WHITE PAPER

Shoreline Modifications

Prepared for
Washington Department of Fish and Wildlife

December 2007 Working Draft

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Executive Summary

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to "preserve, protect, perpetuate, and manage" the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that hydraulic projects are completed in a manner that prevents damage to public fish and shellfish resources and their habitats. To ensure that the HPA program complies with the Endangered Species Act (ESA), WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain an Incidental Take Permit from the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service (also known as NOAA Fisheries), in accordance with Section 10 of the ESA. For WDFW, the objective is to avoid and/or minimize the incidental take of those aquatic species potentially considered for coverage under the HCP (referred to in this white paper as "HCP species") resulting from activities conducted under an HPA.

The HCP will address the impacts, potential for take, and mitigation measures for effects on HCP species from hydraulic projects that require HPAs. WDFW's intent is to build the scientific foundation for the effort to prepare an HCP for hydraulic projects that receive HPAs. To accomplish this, WDFW is compiling the best available scientific information related to the impacts, potential for incidental "take" of species that may be covered in the HCP (as defined in the ESA; see Section 9 of this report for a definition of "take"), adequacy of existing rules (Washington Administrative Code [WAC] 220-110), and possible management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. As the HPA authority covers all waters of the state, this white paper considers hydraulic project impacts in both freshwater and marine environments.

This white paper is one of a suite of white papers prepared to establish the scientific basis for the HCP and assist WDFW decision-making on what specific HPA activities should be covered by the HCP. This particular white paper compiles and synthesizes existing scientific information on shoreline modifications, such as jetties, breakwaters, groins, and bank barbs. Bank protection structures and activities, such as seawalls, levees, and bulkheads, are addressed in Jones & Stokes (2006).

The objectives of this white paper are:

- To compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the construction, maintenance, repair, replacement, modification, and removal (hereafter collectively referred to as construction and repair) of jetties, breakwaters, groins, and bank barbs.

- To use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely to result from the construction and repair of shoreline modifications.

- To assess the extent to which current HPA rules address the potential impacts on covered species, their habitats, and ecological processes.

- To identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid, minimize, or mitigate the risk of incidental take of HCP species.

The literature review conducted for this white paper identified six impact mechanisms that could potentially affect the HCP species. These mechanisms of impact are both direct and indirect and can have temporary, short-term effects or permanent, long-term effects. The impact mechanisms analyzed in this white paper are:

- Construction and maintenance activities
- Hydraulic and geomorphic modifications
- Water quality modifications
- Riparian vegetation modifications
- Aquatic vegetation modifications
- Ecosystem fragmentation.

This white paper includes a discussion of the potential direct and indirect impacts on the 52 HCP species and their habitats due to exposure to the six identified impact mechanisms. Following this discussion, an evaluation of potential for take of the 52 HCP species is included based on a separate analysis conducted using exposure-response matrices for each of the HCP species. This white paper also reviews data gaps and, where there is sufficient information, estimates risk of take. In addition, habitat protection, conservation, mitigation, and management strategies that could avoid or minimize the identified potential impacts are provided. Key elements of the white paper are:

- Identify the distribution of the 52 HCP species (i.e., whether they use fresh water, marine water, or both) and their habitat requirements.

- Based on the distribution information, identify the risk of “take” associated with each of these impacts mechanisms.

- Identify cumulative impacts.

- Identify data gaps.

- Identify habitat protection, conservation, and mitigation strategies.

1.0 Introduction

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to “preserve, protect, perpetuate, and manage” the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that these activities are completed in a manner that prevents damage to public fish and shellfish resources and their habitats. To ensure that the HPA program complies with the Endangered Species Act (ESA), WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain an Incidental Take Permit (ITP), in accordance with Section 10 of the ESA, from the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service (also known as NOAA Fisheries). For WDFW, the benefits of an HCP are to contribute to the long-term conservation of both listed and unlisted species through the minimization and mitigation of impacts on those species and their habitats, while ensuring that WDFW can legally proceed with the issuance of HPAs that might otherwise result in the incidental “take” of ESA-listed species (as defined in the ESA; see Section 9 of this report for a definition of “take”).

The HCP will identify the impacts on those aquatic species considered for coverage under the HCP, the potential for take, and mitigation measures for hydraulic projects that require HPAs. This white paper is part of the effort to compile the best available scientific information to protect these species during the construction, maintenance, repair, operation, replacement, modification, and removal (hereafter referred to as construction, operation, and repair) of shoreline modifications. To accomplish this, WDFW is analyzing the adequacy of existing rules (Washington Administrative Code [WAC] 220-110), as well as possible management directives and mitigation measures, to avoid and/or minimize potential take to the maximum extent practicable. As the HPA authority covers all waters of the state, this white paper considers hydraulic project impacts in both freshwater and marine environments. This white paper is one of a suite of white papers being prepared to establish the scientific basis for the HCP and assist WDFW decision-making regarding what specific HPA activities should be covered by the HCP and what minimization and mitigation measures can be implemented to address the potential effects of hydraulic projects. This white paper addresses impacts and mitigation/minimization measures to be applied to the construction and maintenance of shoreline modifications. Species considered for coverage under the HCP (referred to in this white paper as “HCP species”) are listed in Table 1-1. For the purpose of this white paper, some of the HCP species have been grouped where appropriate (each group is separated by gray line in Table 1-1).

Table 1-1. The 52 HCP species addressed in this white paper.

Common Name	Scientific Name	Status[a]	Habitat
Chinook salmon	*Oncorhynchus tshawytscha*	FE/FT/SC	Freshwater, Estuarine, Marine
Coho salmon	*Oncorhynchus kisutch*	FT/FSC	Freshwater, Estuarine, Marine
Chum salmon	*Oncorhynchus keta*	FT/SC	Freshwater, Estuarine, Marine
Pink salmon	*Oncorhynchus gorbuscha*	SPHS	Freshwater, Estuarine, Marine
Sockeye salmon	*Oncorhynchus nerka*	FE/FT/SC	Freshwater, Estuarine, Marine
Steelhead	*Oncorhynchus mykiss*	FE/FT/SC	Freshwater, Estuarine, Marine
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	FSC	Freshwater, Estuarine, Marine
Redband trout	*Oncorhynchus mykiss*	FSC	Freshwater
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	FSC	Freshwater
Bull trout	*Salvelinus confluentus*	FT/SC	Freshwater, Estuarine
Dolly Varden	*Salvelinus malma*	FP	Freshwater, Estuarine
Pygmy whitefish	*Prosopiim coulteri*	FSC/SS	Freshwater
Olympic mudminnow	*Novumbra hubbsi*	SS	Freshwater
Lake chub	*Couesius plumbeus*	SC	Freshwater
Leopard dace	*Rhinichthys falcatus*	SC	Freshwater
Margined sculpin	*Cottus marginatus*	FSC/SS	Freshwater
Mountain sucker	*Catostomus platyrynchus*	SC	Freshwater
Umatilla dace	*Rhinichthys umatilla*	SC	Freshwater
Pacific lamprey	*Lampetra tridentata*	FSC	Freshwater, Estuarine, Marine
River lamprey	*Lampetra ayresi*	FSC/SC	Freshwater, Estuarine, Marine
Western brook lamprey	*Lampetra richardsoni*	FSC	Freshwater
Green sturgeon	*Acipenser medirostris*	FT/FSC/SPHS	Freshwater, Estuarine, Marine
White sturgeon	*Acipenser transmontanus*	SPHS	Freshwater, Estuarine, Marine
Eulachon	*Thaleichthys pacificus*	FC/SC	Freshwater, Estuarine, Marine
Longfin smelt	*Spirinchus thaleichthys*	SPHS	Freshwater, Estuarine, Marine
Pacific sand lance	*Ammodytes hexapterus*	SPHS	Marine & Estuarine
Surf smelt	*Hypomesus pretiosus*	SPHS	Marine & Estuarine
Pacific herring	*Clupea harengus pallasi*	FC/SC	Marine & Estuarine
Lingcod	*Ophiodon elongatus*	SPHS	Marine & Estuarine
Pacific hake	*Merluccius productus*	FSC/SC	Marine & Estuarine
Pacific cod	*Gadus macrocephalus*	FSC/SC	Marine (occ. Estuarine)
Walleye pollock	*Theragra chalcogramma*	FSC/SC	Marine (occ. Estuarine)
Black rockfish	*Sebastes melanops*	SC	Marine & Estuarine
Bocaccio rockfish	*Sebastes paucispinis*	SC	Marine & Estuarine

Table 1-1 (continued). The 52 HCP species addressed in this white paper.

Common Name	Scientific Name	Status[a]	Habitat
Brown rockfish	*Sebastes auriculatus*	SC	Marine & Estuarine
Canary rockfish	*Sebastes pinniger*	SC	Marine & Estuarine
China rockfish	*Sebastes nebulosis*	SC	Marine & Estuarine
Copper rockfish	*Sebastes caurinus*	FSC/SC	Marine & Estuarine
Greenstriped rockfish	*Sebastes elongates*	SC	Marine & Estuarine
Quillback rockfish	*Sebastes maliger*	FSC/SC	Marine & Estuarine
Redstripe rockfish	*Sebastes proriger*	SC	Marine & Estuarine
Tiger rockfish	*Sebastes nigrocinctus*	SC	Marine & Estuarine
Widow rockfish	*Sebastes entomelas*	SC	Marine & Estuarine
Yelloweye rockfish	*Sebastes ruberrimus*	SC	Marine & Estuarine
Yellowtail rockfish	*Sebastes flavidus*	SC	Marine & Estuarine
Olympia oyster	*Ostrea lurida*	SPHS	Marine & Estuarine
Northern abalone	*Haliotis kamtschatkana*	FSC/SC	Marine
Newcomb’s littorine snail	*Algamorda subrotundata*	FSC/SC	Marine
Giant Columbia River limpet	*Fisherola nuttalli*	SC	Freshwater
Great Columbia River spire snail	*Fluminicola columbiana*	FSC/SC	Freshwater
California floater (mussel)	*Anodonta californiensis*	FSC/SC	Freshwater
Western ridged mussel	*Gonidea angulata*	None	Freshwater

Notes: For the purpose of this white paper, some of the HCP species have been grouped when appropriate (each group is separated by a gray line).

[a] Status:

FE=Federal Endangered
FP=Federal Proposed
FT = Federal Threatened
FC = Federal Candidate
FSC = Federal Species of Concern
SC = State Candidate
SS = State Sensitive
SPHS = State Priority Habitat Species

2.0 Objectives

The objectives of this white paper are:

- To compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the creation, construction, maintenance, installation, repair, replacement, modification, and removal (hereafter collectively referred to as construction and repair) of shoreline modification structures.
- To use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely resulting from the construction and repair of shoreline modifications.
- To identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid and/or minimize the risks of incidental take of HCP species.

3.0 Methods

Information presented in this white paper is based primarily on the compilation and synthesis of the best available scientific information related to human impacts on HCP species, their habitats, and associated ecological processes. The methods used here included the acquisition of existing literature, followed by an analysis of impacts based on a review of the literature. The conceptual framework for assessing potential impacts is described in detail in Section 6; below is a discussion of the literature acquisition and review process.

To acquire literature supporting the best available scientific information, an extensive search of the available literature was conducted using the Thomson Scientific Web of Science (Thomson Scientific Web of Science 2007), which has electronic access to more than 8,500 scientific journals encompassing all fields of environmental science. This yielded several hundred relevant publications, most published within the last 10 years. In addition, literature cited in previous white papers and conference proceedings from the last four Puget Sound–Georgia Basin Research Conferences was reviewed to identify relevant "gray literature" sources. The University of Washington School of Aquatic and Fisheries Sciences, Fisheries Research Institute Reports (UW-FRI) database, a database that includes more than 500 reports pertaining to research conducted by Fisheries Research Institute personnel from its inception to the present, was also searched. A thorough search of theses in the Summit system of libraries was performed to locate relevant student work (Summit is a library catalog that combines information from Pacific Northwest academic libraries, including the Orbis and Cascade systems, into a single database available at URL = http://summit.orbiscascade.org/). Finally, because this white paper was prepared by a diverse group of scientists from a wide range of backgrounds, many other primary resources (consultant reports, textbooks, etc.) were found in the personal collection of Herrera staff (the consulting firm working with WDFW to prepare this white paper).

To identify data gaps and evaluate the state of scientific knowledge applicable to the potential impacts of shoreline modification structures on the HCP species and their habitats, the acquired literature was examined to assess the broader issue of how these species use aquatic habitats and how shoreline modifications and their construction alter habitat functions.

Existing literature reviews, peer-reviewed journal articles, books, theses/dissertations, and technical reports were reviewed for information specific to aquatic species and their interaction with jetties, breakwaters, groins, and bank barbs. Through this process, a collection of information was assembled on the life history, habitat uses, and the potential impacts that these structures pose to HCP species.

Reference material from each of the above databases was compiled in an Endnote personal reference database (Endnote version X). Reference types collected and entered into the database included journal articles, reports, web pages, conference proceedings, theses, statutes, books, and book sections. Each entry in the database included descriptive information, including author(s), year, title, volume, pages, publisher, etc. Whenever an electronic copy of the reference material was available, a link between the reference entry and a PDF copy of the reference material was included in the database. If an electronic (.PDF) copy of a reference was not available, a hardcopy of the material was kept on file. All reference materials cited in the literature review were either linked to the reference database or retained in an associated file as a hardcopy.

Endnote X is the industry standard software for organizing bibliographic information. It features a fully searchable and field sortable database that can contain an unlimited number of references. Reference information is entered into the database either by direct import from online databases or by manually entering the reference information into reference type templates. Once all the references were entered, the database was used for organizational and archival purposes. The final database is included as an electronic appendix to this report (Appendix B).

4.0 Hydraulic Project Description

RCW 77.55.011(7) defines a hydraulic project as "the construction or performance of work that will use, divert, obstruct, or change the natural flow or bed of any of the salt or freshwaters of the state." All shoreline modifications that are oriented perpendicular to the direction of dominant transport are considered in this white paper. These modifications include all jetties, groins, bank barbs, and breakwaters.

However, any shoreline structures that intercept and disrupt normal transport processes, even those which may look different than traditional shoreline protection (e.g., beach-access stairways, boat ramps), would be considered shoreline modifications. This includes any bank protection measures that project beyond the ordinary high water mark (OHWM). This may include many bulkheads, in which case these structures should consider both the recommendations made herein and those in Jones & Stokes (2006). These structures that mimic more common shoreline modifications are hereafter called analogs. Typically, these structures would affect the aquatic environment in a similar manner as groins if they do not influence the flow or exchange of groundwater or surface water with the main water body, or act similar to a jetty if they do.

Because shorelines have a specific, environment-dependent meaning in regulatory language, the subactivities treated in this white paper are organized by the environment (i.e., riverine, lacustrine, and marine) in which they are planned. The environments have been separated in a geomorphic sense. That is, riverine environments are those where flow is significant and confined in a single channel or a set of intersecting channels; lacustrine environments are those freshwater bodies surrounded by land; and marine environments are those where physical processes are dominated by tides and/or waves and salinity is at least occasionally important. Marine environments include deltas and confined estuarine embayments (e.g., Willapa Bay, Commencement Bay). These definitions mean that riverine environments may be tidally modulated, and marine environments may have varying salinity.

The mechanisms of impact on HCP species associated with these projects include the impacts from the ongoing presence of these structures, which in most cases is of a permanent nature, and impacts from construction, maintenance, repair, and removal (including demolition) activities that could result in modifications to riverine, lacustrine, and marine shoreline processes and morphology. This includes modifications to hydraulic and geomorphic characteristics, aquatic and riparian vegetation, and water quality that could result in direct and indirect effects on HCP species. It also includes the effects of fish handling, relocation, and exclusion associated with such activities. In addition, shoreline modification structures alter the alongshore sediment transport processes, circulation, and water chemistry far beyond the structure's actual footprint (Jennings et al. 1999; Williams et al. 2001).

For the purposes of this white paper, shoreline modification structures are defined as follows (and as illustrated in Figure 4-1):

- A **jetty** is a structure constructed at navigational channels to prevent sand from depositing in the channel and to provide wave protection for vessels (Dean and Dalrymple 2002).

- A **breakwater** is a structure that is built seaward of the breaker line parallel to the shoreline to protect nearshore infrastructure and prevent shoreline erosion. Breakwaters are often used in series (Dean and Dalrymple 2002).

- A **piling** or **pile** is driven into the stream, lake, or ocean bed to support shoreline infrastructure. It includes both structural and nonstructural pilings.

- **Groins** are common in marine, lacustrine, and riverine environments; are vertical barriers extending perpendicularly from the shore/bank that impede the downdrift/downstream movement of sediment; are often placed in series; and include analogs that function like a groin but are structurally different (e.g., beach access staircases) (WDFW 2003).

- A **bank barb** is a low-elevation structure projecting from a bank and angled upstream to redirect flow away from a stream bank, thereby controlling erosion of the stream bank (WDFW 2003). Bank barbs are often used in series.

This white paper also summarizes the provisions in the Washington Administrative Code (WAC 220-110) that apply to shoreline modification projects in both fresh and saltwater environments. These are presented in Table 4-1. Table 4-1 also identifies activities that may be related to the topics covered in this white paper, but that are not necessarily an implicit component of the activity as specified in the WAC.

Table 4-1. WAC sections potentially applicable to shoreline modifications.

Freshwater WACs (direct & indirect applicability)	Saltwater WACs (direct & indirect applicability)
220-110-050 (FW banks) 220-110-080 (channel change) 220-110-120 (temporary bypass) 220-110-130* (dredging) 220-110-140* (gravel removal) 220-110-150 (LWD) 220-110-170 (outfalls) 220-110-223 (lake banks)	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (nonSFRM bank) 220-110-285 (SFRM bank) 220-110-320* (dredging)

Note: * indicates that the activity may be related to the topics covered in this white paper, but it is not necessarily an implicit component of the activity as specified in the WAC.
LWD = large woody debris; SFRM = single-family residential marine.

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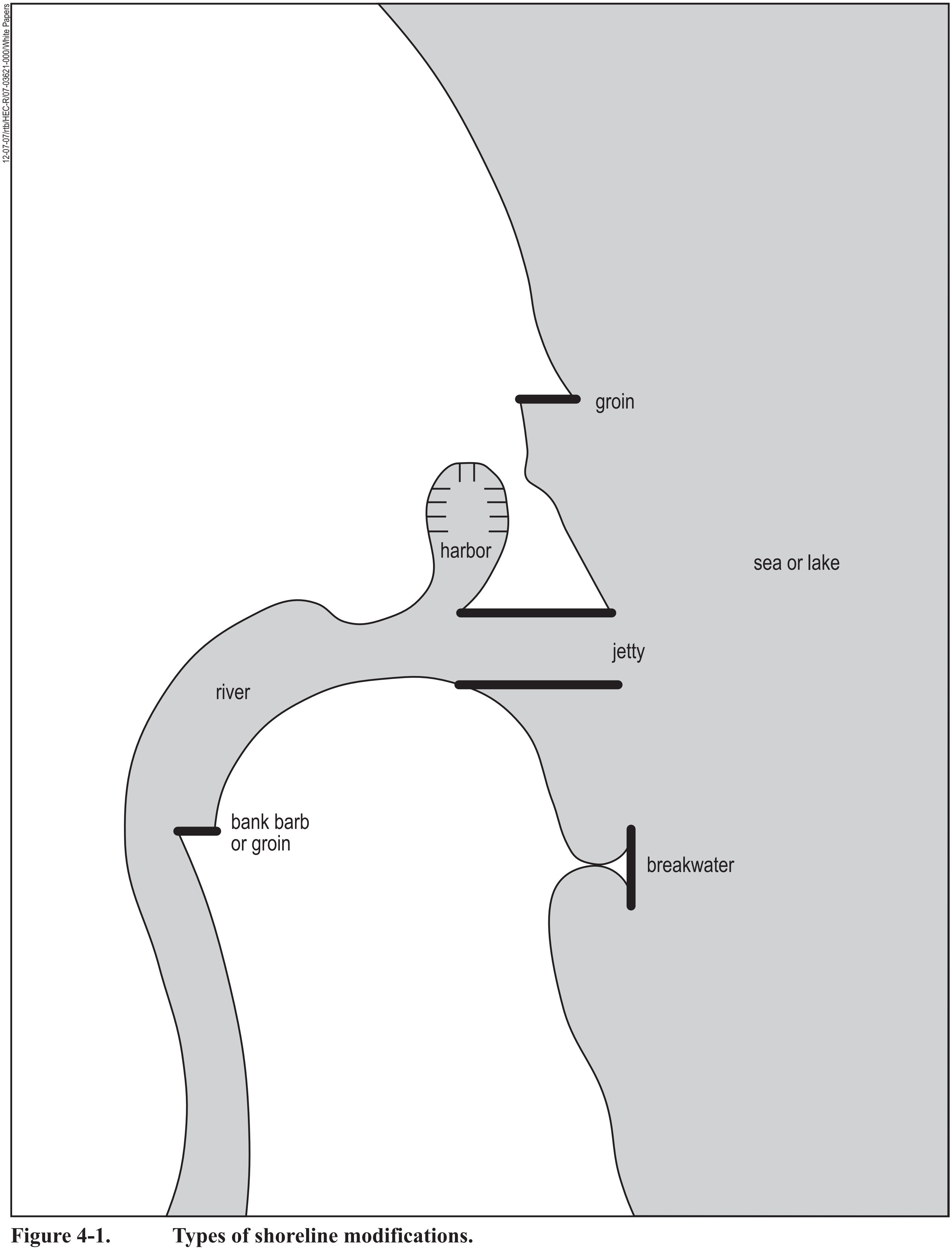


Figure 4-1. **Types of shoreline modifications.**

5.0 Potentially Covered Species and Habitat Use

This white paper identifies what is known about how activities associated with the construction and repair of jetties, breakwaters, groins, bank barbs, and their analogs can pose risks of take for the 52 HCP species. To understand species-specific impacts, it is important to understand the geographic distribution and habitat use of these species. Table 5-1 lists the scientific name, Water Resource Inventory Area (WRIA) of occurrence, tidal reference area, and the reproductive patterns and habitat requirements of each of these HCP species. Through the identification of species-specific habitat needs, the risk of take associated with each mechanism of impact related to the construction and repair of jetties, breakwaters, groins, bank barbs, and their analogs (e.g., beach-access stairways, boat ramps, protruding seawalls) can be identified. Once the potential for take has been identified, it can be avoided. If unavoidable, the risk of take can be minimized by design and/or through conservation and protection measures. (See Section 9 [*Potential Risk of Take*] and the exposure-response matrices for each of these species, presented in Appendix A.)

Table 5-1. Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Chinook salmon	*Oncorhynchus tshawytscha*	01–42, 44–50	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes eight evolutionarily significant units (ESUs) of Chinook salmon in Washington: (1) Upper Columbia River spring-run; (2) Snake River spring/summer run; (3) Snake River fall-run; (4) Puget Sound; (5) lower Columbia River; (6) Washington coast; (7) Mid-Columbia River spring-run; and (8) Upper Columbia River summer/fall-run. Chinook salmon exhibit one of two life-history types, or races: the stream-type and the ocean-type. Stream-type Chinook tend to spend 1 (or less frequently 2) years in freshwater environments as juveniles prior to migrating to salt water as smolts. Stream-type Chinook are much more dependent on freshwater stream ecosystems than ocean-type Chinook. Stream-type Chinook do not extensively rear in estuarine and marine nearshore environments; rather, they head offshore and begin their seaward migrations. Ocean-type Chinook enter salt water at one of three phases: immediate fry migration soon after yolk is absorbed, fry migration 60–150 days after emergence, and fingerling migrants that migrate in the late summer or fall of their first year. Ocean-type Chinook are highly dependent on estuarine habitats to complete their life history. Chinook generally feed on invertebrates but become more piscivorous with age. **Reproduction/Life History** Chinook runs are designated on the basis of adult migration timing: • Spring-run Chinook: Tend to enter fresh water as immature fish, migrate far upriver, and finally spawn in the late summer and early autumn. • Fall-run Chinook: Enter fresh water at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry. • Spring Chinook: Spawning occurs from mid-July to mid-December, and incubation lasts approximately 1.5–7 months, depending on temperature. Emergence follows, 6–8 months from fertilization. • Fall Chinook: Spawning occurs from late October to early December, with incubation occurring for 1–6 months. Emergence follows, approximately 6 months after fertilization. (Healey 1991; Myers et al. 1998; WDNR 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Coho salmon	*Oncorhynchus kisutch*	01–42, 44–48, 50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of coho salmon in Washington: (1) Lower Columbia River; (2) Southwest Washington; (3) Puget Sound and Strait of Georgia; and (4) Olympic Peninsula. This species is found in a broader diversity of habitats than any of the other native anadromous salmonids. Fry feed primarily on aquatic insects and prefer pools and undercut banks with woody debris; adults feed on herring and other forage fish. **Reproduction/Life History** Coho adults spawn from September to late January, generally in the upper watersheds in gravel free of heavy sedimentation. Developing young remain in gravel for up to 3 months after hatching. Fry emerge from early March to late July. Coho rear in fresh water for 12–18 months before moving downstream to the ocean in the spring. Coho spend between 1 and 2 years in the ocean before returning to spawn. (Groot and Margolis 1991; Murphy and Meehan 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Chum salmon	*Oncorhynchus keta*	01, 03–05, 07–29	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of chum salmon in Washington: (1) Hood Canal summer run; (2) Columbia River; (3) Puget Sound/Strait of Georgia; and (4) Pacific Coast. Little is known about their ocean distribution; maturing individuals that return to Washington streams have primarily been found in the Gulf of Alaska. Chum migrate into rivers and streams of Washington coast, Hood Canal, Strait of Juan de Fuca, Puget Sound, and the Columbia River basin to spawn, but their range does not extend upstream above the Dalles Dam in the Columbia River. Fry feed on chironomid and mayfly larvae, as well as other aquatic insects, whereas juvenile fish in the estuary feed on copepods, tunicates, and euphausiids. **Reproduction/Life History** Chum salmon have three distinct run times: summer, fall and winter. Summer chum begin their upstream migration and spawn from mid-August through mid-October, with fry emergence ranging from the beginning of February through mid-April. Chum fry arrive in estuaries earlier than most salmon, and juvenile chum reside in estuaries longer than most other anadromous species. Chum salmon rear in the ocean for the majority of their adult lives. Fall chum adults enter the rivers from late October through November and spawn in November and December. Winter chum adults migrate upstream from December through January and spawn from January through February. Fall and winter chum fry emerge in March and April and quickly emigrate to the estuary. Chum salmon utilize the low-gradient (from 1–2 percent grade), sometimes tidally influenced lower reaches of streams for spawning. (Healey 1982; Johnson et al. 1997; Quinn 2005; Salo 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pink salmon	*Oncorhynchus gorbuscha*	01, 03–05, 07, 09–11, 16–19, 21	1–13	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two ESUs of pink salmon in Washington, neither of which is listed: (1) Odd-year; and (2) Even-year. The most abundant species of salmon, with 13 stocks identified in Washington. They are the smallest of the Pacific salmon and mature and spawn on a 2-year cycle in Washington (primarily spawning during odd years). Adults are opportunistic feeders in marine habitat, foraging on a variety of forage fish, crustaceans, ichthyoplankton, and zooplankton. Juveniles primarily feed on small crustaceans such as euphausiids, amphipods, and cladocerans. **Reproduction/Life History** Pink salmon will spawn in rivers with substantial amounts of silt. Spawning occurs from August through October. Fry emerge from their redds in late February to early May, depending on water temperature, and migrate downstream to the estuary within 1 month. Juveniles remain in estuarine or nearshore waters for several months before moving offshore as they migrate to the Pacific Ocean, where they remain approximately 1 year until the next spawning cycle. (Hard et al. 1996; Heard 1991; WDNR 2005, 2006a)
Sockeye salmon	*Oncorhynchus nerka*	01, 03–05, 07–11, 16, 19–22, 25–33, 35–37, 40, 41, 44–50	5, 8, 14	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes seven ESUs of sockeye salmon in Washington: (1) Snake river; (2) Ozette Lake; (3) Baker river; (4) Okanogan River; (5) Quinault Lake; (6) Lake Pleasant; and (7) Lake Wenatchee. WDFW recognizes an additional sockeye salmon stock in the Big Bear Creek drainage of Lake Washington. Kokanee (landlocked sockeye) occur in many lakes, with the larger populations in Banks and Loon lakes in eastern Washington and Lake Whatcom and Lake Washington-Sammamish in western Washington. Juveniles feed on zooplankton, and adults primarily feed on fish, euphausiids, and copepods. **Reproduction/Life History** Spawn in shallow, gravelly habitat in rivers and lakes during August to October. Juvenile sockeye rear in lakes for 1–2 years before migrating to the ocean. Emergence occurs within 3–5 months. (Gustafson et al. 1997; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Steelhead	*Oncorhynchus mykiss*	01, 03–05, 07–12, 14, 15, 17–41, 44–50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes 15 Distinct Population Segments (DPSs) of steelhead, seven of which occur in Washington. During their ocean phase, steelhead are generally found within 10 and 25 miles of the shore; steelhead remain in the marine environment 2–4 years before returning to fresh water to spawn. Most steelhead spawn at least twice in their lifetimes. Escape cover, such as logs, undercut banks, and deep pools, is important for adult and young steelhead in the freshwater systems. The coastal west-side streams typically support more winter steelhead populations. **Reproduction** A summer spawning run enters fresh water in August and September, and a winter run occurs from December through February. Summer steelhead usually spawn farther upstream than winter populations and dominate inland areas such as the Columbia Basin. Spawning occurs from March to April for both winter and summer run steelhead. After hatching and emergence (approximately 3 months), juveniles establish territories, feeding on microscopic aquatic organisms and then larger organisms such as isopods, amphipods, and aquatic and terrestrial insects. Steelhead rear in fresh water for up to 4 years before migrating to sea. (Busby et al. 1996; McKinnell et al. 1997; WDNR 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	01–05, 07–30	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries has recognized three ESUs in Washington: (1) Puget Sound; (2) Olympic Peninsula; (3) Southwestern Washington/Columbia River. USFWS has assumed sole jurisdiction for this species. No coastal cutthroat trout DPSs are listed under the ESA in Washington. Coastal cutthroat trout exhibit varied life-history forms including: • Resident (stays in streams after rearing in their natal streams) – Resident coastal cutthroat trout utilize small headwater streams for all of their life stages. • Fluvial (migrates to larger rivers after rearing in their natal streams). • Adfluvial (migrates to lakes after rearing in their natal streams). • Anadromous (utilizes estuaries and nearshore habitat but has been caught offshore). Juveniles of all life forms feed primarily on aquatic invertebrates but are opportunistic feeders; adults tend to feed on smaller fish, amphibians, and crustaceans while foraging within the nearshore environment. **Reproduction/Life History** Coastal cutthroat trout are repeat spawners, and juveniles typically rear in the natal streams for up to 2 years. Spawning occurs from late December to February, with incubation lasting approximately 2–4 months. Emergence occurs after 4 months. (Johnson et al. 1999; Pauley et al. 1988; WDNR 2006a)
Redband trout	*Oncorhynchus mykiss gardnerii*	37–40, 45–49, 54–57	NA	**General Information (Habitats and Feeding)** Redband trout is a subspecies of rainbow trout found east of the Cascade Mountains, which prefer cool water that is less than 70°F (21°C), and occupy streams and lakes with high amounts of dissolved oxygen. Their food primarily consists of Daphnia and chironomids as well as fish eggs, fish, and insect larvae and pupae. **Reproduction/Life History** Spawn in streams with clean, small gravel from March through May. Incubation takes approximately 1–3 months, with emergence occurring between June and July. (USFS 2007)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	37–39, 44–55, 58–62	NA	**General Information (Habitats and Feeding/Life-history Types)** Cutthroat trout tend to thrive in streams with extensive pool habitat and cover. The westslope is a subspecies of cutthroat trout with three possible life forms: • Adfluvial (migrates to lakes) • Fluvial (migrates to larger rivers) • Resident (stays in streams). The headwater tributaries used by resident cutthroat are typically cold, nutrient-poor waters that result in slow growth. Fluvial and adfluvial forms can exhibit more growth due to warmer water temperatures and nutrient availability. Fry feed on zooplankton, and fingerlings feed on aquatic insect larvae. Adults feed on terrestrial and aquatic insects. **Reproduction/Life History** Spawning: all three life forms spawn in small gravel substrates of tributary streams in the spring (March to July) when water temperature is about 50°F (10°C); incubation occurs during April to August, and emergence occurs from May through August. Fry spend 1–4 years in their natal stream before migrating to their ultimate habitat. (Liknes and Graham 1988; Shepard et al. 1984; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Bull trout	*Salvelinus confluentus*	01, 03–05, 07–23, 26, 27, 29–41, 44–55, 57–62	All	**General Information (Habitats and Feeding/Life-History Types)** Widely distributed in Washington; exhibit four life-history types: • Resident (stays in streams after rearing in their natal streams) • Fluvial (migrates to larger rivers after rearing in their natal streams) • Adfluvial (migrates to lakes after rearing in their natal streams) • Anadromous (bull trout in the nearshore ecosystem rely on estuarine wetlands and favor irregular shorelines with unconsolidated substrates). Young of the year occupy side channels, with juveniles in pools, runs, and riffles; adults occupy deep pools. Juvenile diet includes larval and adult aquatic insects; subadults and adults primarily feed on fish. **Reproduction/Life History** The migratory forms of bull trout, such as anadromous, adfluvial, and fluvial, move upstream by early fall to spawn in September and October (November at higher elevations). Although resident bull trout are already in stream habitats, they move upstream looking for suitable spawning habitat. They prefer clean, cold water (50°F [10°C]) for spawning. Colder water (36–39°F [2–4°C]) is required for incubation. Preferred spawning areas often include groundwater infiltration. Extended incubation periods (up to 220 days) make eggs and fry particularly susceptible to increases in fine sediments. Bull trout typically rear in natal streams for 2–4 years, although resident fish may remain in these streams for their entire lives; multiple life-history forms may occur in the same habitat environments. (Goetz et al. 2004; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Dolly Varden	*Salvelinus malma*	01, 03, 05, 07, 17–22, 24	6–10, 14–17	**General Information (Habitats and Feeding/Life-History Types)** Species restricted to coastal areas and rivers that empty into them. Juveniles extensively use instream cover; while in the marine systems, they use beaches of sand and gravel. Prefer pool areas and cool temperatures. Feed opportunistically on aquatic insects, crustaceans, salmon eggs, and fish. Closely related to bull trout and exhibit the same life-history traits. Four life-history types occur: • Resident (stays in streams after rearing in their natal streams) • Fluvial (migrates to larger rivers after rearing in their natal streams) • Adfluvial (migrates to lakes after rearing in their natal streams) • Anadromous (migrates to marine waters after rearing in their natal streams). **Reproduction/Life History** Spawn and rear in streams from mid-September through November. Incubation lasts approximately 130 days. Juveniles can spend 2–4 years in their natal streams before migration to marine waters. (Leary and Allendorf 1997; WDNR 2005; Wydoski and Whitney 2003)
Pygmy whitefish	*Prosopium coulteri*	08, 19, 39, 47, 49, 53, 55, 58, 59, 62	NA	**General Information (Habitats and Feeding)** In Washington, pygmy whitefish occur at the extreme southern edge of their natural range; pygmy whitefish were once found in at least 15 Washington lakes but have a current distribution in only nine. They occur most often in deep, oligotrophic lakes with temperatures less than 50°F (10°C), where they feed on zooplankton, such as cladocerans, copepods, and midge larvae. **Reproduction/Life History** Pygmy whitefish spawn in streams or lakes from July through November. They prefer pools, shallow riffles, and pool tail-outs when spawning in streams. Lake spawning by pygmy whitefish occurs at night. Spawning occurs by scattering their eggs over coarse gravel. Incubation and emergence timing are unknown, but eggs are believed to hatch in the spring. (Hallock and Mongillo 1998; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Olympic mudminnow	*Novumbra hubbsi*	08–24	NA	**General Information (Habitats and Feeding)** Occur in the southern and western lowlands of the Olympic Peninsula, the Chehalis River drainage, lower Deschutes River drainage, south Puget Sound lowlands west of the Nisqually River, and in King County. They are generally found in quiet water with mud substrate, preferring bogs and swamps with dense aquatic vegetation. Mudminnows feed on annelids, insects, and crustaceans. **Reproduction/Life History** Adults spawn from November through June (peaking in April and May). Females deposit eggs onto vegetation where fry remain firmly attached for approximately 1 week after hatching. Incubation lasts approximately 8-10 days. (Harris 1974; Mongillo and Hallock 1999; WDNR 2005, 2006a)
Lake chub	*Couesius plumbeus*	48, 61; other locations unknown	NA	**General Information (Habitats and Feeding)** Bottom dwellers inhabiting a variety of habitats in lakes and streams, but are known to prefer small, slow streams. In Washington, they are known only from the northeastern part of the state (small streams and lakes in Okanogan and Stevens counties). Juveniles feed on zooplankton and phytoplankton, whereas adults primarily feed on insects. **Reproduction/Life History** Lake chub move into shallow areas on rocky and gravelly substrates in tributary streams of lakes or lakeshores during the spring to spawn when water temperatures are between 55 and 65°F (13 and 18°C). The eggs are broadcast over large rocks and then settle into the smaller substrate, hatching after approximately 10 days. (WDNR 2005; Wydoski and Whitney 2003)
Leopard dace	*Rhinichthys falcatus*	25–31, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** In Washington, leopard dace inhabit the bottoms of streams and small to mid-sized rivers, specifically the Columbia, Snake, Yakima, and Similkameen Rivers, with velocities less than 1.6 ft/sec (0.5 m/sec); prefer gravel and small cobble substrate covered by fine sediment with summer water temperatures ranging between 59 and 64°F (15 and 18°C). Juveniles feed primarily on aquatic insects; adult leopard dace consume terrestrial insects. **Reproduction/Life History** Breeding habitat for dace generally consists of the gravel or cobble bottoms of shallow riffles; leopard dace breed in slower, deeper waters than the other dace species. The spawning period for dace is from May through July. The eggs adhere to rocky substrates. Fry hatch approximately 6–10 days after fertilization, and juveniles spend 1–3 months rearing in shallow, slow water. (WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Margined sculpin	*Cottus marginatus*	32, 35	NA	**General Information (Habitats and Feeding)** Endemic to southeastern Washington (smaller tributary streams of the Walla Walla and Tucannon River drainages) where habitat is in deeper pools and slow-moving glides in headwater tributaries with silt and small gravel substrate. They prefer cool water less than 68°F (20°C) and avoid high-velocity areas. Food includes immature aquatic insects, invertebrates, small fish, and eggs. **Reproduction/Life History** Spawning occurs in May and June primarily under rocks, root wads, or logs. The female deposits a mass of adhesive eggs in the nest, which is guarded by the male. Incubation duration unknown. (Mongillo and Hallock 1998; WDNR 2005; Wydoski and Whitney 2003)
Mountain sucker	*Catostomus platyrynchus*	25–35, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** Distribution restricted to Columbia River system. Found in clear, cold mountain streams less than 40 ft wide and in some lakes; prefer deep pools in summer with moderate current. Food consists of algae and diatoms. Juveniles prefer slower side channels or weedy backwaters. **Reproduction/Life History** Males reach sexual maturity in 2–3 years and females in 4 years. Spawning in June and July when water temperatures exceed 50°F (10°C). Spawning occurs in gravelly riffles of small streams when suckers move into those reaches to feed on algae. Spawning likely occurs at night when water temperatures are in a range of 51–66°F (10.5–19°C). Fertilized eggs fall into and adhere to the spaces between the gravel composite. Incubation period lasts approximatley 8-14 days. (Wydoski and Whitney 2003)
Umatilla dace	*Rhinichthys umatilla*	31, 36–41, 44–50, 59–61	NA	**General Information (Habitats and Feeding)** Umatilla dace are benthic fish found in relatively productive, low-elevation streams with clean substrates of rock, boulders, and cobbles in reaches where water velocity is less than 1.5 ft/sec (0.5 m/sec). Feeding is similar to that described for leopard dace. Juveniles occupy streams with cobble and rubble substrates, whereas adults occupy deeper water habitats. **Reproduction/Life History** Spawning behaviors are similar to those described for leopard dace, with spawning primarily occurring from early to mid-July. (WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific lamprey	*Lampetra tridentata*	01, 03–05, 07–35, 37–40, 44–50	All	**General Information (Habitats and Feeding)** Found in most large coastal and Puget Sound rivers and Columbia, Snake, and Yakima river basins. The larvae are filter feeders, residing in mud substrates and feeding on algae and other organic matter for at least 5 years. **Reproduction/Life History** From July through October, maturing Pacific lamprey enter fresh water and gradually move upstream to spawn the following spring. The nest usually consists of a shallow depression built in gravel and rock substrates. Eggs hatch in 2–4 weeks, with newly hatched larvae remaining in the nest for 2–3 weeks before moving downstream as larvae (ammocoetes). Juveniles migrate to the Pacific Ocean 4–7 years after hatching and attach to fish in the ocean for 20–40 months before returning to rivers to spawn. (WDNR 2005; Wydoski and Whitney 2003)
River lamprey	*Lampetra ayresi*	01, 03, 05, 07–16, 20–40	1–9, 11–17	**General Information (Habitats and Feeding)** Detailed distribution records are not available for Washington, but they are known to inhabit coastal rivers, estuaries, and the Columbia River system. They have also been observed in Lake Washington and its tributaries. In the marine system, river lamprey inhabit nearshore areas. Adults are anadromous living in the marine system as parasites on fish. Adult river lamprey are believed to occupy deep portions of large river systems. The larvae feed on microscopic plants and animals. **Reproduction/Life History** Adults migrate back into fresh water in the fall. Spawning occurs in winter and spring. Eggs hatch in 2–3 weeks after spawning. Juveniles are believed to migrate from their natal rivers to the Pacific Ocean several years after hatching; adults spend 10–16 weeks between May and September in the ocean before migrating to fresh water. (WDNR 2005; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Western brook lamprey	*Lampetra richardsoni*	01, 03, 05, 07–14, 16, 20–40	NA	**General Information (Habitats and Feeding)** Found in small coastal and Puget Sound rivers and lower Columbia and Yakima river basins; spends entire life in fresh water. Adults are found in cool water (52–64°F [11–17.8°C]) on pebble/rocky substrate. Larvae (ammocoetes) are filter feeders, consuming primarily diatoms. Adults do not feed and die within a month of spawning. **Reproduction/Life History** Spawning generally occurs from April through July, with adults creating nests in coarse gravel at the head of riffles. Eggs hatch after about 10 days in water between 50 and 60°F (10 and 16°C). Within 30 days of hatching, ammocoetes emerge from the nests and move to the stream margin, where they burrow into silty substrates. Larvae remain in the stream bottom—apparently moving little—for approximately 4–6 years. (Wydoski and Whitney 2003)
Green sturgeon	*Acipenser medirostris*	22, 24, 28	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two DPSs of green sturgeon, both of which can be found in Washington. The southern DPS is listed as threatened and the northern DPS is a species of concern. Habits and life history not well known. Washington waters with green sturgeon populations include the Columbia River, Willapa Bay, and Grays Harbor, in addition to marine waters. They spend much of their life in marine nearshore waters and estuaries feeding on fishes and invertebrates. **Reproduction/Life History** Spawning generally occurs in spring in deep, fast-flowing sections of rivers. Spawning habitat includes cobble or boulder substrates. Green sturgeon move upstream during spring to spawn and downstream during fall and winter. Large eggs sink to bottom. (Adams et al. 2002; Emmett et al. 1991; Kynard et al. 2005; Nakamoto and Kisanuki 1995; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
White sturgeon	*Acipenser transmontanus*	01, 03, 05–22, 24–37, 40–42, 44–61	All	**General Information (Habitats and Feeding)** Found in marine waters and major rivers in Washington, including the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. In marine environments, adults and subadults use estuarine and marine nearshore habitats, including some movement into intertidal flats to feed at high tide. Some landlocked populations exist behind dams on the Columbia River. Juveniles feed on mysid shrimp and amphipods; large fish feed on variety of crustaceans, annelid worms, mollusks, and fish. **Reproduction/Life History** Spawn in deep, fast-flowing sections of rivers (prefer swift [2.6–9.2 ft/sec (0.8–2.8 m/sec)] and deep [13–66 ft (4–20 m)] water) on bedrock, cobble, or boulder substrates. Spawning occurs from April through July, with incubation lasting approximately 7 days and emergence following in another 7 days. (Emmett et al. 1991; WDNR 2005; Wydoski and Whitney 2003)
Eulachon	*Thaleichthys pacificus*	01–29 (mouths of major rivers)	14–17	**General Information (Habitats and Feeding)** Eulachon occur from northern California to southwestern Alaska in offshore marine waters. They are plankton-feeders, eating crustaceans such as copepods and euphausiids; larvae and post larvae eat phytoplankton and copepods. They are an important prey species for fish, marine mammals, and birds. **Reproduction/Life History** Spawn in tidal portions of rivers in spring when water temperature is 40–50°F (4–10°C), generally from March through May; use a variety of substrates, but sand and gravel are most common. Eggs stick to substrate and incubation ranges from 20–40 days (dependent on temperature). Larvae drift downstream to salt water where juveniles rear in nearshore marine areas. (Howell et al. 2001; Langer et al. 1977; Lewis et al. 2002; WDFW 2001; WDNR 2005; Willson et al. 2006)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Longfin smelt	*Spirinchus thaleichthys*	01–03, 05–17, 22 and 24	1–9, 15–17	**General Information (Habitats and Feeding)** Marine species that spawns in streams not far from marine waters. They are anadromous, with some populations in Lake Washington that spawn in tributaries, including the Cedar River. Juveniles use nearshore habitats and a variety of substrates; juveniles feed on zooplankton. Adults feed on copepods and euphausiids. Most adults die after spawning. **Reproduction** Spawn in coastal rivers from October through December. Lake Washington populations spawn from January through April. Eggs hatch in approximately 40 days and the larvae drift downstream to salt water. (Gotthardt 2006; WDNR 2005; Wydoski and Whitney 2003)
Pacific sand lance	*Ammodytes hexapterus*	NA	All	**General Information (Habitats and Feeding)** Widespread in Puget Sound, Strait of Juan de Fuca, and coastal estuaries. Schooling plankton feeders. Adults feed during the day and burrow into the sand at night. **Reproduction/Life History** Spawn on sand and beaches with gravel up to 1-inch in diameter at tidal elevations of +4–5 ft (+1.5 meters) to approximately the mean higher high water (MHHW) line from November through February. Emergence occurs from January to April. Larvae and young rear in bays and nearshore areas. (Garrison and Miller 1982; Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997a)
Surf smelt	*Hypomesus pretiosus*	NA	All	**General Information (Habitats and Feeding)** Schooling plankton-feeding forage fish. They feed on a variety of zooplankton, planktonic crustaceans, and fish larvae. Adult surf smelt are pelagic but remain in nearshore habitats. Juveniles rear in nearshore areas, and adults form schools offshore; feed on planktonic organisms. Also an important forage fish. **Reproduction/Life History** Spawning occurs year-round in north Puget Sound, fall and winter in south Puget Sound, and summer along the coast. They spawn at the highest tides during high slack tide on coarse sand and pea gravel. Incubation is 2–5 weeks. Emergence varies with season: 27–56 days in winter, 11–16 days in summer. (Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997c)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific herring	*Clupea harengus pallasi*	NA	1, 2, 4, 5, 8–13, 16, 17	**General Information (Habitats and Feeding)** Eighteen separate stocks in Puget Sound. Widely distributed throughout Puget Sound and coastal wetlands and estuaries. Pacific herring adults feed on small fish, copepods, decapod crab larvae, and euphausiids. Juveniles feed primarily on euphausiids, copepods, and small crustacean larvae. Are also an important forage fish. **Reproduction/Life History** Utilize intertidal and subtidal habitats (between 0 and -40 ft [0 and -12.2 m] mean lower low water [MLLW]) for spawning and juvenile rearing; spawning also occurs above MLLW. Spawning occurs from late January to early April. Eggs are adhered to eelgrass, kelp, seaweed, and sometimes on pilings. Eggs hatch after approximately 10 days. Larvae are pelagic. (Nightingale and Simenstad 2001b; Penttila 2000; Simenstad et al. 1979; WDFW 1997b)
Lingcod	*Ophiodon elongatus*	NA	All	**General Information (Habitats and Feeding)** The lingcod is a large top-level carnivore fish found throughout the West Coast of North America. Adult lingcod have a relatively small home range. Juveniles prefer sand habitats near the mouths of bays and estuaries, while adults prefer rocky substrates. Larvae and juveniles are generally found in upper 115 ft (35 m) of water. Adults prefer slopes of submerged banks with macrophytes and channels with swift currents. Larvae feed on copepods and amphipods; juveniles feed on small fishes; and adults on fish, squid, and octopi. **Reproduction/Life History** Spawn in shallow water and intertidal zone from January through late March. Egg masses adhere to rocks, and incubation is from February to June. Larvae spend 2 months in pelagic nearshore habitat. (Adams and Hardwick 1992; Emmett et al. 1991; Giorgi 1981; NMFS 1990; NRC 2001)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific cod	*Gadus macrocephalus*	NA	All	**General Information (Habitats and Feeding)** Pacific cod are widely distributed in relatively shallow marine waters throughout the northern Pacific Ocean (Washington's inland marine waters are considered the southern limit of populations). Adults and large juveniles are found over clay, mud, and coarse gravel bottoms; juveniles use shallow vegetated habitats such as sand-eelgrass. Feed opportunistically on invertebrates (worms, crabs, shrimp) and fishes (sand lance, pollock, flatfishes). Larvae feed on copepods, amphipods, and mysids. **Reproduction/Life History** Broadcast spawners during late fall through early spring. Eggs sink and adhere to the substrate. Incubate for 1–4 weeks, and larvae spend several months in the water column. Juvenile cod metamorphose and settle to shallow vegetated habitats. (Albers and Anderson 1985; Bargmann 1980; Dunn and Matarese 1987; Garrison and Miller 1982; Hart 1973; Nightingale and Simenstad 2001b; NMFS 1990; NRC 2001)
Pacific hake	*Merluccius productus*	NA	All	**General Information (Habitats and Feeding)** Pacific hake are schooling fish. The coastal stock of hake is migratory; Puget Sound stocks reside in estuaries and rarely migrate. Larvae feed on calanoid copepods; juveniles and small adults feed on euphasiids; adults eat amphipods, squid, herring, and smelt. **Reproduction/Life History** Puget Sound spawning occurs from March through May at mid-water depths of 50–350 ft (15–90 m); may spawn more than once per season. Eggs and larvae are pelagic. (Bailey 1982; McFarlane and Beamish 1986; NMFS 1990; NRC 2001; Quirollo 1992)
Walleye pollock	*Theragra chalcogramma*	NA	All	**General Information (Habitats and Feeding)** Widespread species in northern Pacific. Washington is the southern end of their habitat. Larvae and small juveniles are found at 200-ft (60-m) depth; juveniles use nearshore habitats of a variety of substrates. Juveniles feed on small crustaceans, adults feed on copepods, euphausiids, and young pollock. **Reproduction/Life History** Broadcast spawning occurs from February through April. Eggs are suspended at depths ranging from 330–1,320 ft (100–400 m). Pelagic larvae settle near the bottom and migrate to inshore, shallow habitats for their first year. (Bailey et al. 1999; Garrison and Miller 1982; Livingston 1991; Miller et al. 1976; NRC 2001)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Black rockfish	*Sebastes melanops*	NA	All	**General Information (Habitats and Feeding)** Adults prefer deep and shallow rock substrates in summer, deeper water in winter. Kelp and eelgrass are preferred habitat for juveniles that feed on nekton and zooplankton. Adults feed on amphipods, crabs, copepods, and small fish. **Reproduction/Life History** Spawning occurs from February through April; ovoviviparous incubation as with other rockfish species. Larvae are planktonic for 3–6 months, where they are dispersed by currents, advection, and upwelling. They begin to reappear as young-of-the-year fish in shallow, nearshore waters. (Kramer and O'Connell 1995; WDNR 2006a)
Bocaccio rockfish	*Sebastes paucispinis*	NA	All	**General Information (Habitats and Feeding)** Adults semidemersal in shallow water over rocks with algae, eelgrass, and floating kelp. Larvae feed on diatoms; juveniles feed on copepods and euphausiids. **Reproduction/Life History** Ovoviviparous spawning occurs year-round, with incubation lasting 40–50 days. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kramer and O'Connell 1995; MBC Applied Environmental Sciences 1987; NRC 2001; Sumida and Moser 1984)
Brown rockfish	*Sebastes auriculatus*	NA	All	**General Information (Habitats and Feeding)** Utilize shallow-water bays with natural and artificial reefs and rock piles; estuaries used as nurseries; can tolerate water temperatures to at least 71°F (22°C); eat small fishes, crabs, and isopods. **Reproduction/Life History** Spawning occurs from March through June. Larvae are released from the female into the pelagic environment in May and June (ovoviviparous incubation). Larvae live in the upper zooplankton layer for up to 1 month before they metamorphose into pelagic juveniles. The pelagic juveniles spend 3–6 months in the water column as plankton. They then settle in shallow water nearshore, later migrating to deeper water. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Stein and Hassler 1989)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Canary rockfish	*Sebastes pinniger*	NA	All	**General Information (Habitats and Feeding)** Adults use sharp drop-offs and pinnacles with hard bottoms; often associated with kelp beds; feed on krill and occasionally on fish. Adults are mostly found at depths of 260–660 ft (80–200 meters) (with two recorded at 2,750 ft [838 meters]), tending to collect in groups around pinnacles and similar high-relief rock formations, especially where the current is strong. Young canary rockfish live in relatively shallow water, moving to deeper water as they mature. Juveniles feed on small crustacea such as krill larvae (and eggs), copepods, and amphipods, while adults eat krill and small fish. **Reproduction/Life History** Spawning is ovoviviparous and occurs from January through March. Larvae and juveniles are pelagic. (Boehlert 1980; Boehlert and Kappenman 1980; Boehlert et al. 1989; Hart 1973; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Sampson 1996)
China rockfish	*Sebastes nebulosis*	NA	All	**General Information (Habitats and Feeding)** Occur inshore and on open coast in sheltered crevices. Feed on crustacea (brittle stars and crabs), octopi, and fish. Juveniles are pelagic, but the adults are sedentary associating with rocky reefs or cobble substrates. **Reproduction/Life History** Spawning occurs from January through July; ovoviviparous incubation as with other rockfish species. Individual China rockfish spawn once a year. Larvae settle out of the plankton between 1 and 2 months after release. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Rosenthal et al. 1988)
Copper rockfish	*Sebastes caurinus*	NA	All	**General Information (Habitats and Feeding)** Occur both inshore and on open coast; adults prefer rocky areas in shallower water than other rockfish species. Juveniles use shallow and nearshore macrophytes and eelgrass habitat; feed on crustaceans, fish, and mollusks. **Reproduction/Life History** Spawning occurs from March through May, with ovoviviparous incubation from April to June. Larvae are pelagic in deeper water before moving inshore. Newly spawned fish begin settling near the surface around large algae canopies or eelgrass, when available, or closer to the bottom when lacking canopies. (Eschmeyer et al. 1983; Haldorson and Richards 1986; Kramer and O'Connell 1995; Matthews 1990; NRC 2001; Stein and Hassler 1989)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Greenstriped rockfish	*Sebastes elongates*	NA	All	**General Information (Habitats and Feeding)** Adults found in benthic and mid-water columns. They live at between 330 and 825 ft (100 and 250 m). As they age, greenstriped rockfish move to deeper water. They are solitary and are often found resting on the seafloor and living among cobble, rubble, or mud. Adults feed on euphausiids, small fish, and squid. **Reproduction/Life History** From 10,000 to over 200,000 eggs are produced by the females each season by ovoviviparous spawning. Greenstriped rockfish release one brood of larvae in Washington. Larval release varies, occurring generally from January through July, depending on geographic location. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001)
Quillback rockfish	*Sebastes maliger*	NA	All	**General Information (Habitats and Feeding)** Shallow-water benthic species in inlets near shallow rock piles and reefs. Juveniles use eelgrass, sand, and kelp beds. Feed on amphipods, crabs, and copepods. **Reproduction/Life History** Ovoviviparous spawning from April through July, with larval release from May to July. (Kramer and O'Connell 1995; WDNR 2006a)
Redstripe rockfish	*Sebastes proriger*	NA	All	**General Information (Habitats and Feeding)** Adults found from 330- to 1,000-ft (100- to 300-m) depths, and young often found in estuaries in high- and low-relief rocky areas. Juveniles feed on copepods and euphausiids; adults eat anchovies, herring, and squid. **Reproduction/Life History** Spawning is ovoviviparous, occurring from January through March. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kendall and Lenarz 1986; Kramer and O'Connell 1995; NRC 2001; Starr et al. 1996)
Tiger rockfish	*Sebastes nigrocinctus*	NA	All	**General Information (Habitats and Feeding)** Semidemersal to demersal species occurring at depths ranging from shallows to 1,000 ft (305 m); larvae and juveniles occur near surface and range of depth; adults use rocky reefs, canyons, and headlands; generalized feeders on shrimp, crabs, and small fishes. **Reproduction/Life History** Ovoviviparous spawning peaks in May and June. Juveniles are pelagic. (Garrison and Miller 1982; Kramer and O'Connell 1995; Moulton 1977; NRC 2001; Rosenthal et al. 1988)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Widow rockfish	*Sebastes entomelas*	NA	All	**General Information (Habitats and Feeding)** Adults found from 330- to 1,000-ft (100- to 300-m) depths near rocky banks, ridges, and seamounts; adults feed on pelagic crustaceans, Pacific hake, and squid; juveniles feed on copepods and euphausiids. **Reproduction /Life History** Ovoviviparous spawning occurs from October through December. One brood of 95,000 to 1,113,000 eggs are produced by female widows per year. The season of larval release occurs earlier in the southern parts of their range than in the northern regions, likely January through April in Washington waters. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Laroche and Richardson 1981; NMFS 1990; NRC 2001; Reilly et al. 1992)
Yelloweye rockfish	*Sebastes ruberrimus*	NA	All	**General Information (Habitats and Feeding)** Adults are found from depths of 80–1,800 ft (24–550 m), near reefs and cobble bottom. Juveniles prefer shallow, broken-bottom habitat. Juveniles often hide in rock crevices; adults are demersal and solitary, tending to remain localized and not making extensive migrations. Adults feed on other rockfish species, sand lance, herring, shrimp, rock crabs, and snails. **Reproduction/Life History** Ovoviviparous spawning in late fall or early winter, with the larvae released from May to July. (Eschmeyer et al. 1983; Hart 1973; Kramer and O'Connell 1995; NRC 2001; Rosenthal et al. 1988)
Yellowtail rockfish	*Sebastes flavidus*	NA	All	**General Information (Habitats and Feeding)** Adults found from 165- to 1,000-ft (50- to 300-m) depths; adults semipelagic or pelagic over steep-sloping shores and rocky reefs. Juveniles occur in nearshore areas. Adults are opportunistic feeders on pelagic animals including hake, herring, smelt, squid, krill, and euphausiids. **Reproduction/Life History** Ovoviviparous spawning from October through December. Incubation is between January and March. Larvae and juveniles are pelagic swimmers. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; O'Connell and Carlile 1993)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Olympia oyster	*Ostrea lurida*	NA	1–14, 17	**General Information (Habitats and Feeding)** Species found throughout the inland waters of Puget Sound, as well as in Willapa Bay and possibly Grays Harbor; also grown commercially in Puget Sound. They occupy nearshore ecosystem on mixed substrates with solid attachment surfaces and are found from 1 ft (0.3 m) above MLLW to 2 ft (0.6m) below MLLW. Intolerant of siltation. **Reproduction/Life History** Reproduce spring to fall when water temperatures are between 54 and 61°F (12.5 and 16°C) by broadcast spawning. After 8–12 days, larvae develop into free-swimming larvae. Larvae are free-swimming for 2–3 weeks before they settle onto hard substrate, such as oyster shells and rocks. (Baker 1995; Couch and Hassler 1990; West 1997)
Northern abalone	*Haliotis kamtschatkana*	NA	10	**General Information (Habitats and Feeding)** Also known as pinto abalone. Presence in Washington is limited to the Strait of Juan de Fuca and the San Juan Islands. Occupies bedrock and boulders from extreme low water to 100 ft (30 m) below MLLW; usually associated with kelp beds. The abalone is completely vegetarian and uses its radula to scrape pieces of algae from the surface of rocks. **Reproduction/Life History** Broadcast spawners that release pelagic gametes that develop into free-swimming larvae using cilia to propel themselves. After up to a week, the larvae settle to the bottom, shed their cilia, and start growing a shell to begin sedentary adult life on crustose coralline algae. (Gardner 1981; NMFS 2007a; WDNR 2006b; West 1997)
Newcomb's littorine snail	*Algamorda subrotundata*	NA	14–17	**General Information (Habitats and Feeding)** Found in Grays Harbor and Willapa Bay on Washington coast; current distribution uncertain. Algae feeder occupying narrow band in *Salicornia* salt marshes above MHHW and is not considered a true marine gastropod. **Reproduction/Life History** Broadcast spawning in salt marshes. Other reproductive information unknown. (Larsen et al. 1995)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Giant Columbia River limpet	*Fisherola nuttalli*	35, 36, 40, 45, 47–49	NA	**General Information (Habitats and Feeding)** Also known as the shortface lanx, it occupies fast-moving and well-oxygenated streams. It is found in the Hanford Reach segment of the Columbia River, Wenatchee, Deschutes (OR), Okanogan, Snake, and Methow rivers. Prefers shallow, rocky areas of cobble to boulder substrates and diatom-covered rocks, and feeds by grazing on algae attached to rocks. **Reproduction/Life History** Broadcast external fertilization. Reproduction timing is unknown. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
Great Columbia River spire snail	*Fluminicola columbiana*	35, 45, 48, 49; other locations unknown	NA	**General Information (Habitats and Feeding)** Also known as the Columbia pebblesnail and ashy pebblesnail, its current range is restricted to rivers, streams, and creeks of the Columbia River basin. It requires clear, cold streams with highly oxygenated water and is generally found in shallow water (less than 5 inches [13 cm] deep) with permanent flow on cobble-boulder substrates. Spire snails live on and under rocks and vegetation in the slow to rapid currents of streams where they graze on algae and small crustaceans. **Reproduction/Life History** They are short-lived, usually reaching sexual maturity within a year, at which time they breed and die. Unknown reproduction timing. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
California floater (mussel)	*Anodonta californiensis*	30, 36, 37, 40, 42, 47–49, 52–54, 58–61	NA	**General Information (Habitats and Feeding)** In Washington, it is known to occur in the Columbia and Okanogan rivers and several lakes. Freshwater filter feeder requiring clean, well-oxygenated water for survival that is declining throughout much of its historical range. California floater mussels are intolerant of habitats with shifting substrates, excessive water flow fluctuations, or seasonal hypoxia. **Reproduction/Life History** Spring spawning occurs after adults reach 6–12 years in age. Fertilization takes place within the brood chambers of the female mussel. Fertilized eggs develop into a parasitic stage called glochidia, which attach to species-specific host fish during metamorphosis. After reaching adequate size, juvenile mussels release from the host and attach to gravel and rocks. (Box et al. 2003; Frest and Johannes 1995; Larsen et al. 1995; Nedeau et al. 2005; Watters 1999; WDNR 2006b)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Western ridged mussel	*Gonidea angulata*	01, 03–05, 07–11, 13, 21–42, 44–55, 57–62	NA	**General Information (Habitats and Feeding)** Specific information on this species is generally lacking; reside on substrates ranging from firm mud with the presence of some sand, silt, or clay to coarse gravel in creeks, streams, and rivers. They require constant, well-oxygenated flow, and shallow water (<10 ft [3 m] depth). This species may tolerate seasonal turbidity but is absent from areas with continuous turbidity and is sensitive to water quality changes such as eutrophication or presence of heavy metals. **Reproduction/Life History** During breeding, males release sperm into the water and females must bring this into their shell for fertilization to occur. Larvae called glochidia are released by the female and attach to the gills of fish for 1–6 weeks; postlarval mussels hatch from cysts as free-living juveniles to settle and bury in the substrate. (COSEWIC 2003; WDNR 2006b)

