specific typical beach slopes are automatically copied over from cell D6. The resulting horizontal distance between MHHW and the toe of armoring is automatically copied over into the *ShoreStab* tab.

### *Sea Level Rise*

Climate Change will cause varying levels of sea level rise in Puget Sound. Sea level rise will cause bulkheads to cut off increasing areas of intertidal habitat from aquatic access. Sea level rise at sites with bulkheads means that the water level will move up on the bulkhead; MHHW and HAT will be higher on the beach while the toe of the BH remains in the same location. Effectively, sea level rise lowers the elevation of the toe of the armoring.

The Conservation Calculator includes the effect of average sea level rise for hard armoring for three distinct areas: the Strait of Juan de Fuca, the North Puget Sound marine basin, and the remaining three basins combined (*Tab 5: InputShorel* rows 25 through 29). We chose this breakout based on geographic distribution of basin average rise projections.

- We used a middle of the road (50% exceedance probability) for the sea level rise prediction scenario.
- We used a low Representative Concentrations Pathways (RCP 4.5) greenhouse gas scenario
- We used the sea level rise scenario for 2050 as that is commonly available. This, again, will provide a rather low estimate as it uses a time horizon below the design life of a bulkhead (50 years).
- Including sea level rise predictions for 2050 as though they would occur now provides a conservative estimate because in HEA habitat now is more valuable than habitat in 40 years.

### *Fill Waterward of an Existing Bulkhead*

Replacement/addition of fill (like rip-rap, rocks, ecology blocks) waterward of an existing bulkhead should be entered into the Conservation Calculator as a jetty with dimensions equal to the birds-eye-view length and width (and/or square footage) of habitat covered by the fill.

However, depending on the site-specific scenario, the NOAA biologist will evaluate whether the amount and type of fill is functioning as a new bulkhead. In that case, the new fill may be entered as a bulkhead.

### *Materials Added to the Toe of an Existing Bulkhead*

If new material, such as logs or concrete, is permanently affixed to the toe of an existing bulkhead (to prevent scour or otherwise protect an existing bulkhead), the footprint of that material is entered into the Conservation Calculator in the *BoatR, Jetty* tab as “concrete footings” (‘No’ in cell E10) in the Boat Ramp Installation block. Dimensions entered in the USZ are the birds-eye-view length and width of the attached materials.

Habitat logs with attached root-wads generally don’t have to be entered in the Conservation Calculator. You should discuss the site-specific function of habitat logs with one of the Service’s project biologists. Depending on the site-specific scenario, the NOAA biologist will evaluate whether the amount and type of material anchored may function like a replacement bulkhead. If this is the case, materials anchored to the existing bulkhead toe may be entered as a replacement bulkhead.

*Staircases on Bulkheads*

Impacts to habitat caused by replaced or new solid-structure[22] staircases are similar to the adjacent bulkhead because the stairs, themselves, also function as a bulkhead. Staircases that are in line with a bulkhead (i.e., not extending waterward) are simply added to the total linear feet of the bulkhead.

If stairs extend waterward of a bulkhead, either parallel or in another configuration, we expect additional adverse effects from the footprint of the stairs and landing. In that case, enter the entire bulkhead in the *ShorelStab* tab (if it is being replaced) **and** the length and width of the protruding stairs as a boat ramp in the *BoatR, Jetty* tab.

Stairs that are inset landward of the bulkhead eliminate slightly less habitat than the adjacent bulkhead. This may be accounted for in different ways. The stairs may be entered separately (in a new *ShorelStab* tab in a different calculator) as a bulkhead with a reduced horizontal beach slope distance. Alternatively, both the stairs and bulkhead can be entered as a single bulkhead, the total linear feet (bulkhead + stairs) is then entered with an averaged horizontal beach slope distance based on a weighted horizontal distance accounting for the stair inset.

> Example – A concrete bulkhead of 50 feet (total) will be replaced. The horizontal beach slope distance from the bulkhead toe to MHHW, as determined in the *InputShorel* tab, is 8 and the horizontal distance from MHHW to HAT, also determined in *InputShorel* tab, is 15. The bulkhead has a 5-foot section with an inset stairway. The stairway is inset such that 3 feet of exposed beach exists from the bottom step out to the bulkhead wall. The bulkhead (sans stairs) may be entered separately as a 45 linear foot bulkhead with the site-specific horizontal distances (8 and 15). The 5-foot stairwell can then be entered in Calculator #2 as a 5 linear foot bulkhead with a reduced horizontal distance between the toe and MHHW (5 and 15). Or, the total wall (including stairs) can be entered as 50 linear feet with a weighted average of horizontal slope distances (between the toe and MHHW) between these pieces. ((5x5) + (45x8))/50 = 7.7

## Tab 7: MDredging (Maintenance Dredging)

- The Conservation Calculator currently does not evaluate new dredging/deepening.
- The zone (LSZ or DSZ) is determined by the depth of the existing habitat, not the proposed dredge depth.
- The SAV scenario is usually 0 (zero) for maintenance dredging as very little to no SAV grows in areas frequently disturbed. While maintenance dredging could extend the

[22] Meaning water is not able to flow freely under the staircase. Solid structure staircases are typically rock or concrete, but may be wooden.

duration for which SAV cannot establish, it is usually too speculative to address what type of SAV might be present in the absence of dredging. However, if SAV establishes between dredging, the respective SAV rating should be entered as before condition. Further, if dredging clearly interrupts an eelgrass bed or SAV, then the SAV condition from the surrounding area should be used.

- Credit/debit factors apply to maintenance dredging.
- The Conservation Calculator considers impacts on SAV, sediment quality and forage, and the shallow water migratory corridor to last a combined average of three years.[23] Thus, for multi-year dredge permits, impacts of dredging should be evaluated for every dredging event. This can be done via either summing up the dredged area over the multiple dredge events and entering that sum into the *MDredging* tab or duplicating the dredging tab and entering each dredging event in its own tab.

## Tab 8: BoatR, Jetty (Boat Ramps and Jetties)

- Enter the SAV scenario as noted in the *General Information* section below and in the *Reference* tab of the calculator.
- Use this tab to calculate credit for removal of concrete, rubble and debris.
- Credit/debit factors do apply to boat ramp, jetty and rubble removal work.

### *Marine Rails*

Marine rails resting on the sediment should be entered in the *BoatR, Jetty* tab. Enter the square footage of the solid metal rails as viewed from above (not the open space in between) as a boat ramp. If the square footage of the rail is unknown, use a default of 1 square foot for every 1 foot of length of the two parallel marine rails (based on measurements of terrestrial rails, see Figure 19). For example, if a marine rail system is 50 feet long and 8 feet wide, enter 50 in the length of the boat ramp to be removed field and 1 (2* 0.5 ft) in the width of the boat ramp to be removed. Enter the area of concrete footings and/or stub piles associated with the rails in the *BoatR, Jetty* tab under concrete footings.

Elevated rails should be entered in the *Overwater Structures* tab. Enter the length and width of elevated rails as a solid pier.

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[23] The effects of removal of sediment and invertebrate prey usually extend over two years (Boese et al. 2009, Dethier and Schoch 2005; Jones and Stokes 1998; McCabe et al 1998).
Maintenance dredging occurs at regular intervals; depending on the location every two to five years (pers. com Daniel Krenz, 2020). After dredging, the dredged area starts to silt back in and the habitat functions of the migratory corridor gradually increase. We chose a conservative impact duration for the reduction in migratory corridor function of four years.The average impact duration of three years used for the HEA analysis is based on these two time horizons.

*Figure 21. Rail Width*

## Tab 9: Beach N (Beach Nourishment)

We usually rely on WDFW expertise in determining whether beach nourishment is appropriate for the project location. We welcome WDFW input on site-specific quantities and the technique of placement.

To ensure beach nourishment is ultimately beneficial for juvenile salmonids and will generate conservation credits, the following considerations need to be met:

- Placement of beach nourishment should follow considerations detailed in WDFW Marine Shoreline Design Guidelines (MSDG), 2014.
- Beach nourishment must demonstrate appropriate grain-size profile for target species and sediment supplementation rate according to estimated sediment erosion rates for sites and drift cell reaches.
- Dumping or disposal of non-native material, dredged material, or upland fill is excluded if it does not meet grain size and supplementation rate conditions.
- When placing material in areas known to have forage fish spawning, placement will adhere to timing windows protective of forage fish.
- Place beach nourishment within 9 linear feet of a bulkhead and at 6 inches depth for each foot of shoreline armoring. This recommendation results in 4.5 cubic feet per linear foot (pers. com WDFW).
- Beach nourishment may be piled up against armoring or spread out depending on agency biologists' site-specific instructions.

- Placement and anchoring of large woody material may be required to lengthen the retention of beach nourishment to meet the benefit period used in the Conservation Calculator.
- Material has to be clean and suitable for nearshore habitat enhancement/restoration.

**Beware**:

- Site-specific recommendations will vary.
- Usually, we do not credit placement or beach nourishment in the "No Appreciable Drift" or "Accretion Shoreform" shore types, as shown in WDOE's Coastal Atlas map.
- If the function of the application of beach nourishment appears to be stabilization of structure placement rather than addressing lack of substrate, the activity may not generate credits.

### Tab 10: SAV Planting

To generate conservation credits for SAV planting, submit a planting plan, performance standards, a monitoring plan, and a site protection instrument where applicable with your consultation initiation package. You can find an example of a mitigation plan at: Components of a Mitigation Plan (4) site protections instrument; information on deed restrictions associated with compensatory mitigation here; and an example of a Mitigation Monitoring Report for riparian plantings can be found here.

### Tab 11: Reference

The *Reference* tab provides background information including:

1) The cover categories for submerged aquatic vegetation and USZ vegetation;
2) The delineation of shore zones for the Riparian Zone, Upper Shore, Lower Shore, and Deep Shore Zones
3) Complex float length and width determination for overwater structure (OWS) tab.

## Acknowledgements

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3

# APPENDIX B
# PROTOCOL FOR MARBLED MURRELET MONITORING DURING PILE DRIVING

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# U.S. Fish and Wildlife Service –
## Washington Ecological Services Office
### Protocol for Marbled Murrelet Monitoring During Pile Driving (Revised 7/29/2022)

## 1.0 Objective

The intent of the monitoring protocol is to:

1. Comply with the requirements of the Endangered Species Act Section 7 consultation.
2. Detect all marbled murrelets (*Brachyramphus marmoratus*) (murrelets) within the monitoring area.
3. To minimize take of murrelets from both exposure to potentially injurious underwater sound pressure levels, and from the masking effects of in-air sound by communicating immediately with the pile driver operator.
4. Track incidental take exempted through the Incidental Take Statement found in the final Biological Opinion for the project so that the Lead Federal Action Agency will know when take occurs and/or when take exemptions might be exceeded.

## 2.0 Adaptive Approach

The individuals that implement this protocol will assess its effectiveness during implementation. They will use their best professional judgment throughout implementation and will seek improvements to these methods when deemed appropriate. Any modifications to this protocol will be coordinated between the Lead Federal Action Agency and the Washington Fish and Wildlife Office.

## 3.0 Monitoring

## 3.1 Activities to be Monitored

Application of this protocol is required as specified through the Endangered Species Act consultation process for individual projects. It may apply to projects that involve either in-water impact pile driving when injurious sound pressure levels are expected and to projects that involve either vibratory or impact pile driving when in-air sounds are expected to cause masking effects.

## 3.2 Equipment

- Binoculars - quality 8 or 10 power
- Spotting scopes (optional)
- Two-way radios with earpieces
- Range finder
- Log books

- Seabird identification guide
- Life vest or other personal flotation device for observers in boats
- Cellular phone to contact Lead Federal Action Agency, the Construction Contractor, or Washington Ecological Services Office.

## 3.3 Monitoring Locations

The spacing and placement of monitoring locations must be designed to provide adequate coverage of the entire monitoring area. Locations are determined ahead of time and are identified on the Seabird Monitoring Site/Transect Identification Form. The monitoring design should allow for the entire monitoring area to be fully surveyed within five minutes.

Each land-based observer can cover a 180º arc over a 50 meter (m) area. Each boat observer can cover a 50 m transect on one side of the boat. Using the *Seabird Monitoring Site/Transects Identification Form*, insert an aerial photo of the project site and outline each boat transect or land-based monitoring site. Identify on the aerial photo where each of the two types of monitoring (boat transects and land-based sites) will occur (See Example Dolphin Repair). Construction activity and/or other site specific variables (i.e., topography, pier or barge placement, etc.) can limit visibility. These should be identified on the aerial photo when known ahead of time. If conditions change on-site (e.g., a barge moves into the monitoring zone), monitoring locations can be refined in the field. In that case, note final monitoring locations on an aerial photo or plan sheet, and document the changes in the final monitoring report.

For each land-based monitoring site, draw the shoreline on the *Seabird Land-Based Monitoring Site Form*. Include on-site information such as structures that could be used by seabirds, or fishing piers, which may draw in feeding birds (i.e. gulls). The gridwork will allow the observer to quickly fill in location identifiers during monitoring.

## 3.4 Monitoring Techniques

One qualified biologist shall be identified as the Lead Biologist. The Lead Biologist has the authority to stop pile driving when murrelets are detected in the monitoring area or when visibility impairs monitoring. The Lead Biologist is responsible for:

- Ensuring consistency with the criteria in the consultation;
- Communicating with monitoring crew(s), the pile driver operator, and the Washington Ecological Services Office; and
- Determining monitoring start and end times.

An appropriate number of qualified observers will be positioned on shore and in boats to provide adequate coverage of the monitoring area to ensure no murrelets are in the monitoring area. Monitoring will begin at least 30 minutes prior to commencement of pile driving. Each qualified observer will cover an on shore station or boat transect that is no more than 50 m wide. All observers are responsible for:

- Understanding the requirements in the consultation and monitoring plan;
- Knowing the lines and method of communicating with the Lead Biologist and

boat operator (if an observer on the boat);

- Evaluating the sea conditions and visibility;
- Calibrating their ability to determine a 50 m distance at the beginning of each day. Calibration should be done using a range finder on a stationary object on the water; and
- Determining when conditions for monitoring are not met.

Monitoring will only occur when the sea state is at a Beaufort scale of 2 or less. The Beaufort scale is presented in Table 1 below. Observers should scan without a scope or binoculars; scopes and binoculars should only be used to verify species.

Observers will be positioned at land-based vantage points to scan for murrelets within the monitoring zone. The land-based vantage points must have an unobstructed view of the monitoring zone at all times. Each land-based observer can cover a 50 m area with a 180º arc. At least 2 full sweeps of the monitoring zone shall be conducted prior to pile driving to ensure that no murrelets are in the monitoring zone. Each boat observer is responsible for scanning from 0° (straight ahead of bow) to 90° left or right, depending upon which side of the boat they occupy. Observers should occasionally scan past 90°, looking for murrelets that may have surfaced behind the boat. Boat speed should be no less than 5 knots and no greater than 10 knots. Observer coverage should not be compromised; therefore, observer's ability to scan dictates the speed of the boat. Boat operators will not function as murrelet monitors while operating the boat.

If no murrelets are within the monitoring zone, the observers will notify the Lead Biologist who will communicate to the pile driver operator that pile driving may commence. During pile driving the observers on shore will continue scanning the area for murrelets. The observers in the boats will patrol and scan the monitoring area. All observers will have two-way radios with earpieces to allow for effective communication during pile driving. If murrelets are seen within the monitoring zone during pile driving, the observers will immediately notify the Lead Biologist who will communicate to the pile driver operator that he/she is to cease pile driving. Pile driving will not resume until the murrelets have left the monitoring area and at least 2 full sweeps of the monitoring area have confirmed murrelets are not present.

When a murrelet is detected within the monitoring area, it will be continuously observed until it leaves the monitoring area. If observers lose sight of the murrelet, searches for the murrelet will continue for at least 5 minutes. If the murrelet is still not found, then at least 2 full sweeps of the monitoring area to confirm no murrelets are present will be conducted prior to resumption of pile driving.

***It is the observer's responsibility to determine if he/she is not able to see murrelets and inform the Lead Biologist that the monitoring needs to be terminated until conditions allow for accurate monitoring.***

Murrelets are especially vulnerable to disturbance when they are molting and flightless. Molting occurs after nesting in late summer, typically July through October in Puget Sound populations. Extra precaution should be exercised during this period.