Source: Modified from (Jones & Stokes 2006).

[a] Water Resource Inventory Areas (WRIAs) are administration and planning boundaries for watershed areas, as established and managed by the Washington State Department of Ecology (Ecology). WRIA designations were formalized under WAC 173-500-040 and authorized under the Water Resources Act of 1971, Revised Code of Washington (RCW) 90.54. For WRIA boundary locations and related information, see URL = http://www.ecy.wa.gov/services/gis/maps/wria/wria.htm.

[b] Tidal Reference Areas as follows (from WAC 220-110-240): 1 = Shelton, 2 = Olympia, 3 = South Puget Sound, 4 = Tacoma, 5 = Seattle, 6 = Edmonds, 7 = Everett, 8 = Yokeko Point, 9 = Blaine, 10 = Port Townsend, 11 = Union, 12 = Seabeck, 13 = Bangor, 14 = Ocean Beaches, 15 = Westport, 16 = Aberdeen, 17 = Willapa Bay.

6.0 Conceptual Framework for Assessing Impacts

Jetties, breakwaters, groins, bank barbs, and their analogs are structures located in shallow areas along the shoreline (whether they be lacustrine, marine, or riverine shorelines), at the edge of the terrestrial and aquatic environments where complex interactions between these diverse habitats occur. Alteration of the shoreline for the placement of these structures and the uses associated with them will affect, to varying degrees, the controlling factors of the aquatic ecosystem in which they are located. Furthermore, some activities covered by this white paper will result in additional uses of the shoreline that may comprise shoreline ecosystems.

The conceptual framework process begins with an impact which, in this case, would consist of activities authorized under an HPA for a shoreline modification project. The impact will exert varying degrees of effect on controlling factors within the ecosystem (Williams and Thom 2001). Controlling factors are those physical processes or environmental conditions (e.g., flow conditions or wave energy) that control local habitat structure (e.g., substrate or vegetation). Habitat structure is linked to habitat processes (e.g., shading or cover), which are linked to ecological functions (e.g., refuge and prey production). These linkages form the "**impact pathway**" in which alterations to the environment associated with HPA-authorized activities can lead to impacts on the ecological function of the habitat for HCP species. **Impact mechanisms** are the alterations to any of the conceptual framework components along the impact pathway that can result in an impact on ecological function and therefore on HCP species. Figure 6-1 illustrates the conceptual framework used in this white paper to define shoreline modification impacts on HCP species and their habitats.

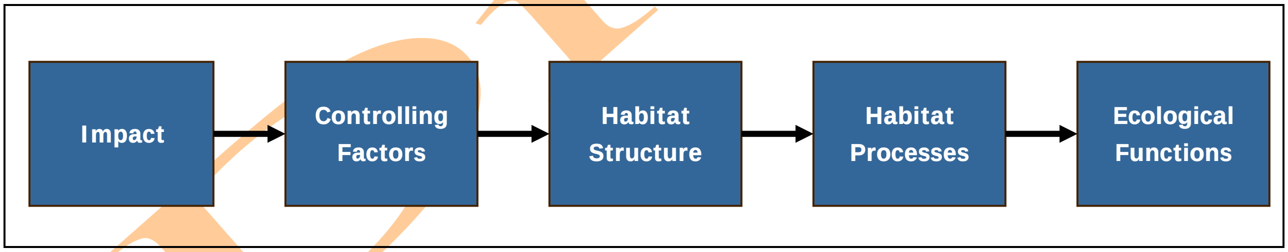


Figure 6-1. Conceptual framework for assessing impacts (Williams and Thom 2001).

Table 6-1 identifies the **mechanisms of impact** that are known to be associated with jetties, breakwaters, groins, and bank barbs. These mechanisms of impact are sorted by the environment (marine, riverine and lacustrine) in which they occur, as the environment plays a fundamental role in the manner by which these mechanisms manifest themselves. In addition, the activity (e.g., jetty installation) typically varies significantly depending on whether it is located in a river or on the coast. This white paper presents what is known about the effects of these mechanisms on those species being considered for coverage under a WDFW multispecies HCP. By identifying these impacts and the nature of the risks these impacts exert on HCP species, measures can be implemented to avoid and, if avoidance is not possible, minimize harmful impacts on these species and the habitats that support their growth and survival.

Table 6-1. Impact mechanisms and submechanisms associated with jetties, breakwaters, groins, and bank barbs.

Environment	Impact Mechanism	Submechanisms
Marine	Construction and Maintenance Activities	Pile driving (underwater noise) Construction vessel operation Work area dewatering Construction/maintenance dredging
	Hydraulic and Geomorphic Modifications	Altered wave energy Altered current velocities Altered nearshore circulation patterns Altered groundwater-surface water exchange Altered sediment supply Altered substrate composition
	Riparian Vegetation Modifications	Altered shading, solar input, and ambient air temperature Altered shoreline stability Altered allochthonous inputs Altered groundwater-surface water exchange Altered habitat complexity
	Aquatic Vegetation Modifications	Altered autochthonous production Altered habitat complexity
	Water Quality Modifications	Altered temperature Altered dissolved oxygen Altered suspended solids and turbidity Altered nutrient and pollutant loading Treated wood pollution
	Ecosystem Fragmentation	Loss of lagoon habitat Loss of LWD Lost opportunities
Lacustrine	Construction and Maintenance Activities	Pile driving (underwater noise) Construction vessel operation Work area dewatering Construction/maintenance dredging
	Hydraulic and Geomorphic Modifications	Altered wave energy Altered current velocities Altered nearshore circulation patterns Altered groundwater-surface water exchange Altered sediment supply Altered substrate composition
	Riparian Vegetation Modifications	Altered shading, solar input and ambient air temperature Altered shoreline stability Altered allochthonous input Altered groundwater-surface water exchange Altered habitat complexity
	Aquatic Vegetation Modifications	Altered autochthonous production Altered habitat complexity

Table 6-1 (continued). Impact mechanisms and submechanisms associated with jetties, breakwaters, groins, and bank barbs.

Environment	Impact Mechanism	Submechanisms
Lacustrine (cont.)	Water Quality Modifications	Altered temperature Altered dissolved oxygen Altered suspended solids and turbidity Altered pH Altered nutrient and pollutant loading Treated wood pollution
	Ecosystem Fragmentation	Loss of nearshore habitat Loss of LWD Lost opportunities
Riverine	Construction and Maintenance Activities	Pile driving (underwater noise) Channel/work area dewatering Construction vessel operation Construction/maintenance dredging
	Hydraulic and Geomorphic Modifications	Altered flow velocities Altered sediment supply Altered substrate composition Altered groundwater-surface water exchange
	Riparian Vegetation Modifications	Altered shading, solar input, and ambient air temperature regime Altered shoreline stability Altered allochthonous input Altered groundwater-surface water exchange Altered habitat complexity
	Aquatic Vegetation Modifications	Altered autochthonous production Altered habitat complexity
	Water Quality Modifications	Altered temperature Altered dissolved oxygen Altered suspended solids and turbidity Altered pH Altered nutrient and pollutant loading Treated wood pollution Loss of channel complexity
	Ecosystem Fragmentation	Loss of LWD Loss of floodplain habitat Lost opportunities

The identification of impact mechanisms associated with HPA-authorized activities that affect habitat is based on a model described by Williams and Thom (2001). For analyzing risk of take and refining the impact analysis as it pertains directly to listed species or species that will be addressed in the HCP, the “exposure-response” model developed by USFWS was used (National Conservation Training Center 2004). Each of these models is discussed in more detail below.

The Williams and Thom model provides the framework for analysis based on the literature search. The goals of this framework are:

- Elucidate impacts associated with each HPA activity
- Determine how those impacts affect habitat structure and function for those habitats utilized by the 52 HCP species
- Develop recommendations for impact avoidance, minimization, and mitigation measures that target the identified impacts.

For each HPA-authorized activity addressed in this white paper, several principal impact mechanisms were identified for each subactivity type, from a geomorphological, engineering, hydrologic, and biological perspective.

This impact analysis serves to identify the direct and indirect impacts that could potentially affect federally listed species and those species that will be addressed in the HCP. To further refine the analysis in each white paper, the exposure-response model (National Conservation Training Center 2004) was incorporated into the impact analysis. The exposure-response model evaluates the likelihood that adverse effects may occur as a result of species exposure to one or more stressors. This model takes into account the life-history stage most likely to be exposed and thereby affected.

The exposure-response model was incorporated as a series of matrices, presented in Appendix A, with results synthesized in Section 7 (*Direct and Indirect Impacts*) and Section 9 (*Potential Risk of Take*) of this white paper. In these species-specific exposure-response matrices, each mechanism and submechanism was initially examined and evaluated to:

- Identify and characterize specific impacts or stressors
- Evaluate the potential for exposure (potential for species to be exposed = identification of stressor, timing/duration/frequency/life-history form presence coincident with impact)
- Identify the species' anticipated response to a stressor
- Identify measures that could reduce or eliminate exposure
- Identify performance standards if appropriate
- Characterize the resulting effects of specific impacts on the various species.

With regard to exposure, standard language was used to indicate when an impact occurs, and for how long and how frequently the stressor or impact occurs. Definitions of those terms used in the analysis are listed in Table 6-2.

Based on life-history information, an analysis of potential exposure was completed for each species. This included an analysis of the direct and indirect impacts (associated with each of the impact mechanisms) on the different lifestages of each species and the likely responses of each species to these stressors. Impact minimization measures to reduce or avoid submechanism impacts were also identified. A final conclusion regarding the overall effect of the submechanism/ stressor on a species is also presented in Appendix A.

Where information was available, the cumulative effects associated with the major impact mechanisms were also identified (see Section 8).

The information generated by the exposure-response analysis is used to summarize the overall risk of take associated with the impact mechanisms produced by each subactivity type. The summary risk of take analysis is presented in Section 9, which presents the risk of take associated with each subactivity type using: (1) a narrative discussion of the risk of take associated with each subactivity type by the specific associated submechanism of impact; and (2) risk of take assessment matrices that rate the risk of take resulting from each subactivity by impact mechanism and environment type. The risk of take ratings presented in the text and matrices in Section 9 are based upon the rating criteria defined in Table 6-3.

Based on the identification of impacts and risk of take analysis, additional recommendations (e.g., conservation, management, protection, BMPs) for minimizing or mitigating project impacts or risk of take were developed and are presented in Section 11.

Table 6-2. Definitions of terms used in the exposure-response analysis for this white paper.

Parameter	Description	Exposure	Definition
When	The timing during which stressor exposure occurs (e.g., time of day, season, associated with activity)	-	Defined flexibly as appropriate for each stressor.
Duration	The length of time the receptor is expected to be exposed to the stressor	Permanent	Stressor is permanent (e.g., conversion of habitat to built environment)
		Long-term	Stressor will last for greater than five years to decades (e.g., time required for complete riparian recovery)
		Intermediate-term	Stressor will last from 6 months to approximately 5 years (e.g., time required for beach substrate to recover from construction equipment)
		Short-term	Stressor will last from days to 6 months (e.g., time required for invertebrate community to recolonize following dewatering)
		Temporary	Stressor associated with transient action (e.g., pile driving noise)
Frequency	The regularity with which stressor exposure is expected to occur and/or the time interval between exposure	Continuous	Stressor is ongoing and occurs constantly (e.g., permanent modification of habitat suitability)
		Intermittent	Stressor occurs routinely on a daily basis
		Daily	Stressor occurs once per day for extended periods (e.g., daytime structural shading)
		Common	Stressor occurs routinely (i.e., at least once per week or several times per month)
		Seasonal	Stressor occurs for extended periods during specific seasons (e.g., temperature effects occurring predominantly in winter and summer)
		Annual	Stressor occurs for annually for a short period of time
		Interannual–decadal	Stressor occurs infrequently (e.g., pile driving associated with project construction and maintenance)

Table 6-3. Definitions of the terminology used for risk of take determinations in this white paper.

Risk of Take Code	Potential for Take	Definition*
H	High	Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to a Likely to Adversely Affect (LTAA) finding.
M	Moderate	Stressor exposure is likely to occur causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or a Not Likely to Adversely Affect (NLTAA) finding depending on specific circumstances.
L	Low	Stressor exposure is likely to occur, causing take in the form of temporary disturbance and minor behavioral alteration. Likely to equate to an NLTAA finding.
I	Insignificant	Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
N	No Risk	No risk of take ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the subactivity type in question, or the impact mechanisms caused by the subactivity type will not produce environmental stressors.
?	Unknown	Unknown risk of take ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

* LTAA = Likely to Adversely Affect; NLTAA = Not Likely to Adversely Affect.

7.0 Direct and Indirect Impacts

This section reviews and synthesizes what is known about the potential impacts of each of the identified impact mechanisms on HCP species. To be consistent with shoreline designations assigned by other governmental agencies and to subdivide effects on HCP species in an efficient manner, the section is organized by environment (i.e., marine, lacustrine, and riverine environments). It reflects the literature findings on both direct and indirect impacts of these HPA activities. The Exposure and Response Matrices (included in Appendix A) provide a synthesis of each of the HCP species (or species groupings). The matrix structure differs slightly from the text below to provide consistency with the Exposure and Response Matrices presented in other WDFW white papers [e.g., (Herrera 2007a)]. In particular, the text below groups the primary mechanisms of impact within the context of each environment, while the matrices introduce the mechanism of impact first and then discuss each environment type.

7.1 Jetties

7.1.1 Marine Environments

Marine habitats are affected by tidal currents, wind, swell, and input from freshwater drainages. Washington's waters support more than 200 species of fishes and a vast array of invertebrates. HCP species dependent on these habitats include several rockfish species, salmon, cod, hake, Pacific herring, walleye pollock, and the Olympia oyster (and other marine species listed in Table 1-1). The controlling factors in these habitats depend upon bathymetry, substrates, circulation and mixing, and sediment transport to provide suitable habitats. Water temperature, salinity, water clarity, nutrients, and dissolved oxygen levels are also important factors in determining the suitability of marine habitats for HCP species.

There are six primary mechanisms of impact in marine environments due to the construction and repair of jetties: construction activities, hydraulic and geomorphic modifications, riparian vegetation modifications, aquatic vegetation modifications, water quality modifications, and ecosystem fragmentation. Jetties are specifically designed to implement hydraulic and geomorphic changes. As a result, most of the direct mechanisms of impact are hydraulic and geomorphic in nature. However, most of the effects on fish and invertebrates arise primarily due to other submechanisms that occur as a result of those hydraulic and geomorphic changes. As a result, there is a substantial amount of cross-referencing in this section.

7.1.1.1 Construction and Maintenance Activities

Construction activities can have direct and indirect effects on HCP species. Activities associated with shoreline modification construction and repair pose the risk of increasing underwater noise levels, increasing suspended solids, removing or disturbing aquatic and riparian vegetation, and releasing toxic substances from construction materials and/or construction equipment to marine waters. Construction activities may also include filling and dredging that can entrain organisms or permanently remove habitat for burrowing and benthic animals. In sum, construction and maintenance activity impacts in marine environments include four submechanisms of impact, capturing a range of activities that are short-lived but intensive and are also required in order to build shoreline modifications as well as to provide or maintain access to infrastructure. The four submechanisms identified for analysis in this white paper include: (1) underwater noise, (2) construction vessel operation and staging, (3) work area dewatering, and (4) construction/maintenance dredging. Direct and indirect effects on fish and invertebrates are summarized below for each of these submechanisms, based on the literature review and subsequent analysis. Note: some of the information presented in this white paper is reproduced from previously prepared white papers addressing the effects of dredging and overwater structures in marine environments (Nightingale and Simenstad 2001a, 2001b).

7.1.1.1.1 Pile Driving (Underwater Noise)

Shoreline modification projects permitted under the WDFW HPA program can produce underwater noise through a variety of mechanisms. These mechanisms include construction-related noise impacts from impulsive sources (i.e., short duration, high intensity noise from sources such as pile driving or materials placement), as well as continuous noise sources (e.g., vessel or equipment operation) that may be associated with the construction of a groin, jetty, or breakwater. The discussion presented in this section provides the noise-related analytical basis for the development of the exposure-response matrices (Appendix A) and the risk of take analysis (Section 9).

This section summarizes existing information on sources of underwater noise, how underwater noise is characterized, existing and proposed effects thresholds, and the magnitude of noise stressors associated with typical project construction and maintenance activities. This discussion is derived in part from a summary of current science on the subject developed by WSDOT (2006).

Measurement of Underwater Noise

Underwater sound levels are measured with a hydrophone, or underwater microphone, which converts sound pressure to voltage, which is then converted back to pressure, expressed in pascals (Pa), pounds per square inch (psi), or decibel (dB) units. Derivatives of dB units are most commonly used to describe the magnitude of sound pressure produced by an underwater noise source, with the two most commonly used measurements being the instantaneous peak sound pressure level (dB_{PEAK}) and the root mean square (dB_{RMS}) pressure level during the impulse, referenced to 1 micropascal (re: 1µPa) (Urick 1983). The dB_{PEAK} measure represents the instantaneous maximum sound pressure observed during each pulse. The RMS level represents the square root of the total sound pressure energy divided by the impulse duration, which provides a measure of the total sound pressure level produced by an impulsive source. The majority of literature uses dB_{PEAK} re: 1µPa sound pressures to evaluate potential injury to fish. However, USFWS and NOAA Fisheries have used both dB_{PEAK} (for injury) and dB_{RMS} (for behavioral effects) threshold values to evaluate adverse injury and disturbance effects on fish, marine mammals, and diving birds (Stadler 2007; Teachout 2007; WSDOT 2006). dB_{RMS} values are used to define disturbance thresholds in fish species, meaning the sound pressure level at which fish noticeably alter their behavior in response to the stimulus (e.g., through avoidance or a "startle" response). dB_{PEAK} values are used to define injury thresholds in salmonids, meaning the sound pressure level at which injury from barotraumas may occur (i.e., physical damage to body tissues caused by a sharp pressure gradient between a gas or fluid-filled space inside the body and the surrounding gas or liquid). Unless otherwise noted, all sound pressure levels cited herein are in dB_{PEAK} or dB_{RMS} re: 1µPa.

Noise behaves in much the same way in air and in water, attenuating gradually over distance as the receptor moves away from the noise source. However, underwater sound exhibits a range of behaviors in response to environmental variables (Urick 1983). For example, sound waves bend upward when propagated upstream into currents and downward when propagated downstream in the direction of currents. Sound waves will also bend toward colder, denser water. Haloclines and other forms of stratification can also influence how sound travels. Noise shadows created by bottom topography and intervening land masses or artificial structures can, under certain circumstances, block the transmission of underwater sound waves. In freshwater systems, sound propagation is often influenced by depth and channel morphology. Underwater noise does not transmit as effectively when water depths are less than 3 feet due to the amplitude of the sound pressure wave (Urick 1983). Because underwater sound does not travel around obstructions, bends in a river or large changes in gradient will truncate sound propagation. This will limit the physical extent of noise related impacts.

Underwater noise attenuation, or transmission loss, is the reduction of the intensity of the acoustic pressure wave as it propagates, or spreads, outward from a source. Propagation can be categorized using two models, spherical spreading and cylindrical spreading. Spherical (free-field) spreading occurs when the source is free to expand with no refraction or reflection from boundaries (e.g., the bottom or the water surface). Cylindrical spreading applies when sound energy spreads outward in a cylindrical fashion bounded by the sediment and water surface. Because neither model applies perfectly in any given situation, most experts agree that a combination of the two best describes sound propagation in real-world conditions (Vagle 2003).

Currently, USFWS and NOAA Fisheries are using a practical spreading loss calculation, which accommodates this view (Stadler 2007; Teachout 2007). This formula accommodates some of the complexity of underwater noise behavior, but it does not account for a number of other factors that can significantly affect sound propagation. For example, decreasing temperature with depth can create significant shadow zones where actual sound pressure levels can be as much as 30 dB lower than calculated because sound bends toward the colder, deeper water (Urick 1983). Haloclines, current mixing, water depth, acoustic wavelength, sound flanking (i.e., sound transmission through bottom sediments), and the reflective properties of the surface and the bottom can all influence sound propagation in ways that are difficult to predict.

Given these complexities, characterizing underwater sound propagation inherently involves a large amount of uncertainty. An alternative calculation approach, known as the Nedwell model (not used by USFWS or NOAA Fisheries), indirectly accounts for some of these factors. Nedwell and Edwards (2002) and Nedwell et al. (2003) measured underwater sound levels associated with pile driving close to and at distance from the source in a number of projects in English rivers. They found that the standard geometric transmission loss formula used in the practical spreading loss model did not fit well to the data, most likely because it does not account for the aforementioned factors that affect sound propagation. They developed an alternative model based on a manufactured formula that produced the best fit to sound attenuation rates measured in the field. This model thereby accounts for uncharacterized site-specific factors that affect noise attenuation, but does not explicitly identify each factor or its specific effects. Because there is considerable uncertainty regarding how to model the many factors affecting underwater noise propagation, and this would require site specific information that cannot practically be obtained in many instances, the Services (i.e., USFWS and NOAA Fisheries) use the more conservative practical spreading loss model in ESA consultations (Stadler 2007; Teachout 2007).

Project-Related Noise Sources

The underwater noise produced by an HPA permitted project, either during construction or operation, is defined by the magnitude and duration of underwater noise above ambient noise levels. The action area for underwater noise effects in ESA consultations is defined by the distance required to attenuate construction noise levels to ambient levels, as calculated using the practical spreading loss calculation or other appropriate formula provided in evolving guidance from USFWS and NOAA Fisheries on this subject.

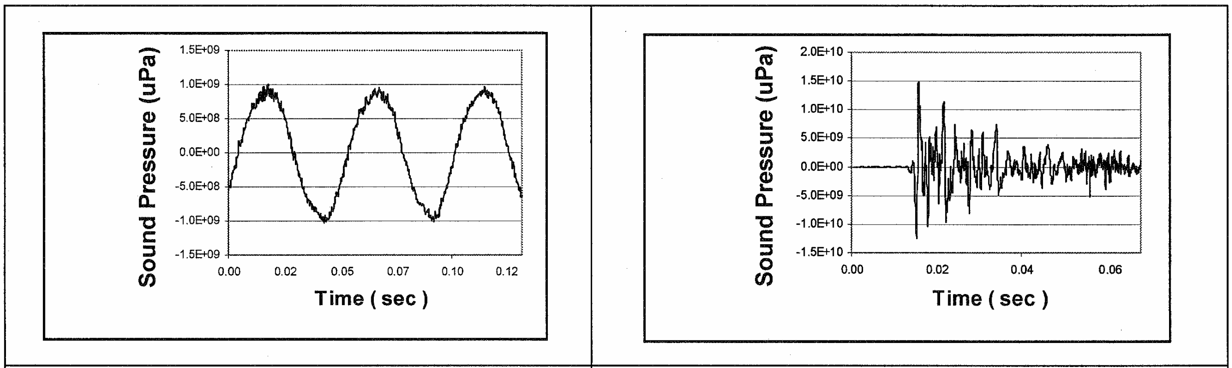
Although there are many sources of noise in the underwater environment, the primary source of underwater noise associated with HPA shoreline modification projects is project-related construction (e.g., equipment operation and materials placement). However, impacts associated with increased vessel operation as a result of shoreline modification should also be considered.

Materials Placement

Most sources of underwater noise potentially resulting from materials placement during HPA permitted projects have received relatively little direct study. Of the potential sources of construction-related noise, pile driving has received the most scrutiny because it produces the highest intensity stressors capable of causing noise-related injury. Other sources of underwater noise, such as dumping of large rock or underwater tool use, have received less study. Therefore, available data on noise levels associated with pile driving are presented here as a basis for comparison.