**Table 1 – Beaufort Wind Scale develop in 1805 by Sir Francis Beaufort of England (0=calm to 12=hurricane)**

| Force | Wind (knots) | Classification | Appearance of wind effects on the water | Appearance of wind effects on land | Notes specific to on-water seabird observations |
|---|---|---|---|---|---|
| 0 | <1 | Calm | Sea surface smooth and mirror like | Calm, smoke rises vertically | Excellent conditions, no wind, small or very smooth swell. You have the impression you could see anything. |
| 1 | 1-3 | Light air | Scaly ripples, no foam crests | Smoke drift indicates wind direction, still wind vanes | Very good conditions, surface could be glassy (Beaufort 0), but with some lumpy swell or reflection from forests, glare, etc. |
| 2 | 4-6 | Light breeze | Small wavelets, crests glassy, no breaking | Wind felt on face, leaves rustle, vanes begin to move | Good conditions, no whitecaps, texture/lighting contrast of water make murrelets more difficult to see. Surface could also be glassy or have small ripples, but with a short, lumpy swell, thick fog, etc. |
| 3 | 7-10 | Gentle breeze | Large wavelets, crests beginning to break, scattered whitecaps | Leaves and small twigs constantly moving, light flags extended | Surveys cease, scattered whitecaps present, detection of murrelets definitely compromised, a hit-or-miss chance of seeing them owing to water choppiness and high contrast. This could also occur at lesser wind with a very short wavelength, choppy swell. |
| 4 | 11-16 | Moderate breeze | Small waves 0.3 to 1.1m becoming longer, numerous whitecaps | Dust, leaves, and loose paper lifted, small tree branches move | Whitecaps abundant, sea chop bouncing the boat around, etc. |
| 5 | 17-21 | Fresh breeze | Moderate waves 1.1 to 2.0 m taking longer form, many whitecaps, some spray | Small trees begin to sway | |

## 3.5 Limitations

No monitoring will be conducted during inclement weather that creates potentially hazardous conditions as determined by the Lead Biologist. Observers must have visibility to at least 50 m. No monitoring will be conducted when visibility is significantly limited such as during heavy rain, fog, glare or in a Beaufort sea state greater than 2.

Glare can significantly limit an observer's ability to detect birds. Boat orientation may be adjusted to reduce glare (e.g. change direction or reduce width of transects to 50 m with observers on only one side of boat). However, if visibility cannot be adjusted, monitoring and pile driving must cease until effective monitoring can be conducted.

Monitoring will not start until after sunrise and will cease prior to sunset. Specific timing restrictions may be in place per the consultation documents.

## 3.6 Documentation

The observers will document the number and general location of all murrelets in the monitoring area. Additional information on other seabirds and behaviors will be collected during documentation to improve general data knowledge on seabird presence and distribution as well as project impacts on various seabirds. Each observer will record information using the *Seabird Monitoring Data Collection Form* and reference completed *Seabird Monitoring Site/Transects Identification* and *Seabird Land-Based Monitoring Site Forms*. Forms are included in the Appendix.

**Data Collection**

All murrelets within transects or monitoring sites will be continuously documented during impacting activities. On the *Seabird Monitoring Data Collection Form*, document the time, number of birds, location, and observed behavior (See Example Dolphin Repair). Update the documentation when a murrelet changes behavior, changes location, or leaves the area. To the extent possible, the observers will also record each murrelet "take" incident observed, as defined in the final Biological Opinion. This may include obvious disturbance responses from pile driving or other construction activities, and injury or mortality that can be attributed to project-related activities.

Observers will also note all seabirds within the area that appear to be acting abnormally during any project activities. For example, if a seabird is listing, paddling in circles, shaking head, or suddenly flushing at the onset of activity, note the information on the *Seabird Monitoring Data Collection Form.* For all birds except murrelets, providing a genus level (grebe, loon, cormorant, scoter, gull, etc.) of identification is sufficient.

General information on other seabird behavior and distribution within the monitoring area will be collected. Every two hours at minimum during pile driving activities, the observer will document other seabird presence, behavior, and distribution in the monitoring area. This information can be collected more frequently. Many seabirds may linger in an area for several hours. If this is the case, note the time, species, and in the comments section identify that this is the same group from earlier and document any notable changes in behavior.

Under location, the data form indicates two separate options for documenting location. Land-based observers can fill out the land-based only or both land-based and boat sections. The land-based location will be based on the grid drawn out on the *Seabird Land-Based Monitoring Site Form* (See Example Dolphin Repair). For the boat transect locations, identify the distance in meters from the boat to the seabird and whether it is landward (toward activity) or seaward (away from activity).

**Example Dolphin Repair**

## Seabird Monitoring Site/Transect Identification Form

**Project Name**

Dolphin repair

**Monitoring Dates**

November 8, 9, 10, 2012

**Number of Monitoring Sites/Transects**

4

Insert aerial photo of entire monitoring project area. Identify each monitoring site/station reflecting 50 meter zones for each observer. For example, if there are two observers on a boat transect, the box will be 100 meters wide. Some monitoring stations will overlap and should be indicated here.

## Seabird Land-Based Monitoring Site Form

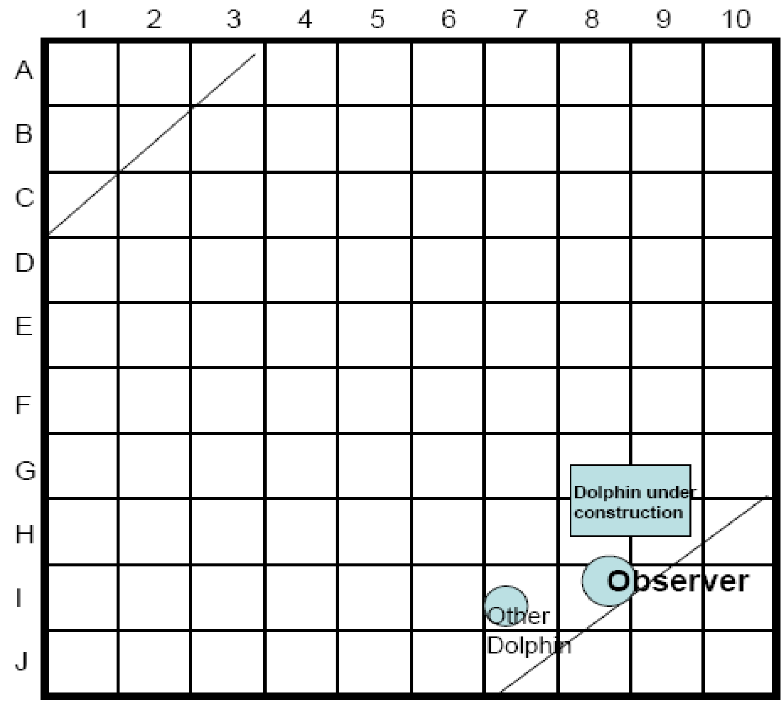
**Project Name** Dolphin Repair **Date** 11/10/12

**Land Based Monitoring Site ID** Station #4

For each monitoring station referenced in the main map grid, sketch the coastline using the 5 meter squares. Indicate the direction to where impacting activities are occurring.

Use space below to describe additional monitoring site details that may be pertinent such as other structures seabirds may use.

Observer located at end of terminal pier adjacent to construction activities. There is another dolphin to the west currently used by cormorants.




## Seabird Monitoring Data Collection Form

**Date** 

**Project Name** Dolphin Repair **Monitoring Site/Transect ID** Land Based Station #4

**Observers** Harry Downy

**Activity** Pile Driving **Time and Duration** 10:30 am to 4:00 pm

| Time | Species | # of birds | Wind speed (Beaufort Marine scale) | Land Observer | Boat Observer | | Observed Behavior* | Comments |
|---|---|---|---|---|---|---|---|---|
| | | | | Grid Location | Distance | Land/Sea Ward | | |
| 10 am | scoters | 10 | 2 | C6 | | | R | |
| 10 am | cormorants | 20 | 2 | I8 | | | R, P | Hanging out on adjacent dolphin |
| 11:15 am | marbled murrelet | 1 | 1 | B4 | | | F | Pile driving ceased, MM left observation area at 11:20 |
| 12:00 pm | grebe | 2 | 2 | G6 | | | P | |
| 12:00 pm | cormorants | 20 | 2 | I8 | | | R, P | Hanging out on adjacent dolphin |
| 2:00 | gulls | 15 | 1 | H10 | | | F | Group attracted by fisherman dumping guts |
| 2:00 | cormorants | 20 | 1 | I8 | | | R, P | Hanging out on adjacent dolphin |
| 4:00 | gulls | 5 | 2 | H10 | | | F | Residuals from earlier |
| 4:00 | cormorants | 20 | 2 | I8 | | | R, P | Hanging out on adjacent dolphin |
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* R=resting, F=feeding/diving, P=preening, Y=flying/flushing, T=transient, N=nesting, O=other