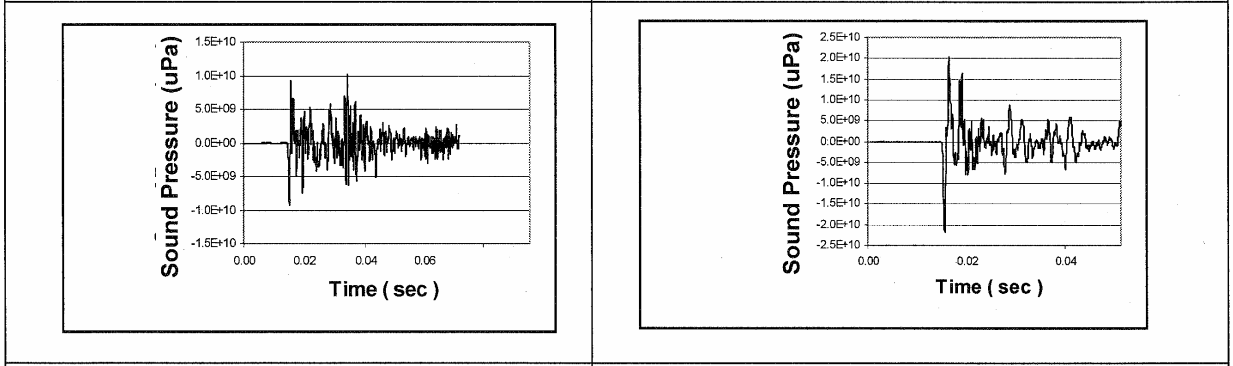
Two major types of pile driving hammers are in common use, vibratory hammers and impact hammers. There are four kinds of impact hammers: diesel, air or steam driven, hydraulic, and drop hammer (typically used for smaller timber piles). Vibratory hammers produce a more rounded sound pressure wave with a slower rise time. In contrast, impact hammers produce sharp sound pressure waves with rapid rise times, the equivalent of a punch versus a push in comparison to vibratory hammers. The sharp sound pressure waves associated with impact hammers represent a rapid change in water pressure level, with greater potential to cause injury or mortality in fish and invertebrates. Because the more rounded sound pressure wave produced by vibratory hammers produces a slower increase in pressure, the potential for injury and mortality is reduced. (Note that while vibratory hammers are often used to drive piles to depth, load-bearing piles must be “proofed” with some form of impact hammer to establish structural integrity.) The changes in pressure or waveform generated by these different types of hammers are pictured in Figure 7-1.

Piling composition also influences the nature and magnitude of underwater noise produced during pile driving. Driven piles are typically composed of one of three basic material types: timber, concrete, or steel (although other special materials such as plastic may be used). Steel piles are often used as casings for pouring concrete piles. Noise levels associated with each of these types of piles are summarized in Table 7-1. Reference noise levels are denoted in both dB_{PEAK} and dB_{RMS} values, at the specified measurement reference distance.



Vibratory Hammer wave form—set duration

Air or steam-driven hammer wave form—single strike



Diesel hammer wave form—single strike

Hydraulic hammer wave form—single strike

Figure 7-1. Sound pressure changes (or waveform) generated by hammer type (WSDOT 2006).

Table 7-1. Reference noise levels, by structure type.

Material Type and Size	Impact Hammer Type	Reference Noise Levels[a]		Environment Type	Source
		dB_{PEAK}	dB_{RMS}		
12-inch timber	Drop	177 @ 10 m	165 @ 10 m	Marine	(Illingworth and Rodkin 2001)
24-inch concrete piles	Unspecified	188 @ 10 m	173 @ 10 m	Unspecified	[DesJardin 2003, personal communication cited by WSDOT (2006); Hastings and Popper 2005]
Steel H-piles	Diesel	190 @10 m	175 @ 10 m	Marine	(Hastings and Popper 2005; Illingworth and Rodkin 2001)
12-inch steel piles	Diesel	190 @ 10 m	190 @ 10 m	Marine	(Illingworth and Rodkin 2001)
14-inch steel piles	Hydraulic	195 @ 30 m	180 @ 30 m	Marine	(Reyff et al. 2003)
16-inch steel piles	Diesel	198 @ 10 m	187 @ 9 m	Freshwater	(Laughlin 2004)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(WSDOT 2006)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(Hastings and Popper 2005)
30-inch steel piles	Diesel	208 @ 10 m	192 @ 10 m	Marine	(Hastings and Popper 2005)
66-inch steel piles	Hydraulic	210 @ 10 m	195 @ 10 m	Marine	(Reyff et al. 2003)
96-inch steel piles	Hydraulic	220 @ 10 m	205 @ 10 m	Marine	(Reyff et al. 2003)
126-inch steel piles	Hydraulic	191 @ 11 m	180-206 @ 11 m	Marine	(Reyff et al. 2003)
150-inch steel piles	Hydraulic	200 @ 100 m	185 @ 100 m	Marine	(Reyff et al. 2003)

[a] Metric distances are listed as they were provided in the source material; 9 m = 29.5 ft; 10 m = 32.8 ft; 11 m = 36 ft; 30 m = 98 ft; 100 m = 328 ft.
All sound pressure values in units re: 1µPa.

Vessel/Equipment Operation and Materials Placement

In comparison to pile driving, data on noise levels produced by placement of other construction-related materials is limited. For example, measured noise levels associated with work on the Friday Harbor ferry terminal ranged between 133 dB_{peak} and 140 dB_{peak}, excluding pile driving. These noise levels were slightly higher than ambient levels, which include routine vessel traffic (Laughlin 2005). Nedwell et al. (1993) measured noise produced by underwater construction tools such as drills, grinders, and impact wrenches at 3.28 ft (1 m) from the source. When corrected for a reference distance 32.8 ft (10 m) from the source using the practical spreading loss model, the noise associated with these sources ranged from approximately 120 to 165 dBpeak. These data suggest that noise associated with these activities, such as in-water tool use, placement of large rock and similar material, vessel operation, and in-water operation of heavy machinery, will generally produce substantially lower noise levels than those associated with pile driving. However, other construction-related noises may generate continuous noise for longer periods, with the effect of elevating ambient noise levels or masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators.

This effect may be of particular concern for shoreline modification projects that will result in changes in vessel operation or equipment use that change ambient noise levels for longer periods (e.g., days to years). For example, vessel operation can significantly influence ambient noise levels. Large vessel engines can produce underwater sound up to 198 dB, and depth sounders can produce noise in excess of 180 dB (Buck 1995; Heathershaw et al. 2001). Hazelwood and Connelly (2005) monitored fishing vessel noise over a broad octave range from 10 Hz–40 kHz and documented noise levels ranging from 140–185 dB_{peak}, with the loudest noise occurring at the lower end of the octave range. Commercial sonar devices operating in a frequency range of 15–200 kHz can produce underwater noise ranging from 150–215 dB at maximum levels (Stocker 2002).

Ambient underwater noise levels serve as the baseline for measuring the disturbance created by project construction or maintenance. Both natural environmental noise sources and mechanical or human-generated noise contribute to the ambient or baseline noise conditions within and surrounding a project site. Many specific measurements of ambient noise levels include sounds produced by vessels and other human activities. Therefore, these noise measurements, particularly those recorded in the vicinity of ferry terminals and other high-activity locations, are indicative of the level of noise levels that could be produced by project construction and operation.

Ambient noise levels have been measured in several different marine environments on the West Coast and are variable depending on a number of factors, such as site bathymetry and human activity. For example, measured ambient levels in Puget Sound are typically around 130 dB_{peak} (Laughlin 2005). However, ambient levels at the Mukilteo ferry terminal reached approximately 145 dB_{peak} in the absence of ferry traffic (WSDOT 2006). Ambient underwater noise levels measured in the vicinity of the Friday Harbor ferry terminal project ranged between 131 and 136 dB_{peak} (Laughlin 2005). Carlson et al. (2005) measured the underwater baseline for the Hood Canal and found it to range from 115 to 135 dB_{RMS}. Heathershaw et al. (2001) reported open-ocean ambient noise levels to be between 74 and 100 $\mathrm{dB}_{\mathrm{peak}}$ off the coast of central California. Note, however, that these ambient noise levels are typical conditions, and typical conditions can be punctuated by atypical natural events. For example, lightning strikes can produce underwater noise levels as high as 260 $\mathrm{dB}_{\mathrm{peak}}$ in the immediate vicinity (Urick 1983).

Limited data are available on ambient noise levels in freshwater environments, but it is reasonable to conclude that they vary considerably based on available information. For example, high-gradient rivers, fast-flowing rivers, and large rivers and lakes with significant human activity are likely to produce more noise than lakes and slow-flowing rivers in more natural environments. Burgess and Blackwell (2003) measured ambient sounds in the Duwamish River in Seattle, Washington, (averaged over 20 seconds to 5 minutes) and found the sound to vary between 110 and 130 dB continuous sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dB re: 1μPa^2/second). Amoser and Ladich (2005) measured ambient noise levels in the mainstem Danube River, a smaller, fast-flowing tributary stream, a small lake, and a quiet river backwater. The river and stream represented fast-flowing habitats, the lake and backwater quiet, slow-flowing habitats. Sound behavior was complex. They found that ambient noise levels ranged from as low as 60 to as high as 120 $\mathrm{dB}_{\mathrm{peak}}$ in the fast-flowing habitats, depending on the sound frequency (lower frequency sound was typically louder). Ambient noise in the slackwater habitats was considerably lower, ranging from 40 to 80 $\mathrm{dB}_{\mathrm{peak}}$ across the frequency range (again with lower frequency sounds being loudest).

Direct and Indirect Effects on Fish

Most fish sense sounds, vibrations, and other displacements of water in their environment through their inner ear and with the lateral line running the length of each side of the fish and on the head. The lateral line is a mechano-sensory system that plays an indirect role in hearing through its sensitivity to pressure changes at close range. The hearing organs and lateral line system are collectively referred to as the acoustico-lateralis system. The hearing thresholds of different fish species vary depending on the structure and sensitivity of this system. Those families of fish known as hearing specialists include cyprinids (dace [e.g., Umatilla and leopard dace], minnows and carp), catastomids (suckers [e.g., mountain sucker]), and ictalurids (catfish), which collectively belong to the Ostariophysan taxonomic grouping of fishes. These fish possess a physical connection between the swim bladder and the inner ear, with the swim bladder acting as an amplifier that transforms the pressure component of sound into particle velocity component, to which the inner ear is sensitive (Moyle and Cech 1988). The hearing capacity of salmonids, on the other hand, is limited both in bandwidth and intensity threshold. The Atlantic salmon, for example, is functionally deaf at sound pressure wavelengths above 380 hertz (Hz) (Hawkins and Johnstone 1978). In these fish, the swim bladder does not likely enhance hearing.

Noise sources such as pile driving that produce high-intensity sound pressure waves can result in direct effects on fish, ranging from effects as limited as temporary stress and behavioral avoidance, to temporary or permanent injury in multiple organ systems (including hearing, heart, kidney, swim bladder, and other vascular tissue), to direct mortality (Popper and Fay 1973; 1993). Another potential effect includes masking of existing ambient noise reducing the ability of fish to sense predators or prey. These activities may also have indirect effects such as reducing the foraging success of these fish by affecting the distribution or viability of potential prey species. Numerous studies have examined the effects on fish associated with underwater noise and are discussed more fully below.

In general, injury and mortality effects from underwater noise are caused by rapid pressure changes, especially on gas-filled spaces in the body. Rapid volume changes of the swim bladder may cause it to tear, resulting in a loss of hearing sensitivity and hydrostatic control. Intense noise may also damage the tissue in hearing organs, as well as the heart, kidneys, and other highly vascular tissue. Susceptibility to injury is variable and depends on species-specific physiology, auditory injury, and auditory thresholds (Popper and Fay 1973, 1993). While species-specific data are limited, the available information indicates variable effects related to physiology, size, and age, as well as the intensity, wavelength, and duration of sound exposure.

Hardyniec and Skeen (2005) and Hastings and Popper (2005) summarized available information on the effects of pile driving-related noise on fish. Pile driving effects observed in the studies reviewed ranged broadly from brief startle responses followed by habituation to instantaneous lethal injury. The difference in effect is dependent on a number of factors, including: piling material, the type and size of equipment used, and mitigation measures; site-specific depth, substrate, and water conditions; and the species, size, and life-history stage of fish exposed.

Popper et al. (2005) exposed three species of fish to high-intensity percussive sounds from a seismic air gun at sound levels ranging between 205 and 209 dB_{Peak}, intending to mimic exposure to pile driving. Subject species included a hearing generalist (broad whitefish), a hearing specialist (lake chub), and a species that is intermediate in hearing (northern pike). They found that the broad whitefish suffered no significant effects from noise exposure, the lake chub demonstrated a pronounced temporary threshold shift in hearing sensitivity (i.e., hearing loss), and the northern pike showed a significant temporary hearing loss but less than that of the lake chub. The hearing sensitivities of lake chub and northern pike returned to their respective normal thresholds after 18 to 24 hours. High-intensity sounds can also permanently damage fish hearing (Cox et al. 1987; Enger 1981; Popper and Clarke 1976).

Enger (1981) found that pulsed sound at 180 dB was sufficient to damage the hearing organs of codfish (genus *Gadus*), resulting in permanent hearing loss. Hastings (1995) found that goldfish exposed to continuous tones of 189, 192, and 204 dB_{peak} at 250 Hz for 1 hour suffered permanent damage to auditory sensory cells. Injury effects may also vary depending on noise frequency and duration. Hastings et al. (1996) found destruction of sensory cells in the inner ears of oscars 4 days after exposure to continuous sound for 1 hour at 180 dB_{peak} at 300 Hz. In contrast, when the two groups of the same species were exposed to continuous sound at 180 dB_{peak} at 60 Hz for 1 hour, and to impulsive sound at 180 dB_{peak} at 300 Hz repeatedly over 1 hour, they showed no apparent injury. Susceptibility to injury may also be life-history specific. Banner and Hyatt (1973) demonstrated increased mortality of sheepshead minnow eggs and embryos when exposed to broadband noise approximately 15 dB above the ambient sound level. However, hatched sheepshead minnow fry were unaffected by the same exposure.

Even in the absence of injury, noise can produce sublethal effects. Behavioral responses to sound stimuli are well established in the literature for many fish species. For example, Moore and Newman (1956) reported that the classic fright response of salmonids to instantaneous sound stimuli was the "startle" or "start" behavior, where a fish rapidly darts away from the noise source. Knudsen et al. (1992) found that in response to low-frequency (10 Hz range) sound, salmonids 1.6–2.4 in (40–60 mm) in length exhibited an initial startle response followed by habituation, while higher frequency sound caused no response even at high intensity. In a study of the effects of observed pile driving activities on the behavior and distribution of juvenile pink and chum salmon, Feist et al. (1992) found that pile-driving operations were associated with changes in the distribution and behavior of fish schools in the vicinity. Fish schools were two-fold more abundant during normal construction days in comparison to periods when pile driving took place. Blaxter et al. (1981) found Atlantic herring to exhibit an avoidance response to both continuous pulsed sound stimuli with habituation to more continuous stimuli occurring over time, and Schwarz and Greer (1984) found similar responses on the part of Pacific herring. Sound has also been shown to affect growth rates, fat stores, and reproduction (Banner and Hyatt 1973; Meier and Horseman 1977).

Prolonged underwater noise can also reduce the sensitivity of fish to underwater noise stimuli, with potentially important effects on survival, growth, and fitness. The fish auditory system is likely one of the most important mechanisms fish use to detect and respond to prey, predators, and social interaction (Amoser and Ladich 2005; Fay 1988; Hawkins 1986; Kalmijn 1988; Myrberg 1972; Myrberg and Riggio 1985; Nelson 1965; Nelson et al. 1969; Richard 1968; Scholik and Yan 2001, 2002; Wisby et al. 1964). Scholik and Yan (2001) studied the auditory responses of the cyprinid fathead minnow to underwater noise levels typical of human-related activities (e.g., a 50 horsepower outboard motor). They found that prolonged exposure decreased noise sensitivity, increasing the threshold level required to elicit a disturbance response for as long as 14 days after the exposure. Amoser and Ladich (2005) reported similar findings in common carp in the Danube River, noting that auditory ability in this hearing specialist species was measurably masked in environments with higher background noise. They reported similar but far less pronounced responses in hearing generalist species such as perch. These data suggest that elevated ambient noise levels have the potential to impair hearing ability in a variety of fish species, which may in turn adversely affect the ability to detect prey and avoid predators, but that this effect is variable depending on the specific sensitivity of the species in question. Feist et al. (1992) similarly theorized that it was possible that auditory masking and habituation to loud continuous noise from machinery may decrease the ability of salmonids to detect approaching predators.

Direct and Indirect Effects on Invertebrates

In general, information on the effects of underwater noise on invertebrates is limited, indicating that additional research on the subject is needed. What little data are available suggest some sensitivity to intense percussive underwater noise. In a study completed by Turnpenny et al. (1994), mussels, periwinkles, amphipods, squid, scallops, and sea urchins were exposed to high air gun and slow-rise-time sounds at between 217 and 260 dB_{peak}, analogous to extremely loud pile driving. One scallop suffered a split shell following exposure to 217 dB_{peak}, suggesting the potential for serious injury when percussive underwater noise exceeds these levels.

No research has been identified regarding the effects of lower intensity continuous underwater noise on invertebrates. However, operational noise is typically associated with sound pressures well below levels that have been observed to cause injury in shellfish, suggesting that HCP invertebrate species might not be subject to these effects. Because HCP invertebrates with the potential for stressor exposure are either filter feeders or grazers and are essentially non-motile, these species are unlikely to be subject to auditory masking effects that would limit the ability to sense predators and prey. Some potential may exist for disturbance-induced interruption of feeding behavior, but more research on this subject is necessary to determine this definitively, and this subject is considered a data gap.

7.1.1.1.2 Construction Vessel Operation

The operation of vessels during construction also presents several potential impacts including: (1) grounding, anchoring, and prop wash; (2) temporary ambient light modification; (3) water quality degradation; and (4) changes in ambient noise levels. Impacts 1 and 2 are addressed in detail in the *Facility Operation and Vessel Activities* section of the Marinas white paper, and water quality impacts are addressed in detail below in Section 7.1.1.5 (*Water Quality Modifications*).

Data on noise levels produced by nonpile driving related construction activities are limited. What data are available tend to show that noise associated with these activities, such as in-water tool use and in-water operation of heavy machinery, will generally produce substantially lower noise levels than those associated with pile driving. However, other construction-related noises may generate continued sound for longer periods, with the effect of elevating ambient noise levels or masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators.

Measured noise levels associated with work on the Friday Harbor ferry terminal ranged between 133 and 140 dB_{peak}, excluding pile driving. These noise levels were slightly higher than ambient levels, which include both ferry and other maritime traffic (Laughlin 2005). Nedwell et al. (1993) measured noise produced by underwater construction tools such as drills, grinders, and impact wrenches at 3.28 ft (1 m) from the source. When corrected for a reference distance 32.8 ft (10 m) from the source using the practical spreading loss model, the noise associated with these sources ranged from approximately 120 to 165 dB_{peak}.

Direct and Indirect Effects on Fish

Other construction-related noises, such as dredge equipment or large vessels, may generate continued sound for longer periods. Direct effects associated with these noises include behavioral responses such as startling or scattering (Mitson and Knudson 2003), even in vessels that are designed to be quiet (Ona et al. 2007), which could compromise the survival of the affected species. Indirect effects include masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators, subsequently reducing foraging success and increasing vulnerability to predation.

Direct and Indirect Effects on Invertebrates

In general, information on the effects of underwater noise on invertebrates is limited, indicating that additional research on the subject is needed.

7.1.1.1.3 Work Area Dewatering

The construction of a jetty does not necessitate dewatering. However, there are several construction activities that may require pile installation, which could potentially require dewatering. Maintenance operations may also require dewatering. These activities would require the same fish-handling practices that are discussed in Section 7.3.2.1.1 (*Channel Dewatering*). See that subsection for details about impacts associated with dewatering work areas.

Direct and Indirect Effects

The direct and indirect effects of work area dewatering will be essentially the same as the dewatering activities discussed in Section 7.3.2 (*Riverine Environments*) for groins and bank barbs, with the primary difference being that marine fish and invertebrates HCP species are affected (rather than freshwater species).

7.1.1.1.4 Construction/Maintenance Dredging

Construction or maintenance dredging converts shallower subtidal habitats to deeper subtidal habitats through periodic deepening to remove accumulated sediments that impede navigation through jetties and around breakwaters. There are several different means by which dredging affects fish and invertebrates, the most significant being alteration of bathymetry, removal of aquatic vegetation, entrainment of benthic organisms, and turbidity and resuspension of contaminated sediments. These stressors are discussed below (see Section 7.1.1.4 [*Aquatic Vegetation Modifications*] for information on removal of aquatic vegetation).

Altered Bathymetry and Substrate Composition

Large channel deepening projects can markedly alter ecological relationships through the change of freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing [e.g., tidal alterations in and near the Columbia River mouth (Sherwood et al. 1990)]. Miller et al. (1990) reported that only through comprehensive areal surveys over a minimum of four seasons before dredging, with follow-up surveys after dredging, can the impacts of channel deepening on aquatic resources be determined.

Depending on site characteristics, maintenance dredging may occur annually or at intervals of 10 years or longer. These different dredging timelines represent different disturbance regimes both in terms of the ability of the benthos to recolonize prior to redisturbance and the magnitude of benthic productivity affected by dredging.

In a study to evaluate the effects of dredged material disposal on biological communities, Hinton et al. (1992) reported a significant increase in benthic invertebrate densities at a disposal site between June 1989 (pre-disposal) and June 1990 (post-disposal). Recolonization could have occurred by invertebrates burrowing up through newly deposited sediments or recruitment from surrounding areas (Richardson et al. 1977).

Direct and Indirect Effects

Dredging required for jetty development in marine environments converts intertidal into subtidal habitats, thereby affecting the plant and animal assemblages that are uniquely adapted to the particular light, current, and substrate regimes of intertidal areas. By altering bathymetry and bottom substrates, such conversions are described as producing a habitat “trade-off” of intertidal and shallow-subtidal communities for deeper, subtidal communities. Dredging activities result in numerous short-term direct effects as well as long-term and food web indirect effects including entrainment and potential mortality; reconfiguration of the benthos and the availability of nutrient and prey resources; periodic removal of potentially suitable habitats for fish and invertebrates; alteration of water circulation and subsequent nutrient, prey, and habitat availability; and increased turbidity and potential resuspension of contaminants. Resulting impacts, as described in detail in Section 7.1.1.5 (*Water Quality Modifications*), Section 7.1.1.4 (*Aquatic Vegetation Modifications*), and Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*), include mortality, injury, decreased foraging opportunity, decreased survival, decreased growth and fitness, and physiological and behavioral responses. Deposition of dredge spoils can smother existing habitats and benthic organisms, resulting in a similar suite of impacts.

Entrainment

Entrainment occurs when an organism is trapped in the uptake of sediments and water being removed by dredging machinery during construction and maintenance activities (Reine and Clarke 1998), or in rapid destabilizing bedload mobilized by altered channel geometry. Benthic infauna and nonmotile life-history stages (e.g., salmonid eggs, lamprey amocoetes) are particularly vulnerable to entrainment, but some motile epibenthic and demersal organisms such as burrowing shrimp, crabs, and rearing larvae and juveniles of many fish species also can be susceptible. Entrainment rates are usually described by the number of organisms entrained per cubic yard (cy) of sediment dredged (Armstrong et al. 1982).

Demersal fish, such as sculpins, suckers, and related species, are hypothesized to have the highest rates of entrainment as they reside on or in the bottom substrates. Lamprey ammocoetes likely have a high risk of vulnerability to entrainment due to the lengthy time of residence this life-history stage spends buried in freshwater sediments. In general, fish eggs and larvae of fish that have no capacity to avoid direct dredge impacts are also at significant risk of entrainment. Of particular concern for the purpose of this analysis are the HCP groundfish (lingcod, rockfish, Pacific cod, pollock, hake) and the forage fishes (herring, sand lance, and surf smelt), which all have larval or juvenile life-history stages with low motility. The juvenile life-history stage of the groundfish species typically rear in shallow nearshore habitats, where dredging is likely to occur. Due to their demersal nature and limited motility, they face a higher risk of entrainment by dredging.

Krueger et al. (2007) studied the effects of suction dredge entrainment on two species of estuarine mussels. The test subjects entrained through the dredge showed no evidence of mortality or significant injury. This suggests that freshwater mollusk species may be relatively insensitive to entrainment-related effects. This is intuitively logical, as these species occur in environments where mobilization of coarse bedload is common. This suggests the likelihood of evolutionary adaptation to protect against mechanical injury from bedload mobility. However, the authors cautioned that their findings were applicable only to the adult life-history stages studied. The sensitivity of juvenile mussel species to entrainment remains unknown. This uncertainty would be expected to extend to the juvenile life-history stages of other HCP invertebrate species as well.

Mollusk larvae and juveniles are expected to be highly sensitive to the effects of entrainment and are assumed to suffer high mortality from mechanical injury, smothering, anoxia, starvation, or desiccation. However, in the case of freshwater mussels, stressor exposure would have to be extensive to result in significant population level effects. As an example, the issue of larval oyster mortality caused by dredge entrainment was studied in detail in the Chesapeake Bay. Lunz (1985) concluded that even if entrained larvae suffered 100 percent mortality, the absolute effects would be relatively limited because the dredge would entrain only a small fraction of larvae in the vicinity. The estimated mortality rate for oyster larvae ranged between 0.005 and 0.3 percent of total abundance. These effects are insignificant in comparison to natural mortality rates. Many species, particularly marine fish and invertebrates, have planktonic larval life-history stages that suffer naturally high mortality rates (in some cases exceeding 99 percent) (Lunz 1985). Therefore, it is likely that larval mortality from burial and/or entrainment is relatively insignificant when viewed from the perspective of natural population dynamics. Moreover, in the case of freshwater mussels, the potential for adverse effects is further limited by the fact that the parasitic glochidia life-history stage reside in the gills of host-fish where stressor exposure is less likely to occur.

The other freshwater mollusks (great Columbia River spire snail and giant Columbia River limpet) hatch from the egg fully formed. As such, these species would be expected to have a higher level of sensitivity to the effects of burial and entrainment.

Direct and Indirect Effects

Direct effects associated with dredging entrainment include injury or mortality of fish and invertebrate species. Indirect effects include alteration of food web configuration or dynamics that could affect overall species survival or fitness.

Turbidity

Several of the studies cited in this section present information in turbidity level units in the place of suspended sediment concentrations to infer effects thresholds. Turbidity is commonly used as a surrogate for suspended sediment concentrations, but the relationship between these measures is site specific. Where available, the equivalent suspended sediment concentration is provided, otherwise the turbidity value is provided. Because this complicates the interpretation of the information, a brief discussion of the relationship between turbidity and suspended sediment concentrations is provided here.

The International Standards Organization (ISO) defines turbidity as the 'reduction of transparency of a liquid caused by the presence of undissolved matter' (Lawler, 2005), as measured by turbidimetry or nephelometry. Turbidity can be caused by a wide range of suspended particles of varying origin and composition. These include inorganic materials like silt and clay, and organic materials such as tannins, algae, plankton, micro-organisms and other organic matter. The term suspended sediments refers to inorganic particulate materials in the water column. Suspended sediments can range in size from fine clay to boulders, but the term applies most commonly to suspended fines (i.e., sand size or finer material). Because suspended sediments are a component of turbidity, turbidity is commonly used as a surrogate measure for this parameter. However, the accuracy of the results is dependent on establishing a clear correlation between turbidity and suspended sediment concentrations to account for the influence of organic materials. This correlation is site specific, given the highly variable nature of organic and inorganic material likely to occur in a given setting.

Although the physics of turbidity generation can be calculated, adequate data do not exist to quantify the biological response in terms of threshold sediment dosages for specific exposure durations that can be tolerated by various marine and estuarine organisms. Some empirical information exists for peak turbidities, but these data are not duration specific. Numerical modeling simulations of dredging-related suspended-sediment plume dynamics are currently being developed under the Dredging Operations and Environmental Research Program (USACE 2007). A large range of turbidities can be expected from dredging operations, ranging from well over 10,000 ppm near dredging operations, decreasing to background levels well away from dredging sites (Black and Parry 1999). As one might expect, concentrations diminish with both distance from the operations and in time after dredging has stopped. The existing data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred parts per million (ppm) in 24 hours and others having no effect at concentrations above 10,000 ppm for 7 days. All of these concentrations are possible within proximity to dredged sites. Studies on east coast species have identified lethal concentration levels, and Newcombe and Jensen (1996) have developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended solid concentrations. However, threshold studies for the temporary impacts of suspended sediment levels specific to aquatic environments in the Pacific Northwest are lacking.

Direct and Indirect Effects

Recent studies have shown that the size and shape of suspended sediments and the duration of exposure are important factors in determining the extent of adverse effects of increased turbidity on salmonids (Martens and Servizi 1993; McLeay et al. 1987; Northcote and Larkin 1989; Servizi and Martens 1987, 1991; Newcombe and MacDonald 1991). Lake and Hinch (1999) found concentrations in excess of 40,000 ppm to elicit stress responses (e.g., decreased leukocrit, indicating reduced immunity response) to correlate with the occurrence of gill damage. Angular sediments, as opposed to rounder sediments, are associated with higher fish stress responses at lower sediment concentrations. This may be due to irritation caused by angular sediments that result in increased mucus production and decreased oxygen transfer. Although the causes of mortality were not clear, Lake and Hinch (1999) found that mortality occurred at concentrations of 100 parts per thousand (ppt), with no differences found in mortality rates in natural or more angular anthropogenically derived sediments.

In addition to size and shape, the concentration of suspended sediments would determine the severity of the responses elicited in aquatic organisms. Effects on aquatic organisms will differ based on their developmental stage. Suspended sediments may affect salmonids by altering their physiology, behavior, and habitat, all of which may lead to physiological stress and reduced survival rates. For example, high levels of suspended solids may be fatal to salmonids due to gill trauma, osmoregulation impairment, and changes in blood chemistry. Lower levels of suspended solids and turbidity may cause chronic sublethal effects, such as loss or reduction of foraging capability, reduced growth, reduced resistance to disease, increased stress, and interference with cues necessary for orientation in homing and migration (Lloyd 1987; Newcombe and MacDonald 1991; Bash et al. 2001).

Sigler et al. (1984) reported that suspended sediments have been shown to affect fish functions such as avoidance responses, territoriality, feeding, and homing behavior. Similarly, Wildish and Power (1985) reported avoidance of suspended sediments by rainbow smelt and Atlantic herring at 20 ppm and 10 ppm, respectively. Berg and Northcote (1985) tested the sensitivity of coho salmon to short-term pulses in suspended sediment over periods lasting from 2.5 days (with sediment levels raised rapidly over a period of 1 hour) to 4.5 days (with sediment levels raised gradually over 2 days). They found that territorial, gill flaring and feeding behaviors were disrupted in the presence of higher turbidity levels. At higher levels of turbidities such as 30–60 nephelometric turbidity units (NTUs), social organization broke down, gill flaring occurred more frequently, and only after a return to turbidity of 1–20 NTUs was the social organization re-established. Similarly, feeding success was found to be linked to turbidity levels, with higher turbidity levels reducing prey capture success. The studies of Gregory (1993) demonstrate that at particular levels of increased turbidity, juvenile salmon actually increase their feeding rates, while at higher levels (such as >200 ppm), they demonstrated pronounced behavioral changes in prey reaction and predator avoidance. In a study of dredging impacts on juvenile chum salmon in Hood Canal, Salo et al. (1980) found that juvenile chum also showed avoidance reactions to low levels of turbidity ranging from 2 to 10 ppm above ambient concentration. However, in related laboratory tests, Salo et al. (1980) found that avoidance was not shown until a concentration of 182 ppm was reached. In general, these behavioral thresholds vary across species and life-history stages.

Consistent with their early reliance upon nearshore estuarine habitats with relatively high turbidities compared to pelagic or freshwater habitats, juvenile chum salmon are classified as turbidity tolerant compared to other fishes (Salo et al. 1980).

Many fish species thrive in rivers and estuaries with naturally high concentrations of suspended sediments. It is currently unknown what behavioral mechanisms are triggered as various fish species encounter patches of increased turbidity, such as dredging plumes. Also unknown is what threshold of turbidity might be a cue to fish to avoid light-reducing turbidity. Simenstad (1990) described the behavioral effects that would affect migrating fishes, such as reduced foraging success, increased risk of predation, and migration delay, to be highly dependent upon the duration of exposure.

The final phase of salmon homing migration requires olfactory cues (Hasler and Scholz 1983; Hasler et al. 1978). In studies of returning Chinook spawners, Whitman et al. (1982) found that suspended volcanic ash at concentrations of 650 ppm did not influence homing performance. Preference experiments indicated that Chinook, when given the choice, preferred clean water from their natal streams (also called “home water”) to municipal drinking water, with the presence of volcanic ash reducing the preference for home water. It was concluded that fish could recognize home water despite the ash suspension and that any reduced home-water preference was due to ash avoidance (Whitman et al. 1982).

Thresholds for lethal effects on clams and eastern oysters have been reported, with negative impacts on eastern oyster egg development occurring at 188 ppm of silt (Cake 1983) compared to a 1,000 ppm threshold for hardshell clam eggs (Mulholland 1984). Dredging operations would be expected to produce these levels of concentrations near the dredge site for nearly the entire operating time period (USACE 1983).

When suspended sediment concentrations rise above the filtering capacities of bivalves, their food becomes diluted (Widdows et al. 1979). In environments with high algal concentrations, Clarke and Wilber (2000) reported that the addition of silt, in relatively low concentrations, showed increased growth on the part of mussels (Kiorboe et al. 1981), surf clams (Mohlenberg and Kiorboe 1981), and eastern oysters. This is because these bivalves are well adapted to some level of turbidity and presumably use suspended sediment as a food source (Kiorboe et al. 1981). Bricelj and Malouf (1984) found that hardshell clams decreased their algal ingestion with increased sediment loads. However, no growth rate differences were observed between clams exposed to algal diets alone and clams with added sediment loads (Bricelj and Malouf 1984). Urban and Kirchman (1992) reported similarly ambiguous findings concerning suspended clay. Suspended clay (20 ppm) appeared to interfere with the ingestion of algae by juvenile eastern oysters by limiting their ability to differentiate between algae and starches (which are normally expelled without ingestion), but it did not reduce the overall amount of algae ingested. Grant et al. (1990) found that the summer growth of European oysters was enhanced at low levels of sediment resuspension and inhibited with increased sediment deposition. It was hypothesized that the chlorophyll in suspended sediments may act as a food supplement that could enhance growth, but higher levels may dilute planktonic food resources, thereby suppressing food ingestion. Behavioral changes in response to sediment loads were also noted for soft-shelled clams under sediment loads of 100–200 ppm (Grant and Thorpe 1991). The clams appeared to cope with increased sediment loading by reducing aerobic respiration rate and feeding behavior. At high concentrations (>100 ppm) over long-term periods (>2 weeks), this led to depleted growth and indicators of nutritional state. Changes in mantle and siphon condition indicative of reduced resistance to infection were also observed.

Collectively, these studies show no clear pattern of sublethal effects from elevated concentrations of suspended solids, and thereby turbidity, that could be generally applied across aquatic mollusks. This uncertainty is further complicated by the fact that many of the HCP invertebrate species are poorly studied. This indicates the need for directed studies on the sensitivity of these species before effects thresholds can be set. In the absence of this information, however, it is useful to consider that HCP invertebrates are all bottom-dwelling mollusks that have evolved to live in dynamic environments under conditions of variable turbidity. Therefore, sensitivity to turbidity-related stressors would be expected to occur only when conditions exceed the range of natural variability occurring in their native habitats.

Further information regarding the direct and indirect effects of turbidity on fish and invertebrates is presented in Section 7.1.1.5 (*Water Quality Modifications*).

7.1.1.2 Hydraulic and Geomorphic Modifications

Shallow nearshore marine habitats (structured by tidal currents, wind, and input from terrestrial and freshwater sources) support at least one life-history strategy of all of the marine and anadromous HCP species [e.g., all salmon, rockfish species, cod, hake, Pacific herring, walleye pollock, Newcomb's littorine snail, and the Olympia oyster (WDNR 2006a, 2006b)]. The controlling factors in these habitats depend upon pre-existing bathymetry, substrate, circulation and mixing, and sediment transport (WDNR 2006a, 2006b). These underlying hydrogeomorphic variables regulate a phenomenon known as alongshore transport, or littoral drift (Komar 1998). Key to understanding littoral drift is the idea of a drift cell (also known as drift sectors), which is a segment of shoreline along which littoral drift moves sediment at noticeable rates. Each drift cell includes: (1) a sediment source, such as a feeder bluff; (2) a driftway along which these sediments move; and (3) an accretion terminal where the drift material is deposited. In this way, a drift cell allows an uninterrupted movement of beach materials (Terich 1987).

It is widely known that jetties alter both the bathymetry and littoral drift of the area around and under such structures both in exposed (Komar 1998) and sheltered settings (NRC 2007). These activities can produce five submechanisms of impact on aquatic life: altered wave energy, altered current velocities, altered nearshore circulation, altered groundwater-surface water exchange, and altered sediment supply. Further, as a part of jetty installation, substrate can be emplaced that is completely artificial, which introduces a sixth submechanism, altered substrate composition (Komar 1998). Impacts associated with each of these mechanisms are described more fully in the following sections.

Weir jetties are jetties that are submerged at most water levels for some portion of their length, usually the landward-most end. These features allow the passage of sediment for localized deposition in some inactive portion of the navigational channel (Seabergh and Kraus 2003). Therefore, they have a tendency to alleviate some of the geomorphic and hydraulic modifications associated with jetties; however, they do initiate change in the substrate and tend to produce geomorphic disturbance, albeit different than their exposed counterparts (Ranasinghe and Turner 2006). Where applicable, these distinctions are noted in the text below.

7.1.1.2.1 Altered Wave Energy

Jetties are designed to limit deposition in a navigable channel and to provide wave protection for vessels (Dean and Dalrymple 2002). They are generally constructed out of rock or poured concrete. Therefore, they result in and cause the retention of wave energy in the surrounding area (Komar 1998). In addition, ship traffic associated with the addition of a pair of jetties can interact with these artificial boundaries, causing a significant increase in wave energy in between the two jetties (Melo and Guza 1991). Regardless of the nature of the alterations, the modified relationship between topography and wave energy results in a shoreline that is out of equilibrium with natural shoreline processes (Komar 1998). The effects are generally independent of jetty design (i.e., weir jetties are as prone to these effects as exposed jetties), although some best management practices (BMPs) can reduce these effects (see Section 11 [*Habitat Protection, Conservation, Mitigation, and Management Strategies*] for details). As a result, wave energy artificially accumulates in some areas and is diminished in others. This redistribution of wave energy can have a number of interrelated indirect and direct effects on fish and invertebrates that are addressed below in two categories: those that result in changes in substrate, and those that change water column characteristics.

Direct and Indirect Effects

Substrate Stressors

Substrate is an important control on the growth of aquatic vegetation in Puget Sound (Koch 2001) and conditions for fish spawning in intertidal waters (typically forage fish). Increased wave energy intensifies bed shear stress; if some of the coarsest material is not mobilized, a generally coarser substrate results (Komar 1998). However, the degree to which this occurs depends on the geologic setting. On the outer coast, for example, substrate is loose, deep, sandy, and unconsolidated. In these areas, increased or displaced wave energy associated with jetties creates wholesale erosion of the shoreline (Miller et al. 2001). In protected, previously glaciated areas, the basin topography is complex and the coarse nature of the substrate slows down erosion dramatically (Nordstrom 1992). In these locales, a lag deposit can result in a near bedrock-like shoreline [e.g., Foulweather Beach (Finlayson 2006)]. Typically, however, the hardening of shoreline bed (beach substrate) manifests by a loss of the pebble veneer that is common throughout much of Puget Sound [(Finlayson 2006); Figure 7-2]. This process is similar to what has occurred on the urbanized shorelines throughout the Great Lakes (Chrzastowski and Thompson 1994). These changes have produced pronounced ecological changes within recent years in the Great Lakes (Meadows et al. 2005), causing the elimination of native species and enabling invasives (zebra mussels) to dominate the nearshore ecosystem (Marsden and Chotkowski 2001).

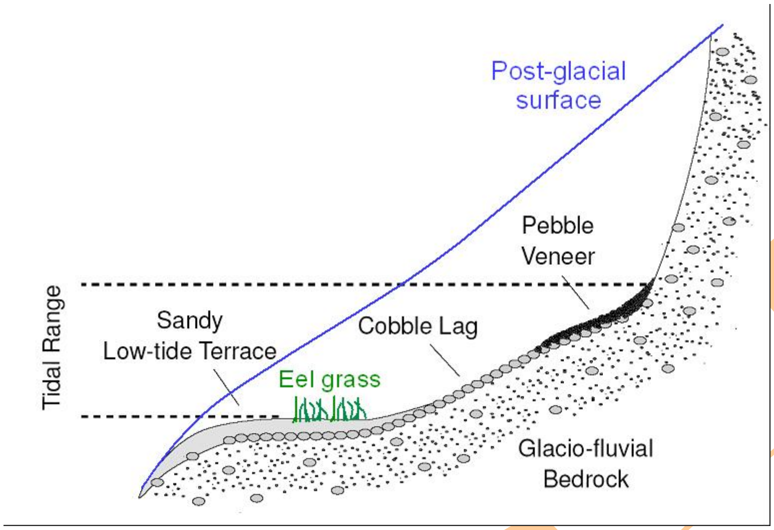


Figure 7-2. A typical Puget Sound beach profile (Finlayson 2006).

Changing substrate can adversely affect the growth of aquatic vegetation. For instance, eelgrass is incapable of growing in dominantly gravelly substrates (Koch 2001). For details regarding the effects on fish and invertebrates as a result of aquatic vegetation disturbances, see Section 7.1.1.4 (*Aquatic Vegetation Modifications*).

Although there have been very few experimental studies of forage fishes, damage to surf smelt spawning areas have been documented in the presence of shoreline hardening (i.e., bulkheads) in Hood Canal (Herrera 2005; Penttila 1978; Thom et al. 1994). Typical spawning substrates consist of fine gravel and coarse sand, characteristic of the pebble veneer found throughout Puget Sound (Finlayson 2006), with broken shells intermixed in some cases (Thom et al. 1994). Surf smelt are quite susceptible to the effects of alterations on shoreline processes (sediment supply, transport, and accretion) due to their reliance on specific beach profiles and substrate compositions for successful spawning (Penttila 1978). Surf smelt make no attempt to bury their demersal, adhesive eggs, but rely on wave action to cover the eggs with a fine layer of substrate (Thom et al. 1994). Therefore, changing the wave environment may change the survivability of surf smelt spawn or suitability of the site for future spawning habitat. The importance of substrate to spawning has also been empirically demonstrated in the closely related Japanese surf smelt (Hirose and Kawaguchi 1998).

Pacific sand lance spawn in the high intertidal zone on substrates varying from sand to sandy gravel. Sand lance also rely on sandy substrates for burrowing at night. As with surf smelt, sand lance spawning is susceptible to the deleterious effects of littoral alterations because sand lance rely on a certain beach profile and specific substrate compositions (Penttila 1995).

Jetties are specifically designed to produce areas of reduced wave activity (Dean and Dalrymple 2002). Reduction in wave energy from natural levels lowers the near bed shear stress, resulting in the deposition of finer sediments (Miller et al. 1977). Considering the large volume of fine-grained sediment supplied to western Washington waters (Downing 1983), even areas that are not part of an active littoral cell can receive a large amount of fine sediment.

Deposition of large amounts of fine sediment can kill aquatic vegetation vital to nearshore HCP species. Recent work has shown that burying eelgrass at depths as little as 25 percent of the total plant height could decrease productivity and increase the mortality of eelgrass (Mills and Fonseca 2003). For more information about the effects of aquatic vegetation on fish and invertebrates, see Section 7.1.1.4 (*Aquatic Vegetation Modifications*). Eelgrass can also be discouraged from colonizing new areas with a high clay content as a result of recent sediment deposition (Koch 2001).

Water Column Stressors

Wave energy is the dominant source of fluid mechanical energy in the nearshore in most Washington waters (Finlayson 2006). Waves are responsible for mixing the upper portion of the water column (Babanin 2006) and producing high shear stresses near the bed (Lamb et al. 2004). These motions can prove harmful to aquatic vegetation and indirectly affect the fish and invertebrates that use and consume it.

Indirect Effects

Attenuation of waves can increase water column stratification in marine waters and lead to dissolved oxygen reduction and temperature anomalies (Qiao et al. 2006); see Section 7.1.1.5 (*Water Quality Modifications*) for details. Surficial mixing and circulation also play an important role in primary productivity, particularly near large river mouths [e.g., Willapa Bay (Roegner et al. 2002)]. Disruption of these processes may have effects on the primary productivity and ultimately on any marine species through food-web interactions.

Wave energy also plays a role in the distribution of aquatic vegetation used by salmonids and other nearshore fishes, particularly in energetic environments. High wave energy has shown to inhibit the colonization and growth of some seagrasses [e.g., eelgrass (Fonseca and Bell 1998); see Section 7.2.1.4 (*Aquatic Vegetation Modifications*) for details], although in more recent work in Puget Sound, no correlation was found between eelgrass prevalence and wave characteristics (Finlayson 2006). High shear stresses associated with waves can also dislodge kelp (Kawamata 2001).