**Seabird Monitoring Data Collection Form** Date 11/10/12

**Project Name** Dolphin Repair **Monitoring Site/Transect ID** Land Based Station #4

**Observers** Jimmy Jones

**Activity** Pile Driving **Time and Duration** 10:30 am to 4:00 pm

| | | | | Land Observer | Boat Observer | | | |
|---|---|---|---|---|---|---|---|---|
| Time | Species | # of birds | Wind speed (Beaufort Marine scale) | Grid Location | Distance | Land/Sea Ward | Observed Behavior* | Comments |
| 10 am | grebe | 1 | 2 | | 25 | sea | T | |
| 11:25 | marbled murrelet | 1 | 1 | | 45 | land | F | Pile driving ceased at 11:15, left monitoring area at 11:45 |
| 12:00 | scoters, loon | 8 | 1 | | 15 | land | R, P | Startled by pile driving re-start, flushed out of area |
| 12:00 | common murre | 2 | 1 | | 25 | sea | T | Startled by pile driving re-start, flushed out of area |
| 2:00 | gulls | 1 | 2 | | 75 | sea | T | |
| 4:00 | gulls | 5 | 2 | | 50 | sea | T | |
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* R=resting, F=feeding/diving, P=preening, Y=flying/flushing, T=transient, N=nesting, O=other

## 3.7 Timing and Duration

Pile driving cannot start until the monitoring pre-sweep has been conducted. The pre-sweep monitoring can commence once there is enough daylight for adequate visibility, and must begin at least 30 minutes before the initiation of pile driving. Monitoring will then continue until pile driving is completed each day. The monitoring set-up (i.e., number and location of observers) should allow for the entire monitoring are to be covered within five minutes.

## 3.8 Contingency

In the unlikely event that a murrelet is perceived to be injured by pile driving, all pile driving will cease and the Washington Ecological Services Office will be contacted as soon as possible.

The Lead Federal Action Agency will work with the Washington Ecological Services Office to make necessary changes to the monitoring plan as described in section 2.0 above. Pile driving cannot resume until the plan has been amended, unless the Washington Ecological Services Office cannot be reached, then the Lead Biologist determines the course of action and continues to ensure consistency with the consultation.

## 4.0 Beach Surveys

Searches for diving seabird carcasses along nearby beaches will be conducted following pile driving activities. The biologist will walk accessible beaches within 0.5 mile of the pile driving location. Beach surveys will be conducted during low or receding tides, if possible, to maximize the chances of finding beached carcasses. Beach surveys will be conducted each day following in-water impact pile driving (as is practical based on the timing of tide events and pile driving activities.) Beach surveys are of secondary priority and will not be conducted if such activities would interfere with the implementation of murrelet monitoring or if the timing of low/receding tides imposes unreasonable schedule demands on the biologist.

Any dead murrelets or other diving seabirds found during the beach surveys (or during monitoring activities) will be collected by monitoring staff and delivered, as soon as possible, to the Washington Ecological Services Office in Lacey, Washington for examination. Collected carcasses will be put in plastic bags, and kept cool (but not frozen) until delivery to the Washington Ecological Services Office. Surveyors will follow the chain-of-custody process included in the consultation documents.

## 5.0 FWS Communication

Prior to the initiation of monitoring the Lead Federal Action Agency and a representative from the Washington Ecological Services Office will meet to review the proposed monitoring locations and any logistical concerns that may have developed during monitoring preparation. The Lead Federal Action Agency will keep the Washington Ecological Services Office informed of the progress and effectiveness of the monitoring activities and of the number and disposition of murrelet take that is documented throughout the duration of the project.

The Lead Federal Action Agency will notify the Washington Ecological Services Office of any problems and/or necessary modification to the monitoring plan. The Lead Federal Action Agency will coordinate with the Washington Ecological Services Office in the development of a modified approach and will obtain Washington Ecological Services Office approval for such modifications.

Primary points of contact at the Washington Ecological Services Office are:

1. Consulting Biologist
2. Lee Corum – phone: (360) 764-3527

## 6.0 Personnel Qualifications and Training

All observers must be certified under the Marbled Murrelet Marine Protocol. Observers will have appropriate qualifications, including education or work experience in biology, ornithology, or a closely related field; at least one season (2-3 months) of work with bird identification being the primary objective (i.e. not incidental to other work). Observers must have experience identifying marine birds in the Pacific Northwest, as well as understanding and documenting bird behavior.

All observers will attend the marbled murrelet marine monitoring protocol training and pass the written and photo examination with 90% proficiency. Upon successful completion, observers will be certified. Certification is valid for one year.

Recertification is required annually, unless the observer can document that he/she implemented the monitoring protocol for at least 25 monitoring days in the previous year. Recertification can then be delayed for one year; however, recertification can only be delayed for one year.

Certifications will be considered expired after one year, unless the Washington Ecological Services Office is notified by the biologist that greater than 25 days of survey were done within one year of their certificate date. If an observer does conduct greater than 25 days of survey the certificate will be valid for an additional year from the certificate date. To extend a certification the biologist sends an email to the attention of Lee Corum (Lee_Corum@fws.gov) with the dates of the surveys they conducted and the date of their original certificate. The Washington Ecological Services Office will maintain a list a certified observers and it will be available on our website.

The Lead Federal Agency is expected to provide all observers with a copy of the consultation documents for the project. Observers must read and understand the contents of the consultation documents related to identifying, minimizing, and reporting "incidental take" of murrelets.

## 7.0 Reporting

At the completion of each in-water work window for which there has been impact pile driving, the Lead Federal Action Agency will forward a monitoring report to the Washington Ecological Services Office within 30 days. Reports shall be sent to the attention of Lee Corum. The report shall include:

- Observation dates, times, and conditions
- Description of the any "take" (as described in the final Biological Opinion) identified by the biologist
- Copies of field data sheets or logs

*Note: Questions and comments regarding this protocol should be directed to Lee Corum at the USFWS, Washington Ecological Services Office (360-764-3527); Lee_Corum@fws.gov.*

# APPENDIX

Print Form

# Seabird Monitoring Site/Transect Identification Form

**Project Name**

**Monitoring Dates**

**Number of Monitoring Sites/Transects**

Insert aerial photo of entire monitoring project area. Identify each monitoring site/transect. Each monitoring station will reflect the 50 meter zone for each observer. For example, if there are two observers on a boat transect, the box will be 100 meters wide. Some monitoring stations will overlap and should be indicated here.

## Seabird Land-Based Monitoring Site Form

**Project Name** ______________________________ **Date** ______________

**Land Based Monitoring Site ID** ______________________________

For each monitoring station referenced in the main map grid, sketch the coastline using the 5 meter squares. Indicate the direction to where impacting activities are occurring.

Use space below to describe additional monitoring site details that may be pertinent such as other structures seabirds may use.

| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| A | | | | | | | | | | |
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| D | | | | | | | | | | |
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| G | | | | | | | | | | |
| H | | | | | | | | | | |
| I | | | | | | | | | | |
| J | | | | | | | | | | |

## Seabird Monitoring Data Collection Form

Date____________________

Project Name ______________________________ Monitoring Site/Transect ID ______________________________

Observers______________________________________________________________

Activity ______________________________ Time and Duration______________________________

| Time | Species | # of birds | Wind speed (Beaufort Marine scale) | Land Observer | Boat Observer | | Observed Behavior* | Comments |
|---|---|---|---|---|---|---|---|---|
| | | | | Grid Location | Distance | Land/Sea Ward | | |
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* R=resting, F=feeding/diving, P=preening, Y=flying/flushing, T=transient, N=nesting, O=other

# APPENDIX C
# STATUS OF THE SPECIES: BULL TROUT

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**Appendix C**
**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).

[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.

<u>*Flathead Geographic Region*</u>

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).

<u>*Kootenai Geographic Region*</u>

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).