Direct Effects

Fish that are planktonic breeders have been shown to produce spatially variable spawn that relies on the combination of wave motion and ambient currents to be transported to appropriate and productive nursery areas (Hernandez-Miranda et al. 2003; Rooper et al. 2006). Waves produce motions and induce transport both in the water column and near the seabed that are capable of transporting particulates large distances (Liang et al. 2007; McCool and Parsons 2004). Altering these mechanical processes alters transport rates (Liang et al. 2007; McCool and Parsons 2004). While no specific studies have analyzed this effect in any of the HCP species, it is possible that changes in wave-induced transport could transport spawn and larvae to less desirable areas and, therefore, contribute to mortality of the larvae of planktonic-breeding species.

Many fish species, such as herring, rockfish, pollock, and cod, have planktonic larvae that are dependent on wave and current patterns for transport to and/or retention in areas favorable for rearing. Developing larvae that are transported away from areas favorable for rearing before they are ready for life in open water face an increased risk of starvation and predation or, in the case of schooling pelagic species, may be permanently isolated from their spawning population (Sinclair 1992).

Invertebrates that cannot tolerate extremely high shear or burial may be directly affected by altered wave energy. Experimental evidence of the mortality limits of large shear stresses on mollusks or other invertebrates is nonexistent. The burial of mollusks and the related stress or mortality resulting from partial and complete burial have been addressed empirically (Hinchey et al. 2006). Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. However, survival of each species examined appeared to decrease exponentially with increasing overburden stress (i.e., depth of burial), with most species being killed once they were completely buried. Most shorelines in Washington do not have the sedimentation rates necessary (e.g., deposition of greater than 1 inch in a single event) to bury mollusks (Hinchey et al. 2006). Near major river mouths, however, sedimentation rates [e.g., on the Skagit as inferred by (Hood 2006)] are possible.

7.1.1.2.2 Altered Current Velocities

Jetties are specifically designed to produce areas of reduced velocity so that maritime traffic can safely pass through and into a navigational channel. In providing protection to boats and ships, velocities can be reduced to the point where the deposition of fine sediment (silt and clay) occurs (Miller et al. 1977), particularly near sources of fine sediment [i.e., large rivers: (Downing 1983)]. These effects are similar those discussed in the preceding subsection, *Altered Wave Energy*, in Section 7.1.1.2.1.

At the other extreme, strong currents can have significant impacts on both aquatic vegetation and the substrate that it is embedded in. The relationship between flow velocity and a change in substrate is related to a quantity, known as the boundary shear stress (Miller et al. 1977). Substrate and aquatic vegetation are removed if a critical shear stress is exceeded. Again, these impacts are comparable to those discussed in the preceding subsection.

<u>Indirect Effects</u>

Nearshore currents, even those in heavily altered environments, do not exceed the threshold for adult salmonid navigation, but high velocities have been shown to exclude some small fishes (e.g., juvenile salmonids and forage fishes) from navigating nearshore waters (Michny and Deibel 1986; Schaffter et al. 1983). This would cause the fragmentation of habitat for these species (see Section 7.1.1.6 [*Ecosystem Fragmentation*] for details).

Eelgrass and many other species of aquatic vegetation (e.g., bull kelp) require some water motion for survival (Fonseca et al. 1983). The degree of sensitivity is species-specific and also dependent on other factors such as pollutant loading, which further disrupts chemical exchange with the water column. Jetties have also been shown to heighten pollutant concentrations by increasing their residence time in the nearshore (Bordalo 2003), exacerbating these effects.

Direct Effects

Similar to the effects associated with alterations in wave energy, alterations in velocity in the water column could alter transport and increase the mortality of planktonic spawn (e.g., rockfish). See the preceding submechanism (Section 7.1.1.2.1 [*Altered Wave Energy*]) for details.

Invertebrates cannot tolerate extremely high shear stress or burial. Experimental evidence of the mortality limits of large shear stresses on mollusks or other invertebrates is not available. Burial of mollusks is discussed in Section 7.1.1.2.1 (*Altered Wave Energy*).

7.1.1.2.3 Altered Nearshore Circulation

Nearshore circulation is a generic phrase that describes the flux of salt, water, and sediment in association with tidal and wave motion near the shoreline. In more exposed, sandy settings, nearshore circulation is dominated the mechanics of wave breaking (Komar 1998). These effects are generally insignificant in Puget Sound (Finlayson 2006), but they can be an important process when swell is present [e.g., on the outer coast (Komar 1998)].

In Puget Sound and near the mouth of large rivers (e.g., the Columbia), tides and freshwater input play a more important role in nearshore currents. Tidal motions are rarely sufficient to mobilize sediment typical of Puget Sound beaches [material of gravel size or larger (Finlayson 2006)], but they can mobilize fine sediments (i.e., silt and clay), particularly in areas of high sediment supply [e.g., (Nittrouer and Sternberg 1975)]. Jetties have been shown to both increase (da Silva and Duck 2001) and decrease (Sherwood et al. 1990) tidal prisms, depending on the characteristics of the tides and freshwater input and the nature and geometry of the alterations.

Indirect Effects

Like wave energy, nearshore circulation patterns are a dominant characteristic that shapes the suitability of nearshore habitats for a range of HCP species. Alteration of nearshore circulation patterns can produce many of the same effects described for altered wave energy in Section 7.1.1.2.1 (*Altered Wave Energy*). Specifically, fish and invertebrate species that are planktonic breeders have been shown to produce spatially variable spawn that relies on the combination of wave motion, ambient currents, and circulation patterns for transport to and retention in productive nursery areas (Hernandez-Miranda et al. 2003; Rooper et al. 2006; Sinclair 1992). As stated, while specific studies on HCP species are lacking, virtually all of the purely marine HCP species have a planktonic egg and/or larval life-history stage dependent on rearing habitat transport and retention dynamics. Developing eggs or larvae that are transported into areas unfavorable for rearing face a high likelihood of starvation and predation or, in the case of schooling pelagic species, may be permanently isolated from their spawning population (Sinclair 1992).

Direct Effects

Changes in nearshore circulation could alter the transport of planktonic spawn, increasing the mortality of those species (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details). Effects on invertebrates would be associated with removal or burial (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details).

7.1.1.2.4 Altered Groundwater-Surface Water Exchange

In the construction of a jetty, it is common for pilings to be placed near the shoreline to ensure that the landward end of the jetty remains intact. In these cases, groundwater connections with the sea are interrupted. Submarine groundwater discharge has been documented to play an important role in the circulation of fluids and nutrients on many coasts throughout the world (Gallardo and Marui 2006; Johannes 1980; Michael et al. 2005). Pilings could interrupt the free exchange of groundwater between the sea and the uplands. If this occurs, deleterious effects on nearshore ecosystems are likely (Nakayama et al. 2007).

Dumped rock or riprap jetties that do not have piles associated with them do not impede or eliminate the exchange of groundwater with supratidal areas. Therefore, these types of jetties or their analogs do not exhibit groundwater impacts.

Indirect Effects

Submarine groundwater discharge serves a number of ecologic functions (Gallardo and Marui 2006). Most work on the subject has focused on the nutrient load that these waters supply to the coastal ocean in sandy, exposed coastal environments [(Gallardo and Marui 2006); see Section 7.1.1.5.4 (*Altered Nutrient and Pollutant Loading*) for details]. However, several important effects have been documented in Puget Sound. For example, the lack of groundwater discharge can lead to increased substrate temperatures at comparable tidal elevations (Dale and Miller 2007; Rice 2006). Another loss of function is the removal of the seepage face at low tide (Gendron 2005). The correlation of the top of the seepage face to the landward limit of eelgrass beds has been anecdotally established in Puget Sound (Finlayson 2006). Although not demonstrated in a systematic study, the loss of the seepage face, as observed by Finlayson (2006) and Gendron (2005), would likely increase the risk of desiccation of aquatic plants. Desiccation has been found to be the dominant control on the growth of eelgrass (*Zostera marina)* in the Pacific Northwest [(Boese et al. 2005); for details, see Section 7.1.1.4 *(Aquatic Vegetation Modifications*)].

Direct Effects

The importance of groundwater seepage to the macro-ecology of the deep ocean (i.e., benthic environments) is also well known (Kiel 2006). Both hydrothermal vents and cold seeps are known to be “hot spots” of biological activity, a direct result of groundwater discharge (Kelley et al. 2002; Kiel 2006). However, the direct effect of submarine groundwater discharge on fish and invertebrates in nearshore areas is less clear (Simmons 1992).

7.1.1.2.5 Altered Sediment Supply

In fulfilling its designed function to prevent sediment from depositing in a navigational channel, the principal effect of a jetty is to obstruct natural littoral transport, thus starving the downdrift shoreline (Dean and Dalrymple 2002). Jetties have even initiated shoreline instability on adjacent shorelines (Dias and Neal 1992) and redistributed turbidity in their vicinity (Sukhodolov et al. 2004). Alteration of sediment transport patterns can present potential barriers to the natural processes that build spits and beaches and provide substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Haas et al. 2002; Penttila 2000; Thom and Shreffler 1996; Thom et al. 1994). In addition, a considerable amount will often deposit on the updrift side of a jetty. This causes the shoreline to protrude into the water body, distorting sediment transport farther up the shoreline (Komar 1998). These impacts primarily manifest as a change in substrate, the impacts of which are discussed in Section 7.1.1.2.1 (*Altered Wave Energy*).

Indirect Effects

The primary indirect effect of changing sediment supply is to change the distribution of substrate within the littoral cell within which the modification occurs (Terich 1987). Therefore, the loss of sediment to a drift cell results in a coarsening of the substrate as fine-grained sediment is lost to deep portions of the basin by resuspension (Finlayson 2006) and are not resupplied by freshly eroded bluff sediments. The coarsening would have the same effects on fish and invertebrates as discussed in Section 7.1.1.2.1 (*Altered Wave Energy*). However, because some drift cells can be extremely long [e.g., more than 20 miles long in the drift cell that extends between Seattle and Mukilteo on the northeastern shore of the main basin of Puget Sound (Terich 1987)], the effects of a modification can extend well beyond the primary activity area.

Direct Effects

The primary direct effect on fish and invertebrates is to alter the degree of turbidity in the nearshore environment (Bash et al. 2001; Berry et al. 2003). (See Section 7.1.1.5.3 [*Altered Suspended Solids and Turbidity*] for a full discussion of the effects of turbidity).

7.1.1.2.6 Altered Substrate Composition

Jetties change the shoreline from a dynamic, loose surface to a rigid, immobile one along their length. Although there are distinct differences between artificial substrates used in jetty construction, they all behave over time like bedrock shorelines, similar to extremely coarse-clastic beaches in Puget Sound (Finlayson 2006). The primary difference between these installments is whether they permit the exchange of groundwater with the marine system. Solid concrete walls and steel pilings allow no flow-through and likely have additional impact as compared to other artificial substrates (e.g., riprap) (Nakayama et al. 2007). For a full discussion of these impacts, see the preceding Section 7.1.1.2.4 *(Altered Groundwater-Surface Water Exchange*).

<u>Indirect Effects</u>

The primary indirect effect on nearshore ecology is to encourage a shift toward hard-substrate, often invasive, communities (Wasson et al. 2005). The immobile substrate also fundamentally changes the mechanics of water motion on the shoreline, increasing wave reflection (Finlayson 2006; Komar 1998) and eliminating the exchange of water into and out of the shoreline if impermeable materials are used (Nakayama et al. 2007). These effects are discussed in preceding subsections (i.e., Section 7.1.1.2.1 [*Altered Wave Energy*], Section 7.1.1.2.3 [*Altered Nearshore Circulation*]).

<u>Direct Effects</u>

In Elliott Bay, Toft et al. (2004) found similar densities of juvenile salmonids at sand/cobble beaches and riprap sites in settings where the riprap extended only into the upper intertidal zone. When riprap extended to the subtidal zone, higher densities of juvenile salmonids were found along riprap than at sand/cobble beaches. Toft et al. (2004) hypothesized that this finding may be based on the fact that the shallow-water habitats preferred by juvenile salmonids were compressed along the highly modified shorelines with steep slopes; therefore, their snorkel observations were able to record all juvenile salmonids present. In comparison, at the sand/cobble beaches, the slopes were gentler, the zone of shallow water was much wider, and densities were therefore lower because the fish were more dispersed.

It is possible that coarser substrate could provide a benefit to some HCP species, such as rockfish. An active debate in the scientific community is whether jetties and other shoreline hardening structures are as productive and diverse as natural hard-rock shorelines, particularly in the Adriatic Sea east of Italy (Bacchiocchi and Airoldi 2003; Bulleri and Chapman 2004; Guidetti, Verginella et al. 2005). In addition to the elimination of mobile, sandy habitats, the Adriatic Sea studies have shown that maritime structures caused weighted abundances in piscivores and urchins, as well as decreased abundances of native species that prefer more mobile substrates (Guidetti, Bussotti et al. 2005). Although species distributions are clearly different in Italy than in Washington State, the steep, paraglacial landscape and relatively short period, locally generated waves make the hydraulic and geomorphic variables essentially identical (Finlayson 2006). This suggests that similar effects on species distribution in Washington State are possible. For example, conversion to coarser substrate types may favor species such as northern abalone, which graze on hard substrates. In contrast, Olympia oysters may be detrimentally affected by the loss of preferred soft substrates. Complicating the debate is that other maritime activities (i.e., fishing and ship traffic) are difficult to separate from other effects and likely limit any gain in the transition of habitat type (Blaber et al. 2000; Guidetti, Bussotti et al. 2005).

However, the conclusions of (Fresh 2006) provide some context for these studies. By examining habitats used by juvenile salmonids, Fresh (2006) concluded that the conversion of sandy, mobile substrates, such as those on natal deltas, would produce a greater impact on salmonid rearing than those on naturally immobile shorelines. Because Puget Sound shorelines are diverse in terms of sediment mobility (Finlayson 2006), the effect on juvenile salmonids from shoreline hardening is highly site specific and could be small in places where the shoreline is naturally immobile. Unfortunately, jetties are often located near river mouths (and deltas) where the transition from mobile, sandy substrate to an immobile, rocky substrate will be most detrimental to juvenile salmonids.

7.1.1.3 Riparian Vegetation Modifications

Marine riparian zones are the upland areas adjacent to water bodies that form the transition zones between terrestrial and aquatic systems. Removal or disturbance of riparian vegetation during the construction of jetties can expose HCP species to stressors caused by a variety of impact mechanisms. These mechanisms include:

- Altered shading, solar input, and ambient air temperature
- Reduced shoreline stability
- Altered allochthonous inputs
- Altered groundwater-surface water exchange
- Altered habitat complexity.

These impact mechanisms and related ecological stressors are discussed below.

7.1.1.3.1 Altered Shading, Solar Input, and Ambient Air Temperature

The influence of shade on nearshore water quality parameters such as temperature is not well established in marine environments. In general, seasonal air temperature conditions, winds, currents, stratification, and tidal exchange play more dominant roles in determining marine water temperatures (Brennan and Culverwell 2004). However, shade may strongly influence temperatures in specific habitat types under specific circumstances, such as the upper intertidal zone, tidal pools, pocket estuaries, and other habitat types that become temporarily isolated or exposed by tidal dynamics. These systems can experience increased variability in temperature and microclimate conditions in the absence of protective shading. Microclimatic conditions in the upper intertidal zone, for example, are demonstrably influenced by riparian vegetation. Rice (2006) compared microclimate parameters at a bulkheaded Puget Sound beach with no overhanging riparian vegetation to those at an adjacent unmodified site with extensive riparian vegetation. He documented significant differences in light intensity, air temperature, substrate temperature, and humidity levels at the modified site. Differences in peak substrate temperatures were particularly striking, averaging nearly 20^{o}F (11^{o}C) higher at the modified site.

Direct and Indirect Effects

Riparian shade strongly influences microclimate conditions in the upper intertidal zone. Loss of riparian shade is correlated with increased substrate temperatures and reduced humidity which in turn are indicative of increased desiccation stress (Rice 2006). This is a significant finding because temperatures and desiccation are significant stressors that limit the survival of many upper intertidal organisms, including HCP forage fish species (Brennan 2004; Brennan and Culverwell 2004). Penttila (2001) reported much higher egg mortality rates among surf smelt for eggs deposited on unshaded beaches compared to those sites with intact overhanging riparian vegetation. The hypothesized mechanism causing the observed higher rate of mortality was increased egg desiccation due to longer periods of direct sun exposure at sites with insufficient riparian vegetation to provide shade and other favorable microclimate conditions. Rice's (2006) findings comparing differences in microclimate conditions and surf smelt spawn survival on shaded versus unshaded beaches strongly support this hypothesis. It is uncertain what the precise thermal limits are for HCP invertebrates species, but it is clear that these exist. For example, Olympia oysters can withstand 86°F (30°C) for several hours (Baker 1995), but it is likely that they experience diminished productivity at these temperatures.

7.1.1.3.2 Altered Shoreline Stability

Although it is unlikely that jetty construction would require the removal of significant areas of riparian vegetation, many related HPA-permitted activities involve the temporary or permanent modification of riparian vegetation structure. Riparian vegetation is an important component of the aquatic ecosystem that serves a variety of important functions for habitat structure, water quality, and biological productivity.

Marine riparian vegetation clearly plays a role in stabilizing marine shorelines, particularly bluffs and steep slopes (Brennan and Culverwell 2004; Desbonnet et al. 1994; Lemieux 2004; Myers 1993), but the specific mechanisms are not as well understood as they are in freshwater environments. The extent to which vegetation affects beach and slope stability varies depending on shoreline characteristics and the types of vegetation present (Lemieux 2004; Myers 1993). On steeper slopes, marine riparian vegetation helps to bind the soils and protect against destabilization, slides, and cave-ins that can imperil structures and disrupt the ecology of the nearshore by increasing sedimentation and burying vegetation (Brennan and Culverwell 2004). On shorelines with shallower slopes, marine riparian vegetation dissipates wave energy, thereby reducing erosion and promoting the accumulation of sediments.

Direct and Indirect Effects

The indirect effects on fish and invertebrates as a result of reduced shoreline stability relate to the increased probability of slope failures. Although slope failures occur naturally and are an important process that maintains proper substrate of adjacent beaches (Finlayson 2006), the immediate and unnatural impacts of riparian vegetation removal can adversely affect HCP species. These effects result from the increased turbidity of adjacent waters (Herrera 2006) and the potential burial of invertebrates. Burial of mollusks is discussed at length in Section 7.1.1.2

(*Hydraulic and Geomorphic Modifications*). Effects due to turbidity are discussed in Section 7.1.1.5 (*Water Quality Modifications*).

7.1.1.3.3 Altered Allochthonous Input

Allochthonous inputs of organic material and large wood from marine riparian systems also have demonstrable effects on nearshore habitat conditions. While the importance of allochthonous inputs of litter is not as well documented as the linkages established for freshwater systems, its importance to marine ecosystems is nonetheless apparent (Brennan and Culverwell 2004; Lemieux 2004).

Direct and Indirect Effects

Marine riparian vegetation is a known source of organic matter, nutrients, and macroinvertebrate prey items for HCP species, and the recruitment of these materials is diminished when riparian vegetation is removed or modified (Brennan et al. 2004; Lemieux 2004; Maser and Sedell 1994; Miller et al. 2001; Sobocinski 2003; Williams et al. 2001). Sobocinski (2003) has documented the importance of insect communities and benthic infauna that are either a direct or indirect result of riparian vegetation. These lower trophic organisms serve as the basis of the food web for HCP fish species that use the upper nearshore environment (Williams and Thom 2001).

7.1.1.3.4 Altered Groundwater-Surface Water Exchange

Alteration or removal of riparian vegetation would appreciably change the interface between plants, soil, and water on and near the bank surface. Riparian vegetation acts as a filter for groundwater, removing sediments and taking up nutrients (Knutson and Naef 1997).

Direct and Indirect Effects

While less well studied, similar effects are likely to occur in marine systems as they do in riverine systems. Jetties that require the removal of riparian vegetation may lead to localized increases in substrate temperature due to the loss of cool groundwater flow (Penttila 2001). A more detailed discussion of the loss of groundwater supply due to jetty emplacement in the marine environment is found in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*).

7.1.1.3.5 Altered Habitat Complexity

By maintaining bank stability and contributing large wood to the aquatic environment, riparian vegetation forms and maintains habitat complexity. Driftwood and/or large woody debris (LWD) helps to build and maintain beach habitat structure. Documented LWD functions for beach stability include its contribution to roughness and sediment trapping (Brennan and Culverwell 2004; Gonor et al. 1988) and to inputs of organic matter, moisture, and nutrients that assist in the establishment and maintenance of dune and marsh plants (Williams and Thom 2001). Eilers (1975) found that piles of downed trees in the Nehalem (Oregon) salt marsh trapped enough sediment to support vegetation, whereby marsh islands that trapped sedge seeds provided an elevated substrate for less salt-tolerant vegetation. Herrera (2005) suggested that driftwood at the top of the beach may also slow littoral drift and reduce wave-induced erosion. It has been suggested that estuarine wood can affect water flow and the subsequent formation of bars and mudbanks (Gonor et al. 1988). The beneficial habitat structure functions of LWD along marine shorelines may be maximized if trees that fall perpendicular to beaches typically remain in place. In a recent study, local fallen trees tended to stay in place along Thurston County shorelines (Herrera 2005). The perpendicular alignment of LWD across the beach provides the LWD structure for the widest possible portion of the aquatic habitat, thus maximizing the potential area for sediment trapping and organic matter contributions.

Marine shorelines that have been modified by human activities tend to have less LWD and driftwood than unmodified beaches (Herrera 2005; Higgins et al. 2005). In particular, jetties and groins redistribute LWD such that it concentrates in certain areas and is absent in others (Miller et al. 2001).

<u>Direct and Indirect Effects</u>

The effects of modification of riparian vegetation and the effects on the structural habitat of fish have not been as well studied in marine systems as in freshwater environments. However, Sobocinski (2003) reports that food sources used by salmonids were directly related to the structural complexity provided by natural LWD-laden shorelines. Therefore, it is expected that the loss of such complexity will compromise salmonid food sources. It is uncertain what role structural complexity plays in the life-history cycle of HCP invertebrate species, if any.

7.1.1.4 Aquatic Vegetation Modifications

The basis for nearly all life in the sea is the photosynthetic activity of aquatic autotrophs such as algae, cyanobacteria, benthic microalgae, benthic macroalgae (kelps and seaweeds), and seed plants (such as seagrasses, mangroves, and salt-marsh plants) (Nybakken and Bertness 2005). The availability of light is a crucial parameter for seagrasses and other aquatic autotrophs (Hall et al. 1999), although other factors (e.g., substrate type) can also play a role in the survivability of aquatic plants (Koch 2001).

Aquatic vegetation, in particular eelgrass, is important cover for juvenile fish and invertebrates (Phillips 1984). Eelgrass also provides a necessary structural surface for a community of epibenthic organisms, making eelgrass communities one of the most productive ecotones in the Pacific Northwest (Ferraro and Cole 2007).

Jetty construction can lead to a number of geomorphic modifications that compromise aquatic vegetation productivity. These are generally related to changes in flow energy, which cause vegetation to either be physically removed or buried. These changes are described in detail in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*).

<u>Direct and Indirect Effects</u>

Marine littoral vegetation is important for the colonization of organisms that are important prey resources for HCP species, such as Newcomb's littorine snail, Pacific sand lance, Pacific herring, Pacific cod, northern abalone, surf smelt, steelhead and coastal cutthroat trout, salmon (pink, chum, coho, and Chinook), Olympia oyster, bull trout, Dolly Varden, rockfish, longfin smelt, eulachon; and walleye pollock (Chambers et al. 1999; Gardner 1981; Goetz et al. 2004; Johnson et al. 1999; Larsen et al. 1995; Myers et al. 1998; Pauley et al. 1988; WDNR 2006a, 2006b; West et al. 1994).

In studies on outmigrating juvenile chum in Hood Canal, Simenstad et al. (1980) found juvenile chum fry (1.2–1.8 in [30–45 mm]) feeding extensively upon small, densely distributed harpacticoid copepods, selecting the largest copepods available. Similarly, Miller et al. (1976) reported that juvenile chum fed predominantly on epibenthic harpacticoid copepods. As the fish grew in size, their diet content was composed more of larger epibenthos and pelagic crustaceans. Consistent with other studies, the highest densities of harpacticoid copepods occurred in magnitudes 4–5 times higher in eelgrass stands than in sand habitat without eelgrass. Specific habitat needs vary across species and life-history stages.

Similarly, in a study of the Drayton Harbor marina, Thom et al. (1989) reported that juvenile salmon density was the highest at the eelgrass habitat site that also supported the highest salmon prey density and epibenthos density. Similarly, total fish density increased dramatically immediately following a peak in maximum epibenthos and the most rapid increase in Zostera biomass (Thom et al. 1989). These epibenthic prey assemblages of copepods, such as the harpacticoids, are known to feed on bacteria, epiphytes, plant detritus, and diatoms. It is consistently documented that vegetation assemblages associated with eelgrass, in particular, support increased magnitudes of juvenile salmonid epibenthic prey (Cordell 1986; Simenstad et al. 1980; Thom et al. 1989).

The limitation of habitat for key prey resources likely affects migration patterns and the survival of many juvenile fish species. For smaller fish less than 1.97 in (50 mm) in length, residence times along particular shorelines are thought to be a function of prey abundance (Simenstad et al. 1980).

Eelgrass also plays a role in protecting invertebrates from both fish and avian predators (Bostrom and Mattila 1999). It is uncertain what role eelgrass plays in the protection of HCP invertebrate species, but the generality of the existing work in the field would suggest that a loss of eelgrass would result in increased predation of those species.

7.1.1.5 Water Quality Modifications

The construction and maintenance of jetties and their hydraulic and geomorphic impacts can give rise to a number of water quality modifications. In general, these modifications alter four primary water quality variables in the marine environment: temperature, dissolved oxygen, turbidity, and nutrient and pollutant loading. Pollution associated with the use of treated wood as a building material is also considered. The buffering capacity of seawater is such that the impacts on pH from concrete placement and other leachable building materials are expected to be small (Webster and Loehr 1996) and are not addressed further in the Marine Environments discussion.

7.1.1.5.1 Altered Temperature

Temperature is a primary metric of aquatic ecosystem health, as aquatic organisms have adapted to live within specific thermal regimes. Alterations to these thermal regimes occur at the detriment of local organisms. Thermal stress can occur through multiple direct and indirect pathways in fish and invertebrates. These include direct mortality, altered migration and distribution, increased susceptibility to disease and toxicity, and altered development, spawning, and swimming speeds (Sullivan et al. 2000). Motile organisms have the ability to avoid or evacuate those areas of extreme temperature, but even then the stress induced from periodic exposure and resulting habitat avoidance can affect organism health and contribute to mortality (Groberg et al. 1978). Each of the HCP species is ectothermic (cold-blooded); consequently, temperature is a resource that organisms use for energetic means. With organism metabolism dependent on water temperature, thermal regime may be the single-most important habitat feature controlling aquatic organisms.

Jetties have been documented to reduce the tidal prism and increase stratification on the Columbia River (Sherwood et al. 1990). This would lead to a reduction of mixing in the estuary, which causes artificial increases in the temperature of surface waters (Fischer et al. 1979), and can occur precisely at the time that fishes are most sensitive to temperature stress (i.e., during the summer).

Direct and Indirect Effects on Fish

The majority of research on temperature impacts on aquatic species has focused on salmonids. Different species of salmonids have evolved to use different thermal regimes. Despite these differences, the majority of salmonids prefer the same temperature ranges during most life-history stages. The primary exception to this is that char (bull trout and Dolly Varden) require lower temperatures for optimal incubation, growth, and spawning (Richter and Kolmes 2005). An optimal temperature matrix is presented in Table 7-2; different species have different requirements at various life-history stages. These same temperature ranges have been adopted by Ecology and incorporated into the state water quality standards (WAC 173-201A 2006). Table 7-3 presents highest 7-day average maximum thresholds as promulgated in the state standards.

Table 7-2 indicates that there are water quality thresholds for different life-history stages which are considerably lower than the lethal limit. Fish are susceptible to a number of sublethal effects related to temperature. For instance, elevated but sublethal temperatures during smolting may result in desmoltification, altered emigration timing, and emigration barriers. Temperatures that impair smolting are above a range of between 52 and 59°F (11 and 15°C) (Poole and Berman 2001a; Wedemeyer et al. 1980). Temperatures in this range have been shown to reduce the activity of gill ATPase (McCullough et al. 2001), an enzyme that prepares juvenile fish for osmoregulation in saline waters (Beeman et al. 1994). Temperature-induced decreased gill ATPase has been correlated with loss of migratory behavior in numerous salmonid species (Ban 2006; Marine and Cech 2004; McCormick et al. 1999) and constitutes a significant impairment to juvenile survival.

Table 7-2. Estimates of thermal conditions known to support various life-history stages and biological functions of bull trout (a species extremely intolerant of warm water) and anadromous (ocean-reared) salmon[a].

Consideration	Anadromous Salmon	Bull Trout
Temperature of common summer habitat use	10–17°C (50–63°F)	6–12°C (43–54°F)
Lethal temperatures (1-week exposure)	Adults: >21–22°C (70–72°F)	—
	Juveniles: >23–24°C (73–75°F)	Juveniles: 22–23°C (72–73°F)
Adult migration	Blocked: >21–22°C (70–72°F)	Cued:10–13°C 50–55°F)
Swimming speed	Reduced: >20°C (68°F)	—
	Optimal: 15–19°C (59–66°F)	—
Gamete viability during holding	Reduced: >13–16°C (55–61°F)	—
Disease rates	Severe: >18–20°C (64–68°F)	—
	Elevated: 14–17°C (57–63°F)	—
	Minimized: <12–13°C (54–55°F)	—
Spawning	Initiated: 7–14°C (45–57°F)	Initiated:<9°C (48°F)
Egg incubation	Optimal: 6–10°C (43–50°F)	Optimal: 2-6°C (36–43°F)
Optimal growth	Unlimited food: 13–19°C (55–66°F)	Unlimited food: 12–16°C (54–61°F)
	Limited food: 10–16°C (50–61°F)	Limited food: 8–12°C (46–54°F)
Smoltification	Suppressed: >11–15°C (52–59°F)	—

Source: (Poole et al. 2001)

[a] These numbers do not represent rigid thresholds, but rather represent temperatures above which adverse effects are more likely to occur. In the interest of simplicity, important differences between various species of anadromous salmon are not reflected in this table, and requirements for other salmonids are not listed. Likewise, important differences in how temperatures are expressed are not included (e.g., instantaneous maximums, daily averages, etc.).

Table 7-3. Aquatic life temperature criteria in fresh water[a].

Category	Highest 7-DADMax
Char spawning	9°C (48.2°F)
Char spawning and rearing	12°C (53.6°F)
Salmon and trout spawning habitat	13°C (55.4°F)
Core summer salmonid habitat	16°C (60.8°F)
Salmonid spawning, rearing, and migration	17.5°C (63.5°F)
Salmonid rearing and migration **Only**	17.5°C (63.5°F)
Nonanadromous interior redband trout	18°C (64.4°F)
Indigenous warm water species	20°C (68°F)

Source: (WAC 173-201A 2006) Table 200(1)(c)

[a] Aquatic life temperature criteria. Water temperature is measured by the 7-day average of the daily maximum temperatures (7-DADMax). Table 200(1)(c) lists the temperature criteria for each of the aquatic life use categories.

Elevated water temperatures can impair adult migration. Adult migration blockages occur consistently when temperatures exceed 69.8–71.6°F (21–22°C) (Poole and Berman 2001a). Elevated temperature regimes also affect salmonid species by altering behavior and reducing resistance to disease and toxic substances. Studies have indicated that under chronic thermal exposure conditions, aquatic organism susceptibility to toxic substances may increase. Because elevated temperatures increase metabolic processes, gill ventilation also rises proportionately (Heath and Hughes 1973). Black et al. (1991) showed that an increase in water flow over the gills, which results from increased gill ventilation at increased temperature, resulted in rapid uptake of toxicants, including metals and organic chemicals, via the gills. Salmonids also become more susceptible to infectious disease at elevated temperatures (57–68°F [14–20°C]) because immune systems are compromised (Harrahy et al. 2001), while bacterial and viral activity is accelerated (Tops et al. 2006). In nearshore areas where temperature (as well as pollutant levels) may be elevated, the combined effect of thermal and water pollution may be a primary driver of salmonid decline.

Direct and Indirect Effects on Invertebrates

Research defining thermal criteria for invertebrates is limited, although marine invertebrates can generally withstand higher temperatures. Gagnaire et al. (2006) noted that elevated temperatures caused blood cell mortality in Pacific oysters but not until temperatures exceeded 104°F (40°C), which is unlikely even in altered settings. It is unclear, however, what sublethal effect(s) may be a significant factor with invertebrate populations.

7.1.1.5.2 Altered Dissolved Oxygen

Dissolved oxygen (DO) content is critical to the growth and survival of all 52 HCP species. The amount of oxygen dissolved in water is dependent on temperature, physical mixing, respiration, photosynthesis, and, to a lesser degree, atmospheric pressure. These parameters can vary diurnally and seasonally and depend on activities such as daytime photosynthesis oxygen inputs and night-time plant respiration processes that deplete dissolved oxygen levels. Dissolved oxygen concentration is temperature dependent; as temperatures rise, the gas-absorbing capacity of the water decreases and the dissolved oxygen saturation level decreases. Reduced dissolved oxygen levels can be due to increased temperature (Snoeyink and Jenkins 1980), organic or nutrient loading (Ahearn et al. 2006), increased benthic sedimentation (Welch et al. 1998), or chemical weathering of iron and other minerals (Schlesinger 1997).

Depressed dissolved oxygen is most commonly associated with increased loading of carbon and the associated increase in biochemical oxygen demand (BOD). Increased BOD is frequently associated with eutrophication brought about by increased nutrient loading and solar radiation. Nutrient cycling has been closely linked to nearshore stratification and circulation (Roegner et al. 2002). By altering tidal exchange and nearshore stratification (Sherwood et al. 1990), jetties have the potential to isolate and concentrate BOD (Fischer et al. 1979).

Direct and Indirect Effects on Fish

Juvenile salmon are highly sensitive to low dissolved oxygen concentrations (USFWS 1986) and, consequently, are among the more vulnerable HCP species with regard to dissolved oxygen impairment. Salmon generally require dissolved oxygen levels of greater than 6 ppm for optimal survival and growth, with lethal one-day minimum concentrations of around 3.9 ppm (Ecology 2002). Different organisms at different life-history stages require different levels of dissolved oxygen to thrive. Table 7-4 lists the minimum recommended dissolved oxygen concentrations for salmonids and stream-dwelling macroinvertebrates (Ecology 2002). The dissolved oxygen thresholds presented in this table were derived from more than 100 studies representing over 40 years of research.

Table 7-4. Summary of recommended dissolved oxygen levels for full protection (approximately less than 1 percent lethality, 5 percent reduction in growth, and 7 percent reduction in swim speed) of salmonid species and associated macroinvertebrates.

Life-history Stage or Activity	Oxygen Concentration (ppm)	Intended Application Conditions
Incubation through emergence	>9.0–11.5 *(30 to 90-DADMin)* and No measurable change when waters are above 52°F (11°C) (weekly average) during incubation.	Applies throughout the period from spawning through emergence Assumes 1-3 ppm will be lost between the water column and the incubating eggs
Growth of juvenile fish	>8.0–8.5 *(30-DADMin)* >5.0-6.0 *(1-DMin)*	In areas and at times where incubation is not occurring
Swimming performance	>8.0-9.0 *(1-DMin)*	Year-round in all salmonid waters
Avoidance	>5.0-6.0 *(1-DMin)*	Year-round in all salmonid waters
Acute lethality	>3.9 *(1-DMin)* >4.6 *(7 to 30-DADMin)*	Year-round in all salmonid waters
Macroinvertebrates *(stream insects)*	>8.5-9.0 *(1-DMin or 1-DAve)*	Mountainous headwater streams
	>7.5-8.0 *(1-DMin or 1-DAve)*	Mid-elevation spawning streams
	>5.5-6.0 *(1-DMin or 1-DAve)*	Low-elevation streams, lakes, and nonsalmonid waters
Synergistic effect protection	>8.5 *(1-DAve)*	Year-round in all salmonid waters to minimize synergistic effect with toxic substances

Source: Ecology 2002.
1-DMin = annual lowest single daily minimum oxygen concentration.
1-DAve = annual lowest single daily average concentration.
7-, 30-, or 90-DADMin = lowest 7-, 30-, or 90-day average of daily minimum concentrations during incubation period, respectively.

It should be noted that recommendations are presented in Table 7-4 for dissolved oxygen thresholds in categories other than lethality. Fish are motile organisms and, where possible, will avoid dissolved oxygen levels that would cause direct mortality. However, this avoidance behavior in and of itself can affect fishes. Stanley and Wilson (2004) found that fish aggregate above the seasonal hypoxic benthic foraging habitat in the Gulf of Mexico, while Eby et al. (2005) found that fish in the Neuse River estuary (North Carolina) were restricted by hypoxic zones to shallow, oxygenated areas, where in the early part of the summer about one-third fewer prey resources were available. Studies such as these reveal how dissolved oxygen can change fish distributions relative to habitat and potentially exclude fishes from reaching foraging and rearing areas. Sublethal dissolved oxygen levels can also cause increased susceptibility to infection (Welker et al. 2007) and reduced swim speeds (Ecology 2002), both of which may cause indirect impacts on HCP fish species.

Direct and Indirect Impacts on Invertebrates

Little consensus exists concerning low dissolved oxygen criteria for macroinvertebrates, and tolerances to hypoxic conditions are taxonomically specific. Many invertebrates are adapted to live in benthic low-energy environments where dissolved oxygen concentrations are naturally low; consequently, these organisms can withstand hypoxic conditions. Other taxa, including Hirudinea, Decapoda, and many aquatic insects, tolerate dissolved oxygen levels below 1.0 ppm (Hart and Fuller 1974; Nebeker et al. 1992). Kaller and Kelso (2007) found benthic macroinvertebrate density, including mollusks, greatest in low dissolved oxygen areas of a Louisiana wetland, while a literature review by Gray et al. (2002) found that in marine environments, invertebrates were not affected by low dissolved oxygen until concentrations fell below 1–2 ppm. Benthic dissolved oxygen levels can seasonally drop below this threshold in productive systems that receive high BOD loadings. For instance, depressed benthic dissolved oxygen levels in Hood Canal, Washington, have been associated with spot shrimp decline (Peterson and Amiotte 2006). This dissolved oxygen decline in turn has been linked to BOD loadings from leaking or improperly functioning onsite wastewater systems. These conditions in Puget Sound highlight the importance of reducing anthropogenically generated BOD. While most BOD increases can be attributed to other human activities, the increased stratification brought about by shoreline modifications (such as jetties) to tidal flows (Sherwood et al. 1990) and subsequent alterations in nutrient loading can have a profound effect on nearshore productivity and diversity (Roegner et al. 2002).

7.1.1.5.3 Altered Suspended Solids and Turbidity

Jetties have the potential to increase suspended solids by causing the shoreline to become out of equilibrium with local sediment supply and transport. These modifications are described in detail in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*). In general, the response of aquatic biota to elevated suspended solids concentrations is highly variable and dependent upon life-history stage, species, background suspended solids concentrations, and ambient water quality. Appendix A provides summary tables of the effects of suspended solids on fish and invertebrates. The following sections provide detailed information on some of those findings.

Effects of Suspended Solids on Fish

Lethal Effects

Although juveniles of many fish species thrive in rivers and estuaries with naturally high concentrations of suspended solids, studies have shown that suspended solids concentration (as well as the duration of exposure) can be important factors in assessing risks posed to salmonid populations (McLeay et al. 1987; Newcombe and MacDonald 1991; Servizi and Martens 1987). Lake and Hinch (1999) found concentrations in excess of 40,000 ppm suspended solids to elicit stress responses in juvenile coho salmon. Suspended solids concentrations this high would likely only be associated with construction activities. However, other studies have shown lethal effects at much lower concentrations.

Servizi and Martens (1991) exposed juvenile coho salmon to natural Fraser River suspended solids and found a 96-hour LC_{50} (the concentration at which 50 percent population mortality was observed) of only 22,700 ppm. Using the identical apparatus and sediment source, juvenile sockeye salmon had a 96-hour LC_{50} of 17,600 ppm (Servizi and Martens 1987), and juvenile Chinook salmon had an LC_{50} of 31,000 ppm (Servizi and Gordon 1990).

Sublethal Effects

Studies on a variety of fishes, including sockeye and Chinook (Newcomb and Flagg 1983), coho, four-spine stickleback, cunner, and sheepshead minnow (Noggle 1978), attribute chronic and acute impacts from high suspended solids to reduced oxygen uptake (Wilber and Clarke 2001). Fish must keep their gills clear for oxygen exchange. In the presence of high loadings of suspended solids, they engage a cough reflex to perform that function. Due to increased metabolic oxygen demand with increased temperatures and the need to keep pathways free of sediments for oxygen uptake, increased temperature and reduced oxygen levels combine to reduce the ability of fish to cough and maintain ventilation rates. The stress induced by these conditions can lead to compromised immune defenses and reduced growth rates (Au et al. 2004). Sigler et al. (1984) noted reduced growth rates in juvenile steelhead and coho salmon at suspended solids concentrations as low as 100 ppm, while Servizi and Martens (1992) noted increased cough frequency in juvenile coho at concentrations of approximately 240 ppm.

Behavioral Effects

Aksnes and Utne (1997), Mazur and Beauchamp (2003), and Vogel and Beauchamp (1999) all report that suspended solids at sublethal concentrations have been shown to affect fish functions such as avoidance responses, territoriality, feeding, and homing behavior. Similarly, Wildish and Power (1985) reported avoidance of suspended solids by rainbow smelt and Atlantic herring to be at 20 ppm and 10 ppm, respectively. However, it also appears that under certain circumstances elevated suspended solids may actually benefit salmonids by providing cover (Gregory and Levings 1998) or triggering a sense of predation cover for salmonids (Gregory 1993). The studies of Gregory (1993) indicated that when suspended solids concentrations exceeded 200 ppm, juvenile salmon increase their feeding rates while demonstrating pronounced behavioral changes in prey reaction and predator avoidance.

Habitat Effects

Increased turbidity is known to compromise the survivalability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. It can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). Eelgrass is associated with important rearing habitats for a suite of marine fishes including Pacific cod, Pacific salmon, rockfish, Pacific herring, and walleye pollock (Nightingale and Simenstad 2001a; Simenstad et al. 1999).

Effects of Suspended Solids on Invertebrates

As with dissolved oxygen, invertebrates tend to thrive across a wide range of suspended solids concentrations. Negative impacts on eastern oyster egg development have been shown to occur at 188 ppm total suspended solids (Cake 1983). Hardshell clam eggs appear to be more resilient, with egg development affected only after total suspended solids concentrations exceeded 1,000 ppm (Mulholland 1984). Mulholland (1984) showed that suspended solids concentrations of <750 ppm allowed for continued larval development but higher concentrations for durations of 10–12 days showed lethal effects for both clams and oysters. Although it is unlikely that suspended solids would exceed 750 ppm strictly from the installation of a jetty, other changes made in association with jetty development could cause this limit to be exceeded. Therefore, suspended solids effects on invertebrates should be considered based on the sum of activities associated with jetty installation.

There is no clear pattern of sublethal effects of elevated suspended sediment on aquatic invertebrates. For bivalves, when suspended solids concentrations rise above their filtering capacities, their food becomes diluted (Widdows et al. 1979). Studies have shown that the addition of silt, in relatively low concentrations in environments with high algal concentrations, can be marked by increased growth of mussels (Kiorboe et al. 1981), surf clams (Mohlenberg and Kiorboe 1981), and eastern oysters (Urban and Langdon 1984). Bricelj and Malouf (1984), however, found that hardshell clams decreased their algal ingestion with increased sediment loads, and no growth rate differences were observed between clams exposed to algal diets alone and clams with added sediment loads (Bricelj et al. 1984). Urban and Kirchman (1992) reported similarly ambiguous results concerning suspended clay. Suspended clay (20 ppm) interfered with juvenile eastern oyster ingestion of algae, but it did not reduce the overall amount of algae ingested. Grant et al. (1990) found that the summer growth of European oysters was enhanced at low levels of sediment resuspension and inhibited with increased deposition. It was hypothesized that the chlorophyll in suspended solids may act as a food supplement that could enhance growth, but higher levels may dilute planktonic food resources, thereby, suppressing food ingestion. Changes in behavior in response to sediment loadings were also noted for soft-shelled clams in sediment loads of 100–200 ppm, with changes in their siphon and mantles over time (Grant and Thorpe 1991).