<u>*Coeur d'Alene Geographic Region*</u>

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*

<u>*Climate Change*</u>

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*

*Conservation Needs*

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 D
# STATUS OF DESIGNATED CRITICAL HABITAT:  BULL TROUT

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**Appendix D**
**Status of Designated Critical Habitat: Bull Trout**

Past designations of critical habitat have used the terms "primary constituent elements" (PCEs), “physical and biological features” (PBFs) or "essential features" to characterize the key components of critical habitat that provide for the conservation of the listed species. The new critical habitat regulations (81 FR 7214) discontinue use of the terms “PCEs” or “essential features” and rely exclusively on use of the term PBFs for that purpose because that term is contained in the statute. To be consistent with that shift in terminology and in recognition that the terms PBFs, PCEs, and essential habit features are synonymous in meaning, we are only referring to PBFs herein. Therefore, if a past critical habitat designation defined essential habitat features or PCEs, they will be referred to as PBFs in this document. This does not change the approach outlined above for conducting the ‘‘destruction or adverse modification’’ analysis, which is the same regardless of whether the original designation identified PCEs, PBFs or essential features.

## Current Legal Status of the Critical Habitat

*Current Designation*

The U.S. Fish and Wildlife Service (Service) published a final critical habitat designation for the coterminous United States population of the bull trout on October 18, 2010 (USFWS 2010, entire); the rule became effective on November 17, 2010. A justification document was also developed to support the rule and is available on the Service’s website: (http://www.fws.gov/pacific/bulltrout). The scope of the designation involved the species’ coterminous range, which includes the Coastal, Klamath, Mid-Columbia, Upper Snake, Columbia Headwaters and St. Mary’s Recovery Unit population segments. Rangewide, the Service designated reservoirs/lakes and stream/shoreline miles as bull trout critical habitat (Table 1). Designated bull trout critical habitat is of two primary use types: 1) spawning and rearing, and 2) foraging, migration, and overwintering (FMO).

Table 1. Stream/Shoreline Distance and Reservoir/Lake Area Designated as Bull Trout Critical Habitat.

| State | Stream/Shoreline Miles | Stream/Shoreline Kilometers | Reservoir/ Lake Acres | Reservoir/ Lake Hectares |
|---|---|---|---|---|
| Idaho | 8,771.6 | 14,116.5 | 170,217.5 | 68,884.9 |
| Montana | 3,056.5 | 4,918.9 | 221,470.7 | 89,626.4 |
| Nevada | 71.8 | 115.6 | - | - |
| Oregon[1] | 2,835.9 | 4,563.9 | 30,255.5 | 12,244.0 |
| Oregon/Idaho[2] | 107.7 | 173.3 | - | - |
| Washington | 3,793.3 | 6,104.8 | 66,308.1 | 26,834.0 |
| Washington (marine) | 753.8 | 1,213.2 | - | - |
| Washington/Idaho | 37.2 | 59.9 | - | - |
| Washington/Oregon | 301.3 | 484.8 | - | - |
| Total[3] | 19,729.0 | 31,750.8 | 488,251.7 | 197,589.2 |

[1] No shore line is included in Oregon
[2] Pine Creek Drainage which falls within Oregon
[3] Total of freshwater streams: 18,975

The 2010 revision increases the amount of designated bull trout critical habitat by approximately 76 percent for miles of stream/shoreline and by approximately 71 percent for acres of lakes and reservoirs compared to the 2005 designation.

The final rule also identifies and designates as critical habitat approximately 1,323.7 km (822.5 miles) of streams/shorelines and 6,758.8 ha (16,701.3 acres) of lakes/reservoirs of unoccupied habitat to address bull trout conservation needs in specific geographic areas in several areas not occupied at the time of listing. No unoccupied habitat was included in the 2005 designation. These unoccupied areas were determined by the Service to be essential for restoring functioning migratory bull trout populations based on currently available scientific information. These unoccupied areas often include lower main stem river environments that can provide seasonally important migration habitat for bull trout. This type of habitat is essential in areas where bull trout habitat and population loss over time necessitates reestablishing bull trout in currently unoccupied habitat areas to achieve recovery.

The final rule continues to exclude some critical habitat segments based on a careful balancing of the benefits of inclusion versus the benefits of exclusion. Critical habitat does not include: 1) waters adjacent to non-Federal lands covered by legally operative incidental take permits for habitat conservation plans (HCPs) issued under section 10(a)(1)(B) of the Endangered Species Act of 1973, as amended (Act), in which bull trout is a covered species on or before the publication of this final rule; 2) waters within or adjacent to Tribal lands subject to certain commitments to conserve bull trout or a conservation program that provides aquatic resource protection and restoration through collaborative efforts, and where the Tribes indicated that inclusion would impair their relationship with the Service; or 3) waters where impacts to national security have been identified (USFWS 2010, p. 63903). Excluded areas are approximately 10 percent of the stream/shoreline miles and 4 percent of the lakes and reservoir acreage of designated critical habitat. Each excluded area is identified in the relevant Critical Habitat Unit

(CHU) text, as identified in paragraphs (e)(8) through (e)(41) of the final rule. It is important to note that the exclusion of waterbodies from designated critical habitat does not negate or diminish their importance for bull trout conservation. Because exclusions reflect the often complex pattern of land ownership, designated critical habitat is often fragmented and interspersed with excluded stream segments.

**The Physical and Biological Features**

*Conservation Role and Description of Critical Habitat*

The conservation role of bull trout critical habitat is to support viable core area populations (USFWS 2010, p. 63898). The core areas reflect the metapopulation structure of bull trout and are the closest approximation of a biologically functioning unit for the purposes of recovery planning and risk analyses. CHUs generally encompass one or more core areas and may include FMO areas, outside of core areas, that are important to the survival and recovery of bull trout.

Thirty-two CHUs within the geographical area occupied by the species at the time of listing are designated under the revised rule. Twenty-nine of the CHUs contain all of the physical or biological features identified in this final rule and support multiple life-history requirements. Three of the mainstem river units in the Columbia and Snake River Basins contain most of the physical or biological features necessary to support the bull trout's particular use of that habitat, other than those physical biological features associated with physical and biological features (PBFs) 5 and 6, which relate to breeding habitat.

The primary function of individual CHUs is to maintain and support core areas, which 1) contain bull trout populations with the demographic characteristics needed to ensure their persistence and contain the habitat needed to sustain those characteristics (Rieman and McIntyre 1993, p. 19); 2) provide for persistence of strong local populations, in part, by providing habitat conditions that encourage movement of migratory fish (MBTSG 1998, pp. 48-49; Rieman and McIntyre 1993, pp. 22-23); 3) are large enough to incorporate genetic and phenotypic diversity, but small enough to ensure connectivity between populations (Hard 1995, pp. 314-315; Healey and Prince 1995, p. 182; MBTSG 1998, pp. 48-49; Rieman and McIntyre 1993, pp. 22-23); and 4) are distributed throughout the historic range of the species to preserve both genetic and phenotypic adaptations (Hard 1995, pp. 321-322; MBTSG 1998, pp. 13-16; Rieman and Allendorf 2001, p. 763; Rieman and McIntyre 1993, p. 23).

*Physical and Biological Features for Bull Trout*

Within the designated critical habitat areas, the PBFs for bull trout are those habitat components that are essential for the primary biological needs of foraging, reproducing, rearing of young, dispersal, genetic exchange, or sheltering. Based on our current knowledge of the life history, biology, and ecology of this species and the characteristics of the habitat necessary to sustain its essential life-history functions, we have determined that the PBFs, as described within USFWS 2010, are essential for the conservation of bull trout. A summary of those PBFs follows.

1. Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia.

2. Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including but not limited to permanent, partial, intermittent, or seasonal barriers.
3. An abundant food base, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish.
4. Complex river, stream, lake, reservoir, and marine shoreline aquatic environments, and processes that establish and maintain these aquatic environments, with features such as large wood, side channels, pools, undercut banks and unembedded substrates, to provide a variety of depths, gradients, velocities, and structure.
5. Water temperatures ranging from 2 °C to 15 °C, with adequate thermal refugia available for temperatures that exceed the upper end of this range. Specific temperatures within this range will depend on bull trout life-history stage and form; geography; elevation; diurnal and seasonal variation; shading, such as that provided by riparian habitat; streamflow; and local groundwater influence.
6. In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival. A minimal amount of fine sediment, generally ranging in size from silt to coarse sand, embedded in larger substrates, is characteristic of these conditions. The size and amounts of fine sediment suitable to bull trout will likely vary from system to system.
7. A natural hydrograph, including peak, high, low, and base flows within historic and seasonal ranges or, if flows are controlled, minimal flow departure from a natural hydrograph.
8. Sufficient water quality and quantity such that normal reproduction, growth, and survival are not inhibited.
9. Sufficiently low levels of occurrence of non-native predatory (e.g., lake trout, walleye, northern pike, smallmouth bass); interbreeding (e.g., brook trout); or competing (e.g., brown trout) species that, if present, are adequately temporally and spatially isolated from bull trout.