In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted that seagrass species richness and community leaf biomass declined sharply with a 15 percent increase in clay content of the sediments. Numerous studies have shown increased biomass of invertebrate (Cardoso et al. 2007; Seitz et al. 2005) and vertebrate species (Ferraro and Cole 2007; Pihl et al. 2006) in association with seagrass presence; thus, sedimentation-related negative impacts on seagrass arising from the construction or presence of jetties would likely affect the HCP species by decreasing available nearshore habitat.

7.1.1.5.4 Altered Nutrient and Pollutant Loading

Jetties can alter nutrient and pollutant loading indirectly by modifying and restricting nearshore circulation. In particular, jetties have the potential to increase the tide range to the extent that previously unflooded areas are then inundated by marine waters. For a complete discussion of how this occurs, see Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*). Jetties (or their analogs) may also directly contribute to metals loading, depending on the construction materials used.

The processing and retention of sediment, nutrients, and pollutants in aquatic systems is also accelerated by the presence of aquatic vegetation (Clarke 2002). Numerous studies have shown that macrophytes and algae in marine environments act to reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). Seagrasses have also been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. But aquatic vegetation does not only reduce nutrient and suspended sediment concentrations; the plants themselves can sequester harmful trace metal pollutants and are frequently planted in wetland treatment systems with that intended function. In a comparative study of heavy metal uptake in terrestrial, emergent, and submerged vegetation, Fritioff and Greger (2003) noted that submerged vegetation was efficient at removing zinc, copper, cadmium, and lead from influent stormwater.

Any activity that mechanically removes or by other means affects aquatic vegetation will reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. With this reduction comes increased loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity.

<u>Direct and Indirect Impacts on Fish and Invertebrates</u>

Eutrophication

Eutrophication occurs when limits to vegetative growth are reduced. In Washington, iron preferentially binds with sulfide, and the associated phosphorus is released; this creates conditions of nitrogen limitation (Blomqvist et al. 2004). When nutrient limitations are eliminated, vegetative growth increases. This process accelerates carbon fixation; the additional carbon loading to the aquatic system increases respiration as heterotrophs use carbon for energy. Through the process of carbon oxidation, oxygen is converted to carbon dioxide (CO_2), and ambient dissolved oxygen levels decrease. Eutrophication-induced hypoxia is a nationwide problem (Scavia and Bricker 2006). In Washington, low dissolved oxygen episodes in Hood Canal have resulted in widespread fish and invertebrate kills (Peterson and Amiotte 2006). These low dissolved oxygen episodes have been linked to excess carbon loading due to nutrient enrichment. Resultant algal blooms may not only impact dissolved oxygen levels but also, if certain species flourish, contribute to paralytic shellfish poisoning (Horner 1998). The ramifications of low dissolved oxygen on the 52 HCP species are addressed in Section 7.1.1.5.2 (*Altered Dissolved Oxygen*) above and in Section 9 (*Potential Risk of Take*).

Metal Toxicity

In urban environments, metals loading to local waterways and water bodies from anthropogenic sources is a major pathway for aquatic habitat degradation. The primary metals of concern in the surface waters of Washington are copper, zinc, arsenic, lead, and nickel (Embrey and Moran 2006). Metals above threshold concentrations act as carcinogens, mutagens, and teratogens in fish and invertebrates (Wohl 2004). Additionally, the sublethal effects of copper toxicity have been extensively studied, with reported effects including impaired predator avoidance and homing behavior (Baldwin et al. 2003). Ecology has established water quality standards for marine waters for each of these constituents. These standards, issued in WAC 173-201a and listed in Table 7-5. The median concentrations were estimated based on an extensive 3-year data set (2001–2003) from the Green River watershed (Herrera 2007b).

Table 7-5. Water quality criteria for metals in marine waters of the state of Washington.

Constituent	Acute (ppb)	Chronic (ppb)
Arsenic	69	36
Copper	4.8	3.1
Lead	210	8.1
Nickel	74	8.2
Zinc	90	81

Source: WAC 173-201A.

7.1.1.5.5 Treated Wood Pollution

Creosote and other wood preservative products used on jetties also pose additional water quality and sediment contamination risks associated with contaminant leaching. The current state of knowledge on the biological effects of creosote-treated wood routes of exposure have been summarized in three major literature reviews: Meador et al. (1995) addressed the bioaccumulation of PAHs in marine fishes and invertebrates; Poston (2001) reviewed treated wood impacts on aquatic environments; and two Stratus documents (2005a, b) presented what is known about the impacts of creosote, chromated copper arsenate (CCA), and ammoniacal copper zinc arsenate (ACZA) treated wood products. The major routes of exposure for marine animals were found to be through the uptake of waterborne chemicals, including the interstitial water of sediments and through trophic transfer; while the direct uptake of sediment-bound chemicals appeared to be negligible (Meador et al. 1995).

Creosote, a distillate of coal tar, can include polycyclic aromatic hydrocarbons (PAHs), alkyl-PAHs, tar acids, phenolics, tarbases/N-heterocyclics (quiolines and carbazoles), S-heterocyclics (thiophenes), O-heterocyclics/furans (dibenzofuran), and aromatic amines (such as aniline). However, 85–90 percent of the mass of creosote is comprised of PAHs. Stratus (2005a) evaluated results from laboratory tests on the leaching of PAHs from creosote-treated pilings. Leaching rates in fresh and salt water both increased with higher water temperatures. In a study of aging effects on leaching (Ingram 1982), it was found that 12 years of field installation in seawater appeared to have reduced leaching rates by only 25 percent. Kang et al. (2003) determined leach rates in fresh water for two flow rates (0.5 and 1.3 in/sec [1.2 cm/sec and 3.3 cm/sec]). The 1.3 in/sec (3.3 cm/sec) flow rate was associated with double the leaching of the 0.5 in/sec (1.2 cm/sec) flow rate. Xiao et al. (2002) found the greatest leaching rates to occur in warm, turbulent water. Poston (2001) reviewed 20 years of research on creosote-treated wood and found that the greatest risks to water quality from creosote-treated wood were in the leaching of trace metals and PAHs over time, with lighter-weight PAHs degrading rapidly and higher-weight PAHs contributing to chronic contamination, particularly to surrounding sediments.

Several models have been developed to estimate PAH leaching rates from creosote-treated wood (Brooks 2004; Poston et al. 1996; Xiao et al. 2002). The models attempt to describe complex interactions and generally rely heavily on site-specific data and assumptions (Stratus 2005a). Evaluations of the CREOSS model (Brooks 2004) and the box plume model (Poston et al. 1996) have shown that although they may not fully explain transient concentrations, such as those immediately following installation or severe disturbance such as abrasion, they are helpful in qualitatively describing the effect of many factors, such as salinity, temperature, wood density, water circulation, surface area to volume ratio, wood grain direction, time from treatment, and whether the wood was treated using BMPs to reduce leaching rate (Stratus 2005b).

Areas of lower pH and reduced water circulation are at a greater risk of contamination. Metals from creosote-treated wood generally become incorporated into the local sediments and are usually undetectable in ambient waters. Stratus (2005a) also reported that a number of jurisdictions have recently put prohibitions in place on the use of creosote-treated wood.

Creosote-treated Wood—Direct and Indirect Effects

Many studies have investigated thresholds for biological effects of PAH concentrations in sediment, but the greatest impacts of creosote-treated wood affect benthic and burrowing organisms on the treated wood structures. Several effects thresholds have been determined using many years of NOAA Fisheries data on the effects of PAH-contaminated sediments on benthic fish in Puget Sound (Stratus 2005a). Thresholds for effect on English sole were determined at 230 ppb for proliferated liver lesions; 630 ppb for spawning inhibition, infertile eggs, and abnormal larvae; and 288 ppb for DNA damage (Johnson et al. 2002). Pacific herring have been shown to have reduced hatching success at PAH concentrations as little as 3 ppb, while 50 percent of the eggs in the same study were viable at concentrations of 100 ppb (Vines et al. 2000).

An important consideration in the analysis of thresholds for biological effects of PAH is potential additive effects. Additive effects of chemicals could have greater detrimental impacts on species than what has been shown to occur by analyzing the effects of chemicals independent of one another.

ACZA and CCA Treated Wood—Direct and Indirect Effects

CCA treatment contains chromium, copper, and arsenic acid. These water-soluble treatments are used to protect wood from wood-boring organisms and fungi. Stratus (2005b) reviewed and evaluated models developed to predict the leaching of metals from treated wood. The Stratus (2005b) review concluded that the chemical processes associated with the chemical fixing process are complex and poorly understood (Lebow and Tippie 2001). Stratus (2005b) found the most important factors affecting the leaching rates of metals from treated wood to be: (1) the metal being considered (Cu, Cr, As, or Zn); (2) post-treatment procedures used to fix the treatment chemical and remove excess treatment solution; (3) duration of post treatment exposure of water; (4) loading or retention of treatment solution in the wood; (5) ambient water quality conditions (including salinity, pH, and temperature); (6) current speed; and (7) wood surface physical features (including surface area-to-volume ratio). Lebow and Tippie. (2001) found that water-repellent stain, latex paint, or oil-based paint greatly reduced arsenic, chromium, and copper leaching rates. Stratus (2005b) compared the applicability of laboratory studies to field conditions and concluded that much higher leaching rates are likely to occur in the field than what is observed in laboratories. However, leaching may not be the primary pathway for contaminant transfer into local food webs.

A study by Brooks (2004) on the Olympic Peninsula found insignificant increases in arsenic, copper, and zinc in sediments and water at three out of four pier sampling sites and minimal uptake by shellfish. Weiss et al. (1993), however, found that oysters growing on CCA-treated wood piles had higher metals concentrations and a greater incidence of histopathological lesions compared to oysters collected from nearby rocks. In a subsequent study, Weis and Weis (1996) fed snails algae grown on CCA-treated docks. The snails in turn suffered mortality. Finally, Weis and Weis (1994) found significantly lower biomass and diversity of sessile epifaunal communities on treated wood panels compared to untreated panels. Studies such as these indicate that the primary trophic pathway for contaminants from treated wood is through invertebrates and algae either growing on or attached to treated wood.

Stratus (2005b) reported that invertebrates in both marine and freshwater environments are the most sensitive to copper, chromium, zinc, and arsenic levels. They concluded that the U.S. Environmental Protection Agency (USEPA) Aquatic Life Criteria (ALC) limits appear to be appropriate to prevent acute lethal impacts of copper and chromium but do not eliminate avoidance responses and olfactory neurotoxicity that salmonids may experience with even brief exposures to sublethal copper concentrations. Metals from treated wood can contaminate sediment and affect benthic communities, which limits food resources for fish and exposes fish to metals contamination through the consumption of contaminated prey (Stratus 2005b).

7.1.1.6 Ecosystem Fragmentation

The construction and maintenance of a jetty can limit the accessibility of fish through changes in both the structure of the shoreline and its characteristics (linear and impermeable versus undulating and covered in large woody debris). This can take the form of complete removal of habitats, such as the elimination of lagoon habitats due to changes in alongshore sediment transport patterns or the tidal prism. However, habitat accessibility can also be compromised if characteristics, such as having different hypsometry (distribution of depths) or elimination of large woody debris, make the shoreline inhospitable to the HCP species.

7.1.1.6.1 Habitat Loss and Fragmentation

A number of processes associated with the installation of a jetty could lead to a loss or fragmentation of existing habitat. These include a loss of accessibility to coastal lagoons, a loss of substrate and appropriate depth necessary for spawning, and a change in the abundance of predators and prey.

Due to a number of hydrogeomorphic changes detailed in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications)*, coastal lagoons could be made inaccessible by the construction of a jetty through a reduction of the tidal prism.

When a shoreline is hardened, erosion down to an underlying coarse cobble lag (i.e., a deposit of coarse cobbles [1–5 inches in diameter] left over from erosion in the geologic past) is possible (Finlayson 2006; Herrera 2005). The result is a nearly vertical shoreline at the location of the hardened structure, often colocated with or near the ordinary high water mark (OHWM).

Jetties and other hard shoreline structures have the potential to both attract and deter use of the nearshore by fish, birds, and people. As a result, the complex interaction between people, birds, and fish can lead to a net loss of HCP species (Roby et al. 2002).

<u>Direct and Indirect Effects</u>

Jetties have been shown to be a locus for shorebird activity, mostly as a result of the availability of food from planktonic breeders (Botton et al. 1994). Pacific herring have been shown to create similar concentrations of shorebirds in natural settings on the western shore of British Columbia (Rodway et al. 2003). The concentrations of spawn and shorebirds, however, were associated with hard-substrate outcrops, similar to artificial hard points commonly associated with jetties. Although these concentrations can occur in the absence of man-made structures, the unnatural concentration of spawn on jetties would artificially enhance these processes. The degree to which the concentration of spawn and shorebirds affects the mortality of spawn and the dwindling numbers of Pacific herring or other planktonic breeders is unknown (Rodway et al. 2003).

Jetties have been shown to concentrate juvenile Chinook salmon in some areas due to strong changes in salinity (Yates 2001). Similar features (groins) have been shown to concentrate chum and coho salmon (Miller et al. 2001), while protruding shoreline structures (i.e., analogous to jetties) have been shown to concentrate juvenile salmonids and all of the HCP forage fish species (Toft et al. 2007). These concentrations likely do not reflect an increase in production of these fishes, but rather a concentration due to bottlenecks in the accessibility to nearshore habitat. Because these concentration mechanisms are exploited by commercial fishermen (Creque et al.

2006; Miller et al. 2001), the risk of increased catch due to jetty installation likely has a negative effect on salmonid survival and navigation near such activities.

The interaction between fishes near man-made structures is more complex than in natural settings and has only been studied in relation to low-crested structures (i.e., breakwaters) and artificial reefs. Jetties likely exhibit some of these impacts, but because the work has been conducted in association with breakwaters, a full discussion of these effects is in Section 7.2.1.6 (*Ecosystem Fragmentation*)

Although erosion associated with hardening of the shoreline is often ephemeral, when the sediment supply is not maintained (Finlayson 2006), it presents the possibility of the elimination of elevations necessary for the proper wave and sediment transport conditions for the survival of surf smelt and sand lance larvae, concomitant with a loss of appropriate spawning substrate (substrate effects are discussed in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*). It is less clear what effects the loss of the upper foreshore would be on HCP invertebrates such as the Olympia oyster.

Lagoons provide important rearing habitat for juvenile salmonids, including Chinook (Busby and Barnhart 1995) and coho (Minakawa and Kraft 2005) as well as Pacific herring (Saiki and Martin 2001), and can be lost due to changes in the tidal prism (Sherwood et al. 1990). Lagoons, sometimes referred to as pocket estuaries, have declined both in terms of size and number due to human modifications to lowland coastal areas in Puget Sound (Beamer et al. 2005). In fact, access to high-quality lagoon habitat has been shown to be the critical path in the restoration of Chinook in the Skagit River system (Beamer et al. 2005). The primary impact of a loss of lagoon habitat would be to expose juvenile salmonids and forage fish to increased risk of predation (Hood 2006; Wagner and Austin 1999).

The Olympia oyster also uses lagoons; if a jetty were placed near or adjacent to a lagoon, the species could be susceptible to losses due to changes in tidal prism (Baker 1995).

7.1.1.6.2 Loss of Large Woody Debris (LWD)

Driftwood and other LWD help build and maintain beach habitat structure in marine environments. As addressed in detail in Section 7.1.1.3.5 (*Altered Habitat Complexity*), documented LWD functions for beach stability include its contribution to roughness and sediment trapping as well as to inputs of organic matter, moisture, and nutrients that assist in the establishment and maintenance of dune and marsh plants. (For more details, see that section.)

In addition, marine shorelines modified by human activities tend to have less LWD and driftwood than unmodified beaches. Marine shorelines modified by human activities tend to have less LWD and driftwood than unmodified beaches (Herrera 2005; Higgins et al. 2005). This occurs through several mechanisms. For example, MacDonald et al.(1994) reported that shoreline armoring limited driftwood accumulation on a beach. Higgins et al. (2005) suggested that the mechanisms for the apparent reduction in LWD appeared to be the removal of adjacent riparian vegetation during and following placement of bank protection; reduced shoreline roughness at armored sites which causes more LWD to be transported away; and limited upper intertidal and backshore areas that allow for LWD deposition above tidal elevations that are routinely inundated. Because LWD is used in some marine soft-shore armoring applications to attenuate wave energy and lessen the potential for erosion, it is assumed that naturally occurring LWD on beaches would do the same, but this has not been empirically tested. Herrera (2005) described how multiple layers of LWD along a shoreline could provide effective energy dissipation, decreasing the amount of wave reflection during high water levels by increasing the roughness of the shoreline and by decreasing its slope relative to a vertical bulkhead.

<u>Direct and Indirect Effects</u>

Although little evidence has shown the direct impacts of the loss of large woody debris on HCP species on marine shorelines, differences in the survivability and diversity of lower trophic species near accumulations of woody debris on marine shorelines has been documented (Storry et al. 2006). Because one of the sources of LWD on marine shorelines is riparian vegetation, the loss of riparian vegetation would represent a loss of productivity on the upper foreshore and may partially explain the dramatic difference in supratidal productivity between modified and unmodified shorelines (Sobocinski 2003). Riparian vegetation can also help accumulate beach wrack by providing a complex surface to collect and retain loose marine debris (see Figure 7-3).

Figure 7-3. Riparian vegetation recruitment of LWD on an undisturbed shoreline in central Puget Sound.

7.1.1.6.3 Lost Opportunities

A potential impact of jetties is related to the loss of opportunities. "Lost-opportunity impacts" result from projects that adversely alter natural nearshore processes important to the ongoing creation of fish and wildlife habitats. Jetties restrict natural geomorphic processes along the shoreline and often fix the location of estuarine exchange (e.g., at the Columbia River mouth). These geomorphic changes will persist for the design-life of the structure and can impose significant impacts on fish and invertebrates.

Likewise, jetties can impose lost-opportunity costs through a number of pathways. Jetties may pose lingering ecological effects when they are not designed properly to account for movement of sediment and wood alongshore. An equally important type of lost-opportunity cost occurs when the structures prohibit migration of fish and invertebrate species or life-history stages. In such instances, the costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly. This may delay the actions necessary to protect the fish migration corridor, perhaps as long as the design-life of the underperforming structure.

7.1.2 Lacustrine Environments

Lacustrine ecosystems, pertaining to lakes, are generally defined by nonflowing inland waters impounded in a topographic depression either by natural or anthropogenic processes without a direct connection to the sea. Physical characteristics such as wave energy, circulation and mixing, sediment types, and water properties (temperature, turbidity, nutrient levels, and dissolved oxygen content) are important habitat-controlling factors in lacustrine environments. As these factors determine the distribution of species in these systems, changes in these factors can, both directly and indirectly, affect the successful reproduction, growth, and survival of the diverse fish and invertebrate species endemic to these habitats.

One of the unique impacts on habitat in lacustrine environments is associated with large, long-term, water-level fluctuations. These can be related to natural hydrologic changes; often on Washington's largest lakes (e.g., Lake Washington), these fluctuations are produced by human manipulation of inlets and outlets. The effects of such manipulations can manifest in a manner similar to natural changes and may complicate any assessment of impacts arising from human activities (Wilcox et al. 2002).

7.1.2.1 Construction and Maintenance Activities

The manner by which jetties are constructed in lacustrine environments is similar to that used in marine environments. The mechanisms of impact are considered to be nearly identical to those presented in Section 7.1.1 (*Marine Environments*).

7.1.2.2 Hydraulic and Geomorphic Modifications

The submechanisms of impact of jetties in lacustrine environments are essentially the same as described for marine environments. In both environments, wave energy and sediment recruitment and transport are altered. However, in lakes these impacts are often exacerbated by differences in human-controlled water-level variability (in the case of reservoirs) and natural lake limnology (Wilcox et al. 2002). This inherent variability makes the differences between natural lakes and reservoirs less pronounced with respect to nearshore processes. However, there are other geomorphic differences with pronounced effects on habitat (discussed at length below).

Regardless of whether a lake is natural or not, the same mechanisms of impact in the marine environment (i.e., altered wave energy, altered current velocities, altered nearshore circulation, altered groundwater-surface water exchange, altered substrate composition, and altered sediment supply) may be considered to occur in lakes, albeit on a different suite of species. Systematic studies of impacts on the habitat in western Washington lakes are extremely limited (Jones & Stokes 2006). Some analysis of habitat types and species distributions has been prepared as part of the development of various shoreline master programs [e.g., (The Watershed Company 2006)], but these typically only catalog species and activity types and do not provide information about their relation to one another. Therefore, the physical processes discussed in depth in Section 7.1.1 (*Marine Environments*) are considered to be relevant here, recognizing that some differences occur (mostly apparent from previous work performed in the Great Lakes). To summarize, the four most important differences between marine and lacustrine environments are the following:

- *Nearshore circulation and water column characteristics*. Nearshore circulation on a lakeshore is fundamentally different than in marine environments. First, wave energy is so small that wave-induced nearshore circulation is negligible (see below); however, wind does play an important role in driving wholesale circulation of the lake (Rao and Schwab 2007). Temperature is the dominant factor in maintaining stratification (unlike in the marine environment, where salinity is typically the most important water column constituent). Finally, the absence of tides means that water level on the time scale of hours to days is stable, and the distribution of wave energy on the shoreline is discrete.

- *Groundwater input*. Because lakes are fundamentally connected (typically phosphorus) to upland environments, the limiting nutrients in a lake are different than in a marine environment (typically nitrogen). Just as in marine environments, benthic productivity and diversity have been linked to groundwater effluent (Hagerthey and Kerfoot 2005; Hunt et al. 2006). However, lacustrine seeps, have high productivity but low species diversity (Hagerthey and Kerfoot 2005; Hunt et al. 2006), as opposed to marine seeps, which are both productive and diverse (Kelley et al. 2002).

- *Short-period waves*. Because lakes are confined, all of their wave energy must be generated from local winds. This makes all of the waves fetch-limited (Komar 1998). Fetch-limited waves have extremely short periods and small wave heights compared to their open, marine counterparts. In this sense, lacustrine littoral processes are similar to those found in Puget Sound (Finlayson 2006).

- *Lack of buffering capacity*. Construction materials used in lacustrine jetties typically have a more pronounced effect on lake water quality because of the relative lack of buffering capacity in lake waters (Webster and Loehr 1996). See Section 7.3.2.5 (*Water Quality Modifications*) for details.

Special Considerations for Reservoirs

Human-operated reservoirs present other special issues. Reservoirs are morphologically, biologically, and hydrologically dissimilar from natural lakes. Morphologically, lakes are often deepest near the middle, while reservoirs are typically deepest at the downstream end. This difference has implications for current strength and direction. The plan view of reservoirs can be quite variable depending on the degree of confinement, but the length of the shoreline is often longer than that of a natural lake. In addition, the extent of shoreline development is much greater than in natural lakes because annual drawdown exposes a larger area to shore processes (Baxter 1977). The location and nature of depositional forms are highly variable with reservoir morphometry, incoming sediment load, and reservoir operation. Reservoirs are also subject to density or turbidity currents resulting from differences in temperature or sediment concentration between inflows and reservoir waters (Snyder et al. 2006). Mixing zones between the water sources influence the usage of reservoir areas by fishes.

Reservoir environments can lack natural habitat due to the loss of riparian forest as a result of flooding, siltation of rocky shorelines, and a paucity of aquatic vegetation resulting from fluctuating water levels (Prince and Maughan 1978). Depending on reservoir operations, drawdown and filling cycles can re-entrain silty deposits in littoral areas. When jetties, barbs, or breakwaters are constructed, the combined footprint of fill materials and pilings obliterates physical habitat and can exacerbate the degradation of littoral areas. The footprint of a jetty also displaces substrate and aquatic vegetation. Littoral areas characterized by submerged and emergent macrophytes are preferred by juvenile fishes (Bryan and Scarnecchia 1992; Meals and Miranda 1991), compared to areas where aquatic vegetation has been removed due to shoreline development. The placement of construction materials can also modify shoreline and substrate erosion rates, circulation and current patterns, and the rate and extent of mixing of dissolved and suspended loads in the water column (NMFS 1998).

7.1.2.2.1 Direct and Indirect Effects

The primary impacts associated with hydraulic and geomorphic modifications are similar to those discussed in Section 7.1.1 (*Marine Environments)* and include mobility alongshore, stratification alterations, changes in emergent and riparian vegetation associated with water level variability, and substrate and hydraulic energy changes.

The presence of structures such as jetties that disrupt either the movement of fishes within the littoral zone or circulation patterns may add to the inherent temperature stressor present in a lake. Littoral zones separated from the larger water body may become significantly warmer and exhibit larger diel temperature fluctuations (Kahler et al. 2000). Similarly, structures that extend into the mixing zone may also present a physical barrier to the movement of fishes in and out of these mixing zones (Altayaran and Madany 1992).

Reservoirs are also subject to density or turbidity currents resulting from differences in temperature or sediment concentrations between inflows and reservoir waters (Snyder et al. 2006). Mixing zones and shallow water littoral habitat were preferred by the razorback sucker in Lake Powell, Utah (Karp and Mueller 2002). They found that these fish primarily use shallow, vegetated habitats in side canyons, but these areas represent <1 percent of the available habitat in Lake Powell. However, temperature gradients in reservoirs can fragment these habitats. Mueller et al. (2000) found that temperature gradients caused the razorback sucker to abandon preferred inshore habitat in Lake Mohave, Arizona.

It is likely that HCP species may also be affected by preferred habitat fragmentation induced by temperature gradients. For example, bull trout in reservoirs behave like adfluvial fish and prefer littoral areas (Block 1955; Chisholm et al. 1989). However, bull trout have very low upper thermal limits, and the formation of thermal barriers may prevent their movement (Selong et al. 2001). Rinne et al. (1981) observed that “fishes introduced into western reservoirs are intrinsically shallow-water, littoral inhabitants,” and any structure in this zone may introduce a physical or thermal stressor.

Fish also respond to habitat characteristics resulting from the association of shoreline and riparian zone modification. In a study of Wisconsin lakes, the habitat characteristics most influenced by this association were depth, substrate size and embeddedness, and amount of woody vegetation and macrophytes (Jennings et al. 1999). Species richness was greatest where there was complexity in this suite of factors. However, jetties disrupt natural associations of depth and substrate and often diminish the total amount of woody vegetation alongshore.

Jetties may also decrease complexity of the shoreline by deflecting wave action from the littoral zone. Wave action creates complex littoral habitat by removing fine or silty sediments (Beauchamp et al. 1994). Wave action may also be a source of desirable spawning substrate. Kokanee salmon were observed to prefer spawning locations characterized by wave action, steep slopes, and an abundance of small, loose particles in Flaming Gorge Reservoir, Wyoming (Gipson and Hubert 1993). Lorang et al. (1993) observed that docks and seawalls intercepted transported gravels in Flathead Lake, Montana, as regulated lake levels rose and fell from early spring to late summer.

7.1.2.3 Riparian Vegetation Modifications

The removal of riparian vegetation as part of jetty construction and maintenance affects water temperature in lacustrine environments through a number of mechanisms that are similar to marine environments (including altered shading, altered shoreline stability, altered allochthonous input, and altered groundwater-surface water exchange). The dominant effect of vegetation removal is the loss of shading and increase in solar radiation exposure. In lakes, there are restrictions in circulation, sometimes a result of seasonal thermal stratification. Therefore, water temperatures generally change gradually through the year with the seasons, and show less change from night to day.

Riparian vegetation also stabilizes shorelines against the erosive forces of wind-driven waves and boat wakes (Carrasquero 2001). The loss of riparian and emergent vegetation promotes shoreline erosion, creating an erosive cycle that further increases vegetation loss, with a resultant adverse effect on nutrient cycles. For example, loss of emergent vegetation can promote erosive cycles that preclude the recovery and reestablishment of such vegetation. Decreased emergent vegetation density results in altered sediment transport patterns. If sediments are not replenished, additional emergent vegetation loss can result, leading to additional shoreline erosion (Rolletschek and Kühl 1997).

Altered habitat complexity in the form of a reduction of shoreline LWD and overhanging vegetation has been linked to removal due to human development (Francis and Schindler 2006). Loss of LWD has a number of effects on the health of the lake food web, all of which are described at length in Section 7.1.2.6 (*Ecosystem Fragmentation*) below.

7.1.2.3.1 Direct and Indirect Effects

Loss of shading of nearshore littoral habitats can result in changes in water temperature of sufficient magnitude to create thermal barriers for various fish species (Carrasquero 2001), perhaps leading to changes in species composition.

As in marine environments, LWD plays a role in the proper functioning of the lacustrine nearshore food web. It has been shown that in areas of intense shoreline development where riparian vegetation has been thinned or removed, littoral coarse woody debris is greatly diminished or entirely absent (Francis and Schindler 2006). Coarse woody debris has been shown to be an essential habitat component for sockeye salmon in lakes and is important for the overall health of lacustrine shoreline ecosystems (Naiman et al. 2000).

7.1.2.4 Aquatic Vegetation Modifications

Jetty construction can lead to a number of geomorphic modifications that can compromise the productivity of aquatic vegetation. These are generally related to changes in flow energy, which cause existing vegetation to be either physically removed or buried. These changes are described in detail in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*).

Aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of freshwater ecosystems. In general, benthic algae occurs in the form of microscopic unicellular algae, forming thin layers or assemblages called periphyton. Macrophytes include angiosperms rooted in the lake bottom, mosses, and other bryophytes.

These include many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998).

The uptake of carbon, nitrogen, and phosphorous by these plants provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous organic matter and part of the source of allochthonous matter within all lakes. Grazing of these primary producers by snails, caddisflies, isopods, minnows, and other grazers is an important pathway of energy flow. For herbivores, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of invertebrate scrapers (Hawkins and Sedell 1981), an important food source for fish. Juvenile salmonids, as drift-feeders, focus on food from autochthonous pathways. Invertebrate scrapers and collector-gatherers are known to be most frequently eaten by salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991).

Freshwater macrophytes are also known to modify their physiochemical environment by slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic content and capturing smaller grain size than substrate in uncolonized areas (Fonseca and Bell 1998).

As with marine aquatic vegetation, lacustrine aquatic vegetation has been shown to dramatically enhance the density of benthic invertebrates in temperate lakes through the relative protection from predation it affords (Beckett et al. 1992). In fact, even invasive vegetation can provide the same functions without compromising invertebrate productivity (Gardner et al. 2001). Although no studies were found that have linked the productivity of invertebrate communities with an increase in fish numbers, these observations may explain the differences in fish density along disturbed lakeshores (Jennings et al. 1999).

7.1.2.4.1 Direct and Indirect Effects

The loss of aquatic vegetation in freshwater lacustrine systems poses both direct and indirect effects on HCP species that depend on aquatic vegetation for any one of their life-history stages, such as white sturgeon, lake chub, pygmy whitefish, bull trout, and Chinook and sockeye salmon (Frest and Johannes 1995; Hughes and Peden 1989; Mongillo and Hallock 1998; Mongillo and Hallock 1999; Watters 1999). Effects primarily include the loss of protection from predators due to a loss of cover, as well as a reduction in primary productivity and food source.

7.1.2.5 Water Quality Modifications

Water quality modifications are generally not specific to different settings and, when they are, they depend upon whether the system is freshwater or saline. A discussion of direct and indirect effects on fish and invertebrate species is provided in the water quality modifications section in riverine environments for groins and bank barbs (Section 7.3.2.5).

7.1.2.6 Ecosystem Fragmentation

Sockeye salmon are the primary HCP species potentially affected by shoreline modifications along lakeshores. Lake shorelines represent crucial spawning habitat for sockeye (Burgner 1991; Scheuerell and Schindler 2004). Several studies of lakes in Washington State have demonstrated a loss of both productivity and diversity in their food webs in the presence of human development (Scheuerell and Schindler 2004). These studies have identified large woody debris and emergent and riparian vegetation as crucial to the maintenance of lake productivity (Smokorowski et al. 2006). The spatial distribution of LWD and emergent and riparian vegetation would likely be altered by the installation and maintenance of a jetty. Collectively, these three submechanisms of impacts (loss of nearshore habitat, altered emergent and riparian vegetation, loss of LWD) have a tendency to fragment the nearshore ecosystem. They are strongly related in their effect on HCP species, although they are discussed separately below.

7.1.2.6.1 Loss of Nearshore Habitat

As described in Section 7.1.1.2 (*Hydraulic and Geomorphic Modifications*), jetties alter nearshore circulation and wave energy in their vicinity. These processes in turn modify the substrate for some distance alongshore. Because lakes and reservoirs exhibit varying water levels, modifications in substrate can change the relationships between different habitat units. These changes can have direct impacts on fishes and invertebrates and indirect impacts on fishes and invertebrates through diminished nearshore productivity.

Direct and Indirect Effects

Sockeye salmon spawn along lakeshores (Burgner 1991). Juvenile Chinook salmon also use lacustrine littoral zones (Sergeant and Beauchamp 2006). Sergeant and Beauchamp (2006) have shown that although substrate preferences for juvenile Chinook are weak, they prefer fine substrates with cover. Because jetties would be expected to change both the local inundation frequency and substrate, they likely also change both the distribution and continuity of spawning and rearing areas, increasing the potential for predation.

7.1.2.6.2 Altered Emergent and Riparian Vegetation

Jetties and other human activities alter the distribution of nearshore substrate and vegetation types by modifying the wave and current environment (see Section 7.1.1.2 [*Hydraulic and Geomorphic Modifications*] for details). Diversity and the interconnectedness of different shoreline types have been found to be more crucial than any particular substrate or vegetation type on lake shorelines for a variety of species (Pratt and Smokorowski 2003). This may explain the dramatic differences observed in ecosystem health when modified and unmodified shorelines are compared (Roth et al. 2007; Scheuerell and Schindler 2004).

While the importance of riparian vegetation to the lacustrine nearshore area has been documented (Scheuerell and Schindler 2004), the role of emergent vegetation on the productivity of habitat has not been explored in detail in Washington waters or with regard to HCP species.

Direct and Indirect Effects

Loss of shading in nearshore littoral habitats can result in changes in water temperature of sufficient magnitude to create thermal barriers for various fish species (Rice 2006; Carrasquero 2001), perhaps leading to changes in species composition. Known indirect effects are related to changes in nearshore substrate and the presence of LWD, as described elsewhere in Section 7.1.2.6.3 (*Loss of Large Woody Debris [LWD]*), below.

7.1.2.6.3 Loss of Large Woody Debris (LWD)

LWD plays a crucial role in the maintenance of biomass in lake food webs (Roth et al. 2007; Smokorowski et al. 2006). In the construction and maintenance of jetties, LWD is often removed from the surrounding jetty site and nearby adjacent navigation channels. In addition, altered wave energy and nearshore circulation can cause a change in the transport energy such that less wrack accumulates on beaches adjacent to the jetty.

Direct and Indirect Effects

The loss of productivity due to the lack or removal of LWD, in conjunction with other human activities (e.g., fishing), can initiate a collapse of piscivorous fish species in freshwater lakes (Roth et al. 2007). It can also reduce the amount of periphyton available for other lower trophic fishes and remove substrate suitable for HCP species (Smokorowski et al. 2006).

7.1.2.6.4 Lost Opportunities

See Section 7.1.1.6 3 (*Lost Opportunities*) in the marine environment discussion for details.

7.1.3 Riverine Environments

Jetty installation in rivers is extremely rare because one of the main purposes of a jetty is to obstruct littoral transport, which does not occur on most rivers. In rivers, transport is not confined to the shoreline, and areas near the bank are generally areas of deposition (Chow 1959). Some projects may erroneously refer to a jetty, when the project is actually a groin. Refer to Section 7.3 (*Groins and Bank Barbs*) for a discussion of impacts on riverine environments from this type of activity.

7.2 Breakwaters

7.2.1 Marine Environments

7.2.1.1 Construction and Maintenance Activities

Breakwaters have many of the same construction impacts as jetties; refer to Section 7.1.1.1 (*Construction and Maintenance Activities*) in Section 7.1 (*Jetties*) for a full discussion of these impacts. However, breakwaters generally require less intrusive construction activities. For example, if material is being delivered to a project site by barge, it is likely that all mechanisms of impact on riparian areas can be ignored. In addition, activities associated with dewatering would be highly unusual in the construction of a breakwater. Pilings are also rarely used with breakwaters. Therefore, impacts associated with the installation and use of pilings would generally not apply to breakwaters.

7.2.1.2 Hydraulic and Geomorphic Modifications

Shallow nearshore marine habitats, structured by tidal currents, wind, and input from terrestrial and freshwater sources, support at least one life-history strategy of all of the marine and anadromous HCP species [e.g., all salmon, rockfish species, cod, hake, Pacific herring, walleye pollock, Newcomb's littorine snail, and the Olympia oyster (WDNR 2006a, 2006b)]. The controlling factors in these habitats depend upon pre-existing bathymetry, substrate, circulation and mixing, and sediment transport (WDNR 2006a, 2006b). These underlying hydrogeomorphic variables regulate a phenomenon known as alongshore transport, or littoral drift (Komar 1998). Key to understanding littoral drift is the idea of a drift cell (also known as drift sectors), which is a segment of shoreline along which littoral drifts moves sediment at noticeable rates. Each drift cell includes: (1) a sediment source, such as a feeder bluff; (2) a driftway along which these sediments move; and (3) an accretion terminal where the drift material is deposited. In this way, a drift cell allows the uninterrupted movement of beach materials (Terich 1987).

Breakwaters limit the amount of wave energy delivered to the shoreline by blocking it with the placement of material in a line parallel to the shoreline. Breakwaters are often used in series to protect a shoreline from erosion, and sometimes to enhance a beach nourishment project (Dean and Dalrymple 2002). They are typically used on sandy, open coastlines (Dean and Dalrymple 2002), although recent work has shown that they are equally effective at shoreline protection in coarse-clastic environments (King et al. 2000) more typical of Puget Sound (Finlayson 2006).

Substantial work has demonstrated that, at least for rockfishes, submerged emplacements of rock and other hard substrate can be used as a habitat enhancement measure (West et al. 1994). Although artificial reefs designed for habitat are usually placed in deeper water, several studies have shown that rock placed in shallower waters, typical of submerged breakwaters (Dean and Dalrymple 2002), can have the same benefits (Pondella and Stephens 1994). However, both submerged and emergent breakwaters have hydraulic and geomorphic impacts on adjacent areas.

7.2.1.2.1 Altered Wave Energy

Breakwaters are generally constructed out of placed rock and are specifically designed to reduce wave energy between them and the shoreline (Dean and Dalrymple 2002). Thus, they diminish wave energy shoreward of the structure while wave energy is generally increased offshore (Dean and Dalrymple 2002). Although the patterns of wave energy produced by emergent and submerged breakwaters are different (Ranasinghe et al. 2006), all breakwaters modify the wave environment in the nearshore. This redistribution of wave energy can have a number of interrelated indirect and direct impacts on fish and invertebrates, and these may be grouped into two categories: those that relate to changes in substrate, and those that change water column characteristics.

Indirect Effects

Substrate Stressors

Substrate is an important factor controlling the growth of aquatic vegetation in Puget Sound (Koch 2001) and the conditions for fish spawning in intertidal waters (typically forage fish). Increased wave energy has been shown to induce scour adjacent to breakwaters (Sumer et al. 2005) and, if some of the coarsest material is not mobilized, a generally coarser substrate will result (Komar 1998). However, the degree to which this occurs depends on the geologic setting. For instance, on the outer coast, substrate is loose, deep, sandy, and unconsolidated. In these areas, increased or displaced wave energy associated with breakwaters creates wholesale erosion of the shoreline (Miller et al. 2001). In protected, previously glaciated areas the basin topography is complex and the coarse nature of the substrate slows down erosion dramatically (Nordstrom 1992). In these locales, a lag deposit can result in a near bedrock-like shoreline [e.g., Foulweather Beach (Finlayson 2006)]. Typically, however, shoreline bed (beach substrate) hardening manifests by a loss of the pebble veneer that is common throughout much of Puget Sound (Finlayson 2006) (Figure 7-2). This process is similar to what has occurred on the urbanized shorelines throughout the Great Lakes (Chrzastowski and Thompson 1994). These changes have produced pronounced ecological changes within recent years in the Great Lakes (Meadows et al. 2005), causing the elimination of native species and enabling invasives (zebra mussels) to dominate the nearshore ecosystem (Marsden and Chotkowski 2001).

Changing substrate can adversely affect the growth of aquatic vegetation. For instance, eelgrass is incapable of growing in dominantly gravelly substrates (Koch 2001). For details regarding impacts on fish and invertebrates as a result of aquatic vegetation disturbances, see Section 7.1.1.4 (*Aquatic Vegetation Modifications*).

Although breakwaters are designed to protect areas from wave energy and therefore initiate deposition, they have been shown to induce scour on the seaward side of their ends (Sumer et al. 2005). This is primarily associated with artificial rip currents developed in these areas. However, there have been no experimental studies that have documented impacts on forage fish spawning areas. The effects associated with scour and coarsening are likely equivalent to the damage associated with shoreline hardening (i.e., bulkheads) in Hood Canal (Herrera 2005; Penttila 1978; Thom et al. 1994). Typical spawning substrates consist of fine gravel and coarse sand characteristic of the pebble veneer found throughout Puget Sound (Finlayson 2006), with broken shells intermixed in some cases (Thom et al. 1994). Surf smelt make no attempt to bury their demersal, adhesive eggs but rely on wave action to cover the eggs with a fine layer of substrate (Thom et al. 1994). Therefore, changing the wave environment may change the survivability of surf smelt spawn. The importance of substrate to spawning has also been empirically demonstrated in the closely related Japanese surf smelt (Hirose and Kawaguchi 1998).

Pacific sand lance also spawn in the high intertidal zone on substrates varying from sand to sandy gravel. Sand lance also rely on sandy substrates for burrowing at night. Because breakwater-induced scour and coarsening generally occurs seaward of the high intertidal zone, it is not expected that spawning areas for these species will be affected significantly by breakwaters. However, if the breakwaters are extremely close to shore, mimicking a seawall or similar protective structure, it is possible that sand lance may be affected.

Breakwaters are specifically designed to produce areas of reduced wave activity (Dean and Dalrymple 2002). Reduction in wave energy from natural levels lowers near bed shear stress, resulting in the deposition of finer sediments (Miller et al. 1977). Considering the large volume of fine-grained sediment supplied to western Washington waters (Downing 1983), even areas that are not within active littoral cells can receive large amounts of fine sediment.

Deposition of large amounts of fine sediment can kill aquatic vegetation vital to nearshore HCP species. Burying eelgrass at a depth as little as 25 percent of the total plant height can decrease productivity and increase the mortality of eelgrass (Mills and Fonseca 2003). For more information about the effects of aquatic vegetation on fish and invertebrates, see Section 7.1.1.4 (*Aquatic Vegetation Modifications*). Eelgrass can also be discouraged from colonizing new areas with a high clay content resulting from recent sediment deposition (Koch 2001).

Water Column Stressors

Wave energy is the dominant source of flow disturbance in the nearshore environment in most Washington waters (Finlayson 2006). Wave energy is responsible for mixing the upper portion of the water column (Babanin 2006; Fischer et al. 1979) and producing high shear stresses near the bed (Lamb et al. 2004). These motions can be harmful to aquatic vegetation as well as the fish and invertebrates that use and consume this vegetation.

Attenuation of waves can increase water column stratification in marine waters and lead to dissolved oxygen reduction and temperature anomalies [(Qiao et al. 2006); see Section 7.1.1.5.2 (*Altered Dissolved Oxygen*)]. Surficial mixing and circulation also play an important role in primary productivity, particularly near large river mouths [e.g., Willapa Bay (Roegner et al. 2002)]. Disruption of these processes will likely have impacts on primary productivity and, ultimately, on a variety of marine species through food-web interactions.

Wave energy also plays a role in the distribution of aquatic vegetation used by salmonids and other nearshore species, particularly in energetic environments. High wave energy has been shown to inhibit the colonization and growth of some seagrasses [e.g., eelgrass (Fonseca and Bell 1998); see Section 7.2.1.4 (*Aquatic Vegetation Modifications*) for details], although in more recent work in Puget Sound, no correlation was found between eelgrass prevalence and wave characteristics (Finlayson 2006). High shear stresses associated with waves can also dislodge kelp (Kawamata 2001).