The revised PBF's are similar to those previously in effect under the 2005 designation. The most significant modification is the addition of a ninth PBF to address the presence of nonnative predatory or competitive fish species. Although this PBF applies to both the freshwater and marine environments, currently no non-native fish species are of concern in the marine environment, though this could change in the future.

Note that only PBFs 2, 3, 4, 5, and 8 apply to marine nearshore waters identified as critical habitat. Also, lakes and reservoirs within the CHUs also contain most of the physical or biological features necessary to support bull trout, with the exception of those associated with PBFs 1 and 6. Additionally, all except PBF 6 apply to FMO habitat designated as critical habitat.

Critical habitat includes the stream channels within the designated stream reaches and has a lateral extent as defined by the bankfull elevation on one bank to the bankfull elevation on the opposite bank. Bankfull elevation is the level at which water begins to leave the channel and move into the floodplain and is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series. If bankfull elevation is not evident on either bank, the ordinary high-water line must be used to determine the lateral extent of critical habitat. The lateral extent of designated lakes is defined by the perimeter of the waterbody as mapped on standard 1:24,000 scale topographic maps. The Service assumes in many cases this is the full-pool level of the waterbody. In areas where only one side of the waterbody is designated (where only one side is excluded), the mid-line of the waterbody represents the lateral extent of critical habitat.

In marine nearshore areas, the inshore extent of critical habitat is the mean higher high-water (MHHW) line, including the uppermost reach of the saltwater wedge within tidally influenced freshwater heads of estuaries. The MHHW line refers to the average of all the higher high-water heights of the two daily tidal levels. Marine critical habitat extends offshore to the depth of 10 meters (m) (33 ft) relative to the mean low low-water (MLLW) line (zero tidal level or average of all the lower low-water heights of the two daily tidal levels). This area between the MHHW line and minus 10 m MLLW line (the average extent of the photic zone) is considered the habitat most consistently used by bull trout in marine waters based on known use, forage fish availability, and ongoing migration studies and captures geological and ecological processes important to maintaining these habitats. This area contains essential foraging habitat and migration corridors such as estuaries, bays, inlets, shallow subtidal areas, and intertidal flats.

Adjacent shoreline riparian areas, bluffs, and uplands are not designated as critical habitat. However, it should be recognized that the quality of marine and freshwater habitat along streams, lakes, and shorelines is intrinsically related to the character of these adjacent features, and that human activities that occur outside of the designated critical habitat can have major effects on physical and biological features of the aquatic environment.

Activities that cause adverse effects to critical habitat are evaluated to determine if they are likely to "destroy or adversely modify" critical habitat by no longer serving the intended conservation role for the species or retaining those PBFs that relate to the ability of the area to at least periodically support the species. Activities that may destroy or adversely modify critical habitat are those that alter the PBFs to such an extent that the conservation value of critical habitat is appreciably reduced (USFWS 2010, pp. 63898:63943; USFWS 2004a, pp. 140-193; USFWS 2004b, pp. 69-114). The Service's evaluation must be conducted at the scale of the entire critical habitat area designated, unless otherwise stated in the final critical habitat rule (USFWS and NMFS 1998, Ch. 4 p. 39). Thus, adverse modification of bull trout critical habitat is evaluated at the scale of the final designation, which includes the critical habitat designated for the Klamath River, Jarbidge River, Columbia River, Coastal-Puget Sound, and Saint Mary-Belly River population segments. However, we consider all 32 CHUs to contain features or areas essential to the conservation of the bull trout (USFWS 2010, pp. 63898:63901, 63944). Therefore, if a proposed action would alter the physical or biological features of critical habitat to an extent that appreciably reduces the conservation function of one or more critical habitat units for bull trout, a finding of adverse modification of the entire designated critical habitat area may be warranted (USFWS 2010, pp. 63898:63943).

*Current Critical Habitat Condition Rangewide*

The condition of bull trout critical habitat varies across its range from poor to good. Although still relatively widely distributed across its historic range, the bull trout occurs in low numbers in many areas, and populations are considered depressed or declining across much of its range (Ratliff and Howell 1992, entire; Schill 1992, p. 40; Thomas 1992, p. 28; Buchanan et al. 1997, p. vii; Rieman et al. 1997, pp. 15-16; Quigley and Arbelbide 1997, pp. 1176-1177). This condition reflects the condition of bull trout habitat. The decline of bull trout is primarily due to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, past fisheries management practices, impoundments, dams, water diversions, and the introduction of nonnative species (USFWS 1998, pp. 31648-31649; USFWS 1999, p. 17111).

There is widespread agreement in the scientific literature that many factors related to human activities have impacted bull trout and their habitat, and continue to do so. Among the many factors that contribute to degraded PBFs, those which appear to be particularly significant and have resulted in a legacy of degraded habitat conditions are as follows: 1) fragmentation and isolation of local populations due to the proliferation of dams and water diversions that have eliminated habitat, altered water flow and temperature regimes, and impeded migratory movements (Dunham and Rieman 1999, p. 652; Rieman and McIntyre 1993, p. 7); 2) degradation of spawning and rearing habitat and upper watershed areas, particularly alterations in sedimentation rates and water temperature, resulting from forest and rangeland practices and intensive development of roads (Fraley and Shepard 1989, p. 141; MBTSG 1998, pp. ii - v, 20-45); 3) the introduction and spread of nonnative fish species, particularly brook trout and lake trout, as a result of fish stocking and degraded habitat conditions, which compete with bull trout for limited resources and, in the case of brook trout, hybridize with bull trout (Leary et al. 1993, p. 857; Rieman et al. 2006, pp. 73-76); 4) in the Coastal-Puget Sound region where amphidromous bull trout occur, degradation of mainstem river FMO habitat, and the degradation and loss of marine nearshore foraging and migration habitat due to urban and residential development; and 5) degradation of FMO habitat resulting from reduced prey base, roads, agriculture, development, and dams.

*Effects of Climate Change on Bull Trout Critical Habitat*

One objective of the final rule was to identify and protect those habitats that provide resiliency for bull trout use in the face of climate change. Over a period of decades, climate change may directly threaten the integrity of the essential physical or biological features described in PBFs 1, 2, 3, 5, 7, 8, and 9. Protecting bull trout strongholds and cold water refugia from disturbance and ensuring connectivity among populations were important considerations in addressing this potential impact. Additionally, climate change may exacerbate habitat degradation impacts both physically (e.g., decreased base flows, increased water temperatures) and biologically (e.g., increased competition with non-native fishes).

Many of the PBFs for bull trout may be affected by the presence of toxics and/or increased water temperatures within the environment. The effects will vary greatly depending on a number of factors which include which toxic substance is present, the amount of temperature increase, the likelihood that critical habitat would be affected (probability), and the severity and intensity of any effects that might occur (magnitude).

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

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# APPENDIX C

# 90% TEMPORARY EROSION AND SEDIMENT CONTROL (TESC)

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# APPENDIX E
# STATUS OF THE SPECIES: MARBLED MURRELET

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**Appendix E**
**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.

<u>Murrelets in the Marine Environment</u>

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. 2021, pp. 22, 24; 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 Washington. 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 often lower during the non-breeding season than they are during the breeding season (Pearson et al. 2022, pp. 7-9). Known areas of winter concentration include and southern and eastern end of Strait of Juan de Fuca (primarily Sequim, Discovery, and Chuckanut Bays) and the San Juan Islands, 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). Murrelets 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).

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).

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). In the Puget Sound–Georgia Basin region, the diet of murrelet nestlings has shifted to include a larger proportion of Pacific sand lance than it did previously (Gutowsky et al. 2009, p. 251). 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).