Direct Effects

Alterations in wave energy could alter planktonic spawn, increasing the mortality of those affected species (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details). Effects on invertebrates would be confined to removal or burial (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details).

7.2.1.2.2 Altered Current Velocities

Breakwaters are not designed to alter nearshore current velocities; however, there is evidence that they can unintentionally cause strong rip currents (Bellotti 2004; Dean and Dalrymple 2002). Also because they function essentially as a new obstacle to flow, they can also reduce velocities in other areas. The relationship between flow velocity and a change in substrate is related to the boundary shear stress (Miller et al. 1977). Substrate and aquatic vegetation are removed if a critical shear stress is exceeded. If the shear stress drops, anomalous deposition can occur. Both of these effects would produce the same stressors discussed in the previous subsection on *Altered Wave Energy*.

Indirect Effects

Nearshore currents, even those in heavily altered environments, do not exceed the threshold for adult salmonid navigation, but high velocities have been shown to exclude some small fishes (e.g., juvenile salmonids, forage fishes) from navigating nearshore waters (Michny and Deibel 1986; Schaffter et al. 1983). This would potentially cause fragmentation of habitat for these species (see Section 7.2.1.6 [*Ecosystem Fragmentation*].

Eelgrass and many other species of aquatic vegetation (e.g., bull kelp) require some water motion for survival (Fonseca et al. 1983). The degree of sensitivity is species-specific and also dependent on other factors such as pollutant loading, which is discussed in detail in Section 7.1.1.5 (*Water Quality Modifications*). The landward side of breakwaters is known to produce unnaturally calm waters that could enhance sedimentation and bury and decrease the productivity of eelgrass (Fonseca et al. 1983).

Direct Effects

Alterations in nearshore currents could alter planktonic spawn, increasing the mortality of affected species (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details). Effects on invertebrates would be confined to removal or burial (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details).

7.2.1.2.3 Altered Nearshore Circulation

Breakwaters alter nearshore circulation by modifying the transport processes associated with a variety of wave and wave-breaking mechanisms (Caceres et al. 2005). However, these alterations would not likely extend to changes in nearshore stratification and dissolved oxygen, unlike jetties.

Indirect Effects

The indirect effects on fish and invertebrates would be similar to those discussed with respect to jetties, although they would not likely be as pronounced.

Direct Effects

Changes in nearshore circulation could alter planktonic spawn, increasing the mortality of affected species (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details). Effects on invertebrates would be confined to removal or burial (see Section 7.1.1.2.1 [*Altered Wave Energy*] for details).

7.2.1.2.4 Altered Groundwater-Surface Water Exchange

In the construction of a breakwater, it is uncommon for structures (e.g., pilings) to be placed that interrupt groundwater transfer between the sea and the shore. However, if the proposed breakwater has this type of design element, groundwater connections with the sea will be interrupted, and submarine groundwater discharge (SGD) impacts would result. See Section 7.1.1.3.4 (*Altered Groundwater-Surface Water Exchange*) for a discussion of these impacts and direct and indirect effects on fish and invertebrates.

7.2.1.2.5 Altered Sediment Supply

Breakwaters have been shown to disrupt the littoral transport of sediments and subsequently cut off downdrift shorelines to a sediment supply (Bowman and Pranzini 2003; Sane et al. 2007; Thomalla and Vincent 2003). Reduction or elimination of the sediment supply can inhibit the proper functioning of spits and beaches and cause the elimination of substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Haas et al. 2002; Parametrix 1996; Penttila 2000; Thom and Shreffler 1996; Thom et al. 1994). These impacts primarily manifest as a change in substrate, the impacts of which are discussed in the preceding subsection on *Altered Wave Energy* (Section 7.2.1.2.1).

Indirect Effects on Fishes and Invertebrates

The primary indirect impact of changing the sediment supply is to change the distribution of substrate within the littoral cell where the modification occurs (Terich 1987). Thus, the loss of sediment to a drift cell results in a coarsening of the substrate as fine-grained sediment is lost to deep portions of the basin by resuspension (Finlayson 2006) and is not resupplied by freshly eroded bluff sediments. The coarsening would have the same impacts on fish and invertebrates as discussed in the *Altered Wave Energy* subsection (Section 7.2.1.2.1). However, because some drift cells can be extremely long [e.g., more than 20 miles long in the drift cell that extends between Seattle and Mukilteo on the northeastern shore of the main basin of Puget Sound (Terich 1987)], the effects of a modification can extend well beyond the primary area of activity.

Direct Effects on Fishes and Invertebrates

The primary direct impact on fish and invertebrates of altering the sediment supply is to alter the degree of turbidity in the nearshore environment (Bash et al. 2001; Berry et al. 2003). See Section 7.1.1.5.3 (*Altered Suspended Solids and Turbidity*) for a full discussion of the effects of turbidity.

7.2.1.2.6 Altered Substrate Composition

Breakwaters create a new shoreline that is rigid and immobile along its length. Many different materials have been used to construct breakwaters including riprap, reinforced concrete, pre-formed concrete elements like dolos, and timber structures (NRC 2007). Regardless, of the material used, the addition of immobile substrate affects fishes and invertebrates (USFWS 2000). These impacts are generally most pronounced if the structure has a vertical wall, rather than a steep slope (Bulleri and Chapman 2004).

Indirect Effects on Fishes and Invertebrates

The primary indirect impact on nearshore ecology is to encourage a shift toward hard-substrate, often invasive, communities (Wasson et al. 2005). However, the immobile substrate also fundamentally changes the mechanics of water motion on the shoreline, increasing wave reflection (Finlayson 2006; Komar 1998) and eliminating the exchange of water into and out of the shoreline if impermeable materials are used (Nakayama et al. 2007). These effects are discussed in preceding subsections (i.e., *Altered Wave Energy*, *Altered Nearshore Circulation Patterns*).

Direct Effects on Fishes and Invertebrates

In general, the addition of immobile substrate decreases the habitat suitability for juvenile salmonids. This is discussed in detail in Section 7.1.1.2.6 (*Altered Substrate Composition*).

It is possible that coarser substrate could provide a benefit to some HCP species, particularly in the case of submerged breakwaters, which act essentially as an artificial reef (Pondella and Stephens 1994). An active debate in the scientific community is whether artificial reefs are as productive and diverse as natural hard-rock shorelines, particularly in the Adriatic Sea west of Italy (Bacchiocchi and Airoldi 2003; Bulleri et al. 2006; Guidetti, Bussotti et al. 2005). In addition to the elimination of mobile, sandy habitats, studies in the Adriatic Sea have shown that maritime structures caused weighted abundances in piscivores and urchins, and decreased abundances of native species that prefer more mobile substrates (Guidetti, Bussotti et al. 2005). Although species distributions are clearly different in Italy than in Washington State, the steep, paraglacial landscape and relatively short period, locally generated waves make hydraulic and geomorphic variables essentially identical (Finlayson 2006). Complicating the debate is that other maritime activities (i.e., fishing, ship traffic) are difficult to separate from other effects (Guidetti, Verginella et al. 2005).

7.2.1.3 Riparian Vegetation Modifications

Breakwaters, by definition, do not intersect the shoreline and are built offshore. As a result, their construction and maintenance do not have associated permanent riparian vegetation impacts. The only impacts would be associated with access during construction (and possibly not even then, if building materials are delivered by barge).

7.2.1.4 Aquatic Vegetation Modifications

Breakwaters and jetties produce impacts on aquatic vegetation via the same mechanisms. Therefore, the reader is referred to Section 7.1.1.4 (*Aquatic Vegetation Modifications)* for details about aquatic vegetation modification associated with breakwaters.

7.2.1.5 Water Quality Modifications

Breakwaters and jetties produce water quality modifications via the same mechanisms—either by initiating circulation changes through their geomorphic modifications, or by polluting waters with building materials. As a result, the reader is referred to Section 7.1.1.5 (*Water Quality Modifications)* in Section 7.1 (*Jetties*) for details about water quality modifications associated with breakwaters.

7.2.1.6 Ecosystem Fragmentation

7.2.1.6.1 Habitat Loss and Fragmentation

Breakwaters have not been shown to disconnect nearshore habitats, alter tidal prisms, or significantly influence nearshore stratification in the same way that jetties do. Therefore, breakwaters do not present the same degree of risk of disconnection to crucial habitats (e.g., lagoons) as jetties.

As offshore hard structures, breakwaters do, however, have a significant impact on the types of ecologic assemblages found in their vicinity (Perez-Ruzafa et al. 2006). They also have a tendency to increase the numbers of piscivores, which, depending on the species of predator and prey could have a negative effect on HCP species (Pondella et al. 2002). As a result, breakwaters could alter predator–prey abundance and compromise certain habitats for HCP species.

Breakwaters and other artificial reefs have been examined in terms of their productivity for Washington fishes and invertebrates for nearly 30 years (Buckley and Hueckel 1985). When properly designed, submerged breakwaters generally increase the numbers of fish, including rockfish and lingcod (West et al. 1994). However, there is significant debate in the scientific community as to whether they increase the total productivity of the nearshore or simply attract those species from surrounding waters [i.e., “attraction versus production” (Pickering and Whitmarsh 1997)].

In some locales, submerged breakwaters have been seen as an ecological loss because the goals of constructing a breakwater are different from those of an artificial reef (Bulleri and Chapman 2004). This is particularly true if those breakwaters are fished or if the species of concern are eaten by the fishes attracted to the reef (Pondella et al. 2002). Because the literature is somewhat contradictory, it is important to understand the three factors that have emerged as crucial variables in the determination of the ecological success of a breakwater:

- *Species and diet of attracted fishes*. If the species attracted to and potentially enhanced by the breakwater structures are HCP species [e.g., bocaccio rockfish (Love et al. 2006)], the installation could be considered to have a net ecological benefit. It is also possible, however, that the introduction of predatory fishes could shift biomass away from smaller fishes (Pondella et al. 2002). If these smaller fishes are HCP species (e.g., Pacific herring), the project could present a net loss.

- *Commercial fishing*. If fishing is allowed on the breakwater, the concentration of HCP species can result in a loss in the number of total fish, while protection from fishing has the potential to increase the total number (Guidetti, Verginella et al. 2005).

- *Geometric complexity of structure*. If the structure is simple (with vertical walls), attraction of fish will be restricted and possibly limiting (Bulleri and Chapman 2004; West et al. 1994). However, if the breakwater provides sheltering sites, its installation may be of net ecological benefit [particularly for rockfishes (Love and York 2006)].

7.2.1.6.2 Loss of Large Woody Debris (LWD)

Breakwaters change the distribution of wave energy, substrate, and wrack material along the shoreline. Therefore, there will be a redistribution of LWD. Large wood will most likely preferentially accumulate in the lee of breakwaters, similar to the substrate changes induced by these structures (Bowman and Pranzini 2003; Thomalla and Vincent 2003). This would cause the shoreline to be discontinuously covered by LWD, potentially compromising natural migration corridors to HCP fish species that use littoral zones.

7.2.1.6.3 Lost Opportunities

A potential impact of breakwaters is related to the loss of opportunities. "Lost-opportunity impacts" result from projects that adversely alter natural nearshore processes important to the ongoing creation of fish and wildlife habitats. Breakwaters restrict deposition and erosion in specific geometries useful for human activities. However, this shoreline geomorphology can be poorly suited for any or all life stages of fish and invertebrates and will be lost for the design-life of the breakwater.

Likewise, breakwaters can impose lost-opportunity costs through a number of pathways. One pathway is analogous to the example provided above. Breakwaters may pose lingering ecological effects when they are not designed properly to account for channel movement or to allow sediment and wood transport. An equally important type of lost-opportunity cost occurs when the breakwater interferes with littoral migration of fish and invertebrate species or life-history stages. In such instances, the costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly.

This may delay the actions necessary to provide appropriate migration for fish, perhaps as long as the design-life of the underperforming structure.

7.2.2 Lacustrine Environments

Breakwaters are structures added to protect the shoreline from wave energy and subsequent erosion. They are generally installed on open, exposed coasts (Komar 1998) and, therefore, are rarely used in lacustrine environments. Most of the work that has been performed on breakwaters in lacustrine environments has been conducted in the Great Lakes (Fitzsimons 1996; Marsden and Chotkowski 2001; Olyphant and Bennett 1994), which are substantially larger (and therefore subject to much larger waves) than any lakes in Washington State.

The primary differences between breakwaters in lacustrine environments and those in marine environments are the following:

- *Short-term stable water levels*. Lakes, although subject to long-term water level variability, are not generally subject to tides. As a result, the size of the breakwater that may be required is significantly smaller and can be placed much closer to shore. This would mean the area of alteration associated with breakwaters would be generally smaller in lakes than in marine waters.

- *Species of interest*. Although some species use both lacustrine and marine shorelines (e.g., salmonids), most fish and invertebrates use only one or the other, with the freshwater-only species having limited scientific study. Further, because studies of the specific impacts on invertebrates in lakes has been extremely limited, there is a data gap in terms of understanding the specific effects of breakwaters on those freshwater species.

- *Invasive species*. Freshwater breakwaters have been shown to harbor invasive species infestations capable of the wholesale disruption of lacustrine nearshore ecosystems (Marsden and Chotkowski 2001). In particular, zebra mussels present an ongoing and serious threat to Washington freshwaters (WDFW 2004).

7.2.3 Riverine Environments

The most common breakwaters found on rivers in Washington State are temporary, floating structures used to protect small, private harbor areas. The analysis in this white paper concentrates on long-term or permanent structures. Therefore, floating, temporary breakwaters are not addressed in this white paper. These structures, if they require an HPA, would be considered overwater structures and are discussed in Carrasquero (2001).

Permanent breakwaters are built to protect the shoreline from wave energy (Dean and Dalrymple 2002). Therefore, nearly all rivers in Washington State are too small (both in width and depth) and fast moving to have shorelines where waves significantly influence the mobility of the shoreline substrate. Only the Columbia River is generally considered large and deep enough to produce wave heights significantly affecting the substrate and erodibility of its banks. Refer to Section 7.2.2 (*Lacustrine Environments*) for an analysis of potential mechanisms of impact and effects on fish and invertebrates in large rivers.

7.3 Groins and Bank Barbs

The primary purpose of a groin is to store sediment on the bank adjacent to the structure and prevent erosion to the updrift shoreline of the groin. Groin structures can be installed in rivers, lakeshores, and marine shorelines, often to prevent erosion (NRC 2007). -Some of the activities permitted by WDFW as a shoreline modification (WDFW 2007) would be closest in character to groins, without having the same primary purpose (e.g., access stairways) in that they disrupt transport of water and sediments. For purposes of this white paper, the generic term “groin” is used for groins and their analogs.

Bank barbs are defined as those flow-perpendicular structures built to inhibit erosion to the banks of a river that are typically submerged at higher flows. As with breakwaters and jetties, the degree of submergence changes the pattern of scour and wave energy (Losada et al. 2005), but it does not eliminate hydraulic and geomorphic modifications that occur (Moschella et al. 2005). Because submerged structures are relatively ineffectual as a result of the large suspended load in high-wave-energy environments, bank barbs are often not associated with marine and lacustrine environments.

7.3.1 Marine and Lacustrine Environments

As with jetties, a primary purpose of groins and bank barbs is to store sediment along the shoreline and prevent shoreline erosion. For a description of the mechanisms of impact and effects on fish, invertebrates, and their habitats, see Section 7.1.1 for jetties in marine environments, and Section 7.1.2 for jetties in lacustrine environments.

There are a few fundamental differences between groins and jetties that should be noted:

- *Maritime traffic*. Unlike jetties, groins and bank barbs discourage maritime activities. Because groins can represent a navigational hazard, they are usually avoided by watercraft. However, some analog structures or activities (e.g., access stairways to floating docks) may encourage maritime traffic. Therefore, such impacts should be considered on a case-by-case basis. When a significant increase in boat traffic is expected with the activity, the *Vessel Activities* section of the Marinas and Shipping/Ferry Terminals white paper (Herrera 2007a) should be consulted.

- *Nearshore circulation.* The only alteration to nearshore circulation from groins is related to the interruption of surf-zone generated alongshore currents and circulation. Groins rarely protrude into depths significantly (more than 10 feet) below mean lower low water (MLLW). As a result, they do not play an important role in tidal and estuarine circulation. Therefore, these hydrogeomorphic impacts (altered nearshore circulation) and the subsequent water quality impacts (dissolved oxygen, temperature) are not considered here in the construction of groins or groin-like structures.

- *Altered groundwater/surface water exchange.* Groins and bank barbs are usually constructed of placed rock or riprap. Therefore, they do not disrupt the exchange of groundwater with the primary water body; thus, these impacts and their subsequent effects on fish and invertebrates are not considered here. However, any groin or groin-like structure that uses piles or other significant, impermeable, embedded elements (e.g., isolating more than 10 lineal feet along the shoreline from groundwater influence) should consider these impacts and effects.

7.3.2 Riverine Environments

Groins, bank barbs, and their analogs are fingerlike, bank-protection structures keyed into one bank and oriented obliquely to the flow. In riverine environments, jetties are the functional equivalent of groins. Groins and bank barbs are typically constructed in sets along the outside of a meander bend, with the primary function of redirecting flow and bed material away from the bank and toward the middle of the channel. Bank barbs are specific types of groins that are typically submerged at or below low water. Weir-type structures, like bank barbs, are intended to redirect flow toward the center of the channel using weir hydraulics over the structure. In contrast, groins are typically exposed above high water and are designed to divert flow (and bed sediment) around the structure. Both classes of structures reduce near-bank velocities, increase centerline velocities, retard bank erosion, cause local bed scour around the groin tip, and trap fine sediment and debris between structures (Li et al. 1984). Because the hydraulic and geomorphic effects of submerged and exposed groins are similar, the direct and indirect impacts on fish and invertebrates from all types of groin structures are considered collectively here.

7.3.2.1 Construction and Maintenance Activities

The submechanisms of impact of the construction of groins are not environment-specific. The primary difference between groin construction in rivers and jetty and breakwater construction in marine and lacustrine environments has to do with the dewatering of a unidirectional channel. As a result, these impacts and their effects on fish and invertebrates are discussed below.

7.3.2.1.1 Channel Dewatering

Channel dewatering is typically associated with groins and bank barbs constructed in riverine environments where working in dry conditions is desirable. Dewatering usually requires the installation of a cofferdam and a bypass system to divert flowing water around the construction site. Prior to dewatering, all fish and other vertebrate aquatic life are removed using a variety of methods. In addition, fish are often excluded from isolated areas for projects where in-water work occurs without dewatering. In-water work and fish handling associated with the practice of channel dewatering can potentially affect fish and invertebrate species.

The direct effects of fish exclusion and dewatering include:

- Direct mortality, injury, and stress from electrical field exposure (i.e., electrofishing)
- Capture by netting, leading to direct mortality, injury, and stress
- Physical and thermal stress and possible trauma associated with handling and transfer during capture and transfer between temporary holding containers and release locations
- Stranding and asphyxiation
- Entrainment or impingement in block nets, dewatering pumps, and bypass equipment
- Increased stress, predation exposure, and habitat competition once relocated
- Increased competition for aquatic species forced to compete with relocated animals.

Exclusion areas may also create temporary barriers to fish passage, with attendant effects on migratory fish species.

Direct and Indirect Effects on Fish

Of the various methods used for dewatering and fish handling, the majority of research has been conducted on incidental mortality and injury rates associated with electrofishing. Much of this research has focused on adult salmonids greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids suggest spinal injury rates lower than those observed for large fish, perhaps because juvenile fish generate less total electrical potential along a shorter body length (Dalbey et al. 1996; Sharber and Carothers 1988; Thompson et al. 1997). Electrofishing-related injury rates are variable, reflecting a range of factors from fish size and sensitivity, individual site conditions, to crew experience and the type of equipment used, with the equipment type being a particularly important factor (Dalbey et al. 1996; Dwyer and White 1997; Sharber and Carothers 1988). Electrofishing equipment typically uses continuous direct current (DC) or low-frequency pulsed DC equipment. The use of low-frequency DC (equal to or less than 30 Hz) is the recommended electrofishing method as it is associated with lower spinal injury rates (Ainslie et al. 1998; Dalbey et al. 1996; Fredenberg 1992). Even with careful selection of equipment, observed injury rates can vary. For example, one study in the Yakima River basin (McMichael et al. 1998) observed a 5.1 percent injury rate for juvenile steelhead captured using 30 Hz pulsed DC equipment, while another study (Ainslie et al. 1998) reported injury rates of 15–39 percent in juvenile rainbow trout using continuous and pulsed DC equipment, with pulsed DC equipment producing a higher frequency of less severe injuries.

It is notable that electrofishing capture typically has a low direct mortality rate, but it is reasonable to conclude that injuries induced by electrofishing could have long-term effects on survival, growth, and fitness. The few studies that have examined this question found that few juvenile salmonids die as a result of electrofishing-induced spinal injury (Ainslie et al. 1998; Dalbey et al. 1996). However, fish with more injuries demonstrated a clear decrease in growth rates, and in some cases growth was entirely arrested (Dalbey et al. 1996). In the absence of additional supporting information, it is reasonable to conclude that these same effects would affect many of the HCP fish species, but this conservative assumption may not be universally accurate. Studies of the effects of electrofishing on other fish species are more limited, but available data indicate that at least some HCP species may be less sensitive to injury-related effects. Holliman et al. (2003) subjected a threatened cyprinid (minnow) species to electrofishing techniques in the laboratory and found that the typical current and voltage parameters used to minimize adverse effects on salmonid species produced no evidence of injury. This suggests that other cyprinids (such as leopard and spotted dace, lake chub, and suckers) may also be less sensitive.

Beyond the effects of electrofishing, the act of capture and handling demonstrably increases physiological stress in fishes (Frisch and Anderson 2000). Primary contributing factors to handling-induced stress and death include exposure to large changes in water temperatures and dissolved oxygen conditions (caused by large differences among the capture, holding, and release environments); duration of time held out of the water; and physical trauma (e.g., due to net abrasion, squeezing, accidental dropping). Even in the absence of injury, stress induced by capture and handling can have a lingering effect on survival and productivity. One study found that stress from handling impaired the salmonids’ ability to evade predators for up to 24 hours following release and caused other forms of mortality(Olla et al. 1995).

Use of a bypass system is a common means of creating exclusion areas via dewatering and flow reduction. Partial dewatering is a technique used to reduce the volume of water in the work area to make capture methods more efficient. In riverine habitats, this method is used to move fish out of affected habitats to reduce the number of individuals exposed to capture and handling stress and potential injury and mortality. NOAA Fisheries has estimated that 50–75 percent of fish in an affected reach will volitionally move out of an affected reach when flows are reduced by 80 percent (NMFS 2006). However, volitional movement will lead to concentration of fish in unaffected habitats, increasing competition for available space and resources.

Failure to capture and remove fish or invertebrates from work areas must also be considered. Organisms left in the exclusion area would potentially be directly exposed to stranding and asphyxiation during dewatering or, if left inundated, to mechanical injury and/or high-intensity noise, turbidity, and other pollutants. Many species of fish, such as salmonids and larval lamprey, are highly cryptic and can avoid being detected even when using multiple-pass electrofishing because they hide in large interstices or are buried in sediments (Peterson et al. 2005; Peterson et al. 2004; Wydoski and Whitney 2003).

NOAA Fisheries has estimated incidental take resulting from dewatering and fish handling associated with stream crossing projects. In calculating incidental take from these activities, the agency applied an estimated stranding rate of 8 percent for ESA-listed salmonids (which equates to 8 percent mortality) (NMFS 2006), based on an expected 45 percent capture efficiency using three pass electrofishing (Peterson et al. 2004), and assuming a 25 percent injury rate.

As noted, research on fish injury and mortality associated with dewatering has focused predominantly on salmonids, relatively large fish species that respond well to this exclusion technique. Other species may have nonmotile or cryptic life-history stages (e.g., lamprey ammocoetes buried in fine sediments) or life-history stages that cannot easily move to adjust to changes in flow or are not easily captured and relocated (e.g., adhesive eggs of eulachon, juvenile rockfish and lingcod). In freshwater environments, examples of species and life-history stages that are sensitive to dewatering impacts include incubating salmonid eggs and alevins; lamprey ammocoetes; and the adhesive eggs of eulachon, sturgeon, and other species. These life-history stages are relatively immobile and also difficult to capture and relocate efficiently. Therefore, they face a higher likelihood of exposure to stranding or entrainment in dewatering pumps, which would be expected to lead to mortality. In marine environments, the larval and juvenile life-history stages of rockfish, lingcod, Pacific cod, hake, pollock, herring, smelt, and sand lance are similarly immobile and difficult to capture and therefore vulnerable to the same effects.

Rewatering of the exclusion area following construction is generally expected to result in a temporary increase in turbidity. The in-water installation and removal work poses the highest risk of disturbing the stream bank and substrate, thereby resuspending sediments and increasing turbidity. Fish may experience short-term, adverse effects as a result of increased turbidity. The effects of increased turbidity are relatively independent of environment type and are discussed in more detail in Section 7.1.1.1.4 (*Construction/Maintenance Dredging)* under *Turbidity*.

Direct and Indirect Effects on Invertebrates

HCP invertebrate species demonstrate different sensitivity to the effects of dewatering and relocation than fish, with many species being relatively insensitive to the effects of handling, at least during adult life-history stages. For example, Krueger et al. (2007) studied the effects of suction dredge entrainment on adult western ridged and western pearlshell mussels in the

Similkameen River (Washington) and found no evidence of mortality or significant injury. Suction dredge entrainment is expected to be a more traumatic stressor than removal and relocation by hand. These findings suggest that careful handling would be unlikely to cause injury. However, the authors cautioned that these findings were limited to adult mussels, and the potential for injury and mortality in juveniles remains unknown.

The sensitivity of other HCP invertebrate species, such as giant Columbia River limpet and great Columbia River spire snail, is somewhat less certain. Adults may be easily removed and relocated during dewatering, but juveniles and eggs may be difficult to locate and remove effectively. This suggests the potential for mortality from stranding.

While handling-related injury and mortality are relatively unlikely, relocation may lead to significant nonlethal effects. For example, scattering of closely packed groups of adult mussels may affect reproductive success. Because female freshwater mussels filter male gametes from the water column, successful fertilization is density dependent (Downing et al. 1993).

Failure to locate and remove small or cryptic invertebrate species or life-history stages may result in stranding or concentrated exposure to other stressors within the exclusion area. Stranding caused by operational water level fluctuations was associated with mass mortality of California floater and western ridged mussels in Snake River reservoir impoundments (Nedeau et al. 2005).

7.3.2.2 Hydraulic and Geomorphic Modifications

River hydrology includes the movement of hyporheic groundwater to the stream and the movement of surface water across land to the stream. Changes to riverine hydrology that reduce or increase the flow of water to the river alter the suitability of habitats within the river. During low-flow periods, alterations to hydrology can result in previously wetted areas going dry, thereby eliminating habitat area for aquatic organisms. Hydrologic alterations that increase overland surface water flow can, on the other hand, increase flooding and substrate scour.

Riverine hydraulics determine the nature, as well as the distribution and deposition, of sediments and other materials along the path of the river’s unilateral movement toward lower elevations. Fishes and invertebrates depend upon the diversity of habitats created by those hydraulic forces that result in the distribution of diverse sediments along the river continuum (Montgomery et al. 1999).

HCP species, such as sturgeon, char, bull trout, salmonids, and freshwater mussels, depend on particular riverine sediment types and habitats. If the flow becomes too strong, reaches of rivers can be made impassable to various fish species or life-history stages, or unsuitable for invertebrates. The reproduction, growth, and survival of these HCP species depend on particular hydraulic regimes to maintain suitable habitats. Alterations to river hydraulics that change the flow of water and the ability of the water to move sediments and nutrients can have direct and indirect effects on HCP species. Projects that alter riverine hydrology can also have direct and indirect effects on HCP species.

Shoreline structures built to protect bank erosion can alter the contribution of sediment to the aquatic environment. Because HCP species depend on the presence or absence of particular substrate types to support important life-history functions, changes in sediment source contributions can have direct and indirect effects on those species. The impacts of substrate modifications for riverine habitats are similar to those discussed in Section 7.1.2.2 (*Hydraulic and Geomorphic Modifications*) for lacustrine habitats. Shoreline structures that are constructed to protect upland properties from erosion can simultaneously modify the substrate available to species for spawning and rearing by blocking the contribution of sediments to the shoreline from the uplands or from upstream areas. Such structures can also increase the scouring of substrates. This scouring action can affect downstream habitats by transporting and depositing fine sediments, thereby compromising downstream spawning habitat. It can also dramatically modify the types and abundance of substrates available to support aquatic vegetation, important to a suite of HCP riverine species.

7.3.2.2.1 Altered Flow Velocities

Groins and barbs alter the velocity field in riverine environments by redirecting flow away from the banks and toward the center of the channel (Lagasse et al. 2001). The formation of flow-separation eddies on the downstream side of groins and barbs results in areas of relatively low velocity in these areas and along the protected bank (Lagasse et al. 2001). The common practice of placing groins and barbs along the outside of a meander bend, where the highest velocity flows typically occur, would cause the greatest alteration of flow velocities.

Because flow velocity in a channel is proportional to the hydraulic radius and inversely proportional to roughness (Leopold et al. 1964), bank hardening and the construction of groins can alter the flow velocity, depth, and width of a channel relative to natural conditions. The net effect of groins is to confine the flow. The primary effects of flow confinement by artificial structures are increased velocity and bed scour. Bank hardening with riprap can reduce the hydraulic roughness and hence can also increase velocity and bed scour.

Direct and Indirect Effects

Fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, and foraging. For example, increases in flow velocities could present potential barriers to fish migration or could exceed thresholds for certain life-history stages of some HCP species. For instance, leopard and Umatilla dace inhabit riverine environments where the velocities are less than 1.6 ft/sec (Wydoski and Whitney 2003). Exceeding this velocity as a result of groin or barb installation would render habitat unsuitable for these species.

7.3.2.2.2 Altered Sediment Supply

Bank stabilization measures can reduce the supply of suitably sized substrates for spawning fish and invertebrates by limiting natural processes of channel migration and bank erosion. Groins can reduce the local supply of coarse sediment by deflecting bed sediment from the river bank to the center of the channel. Because the rate and caliber of sediment supplied to a channel can influence the substrate size (Dietrich et al. 1989), changes in sediment supply can alter the composition of substrate used by HCP species.

Direct and Indirect Effects

Fish and invertebrates require a range of substrate conditions in riverine environments for various life-history stages. These conditions rely on the replenishment of suitably sized substrates to offset natural sediment transport processes that remove sediment. In one California study, the primary cause for the decline of salmon in the Sacramento River was linked to the loss of spawning gravels normally derived from bank erosion before riprap bank stabilization (Buer et al. 1984).

7.3.2.2.3 Altered Groundwater-Surface Water Exchange

Most groins or their analogs do not require the use of stabilizing structures such as pilings or other impermeable structures that impede the exchange of hyporheic water with main river channels. However, when stabilizing structures are necessary, ecological impacts associated with the loss of exchange of hyporheic flow will occur.

Ecological connectivity is essential between riverine and riparian ecosystems (Stanford and Ward 1993). In riverine environments, connectivity is generally expressed in three dimensions: longitudinally (upstream–downstream), laterally (channel–floodplain), and vertically (channel–hyporheic zone [the interface between surface and groundwater]) (Stanford and Ward 1993).

The quality of habitat connectivity in one dimension may affect that in another dimension. For instance, the hyporheic zone serves as a medium across which dissolved organic matter and nutrients are exchanged between the riparian zone and surface water. A high level of substrate fines in channel substrate may hinder the connection between surface and groundwater, limiting vertical and lateral connectivity between these two habitat types (Edwards 1998; Pusch et al. 1998). This lack of connectivity can degrade conditions for riparian zone vegetation, reducing LWD recruitment to the stream channel, subsequently limiting habitat-forming and maintaining processes. Effects on ecological functions and freshwater aquatic species associated with degraded connectivity between different riverine habitat elements are well documented (Bilby and Bisson 1998; Hershey and Lamberti 1992; Karr 1991; Kelsey and West 1998; Montgomery et al. 1999; Naiman et al. 1992; Reiman and McIntyre 1993; Stanford and Ward 1992; Stanford et al. 1996).

Stream temperature is an important factor in determining the suitability of habitats for aquatic species. The interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperatures (Poole and Berman 2001a), so alteration of groundwater flow can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel pattern and depth of the aquifer (Poole and Berman 2001a).

Direct and Indirect Effects

Structures that alter groundwater dynamics in riverine systems can directly affect fish and invertebrates in the short-term by influencing water quality and habitat suitability or availability. In the long-term, changes to groundwater exchange can generate indirect effects on fish and invertebrate species by affecting low flow conditions (i.e., increasing the magnitude of periods of drought resulting in reduced habitat availability and suitability, potential stranding or desiccation), and by affecting water quality through warmer stream temperatures and decreased organic and nutrient inputs.

7.3.2.2.4 Altered Substrate Composition

Hardened banks that replace riparian vegetation can increase the flow velocity and potential for scour and substrate coarsening through a reduction in hydraulic roughness compared to vegetated conditions (Millar and Quick 1998). Groins can alter the composition of bed and bank materials by virtue of adding material coarser than the ambient bed and by redirecting flow and coarse sediment from the river bank to the center of the channel. By deflecting high-velocity flows to the center of the channel and inducing flow separation, groins can also cause fine sediment deposition along the banks and between structures in areas where pools might otherwise form under natural conditions.

Increased velocities associated with flow constrictions created by groins can indirectly affect HCP species by causing local bed scour around structures and corresponding sediment deposition downstream (Richardson and Davies 2001). Bed scour into a substrate of mixed particle sizes (i.e., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, deposition of the finer sediment downstream can bury organisms and cause the substrate to become finer. For groins and barbs, the depth and extent of bed scour depends on the water depth, approach velocity, and shape and size of the obstruction (Richardson and Davies 2001).

Direct and Indirect Effects

Alteration of the substrate composition through coarsening or fining of the bed can have direct and indirect effects on HCP species. The ecological effects of substrate coarsening and fining on salmonids and trout in riverine environments are well known. Far less is known about the effects of these disturbances on the life-history stages of other freshwater fish and invertebrates. Large substrates exceeding the maximum size mobilized by spawning salmonids are avoided during redd building (Kondolf and Wolman 1993). Field observations have shown that salmonids can build redds where the average substrate size (D_{50}) is up to 10 percent of the average body length (Kondolf and Wolman 1993). The optimal range of spawning gravels for salmonids is listed in Table 7-6.

Bed scour and substrate coarsening are often accompanied by an increase in the interlocking strength of bed particles and the threshold force necessary for bed mobility (Church et al. 1998; Konrad 2000; Lane 1955).

Table 7-6. Spawning gravel criteria for salmonids.

Gravel bed criteria	Small-bodied Salmonids <13.8 in (<35 cm)	Large-bodied Salmonids >13.8 in (>35 cm)
Dominant substrate particle size	0.3–2.5 in (8–64 mm)	0.6–5 in (16–128 mm)
Minimum gravel patch size	10.8 ft^2 (1 m^2)	21.5 ft^2 (2 m^2)

Note: Small-bodied salmonids include cutthroat trout. Large-bodied salmonids include coho and Chinook salmon and steelhead trout (after Schuett-Hames et al. 1996).

Substrate fining can adversely affect salmonids and trout by clogging spawning gravels with fine sediment (Zimmermann and Lapointe 2005). Embryo mortality has been found to occur from poor water circulation and lack of oxygenation associated with the filling of intergravel pore spaces by fine sediment (Cooper 1965; Bennett et al. 2003; Chapman 1988; Lisle and Lewis 1992). In a study of spawning chum salmon in low-gradient, gravel-bed channels of Washington and Alaska, Montgomery et al. (1996) found that minor increases in the depth of scour caused by bed fining and a reduction in hydraulic roughness significantly reduced embryo survival. The deposition of fine sediment can also adversely impact invertebrates (Wantzen 2006). Fine sediment particles may clog biological retention mechanisms such as the filtering nets of caddisfly larvae, or the filtering organs of mollusks. Additionally, overburden from increased deposition has been shown to adversely affect invertebrates having low motility (Hinchey et al. 2006).

7.3.2.3 Riparian Vegetation Modifications

Removal or disturbance of riparian vegetation during construction of activities permitted under HPAs can expose HCP species to stressors caused by a variety of impact mechanisms. These mechanisms include:

- Altered shading and solar input
- Altered ambient air temperature
- Reduced shoreline stability
- Altered allochthonous inputs
- Altered groundwater-surface water exchange.

These impact mechanisms and related ecological stressors are discussed below.

7.3.2.3.1 Altered Shading, Solar Input, and Ambient Air Temperature

Removal of riparian vegetation as part of bank protection projects affects water temperature in riverine environments through a number of mechanisms. The dominant mechanism is the effect of reduced shading on solar radiation exposure. The influence of shade on water temperature generally diminishes as the size of the stream increases, because of the proportionally reduced area in which riparian vegetation can insulate against solar radiation and trap air next to the water surface (Knutson and Naef 1997; Murphy and Meehan 1991; Poole and Berman 2001a; Quinn

2005). Alternatively, riparian vegetation removal and alteration can also cause surface waters to gain or lose heat more rapidly because the ability to regulate ambient temperatures is reduced (Bolton and Shellberg 2001; Knutson and Naef 1997; Murphy and Meehan 1991; Poole and Berman 2001a; Quinn 2005).

In addition to the effects of shading, a broad array of research has shown that alterations of riparian vegetation can strongly affect temperatures even when adequate stream shading is still provided. Riparian vegetation restricts air movement, providing an insulating effect that regulates ambient air temperatures. Alterations of the riparian buffer width and vegetation composition can degrade this insulating effect, leading to greater variability in ambient air temperatures that in turn influence water temperatures (AFS and SER 2000; Bartholow 2002; Barton et al. 1985; Beschta 1991, 1997; Beschta et al. 1988; Beschta and Taylor 1988; Brosofske et al. 1997; Brown 1970; Chen et al. 1992, 1993, 1995, 1999; Johnson and Jones 2000; MacDonald et al. 2003; May 2003; Murphy and Meehan 1991; Spence et al. 1996; Sridhar et al. 2004; Sullivan et al. 1990; Theurer et al. 1984; USFS et al. 1993). For example, Chen et al. (1995) found that maximum air temperatures at the margins of old-growth forest stands are elevated 3–29°F (2–16°C) relative to interior temperatures. Riparian buffer widths of 100 to 300 feet may be necessary to provide full ambient temperature regulation (AFS and SER 2000; Brosofske et al. 1997).

Direct and Indirect Effects

Alteration of the temperature regime in riverine and lacustrine systems due to the alternation of riparian vegetation is a well-documented stressor on native fish populations. Loss or degradation of the shading and ambient temperature regulation functions provided by riparian vegetation can lead to increased water temperatures in summer, when solar radiation exposure and ambient air temperatures are highest. In winter, loss or degradation of the insulating capacity of riparian vegetation can lead to decreased water temperatures and increased incidence of ice scour. Increased stream temperatures can also cause a concomitant decrease in dissolved oxygen levels, an additional stressor with additive deleterious effects. Numerous studies of such effects in the field have documented the deleterious effects on fish species of changes in the temperature regime in freshwater stream systems (Poole et al. 2001; Sullivan et al. 2000).

7.3.2.3.2 Altered Shoreline Stability

Bank barb and groin projects often involve the temporary or permanent modification of riparian vegetation structure. Riparian vegetation is an important component of the aquatic ecosystem that serves a variety of important functions for habitat structure, water quality, and biological productivity. The specific nature of these functions varies depending on the type of environment, but increasing bank cohesion plays an important role in regulating channel width and substrate.

The root structure supporting riparian vegetation naturally resists the shear stresses created by flowing water and thus retards bank erosion, stabilizing stream banks and shorelines, and maintaining valuable habitat features along stream margins, such as undercut banks. By dissipating the erosive energy of flood waters, wind and rain, and by filtering sheet flows, riparian vegetation limits the amount of fine sediment entering river and stream systems (Brennan and Culverwell 2004; Knutson and Naef 1997; Levings and Jamieson 2001). If riparian vegetation is removed as part of groin and bank barb projects, stream banks and shorelines will likely be exposed to the erosive effects of wind, rain, and current. The removal of riparian trees and understory can dramatically alter stream bank stability and the filtering of sediments from overland flow (Kondolf and Curry 1986; Shields 1991; Shields and Gray 1992; Simon 1994; Simon and Hupp 1992; Waters 1995), resulting in increased erosion and increased inputs of fine sediment (Bolton and Shellberg 2001).

Direct and Indirect Effects

The direct effects of a loss of riparian vegetation are twofold. First, slope instability can increase turbidity by delivering an excess amount of fine-grained sediment to a river, affecting water quality and habitat conditions. The effects associated with turbidity are discussed in Section 7.3.2.5 (*Water Quality Modifications*). Second, slumping of unstable banks caused by a loss of riparian vegetation can also bury invertebrates and andromous fish eggs.

Burial with fine sediments has been associated with high mortality levels in freshwater mollusk species. Mussel mortality rates exceeding 90 percent have been observed following burial with silt (Ellis 1942), and burial with fines has been implicated in large-scale mortality of western pearlshell mussels in the Salmon River in Idaho (Vannote and Minshall 1982). In a survey of native freshwater mussels in the United States and Canada, it was concluded that declines in populations were caused by habitat destruction, dams, siltation, and channel modifications, with siltation a significant issue in some areas (Williams et al. 1993).

Burial with coarse sediment appears to be less problematic, provided that the stressor is short term in duration. Krueger et al. (2007) studied the effects of burial on western ridged and western pearlshell mussels species in the Similkameen River in Washington State. Interestingly, they found that mussels buried under less than 40 cm (15 inches) of coarse sediment (gravel and cobble) were able to extricate themselves. Test subjects buried at or beyond this depth suffered only a 10 percent mortality rate over the 6-week period. However, none of these individuals were able to extricate themselves. This suggests that burial in coarse sediments caused by bedload scouring could lead to high rates of delayed mortality from starvation as well as other effects.

The other freshwater mollusks, great Columbia River spire snail and giant Columbia River limpet, hatch from the egg fully formed. As such, these species would be expected to have a higher level of sensitivity to the effects of burial.

Burial of redds by fine sediments common to bank failures can also increase egg mortality in salmonid species (Soulsby et al. 2001). The infiltration of the pore spaces in coarse-bedded streams not only influences the physical environment surrounding the eggs, but the chemistry of the newly sealed streambed can undergo changes in the hydrochemistry of the spaces, as interactions with groundwater become severely altered (see Section 7.3.2.2.4 [*Altered Substrate Composition]* for more details).

7.3.2.3.3 Altered Allochthonous Input

Riparian detritus and other externally derived (allochthonous) materials are the primary source of organic fodder in headwater streams, forming the basis for the food chain (MacBroom 1998). This material includes terrestrial macroinvertebrates, along with leaves, branches, and other vegetative materials, the latter providing food sources for benthic macroinvertebrates (Bilby and Bisson 1998; Knutson and Naef 1997; Murphy and Meehan 1991). As rivers increase in order and grow in size, these materials are processed and recycled by an increasing diversity of organisms (Vannote et al. 1980). Without allochthonous inputs, the forage detritus available for benthic macroinvertebrates is compromised, also diminishing habitat and species diversity of these prey items (Murphy and Meehan 1991). Removal of freshwater riparian vegetation as part of groin and bank barb projects would cause an incremental decrease in the input of externally derived (allochthonous) materials to the nearby aquatic environment and food web.

Direct and Indirect Effects

As in marine and lacustrine environments, a reduction in allochthonous food sources to rivers diminishes the ability of the system to support higher trophic organisms. This would extend to most of the freshwater HCP species.

7.3.2.3.4 Altered Groundwater-Surface Water Exchange

Alteration or removal of riparian vegetation would appreciably change the interface between plants, soil, and water on and near the bank surface. Riparian vegetation also acts as a filter for groundwater, removing sediments and taking up nutrients (Knutson and Naef 1997). It also, in conjunction with upland vegetation, moderates streamflow by intercepting rainfall, contributing to water infiltration and using water via evapotranspiration. Plant roots increase soil porosity, and vegetation helps to trap water flowing on the surface, thereby aiding in infiltration as the water stored in the soil is later released to streams through subsurface flows. Through these processes, riparian and upland vegetation help to moderate storm-related flows and reduce the magnitude of peak flows and the frequency of flooding. Riparian vegetation, the litter layer, and silty soils absorb and store water during wet periods and release it slowly over a period of months, maintaining streamflows during low rainfall periods (Knutson and Naef 1997).