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). Within the listed range, murrelet nests have been found in live conifers, specifically, western hemlock (*Tsuga heterophylla*), Sitka spruce (*Picea sitchensis*), Douglas-fir (*Pseudotsuga menziesii*), western red cedar (*Thuja plicata*), and in California, coast redwood (*Sequoia sempervirens*) (Hamer and Nelson 1995, p. 74; 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). At the southern end of the range, nesting occurs in a narrower band within around 15 miles of the coast (Halbert and Singer 2017, pp. 5-6). 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. Substrates such as moss or needles on the nest platform are 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 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, pp. 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 than 100 km inland in Washington (70 miles). The inland distribution in the Siskiyou Mountains portion of the species range (southern Oregon and northern California) 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, pp. 3-4). At the southernmost extent of the range, murrelets are restricted to the western slopes of the Santa Cruz Mountains (Halbert and Singer 2017, pp. 5-6). 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, pp. 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). Among 21 million habitat capable lands in Washington, Oregon, and California, 1.49 million acres (~7 percent) were higher probability nesting habitat for the murrelet in 2017 (Lorenz et al. 2021, p. 48).

Monitoring of murrelet nesting habitat within the Northwest Forest Plan area indicates higher probability nesting habitat has decreased from an estimated 1.51 million acres in 1993 to an estimated 1.49 million acres in 2017, a total decrease of about 1.4 percent (Lorenz et al. 2021, p. 28). Timber harvest is the primary cause of nesting habitat loss on both Federal and non-Federal lands (Lorenz et al. 2021, p. 33). While most (71 percent) of the potential habitat is located on federal lands, a substantial amount of nesting habitat occurs on nonfederal (29 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 probability murrelet nesting habitat by State and major land ownership within the area of the NWFP – derived from 2017 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 higher probability nesting habitat (all lands) (1,000s of acres) | Percent of habitat capable land that is currently in habitat |
|---|---|---|---|---|---|---|
| WA | 10,849.3 | 702.4 | 39.6 | 194.0 | 936.0 | 9 % |
| OR | 6,609.5 | 273.8 | 38.3 | 205.7 | 517.8 | 8 % |
| CA | 3,250.1 | 11.2 | 0.5 | 26.9 | 38.6 | 1 % |
| Totals | 20,708.9 | 987.4 | 78.4 | 426.5 | 1,492.2 | 7 % |
| Percent | | 66 % | 5 % | 29 % | 100 % | - |

Source: (Lorenz et al. 2021, pp. 3, 28).

**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.

<u>Population Status and Trends</u>

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 2020 indicated that the murrelet population in Conservation Zones 1 through 5 (NWFP area) increased at a rate of 0.3 percent per year (McIver et al. 2022, p. 4). While the trend estimate across this period is slightly positive, the confidence intervals are tight around zero (95 percent confidence interval [CI]: -0.6 to 1.2 percent), indicating that at the scale of the NWFP area, the population is changing very little (McIver et al. 2022, p. 4) (Table 2). At the state scale, Washington exhibited a significant declining trend between 2001 and 2018 (4.1 percent decrease per year, while Oregon and California showed significant positive trends (OR = 2.0 percent increase per year; CA = 3.8 percent increase per year) (McIver et al. 2022, p. 4) (Table 2). Zone 1 shows the greatest decline of 5.0 percent per year, while the decline in Zone 2 is smaller, 3.3 percent per year, and less statistically certain, though still reaching the traditional threshold for statistical significance (Table 2). Zone 4 shows the greatest increase of 2.8 percent per year, while Zone 3 shows a smaller, and less statistically certain (though still statistically significant), 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. 2021, p. 26). 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 2020 was 19,700 murrelets (95 percent confidence interval [CI]: 15,500 to 23,900 birds) (McIver et. al 2022, p. 3). 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 2020 (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 slightly different at-sea survey methods (Felis et al. 2022, pp. 2-3). Surveys in Zone 6 indicate a small population of murrelets with no clear trends. Population estimates from 2001 to 2021 have fluctuated from a high of 699 murrelets in 2003, to a low of 174 murrelets in 2008 (Felis et al. 2022 p. 8). In 2021, surveys indicated an estimated population of 402 murrelets in Zone 6 (95 percent CI: 219-737) (Felis et al. 2022, p. 8) (Table 2). Any effect of the major loss of nesting habitat in Zone 6 is not yet evident in the population estimate, although 2021 survey results were more variable than usual from one survey to the next (Felis et al. 2022, p. 10).

Table 2. Summary of murrelet population estimates and trends (2001-2020/2021) 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 | 2021 | 1,018 | 564 | 1,428 | 0.617 | -3.3 | -6.1 | +0.4 |
| 3 | 2020 | 8,359 | 5,569 | 11,323 | 5.239 | +1.5 | +0.02 | +3.1 |
| 4 | 2021 | 5,132 | 3,739 | 8,243 | 4.427 | +2.8 | +0.9 | +4.6 |
| 5 | 2021 | 42 | 0 | 79 | 0.473 | +1.5 | -7.1 | +11.7 |
| Zones 1-5 | 2020 | 19,685 | 15,493 | 23,877 | 2.24 | +0.3 | -0.6 | +1.2 |
| Zone 6 | 2021 | 402 | 219 | 737 | na | na | na | na |
| | | | | | | | | |
| WA | 2020 | 4,481 | 2,997 | 5,965 | 0.87 | -4.1 | -5.5 | -2.8 |
| OR | 2020 | 10,742 | 7,565 | 13,919 | 4.69 | +2.0 | +0.8 | +3.2 |
| CA Zones 4 & 5 | 2021 | 3,870 | 2,727 | 5,014 | 2.47 | +3.9 | +2.2 | +5.6 |

Sources: (McIver et al. 2022, pp. 16-20, Felis et al. 2022, p. 8).

Factors Influencing Population Trends

Population monitoring data show murrelet populations declining in Washington, but increasing in Oregon and northern California (McIver et al. 2022, 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 2017, there was a net gain of about 2.9 percent of higher probability potential nesting habitat on federal lands, compared to a net loss of about 10.7 percent on nonfederal lands, for a total cumulative loss of about 7.8 percent of higher probability habitat across the NWFP area (Lorenz et al. 2021, p. 28). 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 (Lorenz et al. 2021, p. 31) (Table 3).

Table 3. Distribution of higher probability murrelet nesting habitat by Conservation Zone, and summary of net habitat changes from 1993 to 2017 within the NWFP area.

| Conservation Zone | 1993 | 2017 | Change (acres) | Change (percent) |
|---|---|---|---|---|
| Zone 1 - Puget Sound/Strait of Juan de Fuca | 512,645 | 476,793 | -35,852 | -7.0 % |
| Zone 2 - Washington Coast | 487,372 | 459,186 | -28,186 | -5.8 % |
| Zone 3 - Northern to central Oregon | 439,852 | 474,561 | +34,709 | +7.9 % |
| Zone 4 - Southern Oregon - northern California | 71,100 | 79,611 | +8,511 | +12.0 % |
| Zone 5 - North-central California | 2,107 | 2,077 | -30 | -1.5 % |

Source: (Lorenz et al. 2021, pp. 39, 41).

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, and 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. 2022, p. 20).

Population Models

Prior to the use of survey data to estimate trends, 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. 2022, 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 2001-2020 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 USFWS (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 proportion of 158 tagged adult birds actually attempted to nest (13 to 20 percent) (Lorenz et al. 2017, p. 316). A recent study in Oregon reported a similar result: 33 of 239 tagged birds (13.8 percent) attempted nesting (Woodis et al. 2022, p. 121). Studies from California also report low rates, though higher than those reported in Washington and Oregon. 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). 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 individual will attempt to breed, but studies reporting higher rates of attempted nesting used similar radio tags, so radio-telemetry methods do not fully account for differences between

[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.

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 Oregon (Woodis et al. 2022, p. 121). 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, out of 33 nesting attempts, chicks successfully fledged from 10 nests, a rate of 0.33 (Woodis et al. 2022, p. 121).

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); in Oregon, of 239 tagged birds, ten produced fledglings; 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, 0.045 in Oregon, 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 0.179, depending on the year and area surveyed (Strong 2018, p. 7; Strong 2020, p. 21; Strong 2021, p. 17). 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 0.052, over 20 years of survey between 1996 and 2021 (Felis et al. 2022, 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 of **Ŕ** for Oregon and California may be unreliable, because at-sea observations are not made in the optimal time period for observing recently-hatched juveniles. Estimates for **Ŕ** in the San Juan Islands in Washington, which include observations better timed to observe juveniles, 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).

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

[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.

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.

<u>Integration and Summary: Murrelet Abundance, Distribution, Trend, and Reproduction</u>

A statistically significant decline was detected in Conservation Zones 1 and 2 for the 2001-2020 period (Table 2). The overall population trend for the NWFP area from the combined 2001-2020 population estimates (Conservation Zones 1 - 5) indicates a potentially stable population with a 0.3 percent increase per year (McIver et al. 2022, 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. 2022, 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; Strong 2022, p. 2); 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. 2022, p. 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. 2021 pp. 28, 30).

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-21$^{st}$ century projected to be approximately four to six degrees Fahrenheit (°F) (2.2 to 3.3 degrees Celsius [°C]) warmer than the late 20$^{th}$ 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 20$^{th}$ century and mid-or late-21$^{st}$ 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 21$^{st}$ century than they were during the late 20$^{th}$ 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 20$^{th}$ 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 exacerbated by climate change (Goss et al. 2020, p. 11; Higuera and Abatzoglou 2021, entire; Romps et al. 2014, p. 853).

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 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). An even more extreme marine heatwave in the Northeastern Pacific, sometimes called "the Blob 2.0," occurred in 2019 to 2021; this was also the longest Northeastern Pacific heatwave on record (Barkhordarian et al. 2022, pp. 2-4). Anthropogenic climate change contributed to the development of these extreme heatwaves, and even more extreme heatwaves are likely to occur as climate change continues (Barkhordarian et al. 2022, p. 9).

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). Fish growth and body composition may also be sensitive to sea surface temperature; for example, one-year-old sand lance (the age typically consumed by murrelet nestlings) were dramatically smaller and less energy-dense during warm water years (2014 through 2016) than during the immediately preceding cool years (2012 through 2013) (von Biela et al. 2019, pp. 176-179). 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 diseased 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. Following the establishment of the NWFP, higher probability habitat has decreased plan-wide between 1993 and 2017 (Lorenz et al. 2021, p. 28). This does not support the goal of the NFWP to increase high quality habitat for the marbled murrelet, for which high quality habitat is defined as higher probability habitat that is also core habitat (Lorenz et al. 2021, p. 51). Furthermore, moderate suitability habitat growth occurred primarily on Federal lands, while non-Federal lands experienced overall habitat loss (Lorenz et al. 2021, p. 48). 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 waters of 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. Higher probability 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, contiguous blocks of moderate and higher probability habitat remain near the coast along the Strait of Juan de Fuca, where there is a lower human footprint (Lorenz et al. 2021, p. 23; 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 in 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. 2021, pp. 22, 24). Zone 2 includes the rich waters of the Olympic Coast National Marine Sanctuary, which are adjacent to areas of contiguous, high-quality habitat along the coast of the Olympic Peninsula, as well as relatively large quantities of higher probability habitat farther inland (Lorenz et al. 2021, pp. 23, 26). Even more than 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 northernmost 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. 2022, 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. Habitat modeling shows an increase in higher probability habitat in Zone 3, but most of the additional habitat is scattered or along forest edges, and some of this increase may be an artifact of the modeling process rather than reflecting actual growth of new nesting opportunities (Lorenz et al. 2021, pp. 42, 49). 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). Between 1993 and 2012, habitat modeling showed that this Zone 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); however, the most recent habitat modeling effort shows a small net increase in higher probability habitat, mainly in scattered patches (Lorenz et al. 2021, p. 42). As in Zone 3, some of the modeled ingrowth may be an artifact of the modeling process rather than reflecting actual growth of new nesting opportunities (Lorenz et al. 2021, p. 49). 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. 2021, p. 28). 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, and a 1 percent reduction in higher probability nesting habitat was observed between 1993 and 2017 (Lorenz et al. 2021, pp. 36-37; 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. 2022, pp. 11-17). One 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. 2021, p. 28; Strong 2018, pp. 6-7), and the most recent estimate, from 2021, was of 42 individuals (McIver et al. 2022, pp. 16-17). 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 2021, with a range between 174 and 699 birds across the years (Felis et al. 2022, p. 8). 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 range 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, including 1) 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 (+0.3 percent per year, 95 percent CI: -0.6 to +1.2 percent) (McIver et al. 2022, p. 4). The most recent extrapolated population estimate for the entire NWFP area was 19,700 murrelets (95 percent CI: 15,500 to 23,900 birds) in 2020 (McIver et al. 2022, 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 (-4.1 percent per year; 95% CI: -5.5 to -2.8%) (McIver et al. 2022, p. 4).

Monitoring of murrelet nesting habitat within the Northwest Forest Plan area indicates high probability nesting habitat has decreased from an estimated 1.51 million acres in 1993 to an estimated 1.49 million acres in 2017, a total decrease of about 1.4 percent (Lorenz et al. 2021, p. 28). 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.

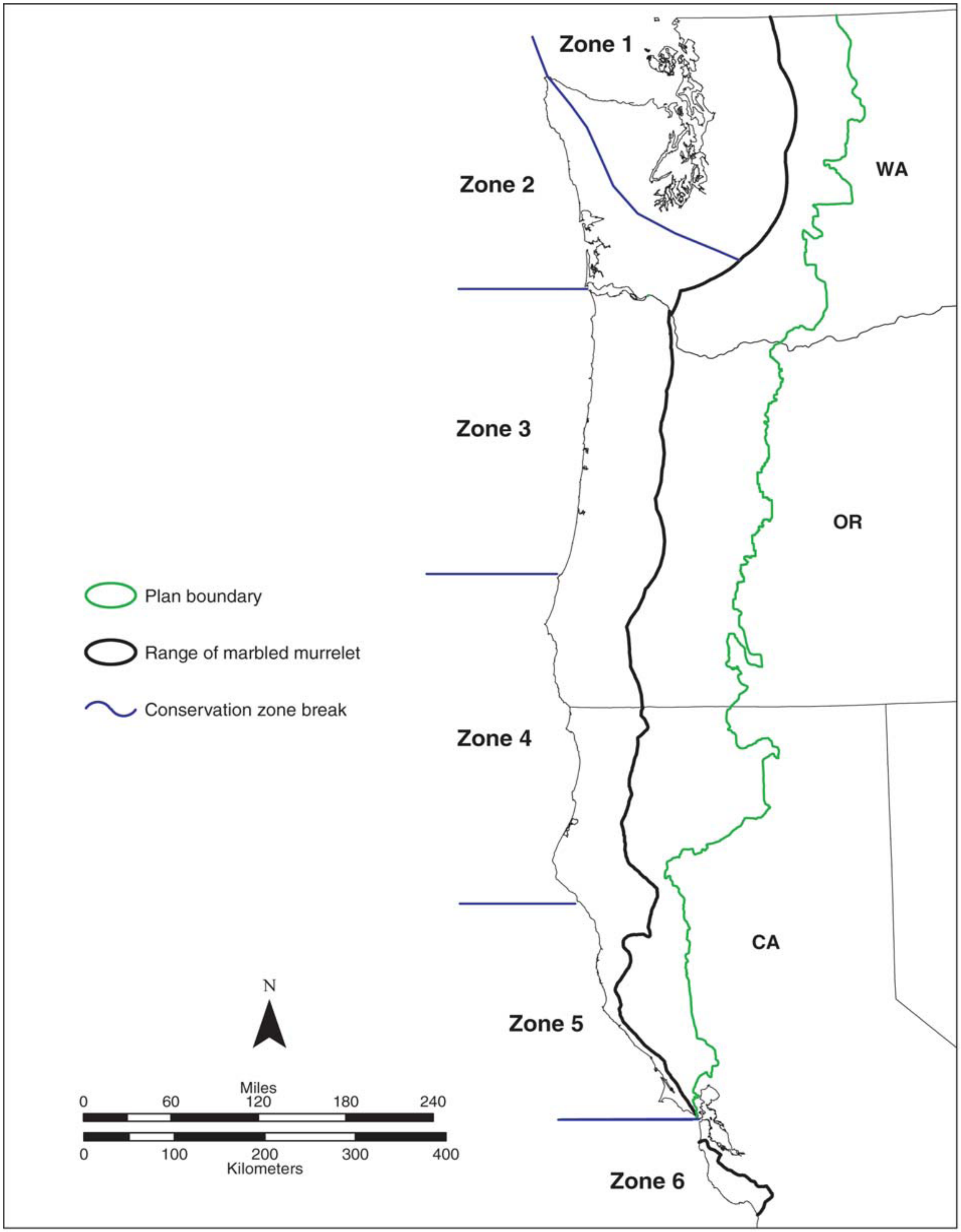


Figure 1. The six geographic areas identified as Conservation Zones in the recovery plan for the marbled murrelet (USFWS 1997). Note: “Plan boundary” refers to the NWFP. Figure adapted from Huff et al. (2006, p. 6).

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