Groin and bank barb projects that create a physical barrier between the bank and hyporheic flow may prevent exchange between the bank and with the aquatic ecosystem. The interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperatures (Poole and Berman 2001b), so alteration of groundwater flow can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel pattern and depth of the aquifer (Poole and Berman 2001b).

Direct and Indirect Effects

The effects of altering groundwater input are discussed in depth in the Section 7.3.2.2 (*Hydraulic and Geomorphic Modifications*).

7.3.2.4 Aquatic Vegetation Modifications

Aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of stream ecosystems. In general, benthic algae occur in the form of microscopic unicellular algae, forming thin layers or assemblages called periphyton. Macrophytes include angiosperms rooted in the stream bottom, along with mosses, and other bryophytes. These include many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998). A small algal biomass in a stream can support a much larger biomass of consumers due to the rapid turnover in biomass (Hershey and Lamberti 1992; Murphy 1998). Although aquatic primary production is sometimes underrated due to the small amount of algae and plants present in many streams, it is a basic energy source for freshwater ecosystems.

7.3.2.4.1 Direct and Indirect Effects

The loss of aquatic vegetation in riverine environments poses both direct and indirect effects on HCP species that depend on aquatic vegetation for any one of their life-history stages, such as green and white sturgeon, California floater and western ridged mussels, mountain sucker, giant Columbia River limpet, pygmy whitefish, leopard and umatilla dace, bull trout, Dolly Varden, and Pacific salmon (Frest and Johannes 1995; Hughes and Peden 1989; Mongillo and Hallock 1998; Mongillo and Hallock 1999; Watters 1999).

The uptake of carbon, nitrogen, and phosphorous by aquatic vegetation provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous organic matter and part of the source of allochthonous matter in each stream reach. Although terrestrial and adult aquatic insects are important (Bjornn and Reiser 1991), juvenile salmon in streams have been found to be primarily supported by autochthonous organic matter (Bilby and Bisson 1992). A further general discussion of the effects of aquatic vegetation alterations on autochthonous organic production is presented for jetties in lacustrine environments in Section 7.1.2.4 (*Aquatic Vegetation Modifications*).

The density of coho salmon fry in the summer has been found to be directly related to the abundance of algae. A high density of fry can result from smaller feeding territories (Dill et al. 1981) due to increased invertebrate prey (Hawkins et al. 1983; Murphy et al. 1981). Increases in vertebrate production have been found to occur primarily in the spring and early summer, coincident with the primary production cycle of benthic algae (Murphy 1998). Therefore, removal of or permanent disturbance to algal communities could have an adverse effect on local freshwater ecosystems and the HCP species that depend upon these ecosystems. In the case of coho salmon fry, the reduction in prey area (i.e., smaller feeding territories) results in a direct effect on the fitness, growth, and survival of the affected fry.

7.3.2.5 Water Quality Modifications

A number of water quality modifications can result from the installation of shoreline modifications. Aside from the installation of potentially toxic materials (discussed briefly below, but in more depth in Section 7.1.1 (*Marine Environments*) of Section 7.1 (*Jetties*), most water quality modifications occur as result of the inhibition of mixing and heightened (thermal) stratification. Heightened stratification can ultimately lead to eutrophication (Fischer et al. 1979).

Eutrophication of receiving waters across Washington State has been identified as a major source of environmental degradation (Nelson et al. 2003; Pickett 1997). Any activity that decreases in-channel processing of nutrients or riparian buffer widths will contribute to the increased export of nutrients to downstream receiving waters, potentially affecting many of the HCP species. Excess nutrients in a water body may lead to eutrophication. Eutrophication is characterized by elevated primary production and associated elevated respiration. In eutrophic systems, nighttime respiration drives down dissolved oxygen to levels that would adversely affect many of the HCP species.

7.3.2.5.1 Altered Temperature

Water temperature is strongly dependent on mixing in rivers and streams (Fischer et al. 1979). Placement of groins affects these mixing processes, often reducing mixing and increasing thermal stratification. Stratified waters can lead to elevated surface temperatures, particularly during the summer months (Fischer et al. 1979).

Direct and Indirect Effects

A substantial amount of information is available about tolerances of HCP fish species (particularly salmonids) to thermal stress. For instance, it has been found that coho egg, alevin, and fry development is most rapid at 39^{o}F (4^{o}C), while alevin and fry of pink and chum salmon develop fastest at 46^{o}F (8^{o}C) (Beacham and Murray 1990). Despite these and other small differences, the majority of salmonids prefer the same temperature ranges during most life-history stages. The primary exception to this is that char (bull trout and Dolly Varden) require lower temperatures for optimal incubation, growth, and spawning (Richter and Kolmes 2005).

Elevated water temperatures can also impair adult migration and spawning. Adult migration blockages occur consistently when temperatures exceed 70–72^{o}F (21–22^{o}C) (Poole and Berman 2001a). Thermal barriers to migration can isolate extensive areas of potentially suitable spawning habitat and contribute to prespawning mortality. If salmon are exposed to temperatures above 57^{o}F (14^{o}C) during spawning, gametes can be severely affected, resulting in reduced fertilization rates and embryo survival (Flett et al. 1996). Ideal temperatures for salmonid spawning are in the range of 44–57^{o}F (7–14^{o}C) (Brannon et al. 2004; McCullough et al. 2001). See Tables 7-2 and 7-3 in Section 7.1.1.5 (*Water Quality Modifications*) for details.

Considerably less research exists defining thermal criteria for invertebrates, although marine invertebrates can generally withstand higher temperatures. Gagnaire et al. (2006) noted that elevated temperatures caused blood cell mortality in Pacific oysters but not until temperatures exceeded 104°F (40°C), which is unlikely even in altered settings. In studies on northern abalone, optimal growth rates were found between 44.6 and 62.6°F (7 and 17°C) (Hoshikawa et al. 1998), with significant mortality at 32.9°F (0.5°C) and 79.7°F (26.5°C) (Paul and Paul 1998). It is unclear, however, what sublethal effect(s) may be significant with invertebrate populations. In general, an altered temperature regime will result in blocked migrations, increasing the chances of infection, deformities in developing eggs, stress, and mortality of several HCP species.

7.3.2.5.2 Suspended Solids and Turbidity

The presence of suspended solids and turbidity will result from any geomorphic change as well as various construction activities associated with the installation of a groin or bank barb. Turbidity has a number of effects on HCP species. Because these effects are essentially independent of the environment, a full discussion of them is included in Section 7.1.1.5 (*Water Quality Modifications*).

7.3.2.5.3 Dissolved Oxygen

Eutrophication is characterized by elevated primary production and associated elevated respiration. In eutrophic systems, nighttime respiration drives down dissolved oxygen to levels that would adversely affect many of the HCP species. The effects on HCP species from reduced dissolved oxygen are discussed in the *Eutrophication* subsection below.

7.3.2.5.4 Nutrient and Pollutant Loading

Numerous studies have shown that macrophytes and algae in both marine and freshwater environments act to reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). In a study of macrophyte impact on sediment and nutrient retention in Danish streams, Sand-Jensen (1998) reported that dense-stemmed macrophytes created conditions conducive to sediment deposition and that the sediments retained within the macrophyte stands were fine-grained and nutrient-rich. He noted that enrichment of sediment within macrophyte beds relative to the surrounding substratum was 0.1597 lb organic matter per ft^2 (780 g /m^2), 0.006 lb nitrogen per ft^2 (30 g/m^2) and 0.005 lb phosphorus per ft^2 (25 g per m^2).

Channel complexity promotes the retention of water and organic material. This retentiveness plays an important role in the fate of nutrients in the stream channel. In a study by Mulholland et al. (1985), it was suggested that leaf litter in streams promotes nutrient retention as the leaf pack acts as a substrate for nutrient-hungry microbes. Using solute injection techniques, Valett et al. (2002) found that phosphorus uptake in channels with high LWD volumes, frequent debris dams, and fine-grained sediments was significantly greater than in channels in younger forests without these characteristics. Corroborating this finding, Ensign and Doyle (2005) conducted phosphorus injections in streams both before and after the removal of LWD and coarse-particulate organic matter (CPOM) in the channels and found that phosphate uptake decreased by up to 88 percent after LWD removal. These studies show that channel complexity increases water retention and, through CPOM and LWD retention, provides a substrate for biofilm growth. Groins typically simplify channel structure, causing a drop in complexity. Decreased nutrient retention will affect both local waterways and downstream receiving waters. Local waterways will be affected through the associated reduction in primary production, and receiving waters (which are primarily located in more nutrient-impacted lowland areas) will be affected through additional nutrient loading, which may lead to eutrophication.

Direct and Indirect Effects on Fish and Invertebrates

Eutrophication

When riparian canopies are opened, increased photosynthetic active radiation reaches the channel, temperatures increase, and nutrient loading increases. These alterations have been shown to increase macroinvertebrate abundance and biomass as well as algal biomass (Fuchs et al. 2003; Hetrick et al. 1998). However, the cumulative effect of increased nutrient loading will contribute to eutrophication in downstream receiving waters. Eutrophication occurs when limits to vegetative growth are reduced. In Washington, the primary limiting nutrient in fresh water is phosphorus. This is due to the fact that abundant iron in freshwater systems binds with phosphorus (P) and reduces P availability for biotic assimilation. When nutrient limitations are eliminated, vegetative growth increases. This process accelerates carbon fixation; the additional carbon loading to the aquatic system increases respiration as heterotrophs use carbon for energy. Through the process of carbon oxidation, oxygen is converted to CO_2, and ambient dissolved oxygen levels decrease. Eutrophication-induced hypoxia is a nationwide problem (Scavia and Bricker 2006). As all of the HCP species are aerobic respirators, reduction in dissolved oxygen levels can cause stress and mortality. See Section 7.1.1.5 (*Water Quality Modifications*) for details.

Any activity that affects riparian areas will also degrade the buffering capability of the terrestrial–aquatic ecotone. Numerous studies have shown that wide stream buffers are effective at attenuating nutrients (Feller 2005; Mayer et al. 2005), herbicides (Gay et al. 2006), and sediment loading (Jackson et al. 2001). Riparian vegetation retards overland flow, promotes infiltration, and assimilates shallow groundwater nutrients. When this vegetation is removed through any HPA-permitted activity, nutrients and pollutants will be more efficiently transported from upland sources to downgradient water bodies. Forested buffers can effectively remove nutrients in shallow groundwater. In a study of a forested buffer in Alabama, a 33-ft (10-m) buffer reduced the groundwater nitrate concentration by 61 percent (Schoonover and Williard 2003). In a subsequent study of a forested wetland buffer, a buffer averaging 125 ft (38 m) in width reduced the nitrate concentration by 78 percent and total phosphorus by 66 percent (Vellidis et al. 2003).

Metal Toxicity

In urban environments, metals loadings to local waterways and water bodies from anthropogenic sources is a major pathway for aquatic habitat degradation. The primary metals of concern in the surface waters of Washington are copper, zinc, arsenic, lead, and nickel (Embrey and Moran 2006). Metals above threshold concentrations act as carcinogens, mutagens, and teratogens in fish and invertebrates (Wohl 2004). Additionally, the sublethal effects of copper toxicity have been extensively studied, with some reported effects including impaired predator avoidance and homing behavior (Baldwin et al. 2003). Ecology has established water quality standards for fresh waters for each of these constituents. These standards, issued in WAC 173-201a, are listed in Table 7-7. Freshwater toxicity thresholds are hardness dependent and can vary widely depending on alkalinity. The standards presented here are based on median freshwater hardness concentrations estimated from an extensive 3-year data set (2001–2003) from the Green River watershed (Herrera 2007b).

Table 7-7. Water quality criteria for fresh waters of the state of Washington based on median hardness values.

Constituent	Acute (ppb)	Chronic (ppb)
Arsenic	360[a]	190[b]
Copper	7.0[a]	7.5[b]
Lead	22.9[a]	1.5[b]
Nickel	640[a]	104[b]
Zinc	51.6[a]	69.2[b]

[a] Criterion varies with hardness. Acute criterion is based on an estimated median storm flow hardness of 39.1 ppm as $CaCO_3$.

[b] Criterion varies with hardness. Chronic criterion is based on an estimated median base flow hardness of 61.5 ppm as $CaCO_3$.

7.3.2.5.5 Altered pH Levels

The pH of fresh and salt water normally ranges from 6.5–8.5 (Schlesinger 1997). The construction of groins or other shoreline structures using concrete can affect the pH of surrounding waters if the uncured concrete is allowed to contact the receiving water body. Uncured concrete can dissolve in water and, depending on temperatures, can raise the pH level to as high as 12, which is far outside the livable range for all of the HCP species (Ecology 1999). This impact will be greatest during construction when concrete wash-off and slurries come into contact with water (Dooley et al. 1999); however, once construction is complete, concrete may still affect the surrounding environment. Curing concrete surfaces can exhibit pH values as high as 13 during the 3 to 6 months it takes for concrete to cure underwater (Dooley et al. 1999). This elevated pH prevents attached macroalgae growth during this period.

Altered pH from curing concrete will increase pH to levels that can affect fish, invertebrates, and their food. But this effect is localized and, as stated above, should last no more than 6 months. Consequently, it is estimated that this impact mechanism will be most significant for large projects in areas with poor water circulation.

Direct and Indirect Effects

Fish have adapted to the ambient pH levels of their particular habitat, and they tend to have narrow ranges of pH tolerance. The effects of high pH levels that are outside of their tolerance range can include death; damage to gills, eyes, and skin; and an inability to excrete metabolic wastes (DFO 2007). Elevated pH has been shown to increase ammonia toxicity in fish because the organisms have difficulty excreting ammonia waste through their gills when ambient conditions are characterized by elevated ammonia and pH. It has been shown that at ambient ammonia concentrations of 5.0 ppm, mortality of tambaqui (*Colosoma macropomum*; also known as pacu), increased from 0 to 15 to 100 percent at a pH of 7, 8, and 9, respectively (de Croux et al. 2004). Consequently, if ammonia concentrations are elevated due to waste dumping from recreational vessels or from upland sources, the toxicity may be compounded by elevated pH from construction activities.

It is known that pH alone can affect fish exposed to alkaline conditions. In a rainbow trout toxicity study, a pH above 8.4 caused an increase in glucose and cortisol levels, and a pH above 9.3 caused mortality (Wagner et al. 1997). Alterations in pH can also affect invertebrates. The majority of research on the effect of pH on invertebrates is related to the impact of acidification on abundance and diversity; consequently, there is little research on the impact of elevated pH on invertebrates. In a study of the freshwater Malaysian prawn, Cheng and Chen (2000) noted a 38 percent decrease in haemocyte (invertebrate blood cell) count when pH dropped below 5 or rose above 9. In another study, Bowman and Bailey (1998) found that zebra mussels have an upper pH tolerance limit of 9.3 through 9.6. From these studies, it can be assumed that pH levels that exceed a pH of between 9 and 10 will have a negative impact on invertebrate HCP species. As indicated above, pH levels on and around curing concrete can exceed this pH threshold and thus there is the potential for impact on local invertebrate communities.

7.3.2.5.6 Treated Wood Pollution

Treated wood is not often used in riverine environments, particularly in new construction, but it can often be found in older structures and re-exposed during maintenance operations. For a detailed discussion of the types of pollutants typically encountered in shoreline modifications, see Section 7.1.1 (*Marine Environments*) in Section 7.1 (*Jetties*).

Creosote-Treated Wood–Direct and Indirect Effects

The greatest impacts of creosote-treated wood are on those benthic and burrowing organisms present on the treated wood structures. Habitat areas of lower pH and reduced water circulation are at a greater risk of contamination. Metals from creosote-treated wood generally become incorporated into the local sediments and are usually undetectable in ambient waters. Poston

(2001) concluded that riverine salmon spawning substrates do not typically accumulate PAHs or metals but that salmonids are potentially at some risk of exposure from the consumption of contaminated prey. Stratus (2005a) reported that a number of jurisdictions have recently put prohibitions in place on the use of creosote-treated wood.

<u>ACZA and CCA Treated Wood–Direct and Indirect Effects</u>

The U.S. Environmental Protection Agency (USEPA) has established aquatic life criteria (ALC) (i.e., concentration criteria) for the constituent metals that may leach from ACZA− or CCA Type C−treated wood (USEPA 2002, in Stratus 2005b). The ALC have been established for criterion maximum concentrations (CMCs) for acute exposure and criterion chronic concentrations (CCCs) for chronic exposure for both salt and fresh water (refer to Table 7-8). In both fresh and salt water, invertebrates are the species most sensitive to copper, chromium VI, zinc, and arsenic (Stratus 2005b). These ALC appear to be appropriate for acute lethal impacts of copper and chromium VI (Stratus 2005b), but avoidance responses and olfactory neurotoxicity may occur in salmonids at sublethal copper concentrations, even with brief exposure (Hansen et al. 1999a, 1999b; Baldwin et al. 2003; Sandahl et al. 2004; all in Stratus 2005b), and there may be a risk of bioaccumulated toxicity in salmonid prey species at the chronic chromium VI criterion (Stratus 2005b).

Table 7-8 U.S. water quality criteria for the protection of aquatic life (“aquatic life criteria”) for water soluble chemicals used in treating wood.

Chemical	Freshwater CMC (ppb)	Freshwater CCC (ppb)	Saltwater CMC (ppb)	Saltwater CCC (ppb)
Arsenic	340	150	69	36
Copper[e]	7.0[a]	5.0[a]	4.8	3.1
Copper (2003)	BLM[b]	BLM[b]	3.1	1.9
Chromium III	323	42	None (850)[c]	None (88)[d]
Chromium VI	16	11	1,100	50
Zinc	65[a]	65[a]	90	81

Source: USEPA 2002, except as noted, as taken from Stratus 2005b.

[a] Criteria are hardness-dependent. Criteria values calculated using site-specific hardness based on the equations presented in USEPA (2002). Hardness-dependent criteria values are presented for a hardness of 50 ppm (as CaCO3).

[b] Criteria developed using site-specific chemistry and the Biotic Ligand Model (BLM).

[c] No saltwater CMC. As a proxy, we report the lowest reported LC_{50} from the USEPA database (Lussier et al. 1985) divided by a factor of two. See text for additional details.

[d] No saltwater CCC. As a proxy, we report the lowest reported chronic value from the USEPA database (Lussier et al. 1985) divided by a factor of two. See text for additional details.

[e] From USEPA 2002.

From draft ALC guidance on copper provided by USEPA in 2003 that relies on the BLM for calculating freshwater criteria based on site-specific water chemistry.

CMC = criterion maximum concentration.

CCC = criterion chronic concentration.

ppb = parts per billion.

There does not appear to be a pattern of sensitivity among species with respect to chromium III, but the ALC, although established only for fresh water, appears to be protective of fish, particularly salmonids (Stratus 2005b). If chromium III toxicity is related to salinity (similar to chromium VI and copper), then the application of the freshwater criteria to salt water would include a margin of safety. The ALC for zinc are water hardness-dependent and do not appear to be protective of salmonids in fresh water of low hardness (30 ppm) (Hansen et al. 2002, in Stratus 2005b); however, the zinc ALC for salt water are likely protective of salmonids (Stratus 2005b).

Avoidance behavior has also been observed among salmonids at zinc concentrations below or slightly above the ALC (Sprague 1964, 1968; Black and Birge 1980, all in Stratus 2005b). The ALC for arsenic are likely to be protective of salmonids (Stratus 2005b). Overall, the ALC are suitable for assessing the impacts of ACZA and CCA Type C-treated wood on water quality and the potential risk to HCP species (Stratus 2005b).

Metals from treated wood can contaminate sediment and affect benthic communities, limiting food resources for fish and exposing fish to metals contamination through the consumption of contaminated prey (Stratus 2005b). Table 7-9 lists the threshold effects concentrations for freshwater sediment and possible concentrations for freshwater sediment.

Table 7-9. Probable effects concentrations for arsenic, chromium, copper, and zinc in freshwater sediment.

Name	Definition	Concentration (mg/kg dry wt [ppm])					Reference
		Basis	As	Cr	Cu	Zn	
Lowest effects level	Level that can be tolerated by the majority of benthic organisms	Field data on benthic communities	6	26	16	120	(Persaud et al. 1991)
Biological threshold effects level	Concentration that is rarely associated with adverse biological effects	Compiled results of modeling, laboratory, and field studies on aquatic invertebrates and fish	5.9	37.3	35.7	123	(Smith et al. 1996)
Minimal effects threshold	Concentration at which minimal effects are observed on benthic organisms	Field data on benthic communities	7	55	28	150	(Environment Canada 1992)
Effects range low	Concentration below which adverse effects would rarely be observed	Field data on benthic communities and spiked laboratory toxicity test data	33	80	70	120	(Long and Morgan 1991)
Survival and growth threshold effects level	Concentration below which adverse effects on survival or growth are expected to occur only rarely	Laboratory toxicity tests on the amphipod *Hyalella azteca* using field-collected sediment	11	36	28	98	(Ingersoll et al. 1996; USEPA 1996)
Consensus threshold effects concentration	Concentration below which adverse effects are expected to occur only rarely	Geometric mean of above published effect concentrations	9.79	43.4	31.6	121	(MacDonald et al. 2000)

Sources: (Jones & Stokes 2006) and (Stratus 2005b).

7.3.2.6 Ecosystem Fragmentation

7.3.2.6.1 Habitat Loss and Fragmentation

Floodplains have been shown to act as nutrient sinks and carbon sources for adjacent channels (Tockner et al. 1999; Valett et al. 2005). Consequently, floodplain–channel connection augments allochthonous carbon budgets and expands the habitat accessible to HCP species. By forcing flow into narrow corridors, groins can interrupt floodplain-channel connection and thereby cause the loss of habitat area and productivity.

Direct and Indirect Effects

Floodplain connectivity creates fish forage and refuge habitat for several of the HCP species (Feyrer et al. 2006; Henning 2004). Chinook that rear on floodplains have been shown to grow faster than those rearing in adjacent channels (Sommer et al. 2001). Additionally, in a 2004 study of the Sacramento splittail (Ribeiro et al. 2004), fishes rearing in floodplain habitat were healthier and longer. Juvenile coho have also been shown to use floodplain habitats when they are available, even in locations where they have disconnected in the past (Henning et al. 2006; Morley et al. 2005).

7.3.2.6.2 Loss of Large Woody Debris (LWD)

Woody debris in freshwater streams controls channel morphology, regulates the storage and transport of sediment and particulate organic matter, and creates and maintains hydraulic complexity that contributes to fish habitat (Murphy and Meehan 1991; Naiman et al. 2002). Within channels, approximately 70 percent of structural diversity is derived from root wads, trees, and limbs that fall into the stream as a result of bank undercutting, mass slope movement, normal tree mortality, or windthrow (Knutson and Naef 1997). In small streams, LWD is a major factor influencing pool formation in plane–bed and step–pool channels. Bilby (1984) and Sedell et al. (1985) found that approximately 80 percent of the pools in several small streams in southwest Washington and Idaho were associated with wood. In larger streams, the position of LWD strongly influences the size and location of pools (Naiman et al. 2002). In larger streams, LWD is typically oriented downstream due to powerful streamflow, which favors the formation of backwater pools along margins of the mainstem (Naiman et al. 2002). The hydraulic complexity created by LWD encourages the capture and sequestration of other allochthonous inputs, making these materials more available to the food chain through grazing and decomposition (Knutson and Naef 1997; Murphy and Meehan 1991; Naiman et al. 2002; Quinn 2005).

LWD accumulation creates habitat through two primary mechanisms. First, the wood itself provides cover and creates local scour pool habitat; second, wood in channels promotes lateral and vertical energy transfer, thus altering the composition and/or abundance of accessible food sources for HCP species and creating more accessible habitat for foraging and rearing.

The presence of LWD within channels creates locally complex geomorphic features, including scour pools and depositional bars. Depending on channel form and wood size and orientation, LWD-induced pools and bars can occur upstream, downstream, and/or lateral to wood structures. Groins simplify stream structure and focus flow energy into limited areas, causing LWD to be flushed downstream.

LWD itself serves as a substrate for algal growth and macroinvertebrate habitat (Bowen et al. 1998). In German streams, Hoffman (2000) showed an intimate connection between LWD and all life-history stages of the lepidostomatid caddisfly, *Lasiocephala basalis*. Meanwhile, Hilderbrand et al. (1997) noted no change in macroinvertebrate populations following a wood addition experiment in Virginia, and Spanhoff et al. (2006) noted a net negative impact of wood addition on stream macroinvertebrates. Despite these findings, other studies have shown that wood in channels and on shorelines can serve as a substrate for both algal growth (Bowen et al. 1998) and macroinvertebrate habitat (Rolauffs et al. 2001; Warmke and Hering 2000).

The presence of LWD within channels has been shown to promote floodplain connection during storm flow conditions by increasing flow resistance within the channel (Dudley et al. 1998). Increased channel roughness promotes backwater conditions that locally connect floodplain and channel habitat. As addressed previously, LWD also promotes floodplain connection by diverting flow into side channels (Abbe and Montgomery 1996).

<u>Direct and Indirect Effects</u>

Fish use these complex environments and the structure of the LWD itself for cover and refuge (Cederholm et al. 1997; Everett and Ruiz 1993; Harvey et al. 1999). In a study of Smith Creek in northwest California, Harvey et al. (1999) found that tagged adult coastal cutthroat trout moved more frequently from pools without LWD than from pools with LWD. They hypothesized that the habitat created by LWD attracts fish, and once fish establish territory within the desirable habitat, they remain there longer. A study by Cederholm et al. (1997) on a tributary of the Chehalis River found that LWD additions caused an increase in winter populations of juvenile coho salmon and age-0 steelhead populations. It should be noted that Fausch et al. (1995) and others have criticized studies such as Harvey et al. (1999) because it is difficult to determine if increased abundance in treatment sites is due to increased populations or simply just concentrations of fishes that would have thrived equally well in other habitat.

LWD increases access to floodplain habitat. The addition of LWD to a channel promotes floodplain–channel resource exchange and habitat accessibility; this will likely result in a benefit to many of the 52 HCP species, especially those that favor floodplain habitat.

7.3.2.6.3 Lost Opportunities

A potential impact of groins and bank barbs relates to the loss of opportunities. “Lost-opportunity impacts” result from projects that adversely alter natural fluvial processes important to the ongoing creation of fish and wildlife habitats (WDFW 2003). The following quote from WDFW (2003) provides a definition of lost-opportunity impacts:

> *"Preventing a channel from naturally migrating across the floodplain usually eliminates sources of woody debris, sediment, and side channels; these losses are defined as "lost opportunities." Natural channels evolve over time and migrate across their floodplains. When a channel naturally moves to a new alignment, it leaves behind vital habitat, such as floodplain sloughs and side channels. Those habitats have a finite productive longevity, some likely less than 20 years. If the natural fluvial processes of a stream are restricted or interrupted, these side-channel habitats will diminish in productivity and will not be replaced. These habitats cannot be mitigated by the design of a project. They are lost when a channel is fixed in a specific location, regardless of the bank-protection technique. Lost-opportunity impacts last as long as channel migration is halted."*

Likewise, groins and bank barbs can impose lost-opportunity costs through a number of pathways. Certain types of shoreline protection measures, like groins and bank barbs, may pose lingering ecological effects when they are not designed properly to account for channel movement or to allow sediment and wood transport. An equally important type of lost-opportunity cost occurs when the structures do not provide adequate passage for all fish and invertebrate species or life-history stages. In such instances, the costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly. This may delay the actions necessary to provide appropriate fish passage, perhaps as long as the design-life of the underperforming structure.

8.0 Cumulative Effects

Increasing evidence indicates that the most devastating environmental effects are most likely not the direct effects of a particular action, but the combination of individually minor effects of multiple actions over time (CEQ 1997).

In general, as the number of shoreline modification structures increases in a given area, impacts will accrue producing a net loss in vegetation production and a concomitant reduction in epibenthic and benthic nearshore habitat. The type and extent of each of these alterations depend on site-specific characteristics and structure types. The bathymetry of Washington's inland Puget Sound, as a fjord surrounded by a narrow strip of shallow, vegetated habitat, magnifies the need to protect the integrity and continuity of this limited area of nearshore habitat.

Numerous studies throughout the world have documented the cumulative impact of shoreline modifications on shoreline ecological communities (Byrnes and Hiland 1995; Guidetti 2004; Meadows et al. 2005; Penland et al. 2005; Wijnberg 2002). Because of the nature of these studies, they have not focused strictly on one activity type. The primary impacts identified by these studies are the disruption of littoral processes as well as hardening of the shoreline and consequent coarsening of the substrate, although other maritime activities likely also play a role [e.g., fishing: (Blaber et al. 2000; Guidetti, Verginella et al. 2005)]. As a result, the discussion of cumulative impacts in this white paper is not organized by activity type, but rather by mechanism of impact.

8.1 Construction and Maintenance Activities

Of the four submechanisms associated with construction and maintenance activities, the two that present the greatest risk of cumulative effects are navigation/maintenance dredging and channel/work area dewatering. These submechanisms are discussed in greater detail below.

Analysis of cumulative effects of landscape-scale bathymetry modifications and changes to habitat structure should include the overall scope of dredging activities undertaken in the region. Understanding the scope of current dredging activities requires a breakdown and comparison of the aerial extent of maintenance dredging undertaken annually compared to new project dredging, as well as the extent to which this dredging alters the nature of existing habitats in marine, riverine, and lacustrine environments. An analysis of the scope and nature of current dredging activities can lay the groundwork for assessing the long-term, cumulative effects that dredging activities can pose on existing ecosystem dynamics and the effects such changes may have on a variety of species.

The scope of such an assessment varies depending on the environment type. For example, in marine environments, an assessment might focus on the aerial extent of dredging activities within each of the oceanographically distinct basins in Puget Sound, differentiating habitat impacts in terms of the depth, substrate composition, and bathymetric profile of the affected areas. In lacustrine habitats, the analysis might have a similar focus but would be limited to the individual lake. In riverine environments, a watershed-scale approach or a more targeted approach differentiating estuarine impacts may be appropriate.

Although there are no available studies on the cumulative effects of temporary activities associated with channel dewatering, cumulative effects could result from the permitting of numerous dewatering activities within a watershed over a relatively short period of time. The cumulative impacts on a particular species' population would depend on the number of concurrent projects at a watershed scale, as well as the population size of a given species. The threshold for watershed and population size and the number of activities that must occur within a particular watershed to have a measurable cumulative impact are not yet established in the literature.

8.2 Water Quality Modifications

Water quality impacts, such as those discussed in Section 7.3 (*Groins and Bank Barbs*), are dependent on the level of use and design of the shoreline modification structure, the hydrography and geomorphology of the surroundings, as well as proximity of the structure to other affected habitat. Studies have shown that shoreline modifications in areas where there is poor tidal exchange are characterized by higher concentrations of pathogens than altered areas with elevated water circulation (Bordalo 2003), but there is no clear pattern indicating that shoreline modification structures consistently degrade water quality in every application.

8.3 Riparian Vegetation Modifications

Substantial loss and fragmentation of riparian habitat have occurred in the Puget Sound region over the last 100 years. Although, empirical data are lacking to quantify the extent and quality of riparian habitat, existing data suggest that riparian areas within urbanized shoreline areas such as King County have been significantly altered (up to 100 percent) by upland development and increasing levels of urbanization (Brennan and Culverwell 2004). Francis and Schindler (2006) described the process of urbanization in the Pacific Northwest as a trend that moves toward deforestation without replanting.

Naiman et al. (2000) reports that although riparian communities are being managed for a wider-than-ever variety of ecological functions, riparian communities in heavily urbanized environments constrained by pavement are precluded from the full restoration of natural functions. Shoreline modification structures require substantial impervious surface and encourage further shoreline development.

The Seattle-Tacoma area is an area of intense urbanization. A major finding in a study of the cumulative effects of urbanization on 22 Puget Sound streams found that mature forested riparian corridors were effective in mitigating some of the cumulative effects of adjacent development. In riparian corridors found in highly urbanized areas, poor stream quality is common (May 1998).

Site-specific habitat functions are determined by whether an existing shoreline is in a relatively natural state or whether it is affected by urban development; cumulative effects from additional shoreline modification structures also influence habitat functions. A natural environment that supports fish and shellfish spawning, rearing, and refugia is highly valuable from a biological perspective. Any alteration to that specific environment could influence the recruitment of fish and shellfish stocks in the larger ecosystem. As a result, the cumulative impact of additional marina or terminal structures along an ecologically intact shoreline could generate potentially significant cumulative effects.

In contrast, an urban, industrialized shoreline area may have, over a long period of time, lost its native vegetation and suffered major changes to its historical substrates. In that scenario, the addition of a new structure may pose a qualitatively different set of cumulative effects than the effects of the same new structure in a more natural environment.

8.4 Aquatic Vegetation Modifications

If the shoreline modification structure encourages vessel traffic, benthic disturbance, turbidity, and eelgrass and macroalgae disturbance will all increase. These potential effects, which are primarily associated with grounding, anchoring, and/or prop wash, are described in detail in the Marinas and Shipping/Ferry Terminals white paper (Herrera 2007a). In freshwater environments, boat use will likely increase as the state population increases, resulting in similar disturbance of bottom substrates and vegetation generated by prop wash.

Existing structures will continue to modify ambient light conditions and subsequently aquatic vegetation via shading and turbidity. An increase in facilities or facility capacity and an overall increase in vessel traffic will likely magnify these impacts. Future construction of new facilities could result in the removal of existing aquatic vegetation, further affecting these resources.

8.5 Hydraulic and Geomorphic Modifications

The hydraulic and geomorphic impact of cumulative shoreline hardening and maritime activities on the coastal ecological communities has been shown to be significant in a number of settings around the world (Byrnes and Hiland 1995; Guidetti 2004; Meadows et al. 2005; Penland et al. 2005; Wijnberg 2002). The primary results of these studies include the disruption of littoral processes as well as hardening of the shoreline and consequent coarsening of the substrate, although other maritime activities (e.g., fishing) likely play a role as well (Blaber et al. 2000; Guidetti, Bussotti et al. 2005). Although the notion of cumulative environmental impacts has been hypothesized to be important in the marine environment in Washington State (e.g., in Puget

Sound [Gelfenbaum et al. 2006]), there have been no systematic, peer-reviewed studies that have investigated the phenomenon in Washington waters. Despite this lack of local data, the sum of work performed outside of Washington State documents a general pattern of ecological change due to the construction of shoreline protection structures. In particular, the switch from biological communities preferring soft substrates and relatively quiescent conditions to those preferring higher wave-energies and harder substrates is almost always identified (Meadows et al. 2005; Guidetti 2004; Guidetti, Bussotti et al. 2005). For the outer coast of Washington, California is a relevant comparison. Although development has been more recent, there has been some documentation of the general hardening of shorelines in California. For instance, Wasson et al. (2005) described the increased prevalence of coarse substrate-dependent (invasive) communities on shoreline works.

Although many of these locales are superficially different from Washington State, some of these studies are particularly germane to anthropogenic environmental degradation. In particular, the paraglacial landscape of the Great Lakes and the Adriatic Sea provide similar templates to the geomorphic variables responsible for nearshore change in the Puget Sound, the Strait of Juan de Fuca, and the large lakes of western Washington (Finlayson 2006). These areas have also been developed for a much longer time (in the case of the Great Lakes, hundreds of years; in northern Italy, millennia), such that the cumulative effect has been made much clearer. For instance, Bearzi et al. (2004) documented the historical loss of marine mammals in the Adriatic and attributed the loss to human activities (in general).

Specific cumulative effects of groins on river systems have not been identified in the literature; however, several studies have documented the cumulative effects of bank hardening on riverine ecology. Riprap stabilization of one 15.5-mile (25-km) reach of the Sacramento River was cited as the primary cause of salmon decline in this river due to the loss of spawning gravels previously supplied from bank erosion (Buer et al. 1984; Shields 1991). In a comprehensive study of the historical decline of coho salmon smolt production in the lower Skagit River, Washington, Beechie et al. (1994) found that hydraulic modification from the combined effects of levee construction, bank hardening, and dredging accounted for 73 percent of summer habitat losses and 91 percent of winter habitat losses. Bank stabilization along 25 percent of the 99-mile (160-km) Garrison Reach of the Missouri River in North Dakota nearly eliminated the positive effect of riparian forest on the density of instream woody debris (Angradi et al. 2004).

The cumulative effects of bank hardening and historical removal of riparian forests throughout the lower Skagit River, Washington, have prevented wood recruitment from the natural processes of channel migration, thereby reducing the delivery of large wood to the estuary (Collins 2000). The loss of this wood can disrupt food webs for juvenile salmonids in estuarine marshes.

Previous studies on the cumulative effects of increased impervious areas have focused on the effects of urbanization on the ecology of urban streams. The condition of urban streams is controlled by the altered timing and volume of water, sediment, nutrients, and contaminants resulting from the urbanized catchment (Bernhardt and Palmer 2007). The most noticeable determinant of channel change in urban watersheds was the increase in streamflow discharges

(Booth and Henshaw 2001). Increased peak streamflows from urban development cause streams to incise deeper and wider channels (Hammer 1972; Booth 1990; Leopold et al. 2005). The consequence of this channelization is local bank failure, an increase in sediment supply, and sediment deposition in lower gradient, downstream reaches (Booth 1990). Konrad (2000) examined urban watersheds in the Puget Lowland and found that urban development increased peak discharge magnitudes and decreased storm flow recession rates, causing "flashy" runoff conditions. Consequently, substrate reworking by flow was more frequent and extensive in urban streams than in suburban streams draining less-developed watersheds. Summer base flow was also suppressed relative to suburban streams. The ecological effects of urbanization on urban streams are species-poor assemblages of fish and invertebrates (Freeman and Schorr 2004).

9.0 Potential Risk of Take

This section provides an assessment of the risk of take resulting from the impact mechanisms associated with shoreline modifications.

The following context is used as a basis for the risk of take assessment. First, it is clear that shoreline development will result in significant modification of the project site and the surrounding area, causing a fundamental alteration in the environmental characteristics of the natural shoreline. The impact mechanisms produced by development of the structures associated with shoreline modifications create environmental stressors. The risk of take resulting from stressor exposure will vary by species, depending on the nature of stressor exposure, as well as the sensitivity of the species and life-history stage exposed to the stressor. The magnitude, timing, duration, and frequency of each impact mechanism will vary widely with the project scale and location. Therefore, the assessment of risk of take associated with each impact mechanism is necessarily broad and applies a “worst-case scenario” standard. This assessment is conditioned by the species occurrence and life-history specific uses of habitats where jetties, breakwaters, and groins and bank barbs are typically developed. For example, for the purpose of this analysis, river environments are considered unsuitable for jetties and most breakwaters (which are appropriate only for the largest rivers where wind-driven waves and vessel wakes are sufficiently large to cause bank erosion). Therefore, many species and species life-history stages will not be exposed to related impact mechanisms and stressors. In contrast, groins and bank barbs can occur in lacustrine, marine, and riverine environments, including most small rivers and streams. The risk of take is rated for each species by impact mechanism using the criteria presented in Table 6-3. The following criteria are used to rate the risk of take:

- **High risk of take (H)** ratings are associated with:

    ☐ Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to an LTAA finding.

- **Moderate risk of take (M)** ratings are associated with:

    ☐ Stressor exposure is likely to occur causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or an NLTAA finding depending on specific circumstances.

- **Low risk of take (L)** ratings are associated with:
    - ☐ Stressor exposure is likely to occur causing take in the form of temporary disturbance and minor behavioral alteration. Likely to equate to an NLTAA finding.
- **Insignificant or discountable risk of take (I)** ratings apply to:
    - ☐ Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
- **No risk of take (N)** ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the subactivity type in question, or the impact mechanisms caused by the subactivity type will not produce environmental stressors.
- **Unknown risk of take (?)** ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

The risk of take assessment is organized by impact mechanism category and environment type. In cases where the physical effects and related risk of take are similar between environment types, the risk of take discussion is grouped to avoid redundancy. Each of the following subsections provides a general description of the risk of take associated with each impact mechanism and submechanism, as well as a discussion of the differences in the resulting risk of take for each subactivity type. This assessment is supported by risk of take matrices (Tables 9-1 through 9-3, presented at the end of the narrative) identifying the risk of take for each of the 52 HCP species by impact mechanism and environment type. These matrices consolidate the risk of take at the mechanism of impact level.

9.1 Construction and Maintenance Activities

Construction and maintenance of shoreline modification projects involve a diverse array of activities, including driving pilings, placement of materials, construction vessel operation, maintenance dredging, and work area dewatering. The majority of these activities are temporary in nature, lasting from a few days to several weeks, depending on the size of the facility in question. The risk of take associated with construction activity varies by impact mechanism and is dependent on the project-specific magnitude of that impact mechanism. As discussed below, some mechanisms may produce a high risk of individual take due to their intensity, while others may result in a low risk of take due to their limited magnitude and duration.

The risk of take resulting from these impact mechanisms also varies by subactivity type, based on the scale of the project and the type of environment where it is implemented. For example, a large jetty project at a river mouth may have a significant impact on the nearshore and estuarine environment, but the risk of take associated with the structure would be limited to those species and individual life-history stages that occur in those habitat types. Breakwaters are typically developed in marine and lacustrine habitats and would only occur in the largest of rivers, specifically the Columbia River, with sufficient open water to allow the formation of wind-driven waves, or supporting vessel traffic producing wakes large enough to cause bank erosion. The distribution of these project types limits the potential for related stressor exposure to the species and life-history stages that occur in these environments. In contrast, groins and bank barbs are commonly implemented in marine, riverine, and lacustrine environment types, and are often used in smaller streams and rivers, so the range of species and life-history stages exposed to stressors from these project types is much broader.

For example, bull trout would be exposed to jetty and breakwater related stressors during subadult and adult life-history stages, but the egg, alevin, and juvenile stages would not, as they occur in upriver environments that are not suitable for these subactivity types. Other HCP species, such as the western brook lamprey and the Columbia River spire snail, are limited in distribution to free-flowing rivers and streams inappropriate for jetties and breakwaters, so these species will never be exposed to related stressors. In contrast, groins or bank barbs are used across all environment types, including the smaller streams and free-flowing rivers where these species occur.

The species-specific risk of take ratings for construction and maintenance of jetties, breakwaters, and groins and bank barbs reflects this perspective. The rankings for these three subactivity types are presented at the end of this narrative in Tables 9-1, 9-2, and 9-3, respectively. The risk of take resulting from each associated submechanism of impact is described in further detail below.

9.1.1 Pile Driving (Underwater Noise)

Jetties or breakwaters may incorporate structural pilings as a component of design. Groins often incorporate wooden or steel pilings as design elements, particularly where increased structural stability is needed in higher velocity flowing water environments. Pile installation typically requires pile driving, an activity with a high potential for direct individual take caused by the sound pressure waves propagated through the water column. As detailed in Section 7.1.1 (*Marine Environments*), sound pressure from pile driving has the potential to cause injury and mortality of aquatic receptors, particularly fish. The potential for injury or mortality varies depending on piling size and composition, pile driving methods, and site-specific environmental characteristics such as bathymetry, intervening land masses, and substrate composition.

Until recently, NOAA Fisheries and USFWS recognized underwater noise levels of 150 dB_{RMS} and 180 dB_{peak} as thresholds for disturbance and injury, respectively, of federally listed salmonid species (Stadler 2007, Teachout 2007). While the disturbance threshold still stands, on April 30, 2007, NOAA Fisheries established the following dual criteria to evaluate the onset of physical injury to fishes exposed to underwater noise from impact hammer pile driving (NMFS 2007b) (exceeding either criterion equals injury):

> ***SEL:*** *A fish receiving an accumulated Sound Exposure Level (SEL) at or above 187 dB re: one micropascal squared-second during the driving of piles likely results in the onset of physical injury; a simple accumulation method shall be used to sum the energy produced during multiple hammer strikes.*
>
> ***Peak SPL:*** *A fish receiving a peak sound pressure level (SPL) at or above 208 dB re: one micropascal from a single hammer strike likely results in the onset of physical injury.*

While these new criteria accommodate a more comprehensive evaluation of the effects of sound exposure, it is difficult to compare the SEL threshold to established reference values, which are typically reported in dB_{RMS} or dB_{peak} units. In general, pile driving activities with the greatest potential to cause injury involve large diameter steel pilings placed with an impact hammer. Smaller diameter wooden pilings placed with a vibratory hammer present the lowest potential for injury and are likely to result in take only in the form of temporary disturbance and behavioral alteration.

Project scale and location are primary determinants of the piling material types and placement methods selected. For example, the engineering and longevity requirements for developing a marine jetty or breakwater may effectively demand use of large diameter steel or concrete pilings placed using an impact hammer. Wooden pilings may be suitable for groins or bank barbs placed in riverine or lacustrine environments, and substrate conditions may allow placement using a vibratory hammer. In contrast, some riverine or lacustrine environments may demand more robust steel pilings. Material selection, pile driving methods, and site characteristics are all important determinants governing the potential for take resulting from this impact mechanism. Applying a worst-case scenario perspective, pile driving must be associated with a high risk of take due to the potential for injury or mortality for the majority of HCP species experiencing possible exposure.

9.1.2 Equipment Operation and Materials Placement

Equipment operation and materials placement results in increased ambient noise levels in and around the project vicinity. Noise associated with vessels and materials placement is usually of insufficient magnitude to cause direct injury to fish. Although this would presumably also apply to invertebrates, insufficient data are available to definitively support this conclusion. As stated in Section 7.1.2 (*Lacustrine Environments*), increased ambient noise levels can result in auditory masking or can even cause short-term decreases in fish hearing sensitivity, potentially decreasing their ability to sense predators and prey. Increased ambient noise may also result in habitat avoidance, leading to increased stress and competition as displaced individuals seek out other suitable habitats. Increased predation exposure, increased competition, and decreased foraging success can lead to decreased survival, growth, and fitness. These effects constitute a moderate risk of take due to their short-term duration.

9.1.3 Channel/Work Area Dewatering

Channel or work area dewatering is most often required for groin and bank barb construction. Dewatering is not commonly used in jetty and breakwater construction due to the large scale of these structures and the environments where they are typically constructed. Thus, the risk of take resulting from this impact mechanism is most commonly associated with groins and bank barbs.

This construction and maintenance activity produces stressors that are likely to cause injury or mortality for a broad range of HCP species. Well-designed protocols and trained personnel are necessary to implement this activity properly, and high rates of mortality can result even when well-designed protocols and experienced field crews are used. For example, NOAA Fisheries evaluated take associated with dewatering and handling in a recent biological opinion. They estimated that salmonid mortality rates in the range of 8 to as high as 20 percent may occur even when trained personnel are used, and have assumed an injury rate of 25 percent (NMFS 2006).

Mortality rates may be even higher in areas with complex substrate and bathymetry. During the egg, larval, or juvenile life-history stage of many species, individuals may be too small or too well concealed to collect and relocate effectively (e.g., juvenile salmonids hiding in cobble interstices, river lamprey ammocoetes buried in fine substrate), meaning that mortality is likely for any individuals trapped within the exclusion area. Even in the absence of mortality, fish handling and relocation may result in stress and injury, as well as increased competition for forage and refuge in the relocation habitat. Even in the absence of these effects, the act of capturing and handling an ESA-listed species constitutes harassment, which is considered a form of take.

Given the potential for direct injury and mortality for essentially all species exposed to dewatering and fish handling, the permitting of channel and work area dewatering poses a high risk of take for all species potentially exposed to this activity.

9.1.4 Construction and Maintenance Dredging

Development of shoreline modification structures may involve dredging during construction and maintenance. For example, groins and bank barb structures often extend below the substrate surface, requiring dredging to excavate the foundation (although these activities are usually conducted within a dewatered exclusion area). Dredging activities are typically temporary to short-term in duration, lasting from days to weeks, with maintenance recurring at interannual to decadal frequencies. Stressors associated with dredging include direct disturbance and the potential for injury or mortality from physical entrainment. The potential for take associated with this stressor varies by species and life-history stage, ranging from a moderate risk of take (e.g., from limited exposure to disturbance and displacement) to a high risk (e.g., exposure to entrainment resulting in injury and/or mortality). Generally, many juvenile and most adult fish are sufficiently motile to avoid entrainment and injury. In combination with timing restrictions, this will limit exposure so that only a low to moderate risk of take will result from activity-related disturbance and temporary displacement. In contrast, fish eggs and demersal larvae and the HCP invertebrate species are effectively nonmotile and therefore are vulnerable to entrainment. These species would face a high risk of take.

In addition to the potential for direct physical impacts, dredging produces related stressors that create the potential for take. Specifically, dredging causes increased suspended solids (turbidity), altered substrate composition, and changes in bathymetry that alter habitat suitability and potentially alter wave energy, current, and circulation patterns. In specific cases, dredging may also result in the resuspension of contaminated sediments. These stressors are associated with a moderate to high risk of take, as described below in Section 9.2 (*Hydraulic and Geomorphic Modifications)* and Section 9.5 (*Water Quality Modifications).*

9.2 Hydraulic and Geomorphic Modifications

Shoreline modifications may create large structures perpendicular to the direction of water flow. Therefore, these structures inherently involve modification of the hydraulic and geomorphic conditions in the project vicinity, and the subsequent imposition of a number of impact mechanisms and related stressors on the aquatic environment. As with the other categories of impact mechanisms addressed in this white paper, the nature and scale of hydraulic and geomorphic modification, and the associated risk of take, vary by project type and scale. Jetties, by virtue of their size and location, typically have the most significant effects. In Washington State, these projects are typically implemented at river mouths and are intended to focus river flow to prevent the accumulation of bars. By their very nature, they have dramatic effects on hydraulic and geomorphic conditions in the nearshore marine and lacustrine environments where they are implemented.

Breakwaters have similar impacts, although they may manifest differently because of differences in orientation to the shoreline (i.e., breakwaters are typically oriented parallel to the shore, while jetties are oriented perpendicularly). In the absence of other shoreline structures, breakwaters are also less prone to interrupt alongshore drift, thereby having lesser effects on substrate conditions. As noted previously, jetties are generally limited to river mouths along the marine shoreline while breakwaters, groins, and bank barbs can occur in marine, lacustrine, and riverine habitats. Groins and bank barbs in marine and lacustrine environments will have effects similar to jetties. However, the associated impact mechanisms are more limited in scale because these structures are usually far less extensive and intrusive. The nature of these mechanisms and stressors varies slightly between riverine, marine, and lacustrine environments. Therefore, the risk of take in these different habitat types is discussed separately below, with the level of risk resulting from these structures tailored to the subactivity types likely to occur there.

The species-specific risk of take resulting from hydraulic and geomorphic modification caused by jetties, breakwaters, and groins and bank barbs is presented at the end of this narrative in Tables 9-1, 9-2, and 9-3, respectively.

9.2.1 Marine Environments

Jetties, breakwaters, and groins and bank barbs are all project types that could potentially be implemented in the marine environment. These projects unavoidably modify hydraulic and geomorphic conditions in the nearshore marine environment, resulting in the imposition of several impact mechanisms and related stressors. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore environment.

9.2.1.1 Altered Wave Energy, Altered Current Velocities, and Altered Nearshore Circulation Patterns

Wave energy, current velocities, and circulation patterns are all important determinants governing nearshore marine habitat characteristics. These factors determine habitat suitability for a number of species-specific life history processes. For example, wave energy conditions, currents, and circulation patterns have a strong influence on nearshore water temperatures and on the sorting and transport of sediments. Many fish species selectively spawn in locations where current and circulation patterns promote the settling of planktonic larvae in favorable environments for rearing. Alteration of these patterns can cause larvae to be transported to unfavorable environments. Similarly, juvenile fish rearing in nearshore environments selectively choose environments with suitable wave energy and current conditions. These impact mechanisms can fundamentally alter habitat suitability for these uses, leading to effectively permanent changes in the availability of suitable habitats, causing decreased survival, growth, and fitness. This equates to a high risk of take for species that are dependent on these habitats during some phase of their life history (see Tables 9-1, 9-2, and 9-3, presented following this narrative).

9.2.1.2 Altered Sediment Supply and Altered Substrate Composition

Sediment supply and substrate composition are fundamental components of the nearshore ecosystem structure. The physical alteration of the shoreline environment that accompanies shoreline modifications can lead to alterations in sediment supply and substrate conditions through fragmentation of the shoreline environment from sources of sediment recruitment, as well as the interruption or alteration of alongshore sediment transport. In conjunction with altered wave energy, this can lead to changes in substrate conditions that may be beneficial or detrimental to individual species. Because substrate composition is an important determinant of community structure in the nearshore environment, these habitat changes can fundamentally alter community structure and habitat suitability for species dependent on the original habitat condition. This equates to a high risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages (see Tables 9-1, 9-2, and 9-3, presented following this narrative).

9.2.1.3 Altered Freshwater Inputs

Freshwater inputs to the nearshore environment are demonstrably linked to a number of important habitat parameters such as temperatures in forage fish spawning substrates, eelgrass distribution, and habitat selection by certain fish species. Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability for species dependent on the nearshore environment. This equates to a moderate to high risk of take for species with demonstrable dependence on these habitats because freshwater inputs will likely still occur; however, they will be modified, resulting in a potential reduction in suitable habitat area, which in turn will lead to reduced survival, growth, and fitness (see Tables 9-1, 9-2, and 9-3, presented following this narrative).

9.2.2 Lacustrine Environments

Shoreline modification projects implemented in the lacustrine environment unavoidably modify hydraulic and geomorphic conditions, resulting in the imposition of several impact mechanisms and related stressors. As with the marine environment, all three subactivity types potentially occur in lacustrine environments, depending on the size of the lake and the types of human activities present. Risk of take associated with the resulting impact mechanisms is determined by the size and scale of the project in question, and species-specific dependence on the nearshore lacustrine environment where the stressors associated with these impact mechanisms are manifest. In general, jetties are implemented at the mouths of navigable rivers in larger lakes. There are very few lakes in Washington State suitable for these types of structures. In contrast, many lakes in the state may be suitable for breakwaters or groins and bank barbs. Therefore, the risk of take associated with jetties presented in Table 9-1 is tailored to the likelihood of species occurrence in larger lake and reservoir systems where jetties could potentially be implemented. Risk of take resulting from breakwaters and groin and bank barb development (see Tables 9-2 and 9-3, respectively) recognizes the broad potential application of these subactivity types.

9.2.2.1 Altered Wave Energy, Altered Current Velocities, and Altered Nearshore Circulation Patterns

Wave energy, current velocities, and circulation patterns are all important determinants governing nearshore lacustrine habitat characteristics. These processes strongly influence nearshore water temperatures and other water quality parameters, shoreline stability, and the accumulation of allochthonous and autochthonous materials. Alteration of these parameters can fundamentally alter the suitability of nearshore habitats for species dependent on these habitats, leading to decreased survival, growth, and fitness. This equates to a high risk of take for species that are dependent on these habitats during some phase of their life history (see Tables 9-2 and 9-3, presented following this narrative).

9.2.2.2 Altered Sediment Supply and Altered Substrate Composition

Sediment supply and substrate composition are also fundamental components of the nearshore ecosystem structure. Physical alteration of the shoreline environment that accompanies shoreline modification projects can lead to alterations in sediment supply and substrate conditions through fragmentation of the shoreline environment from sources of sediment recruitment, as well as the interruption or alteration of alongshore sediment transport. In conjunction with altered wave energy, this can lead to changes in substrate conditions that may be beneficial or detrimental to individual species. Because substrate composition is an important determinant of community structure in the lacustrine environment, these habitat changes can fundamentally alter community structure and habitat suitability for species dependent on the original habitat condition. This equates to a high risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages (see Tables 9-2 and 9-3, presented following this narrative).

9.2.2.3 Altered Groundwater–Surface Water Exchange

Hydraulic and geomorphic modifications can influence and alter groundwater and surface water exchange in the vicinity of the modifications. Groundwater and surface water exchange is an important component of ecosystem function (including water quality moderation) in lacustrine environments. Thus, this impact mechanism has the potential to affect survival, growth, fitness, and (in some cases) the spawning productivity of a range of species. Therefore, this mechanism is generally equated with a high risk of take for species exposed to this stressor, depending on species-specific life-history characteristics (Tables 9-2 and 9-3, presented following this narrative).

9.2.3 Riverine Environments

Shoreline modification projects implemented in riverine environments include breakwaters and groins and bank barbs. Jetties are typically placed at river mouths where they enter the ocean or large lacustrine environments. Therefore, for the purpose of this assessment, impact mechanisms resulting from jetty development are considered not to affect the riverine environment. Thus, there is effectively no risk of take from jetty-related impact mechanisms for HCP species occurring in riverine environments (see Table 9-1 presented following this narrative).

The effects of hydraulic and geomorphic modifications caused by breakwaters versus groins and bank barbs, and the related risk of take, are limited to specific segments of the river continuum where these activities are likely to take place. For example, breakwaters are most likely to be implemented in higher order rivers where wind-driven waves or boat wakes are sufficiently large to warrant these structures to protect marinas, boat launches, or other infrastructure. In contrast, groins and bank barbs are used in a broad array of river environments, from small mountain streams to large river mainstems. Therefore, the range of HCP species and life-history stages that could be exposed to breakwater-related stressors in riverine environments is limited, whereas effectively all riverine species and life-history stages could be exposed to stressors resulting from groins and bank barbs.

9.2.3.1 Altered Channel Geometry, Altered Flow Velocity, and Altered Substrate Composition

Channel geometry, flow conditions, and substrate composition are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects are in turn likely to lead to reduced spawning success, as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. This equates to a high risk of take for species with exposure to these impact mechanisms (see Tables 9-2 and 9-3, presented following this narrative).

9.2.3.2 Altered Groundwater–Surface Water Exchange

Hydraulic and geomorphic modifications can influence and alter groundwater and surface water exchange in the vicinity of the modification. This hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Therefore, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. This mechanism is generally equated with a high risk of take for species exposed to this stressor, depending on species-specific, life-history characteristics (see Tables 9-2 and 9-3, presented following this narrative). Species with a high risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels. For example, most salmonids preferentially spawn in areas with groundwater-induced upwelling, which promotes oxygenation of spawning gravels. Alteration of hyporheic exchange in environments suitable for spawning could potentially affect egg survival and reduce the availability of suitable spawning habitat, resulting in reduced spawning success. Similarly, groundwater inflow can provide important thermal refugia for migrating adult and rearing juvenile salmonids during periods with high water temperatures. A reduction in the amount of thermal refugia may negatively affect survival during these life-history stages. Similar effects would be expected for other coldwater fish species with low thermal tolerance thresholds, such as pygmy whitefish. More generally, hyporheic exchange also plays a key role in nutrient cycling and food web productivity in alluvial bed rivers. Projects resulting in significant alteration of hyporheic exchange could adversely affect food web productivity, limiting foraging opportunities for fish and invertebrate species dependent on these types of environments.

9.3 Riparian Vegetation Modifications

The development of shoreline modification projects in many cases involves the modification of riparian vegetation in the project area, as well as the subsequent imposition of a number of impact mechanisms and related stressors on the aquatic environment. However, because jetties and groins and bank barbs are most typically oriented perpendicular to the shoreline, the extent of riparian impacts during construction and the amount of habitat permanently modified will be relatively minor in comparison to activities such as bank protection. The potential for riparian vegetation modification caused by breakwaters is even more limited, as these structures have no onshore component intersecting the riparian environment. These structures are constructed primarily from barges or floating platforms accessed from established landings. Therefore, effectively no riparian vegetation modification is associated with this subactivity type, and no risk of take is expected (see Table 9-2 following this narrative).

As noted previously, jetties are generally restricted to marine and large lacustrine environments and, therefore, no riparian vegetation modification will occur in riverine environments as a result of these structures, and no risk of take is expected (see Table 9-1 following this narrative). In marine and lacustrine environments, the onshore component of jetties results in both short-term and effectively permanent modification of riparian vegetation. It is important to note, however, that because the onshore component of jetties is relatively small in comparison to the overall footprint of the structure, and the majority of these structures are away from shore and oriented perpendicular to the shoreline, the overall magnitude of riparian vegetation modification in most cases will be relatively limited. Therefore, while the full suite of impact mechanisms will occur and these impact mechanisms will produce stressors, the related risk of take is expected to be low for most species due to the limited area affected.

Groins and bank barbs can occur in all three environment types and, as such, have the potential to result in riparian vegetation modification. Like jetties, the riparian footprint of an individual structure is typically limited as groins and bank barbs are oriented perpendicular to the shoreline. However, because a groin or bank barb project often incorporates a series of several structures, the resulting short-term to intermediate-term construction impacts can be fairly extensive, affecting a larger riparian footprint. While the risk of take from this subactivity type resulting from riparian vegetation modification is similar to that described for jetties (i.e., low) (see Table 9-3 following this narrative), it is important to note that the number of individuals affected may potentially be larger in cases where the affected riparian footprint is more extensive.

9.3.1 Marine Environments

Shoreline modification projects in the marine environment have variable effects on marine riparian vegetation, depending on the subactivity type and nature of the project. As noted above, breakwaters are not expected to result in any riparian vegetation related stressors. In contrast, jetties and groins and bank barbs have the potential to result in riparian vegetation modification, but the extent of these modifications is limited to the onshore footprint of the structures and the construction area. Therefore, the overall risk of take from this impact mechanism is likely to be low given the limited onshore footprint of these structures.

This caveat must be taken into account when considering the effects of the stressors resulting from marine riparian vegetation modification. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore environment and, where supported by available science, the demonstrated dependence on riparian functions of the species in question. For many species, the risk of take associated with marine riparian impact mechanisms is unknown because scientific understanding of the related ecological processes is in its infancy, and the extent to which many marine or anadromous species rely on the nearshore environment during their life history is unclear.

9.3.1.1 Altered Riparian Shading and Altered Ambient Air Temperature Regime

The influence of riparian shading on water temperatures in the nearshore marine environment is limited in most circumstances. Moreover, the perpendicular orientation of jetties, groins, and bank barbs to the shoreline tends to limit their riparian footprint, further reducing the effects of riparian vegetation modifications on the nearshore environment. As such, the risk of take from riparian vegetation modifications associated with these structure types is considered to be low. Breakwaters have no onshore components and therefore have no effects on riparian conditions, and they will impose no related risk of take.

9.3.1.2 Altered Shoreline and Bluff Stability

Depending on site-specific conditions, modifications of marine riparian vegetation can lead to physical alteration of the shoreline and bluff instability. In the context of shoreline modification projects, this effect is expected to be small because the onshore footprint of these structures is limited. In addition, the structure itself will stabilize the shoreline where vegetation has been removed. The effects of this stressor on receptors can be similarly variable. In general, however, this stressor would be expected to alter shoreline habitat conditions and habitat suitability for species dependent on the nearshore environment during some portion of their life history. This equates to a low risk of take for species with demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness.

9.3.1.3 Altered Allochthonous Inputs

Allochthonous inputs to the nearshore environment from marine riparian vegetation include leaf litter and terrestrial insect-fall, as well as inputs of LWD. These inputs clearly contribute to aquatic food web productivity, but the science regarding the significance of these inputs in the marine environment is relatively limited. LWD recruitment is an important contributor to habitat structure. Because this stressor has the potential to alter food web productivity and habitat complexity, it is likely to affect the survival, growth, and fitness of species dependent on the nearshore environment for foraging and rearing during some portion of their life history. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.1.4 Altered Habitat Complexity

The physical structure of marine riparian vegetation, allochthonous inputs of LWD, shoreline stability, and effects on localized microhabitat conditions all contribute to habitat structure and complexity of the nearshore environment. Alteration of habitat complexity can have demonstrable effects on the productivity of aquatic species dependent on the nearshore environment, particularly fish species that spawn and rear in these areas, through effects on survival, growth, and fitness. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.1.5 Altered Freshwater Inputs

Freshwater inputs to the nearshore environment are demonstrably linked to a number of important habitat parameters such as temperature in forage fish spawning substrates, eelgrass distribution, and habitat selection by certain fish species. Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability for species dependent on the nearshore environment. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.2 Lacustrine Environments

Shoreline modification projects in lacustrine environments have variable effects on riparian vegetation, depending on the subactivity type and nature of the project. As noted above, breakwaters are not expected to result in any riparian vegetation related stressors (Table 9-2). In contrast, jetties and groins and bank barbs potentially result in riparian vegetation modification, but the extent of these modifications is limited to the onshore footprint of the structures and the construction area. Therefore, the overall risk of take from this impact mechanism for jetties and groins and bank barbs is likely to be relatively limited in terms of the overall number of individuals affected (see Tables 9-1 and 9-3, respectively).

This caveat must be taken into account when considering the effects of the stressors resulting from lacustrine riparian vegetation modification. The risk of take from these impact mechanisms is strongly linked to species-specific dependence on the nearshore lacustrine environment and, where supported by available science, the demonstrated dependence of the species in question on the affected riparian functions.

9.3.2.1 Altered Riparian Shading and Altered Ambient Air Temperature Regime

Riparian shading in lacustrine environments can have a pronounced effect on nearshore water temperatures. The effect of riparian modification on the ambient air temperature regime is less clear and depends on a range of site-specific environmental factors. In general, water temperatures in lacustrine environments are predominantly driven by solar radiation exposure, seasonal stratification, turnover rate, and the temperature of source water. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.2.2 Altered Shoreline Stability

Depending on site-specific conditions, modification of lacustrine riparian vegetation can lead to alteration of shoreline stability conditions. In general, shoreline modification projects are intended to increase shoreline stability. However, unmitigated vegetation alteration may lead to localized decreases in shoreline stability and cyclical erosion. Where this impact mechanism occurs, it would be expected to alter shoreline habitat conditions and habitat suitability for species dependent on the nearshore environment during some portion of their life history. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.2.3 Altered Allochthonous Inputs

Allochthonous inputs to the lacustrine environment from riparian vegetation include leaf litter, other organic debris, and terrestrial insect-fall, as well as inputs of LWD. These inputs clearly contribute to aquatic food web productivity, and LWD recruitment is an important contributor to habitat structure. Because this stressor has the potential to alter food web productivity and habitat complexity, it is likely to affect the survival, growth, and fitness of those species dependent on the nearshore environment for foraging and rearing during some portion of their life history. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.2.4 Altered Habitat Complexity

The physical structure of lacustrine riparian vegetation, allochthonous inputs of LWD, shoreline stability, and effects on localized microhabitat conditions all contribute to habitat structure and complexity of the nearshore environment. Alteration of habitat complexity can have demonstrable effects on the productivity of aquatic species dependent on the nearshore environment, particularly fish species that spawn and rear in these areas, through effects on their survival, growth, and fitness. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.2.5 Altered Groundwater-Surface Water Exchange

Groundwater inputs to the lacustrine nearshore environment provide beneficial microhabitat conditions for a range of species. For example, beach spawning sockeye salmon populations are dependent on groundwater-fed beaches for spawning habitat. Juvenile salmonids rearing in nearshore environments may also depend on thermal refugia provided by groundwater inflow. Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability for species dependent on the nearshore environment. However, because the footprint of jetties, groins, and bank barbs within the riparian area is limited, the extent of the related effects is likely to be low. Accordingly, this equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component and therefore no potential to impose this submechanism.

9.3.3 Riverine Environments

Shoreline modification projects in riverine environments are limited to breakwater and groin and bank-barb subactivity types. Jetties are limited to marine and larger lacustrine habitats. Therefore, there is effectively no risk of take resulting from this impact mechanism in the riverine environment (see Table 9-1). Breakwaters are primarily limited to the lower reaches of higher order river systems where they can protect infrastructure or habitat from wind-driven waves or vessel wakes. As noted, however, this subactivity type does not involve an onshore component; therefore, the riparian vegetation impacts are negligible and no risk of take is expected to result (Table 9-2).

Therefore, for riverine environments, the risk of take associated with the impact submechanisms and related stressors described below apply only to groins and bank barbs. These subactivity types can be implemented across a range of riverine environments, from smaller, low-order streams to larger rivers. Thus, a broader range of species and life-history stages are exposed to risk of take resulting from stressor exposure (Table 9-3).

9.3.3.1 Altered Riparian Shading and Altered Ambient Air Temperature Regime

Removal of riparian vegetation can demonstrably affect the temperature of streams and lower order river environments, producing a range of potential effects on fish and wildlife species. In higher order river environments, this effect is far less pronounced. Water temperatures in systems of this nature are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas of the watershed, hydromodification (e.g., dam and reservoir development), and other factors that affect water temperatures flowing through the affected area. In contrast, in smaller rivers and streams, removal of riparian vegetation for a shoreline modification project may have a measurable effect on stream temperature. No associated risk of take is anticipated for breakwaters and jetties because riverine environments are not appropriate for these types of structures.

On this basis, the risk of take associated with this impact mechanism is viewed as variable depending on the type of environment in which it is implemented. In larger river systems, the temperature effect may not be measurable and the resulting risk of take discountable. In smaller streams, stream temperature effects may influence local habitat suitability and by, extension affect the survival, growth, and fitness of exposed species and life-history stages.

9.3.3.2 Altered Stream Bank Stability

Removal of riparian vegetation can affect shoreline stability through the reduction in root cohesion and the loss of LWD inputs that affect localized erosion and scour conditions. Groins and bank barbs are typically intended to increase local bank stability. In the worst-case scenario, however, riparian vegetation modification associated with a permitted project could result in decreased stream bank and shoreline stability, as well as increased erosion and turbidity. These effects are localized and predominant during seasonal high-flow conditions. The risk of take associated with this stressor varies depending on species-specific sensitivity to increased turbidity. In general, more motile fish species experience only temporary behavioral alteration and low risk of take. In contrast, less motile fish life-history stages or sessile invertebrates could experience a high risk of take from decreased survival due to substrate sedimentation and smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

9.3.3.3 Altered Allochthonous Inputs

Riparian vegetation is an important source of nutrient input to the aquatic environment, strongly influencing the productivity of the aquatic food chain. Allochthonous nutrient inputs include sources such as insect-fall, leaf litter and other organic debris, and LWD inputs that contribute both organic material and habitat complexity. The importance of allochthonous inputs to riverine food web productivity decreases along a downstream gradient. However, as rivers grow in size, the contributions of allochthonous production and nutrient cycling to the food web increase.

Therefore, the magnitude of this impact mechanism varies depending on the scale of the groin or barb project and its position in the watershed. In lower order streams, allochthonous inputs are more important to food web productivity, while they provide a minor contribution in the lower reaches of large river systems. On this basis, the loss of allochthonous production from a groin project near the mouth of a large river will produce related stressors of potentially far lower magnitude than a series of bank barbs in a small, higher elevation stream. In such cases, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a high risk of take for a range of HCP species that are dependent on riverine rearing conditions due to the long-term nature of the effect.

9.3.3.4 Altered Habitat Complexity

The influence of riparian vegetation on riverine habitat complexity is broadly recognized. Modification of riparian vegetation alters habitat complexity in a number of ways, primarily through the loss of undercut banks, root structure, and LWD inputs to the channel. The hydraulic and geomorphic effects of riparian vegetation modification can lead to further alterations in habitat complexity. This impact mechanism presents a potential risk of take for a broad range of species dependent on riverine aquatic ecosystems through a variety of species-specific stressors. Depending on the particular life history of the affected species, alteration in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected through this mechanism of impact by groin and bank barb development are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. This equates to a high risk of take, which applies broadly across all species exposed to the stressor.

9.3.3.5 Altered Groundwater–Surface Water Exchange

The influence of riparian vegetation on hyporheic exchange is well documented as an important component of ecosystem health. Alteration of riparian vegetation can in turn lead to alteration of surface water and groundwater exchange, with important effects on the riverine ecosystem. The potential risk of take caused by this mechanism is significant for many species and life-history stages potentially exposed to this stressor. For example, many salmonid species preferentially select spawning locations with active groundwater inflow to maximize incubation success. For rearing salmonids and other temperature-sensitive species, groundwater inflow may provide thermal refugia important for survival during summer rearing periods. Hyporheic connectivity is also an important component of food web productivity. Thus, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Therefore, this mechanism is generally equated with a high risk of take for species exposed to this stressor, depending on species-specific life-history characteristics.

9.4 Aquatic Vegetation Modifications

Shoreline modification projects can result in aquatic vegetation modification through the alteration or elimination of vegetation in the construction footprint, as well as the subsequent effects of the structure on hydraulic and geomorphic conditions. During construction, vegetation in the structural footprint of the project can be eradicated or buried by the placement of fill or structural material. After construction, changes in wave energy, circulation patterns, flow and/or current velocities, and substrate composition can also alter the vegetation community.

Alteration of aquatic vegetation imposes impact mechanisms on the nearshore environment in the form of changes in autochthonous production and altered habitat complexity. The nature of these mechanisms and related stressors varies slightly between riverine and lacustrine habitats versus marine habitats, primarily because the role of aquatic vegetation differs between these systems. Therefore, the risk of take in these different environments is discussed separately. Species-specific risk of take ratings for the impact mechanisms resulting from aquatic vegetation modification are presented by subactivity and environment in Tables 9-1, 9-2, and 9-3.

As with the other categories of impact mechanisms addressed in this white paper, the nature and scale of aquatic vegetation modification are dependent on the size and design of the individual project in combination with site-specific conditions.

9.4.1 Marine Littoral Environments

Submerged aquatic vegetation (including eelgrass, kelp, and other forms of marine algae) is an important component of the marine littoral ecosystem relied upon by many species during critical life-history stages.

9.4.1.1 Altered Autochthonous Production

Autochthonous production by submerged aquatic vegetation is a source of primary and secondary production in the aquatic food web of the marine littoral zone. A diversity of species feed directly on live and fragmented submerged aquatic vegetation, forming the basis of the food web for a number of other species. Alteration of marine littoral vegetation caused by shoreline development projects may in some cases lead to localized shifts in food web productivity, possibly affecting foraging opportunities for dependent species and life-history stages. This equates to a high risk of take resulting from decreased growth and fitness.

9.4.1.2 Altered Habitat Complexity

The contribution of submerged aquatic vegetation to habitat structure in nearshore marine environments is well recognized. Numerous species use these habitats for cover and rearing during larval and juvenile life-history stages. Submerged aquatic vegetation also provides spawning habitat for Pacific herring. Alterations of the submerged aquatic vegetation community through reduction in aerial extent or conversion to other habitat types (e.g., conversion of eelgrass habitat to algae and kelp) can reduce the productivity of these habitats for dependent life-history stages. This equates to a high risk of take for species dependent on these habitats through reduced survival, spawning success, or growth and fitness.

9.4.2 Lacustrine and Riverine and Environments

Aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, the majority of aquatic vegetation species in rivers and lakes are invasive species. Thus, the risk of take resulting from this impact mechanism is relatively minor in comparison to that occurring in the marine environment. In riverine systems, protected slow-water areas created by groins and bank barbs may increase suitable habitat for emergent vegetation.

9.4.2.1 Altered Autochthonous Production

Modification of the submerged aquatic vegetation community in lakes and rivers can lead to decreased primary and secondary productivity, which in turn may affect overall food web productivity in the nearshore environment. In systems where the aquatic vegetation community is an important component of food web productivity, this can lead to a high risk of take through indirect effects on foraging success, growth, and fitness of species and life-history stages that depend on forage in the nearshore environment.

9.4.2.2 Altered Habitat Complexity

Submerged aquatic vegetation provides habitat structure in nearshore environments, creating vertical dimension and overhead cover. Alteration of habitat complexity can decrease the availability of suitable rearing habitat for species and life-history stages dependent on the nearshore environment, leading to increased predation risk and increased competition for suitable space, leading to effects on survival, growth, and fitness. This equates to a high risk of take for species dependent on aquatic vegetation functions in these environments.

9.5 Water Quality Modifications

Water quality modification is a broad category covering a number of impact mechanisms and related stressors. These impact mechanisms can be produced by a variety of activities associated with the development of shoreline modification structures. The severity of individual stressors will vary depending on the nature of the effect; its magnitude, duration, and frequency; and the sensitivity of the species and life-history stage exposed.

It is not practicable to clearly delineate a difference between shoreline modification subactivity types in terms of the magnitude of water quality related impact mechanisms they produce. The size of the structure, its construction and maintenance requirements, and the level of associated development and activity will determine stressor intensity. Again, to assess the risk of take associated with these facilities, a "worst-case scenario" approach is taken in this white paper, with consideration of the scale of the structure and related water quality effect that a given species is likely exposed to in each environment. Species-specific risk of take ratings by impact mechanism are shown for jetties, breakwaters, and groins and bank barbs in Tables 9-1, 9-2, and 9-3 (presented at the end of this narrative).

9.5.1 Altered Temperature

Shoreline modifications have the potential to alter temperature conditions through the hydraulic and geomorphic mechanisms they impose. Shoreline modification structures can alter waves, currents, and circulation patterns in marine and lacustrine environments, leading to increased stratification. In riverine environments, groins and bank barbs can slow water flows in the lee of the structures, creating slow water areas prone to stratification and elevated temperature conditions.

These effects can be magnified when stratified areas experience decreased shading due to modification of shading riparian vegetation, which may occur in association with jetties, groins, and bank barbs. However, because these structures are typically oriented perpendicular to the shoreline and their onshore footprint is small, the extent of vegetation modification is usually limited, and these effects are small.

Modification of temperature conditions can change the suitability of nearshore habitats. This may in turn affect the survival, growth, and fitness of HCP species that use the affected habitats. Because these effects are essentially permanent, they must be associated with a high risk of take.

9.5.2 Suspended Solids and Turbidity

Increased suspended solids can result from several different impact mechanisms. The severity of this stressor varies depending on its magnitude, duration, and frequency, as well as the sensitivity of the species and life-history stage exposed. In general, motile species and life-history stages exposed to temporary sediment impacts at low occurrence frequency experience only temporary disturbance, behavioral alteration, and low risk of take. In contrast, sessile invertebrates or relatively immobile life-history stages exposed to the same stressor may experience decreased survival and reduced foraging opportunities, leading to a moderate to high risk of take. For example, increased fine sediment levels in spawning gravels demonstrably affect the survival of salmonid eggs. In the marine environment, larval rockfish, cod, pollock, and lingcod reside in microlayer habitat. Increased suspended solids in microlayer habitat can cause direct mortality of exposed larvae, which are relatively immobile and thereby incapable of escaping acute water quality degradation. Sublethal levels of suspended sediments may affect the foraging success of planktonic herring larvae, leading to decreased foraging success and decreased survival, growth, and fitness. More frequent or longer duration sediment impacts have more pronounced effects on even motile species. Such exposure can cause behavioral alteration for longer periods, potentially increasing stress, exertion, and predation exposure; decreasing foraging opportunities; or even causing injury in extreme events.

9.5.3 Dissolved Oxygen

There are limited pathways through which shoreline modification projects can lead to alterations in surface water dissolved oxygen levels that are not implicitly addressed by other impact mechanisms. A primary area of concern related to the effects of shoreline modifications in marine and lacustrine environments is their potential to alter wave energy, current, and circulation patterns sufficiently to change stratification, isolating biochemical oxygen demand (BOD) and contributing to eutrophication. In extreme circumstances, this could lead to eutrophication-driven DO depletion in affected habitats. This effect equates to a high risk of take from changes in DO conditions in these environment types, due to the effectively permanent nature of the change in habitat conditions that shoreline modifications impose. These effects would not be anticipated in riverine environments due to the continuous, unidirectional flow path imposed by riverine environments.

Other potential causes of altered DO conditions include inputs of nutrient-rich discharge from construction vessel sanitary systems or ballast water that could cause temporary or short-term decreases in dissolved oxygen levels through discrete eutrophication effects. A large decrease in aquatic vegetation may limit photosynthetic production of oxygen, but the likelihood of this effect substantially decreasing dissolved oxygen levels is quite limited. In general, the likelihood of this stressor occurring as a direct or indirect result of a shoreline modification project is low. Fish species that are highly motile are generally able to avoid adverse effects through behavioral avoidance, equating to a low risk of take. In contrast, sessile invertebrates and less motile life-history stages could experience direct mortality as a result of exposure, equating to a moderate or even high risk of take depending on species-specific life history. However, because of the low likelihood of occurrence, the overall risk of take associated with this stressor is considered low for all species.

9.5.4 Nutrient and Pollutant Loading

Shoreline modification projects present multiple pathways for the introduction of a range of toxic substances to the aquatic environment, primarily through construction activities and, in some cases, the use of treated wood materials in the structure. Shoreline modification projects may also indirectly encourage pollutant and nutrient loading by supporting the development of additional infrastructure. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of immobile rockfish larvae in the demersal microlayer). More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased survival, growth, and fitness. This presents a moderate risk of take to species potentially exposed to this stressor.

9.5.5 Altered pH Levels

There are limited pathways through which shoreline modification projects can lead to alterations in surface water pH. A primary pathway is the in-water curing of concrete and discharge of concrete leachate to surface waters. Concrete, cement, mortars, grouts, and other portland cement or lime-containing construction materials are alkaline in nature and are capable of measurably raising the pH of surface waters. Operational discharges and accidental spills of acidic or caustic materials may also lead to the alteration of normal pH levels.

Increases or decreases in pH that exceed the normal tolerance thresholds of fish and invertebrates are highly toxic and can rapidly cause acute mortality. Risk of take from this impact mechanism is generally higher in fresh water than in marine waters. Salt water has an inherent buffering capacity that limits the changes in alkalinity. In general, this stressor is limited to low-frequency events that are temporary to short-term in duration. Fish species that are highly motile are generally able to avoid adverse effects through behavioral avoidance, equating to a low risk of take. In contrast, sessile invertebrates and less motile life-history stages could experience direct mortality as a result of exposure, equating to a high risk of take depending on species-specific life history.

9.5.6 Treated Wood Pollution

Creosote-treated wood was often used historically in shoreline modification projects and other structures in marine and freshwater environments. This substance is still permitted in some circumstances. Creosote is a wood preservative with a complex formula composed of more than 150 toxic chemical substances. The Hydraulic Code prohibits use of creosote- and pentachlorophenol-treated wood in lakes; therefore, exposure to this stressor exposure will not occur in most lacustrine habitats. There is some uncertainty about potential exposure in lacustrine environments because the applicability of this statute to reservoirs is not clear.

Prohibitions on the use of creosote, pentachlorophenol, and other wood preservatives have prompted the development of alternatives. ACZA and CCA type C are alternative wood preservatives that are less toxic than prohibited materials but are still effective against undesirable invertebrates. These substances, which slowly leach out of treated wood over time, are toxic to other forms of aquatic life than the intended target species and also have the potential to bioaccumulate.

These substances are expected to produce similar risk of take as described above under Nutrient and Pollutant Loading (i.e., a moderate risk of take for species potentially exposed to this stressor). It is worthwhile to note, however, that this submechanism poses greater potential for chronic exposure as leaching of toxics occurs over extended periods.

9.6 Ecosystem Fragmentation

Ecosystem fragmentation is an impact mechanism that incorporates the collective effects of the loss of habitat within the footprint of the structure, the physical barrier the structure presents in terms of the migration and dispersal of organisms, the transport of LWD and other organic material, and the impact mechanisms imposed by hydraulic and geomorphic modification. This impact mechanism operates differently in riverine, marine, and lacustrine environments due to the differences in characteristic physical and biological processes.

9.6.1 Marine Environments

Jetties, breakwaters, and groins and bank barbs are all capable of causing ecosystem fragmentation in the marine environment. The magnitude of this impact sub-mechanism and the related risk of take are driven by the scale of the project in question, with larger projects having the most potential for adverse effects.

9.6.1.1 Habitat Loss and Fragmentation

Jetties and groins and bank barbs (depending on their scale and location) present significant potential for habitat loss and fragmentation in the marine environment. By design, jetties are intended to accelerate the flow of water from estuaries into open ocean waters, thereby keeping shallow bar areas from forming. As a consequence, they can alter bathymetric, salinity, tidal exchange, and circulation patterns within the estuarine and nearshore environment, altering habitat conditions and potentially eliminating certain desirable habitat types. Habitats in the physical footprint of the structure are permanently lost as a result of construction. Due to their perpendicular orientation to the shore, jetties and groins and bank barbs present a physical barrier to the migration of many species. For example, many salmonid species typically migrate as juveniles in shallow water along the shoreline. These structures effectively force these individuals to migrate around the structure into deeper water where predation risk and foraging opportunities are less favorable to survival. Because jetties are typically larger in size, these effects are more pronounced. Because breakwaters are constructed offshore, typically parallel to the shoreline, they present less of a barrier to migration overall.

As discussed in Section 9.2 (*Hydraulic and Geomorphic Modifications*), all three structure types can alter wave energy, current, and circulation patterns in the nearshore and offshore environment. These effects can in many cases result in habitat fragmentation through various pathways. Alteration of these habitat characteristics may render productive habitats less suitable for a given species or, in the case of organisms with a planktonic life-history stage, may hinder the dispersal and retention of eggs and larvae in areas suitable for rearing. Collectively, this can result in take through long-term effects on survival, growth, and fitness of affected populations, which equates to a high risk of take for exposed species.

9.6.1.2 Loss of LWD Recruitment

Placement of shoreline modification structures in marine environments can alter the transport of drift wood to beach environments, limiting the recruitment of this important habitat element. Many large jetties and breakwaters are intentionally cleared of driftwood accumulations for maintenance purposes, which may further limit the potential for recruitment to nearby beach areas. Groins and bank barbs may similarly alter the transport of woody material along the shoreline. Shorelines with limited LWD recruitment potential due to natural conditions or existing riparian vegetation modifications may become increasingly starved of LWD as a result. The risk of take associated with this submechanism is effectively the same as the risk of take from the loss of LWD recruitment directly from the riparian zone (versus downstream transport), as discussed above in Section 9.3 (*Riparian Vegetation Modifications*) under *Altered Habitat Complexity*.

9.6.2 Lacustrine Environments

Jetties, breakwaters, and groins and bank barbs are all capable of causing ecosystem fragmentation in lacustrine environments. The magnitude of this submechanism and the related risk of take are driven by the scale of the project in question, with larger projects having the most potential for adverse effects.

9.6.2.1 Loss of Nearshore Habitat

Jetties and (depending on their scale and location) groins and bank barbs present significant potential for habitat loss and fragmentation in lacustrine environments. By design, jetties are intended to accelerate the flow of water from river mouths into open water, thereby keeping shallow bar areas from forming. As a consequence, they can alter the bathymetric profile and circulation patterns within the lake environment, altering habitat conditions and potentially eliminating certain desirable habitat types. Habitats in the physical footprint of the structure are permanently lost as a result of construction. Due to their perpendicular orientation to the shore, jetties and groins and bank barbs present a physical barrier to the migration of many species. For example, many salmonid species typically migrate as juveniles in shallow water along the shoreline. These structures effectively force these individuals to migrate around the structure into deeper water where predation risk and foraging opportunities are less favorable to survival. Because jetties are typically larger in size, these effects are more pronounced. Because breakwaters are constructed offshore, typically parallel to the shoreline, they present less of barrier to migration overall.

As discussed in Section 9.2 (*Hydraulic and Geomorphic Modifications*), all three structure types can alter wave energy, current, and circulation patterns in the nearshore and offshore environment. These effects can in many cases result in habitat fragmentation through various pathways. Alteration of these habitat characteristics may render productive habitats less suitable for a given species. Collectively, this can result in long-term effects on survival, growth, and fitness of affected populations, which equates to a high risk of take for exposed species.

9.6.2.2 Loss of LWD Recruitment

Placement of shoreline modification structures in lacustrine environments can alter the transport of drift wood to beach environments, limiting the recruitment of this important habitat element. Many large jetties and breakwaters are intentionally cleared of drift wood accumulations for maintenance purposes, which may further limit the potential for recruitment to nearby beach areas. The risk of take associated with this submechanism is effectively the same as the risk of take from the loss of LWD recruitment directly from the riparian zone (versus downstream transport), as discussed above in Section 9.3 (*Riparian Vegetation Modifications)* under *Altered Habitat Complexity*.

9.6.3 Riverine Environments

Breakwaters and groins and bank barbs are all capable of causing ecosystem fragmentation in riverine environments. Jetties can have similar effects, although for the purpose of this white paper these effects are addressed relative to marine and lacustrine habitats where they predominantly manifest.

9.6.3.1 Habitat Loss and Fragmentation

Groins and bank barbs implemented in riverine environments, particularly those with higher velocity flows, often cause localized changes in river geomorphology. In addition to the loss of habitat area within the structural footprint, these structures can concentrate and accelerate river flows, causing localized channel downcutting that can lead to a lowering of mean water surface and the consequent fragmentation of side channels and other floodplain habitats. This hydraulic and geomorphic effect is most prevalent in higher gradient reaches with sufficient velocity to transport bedload, and less prevalent in the lower gradient depositional reaches of large river mainstems. Therefore, this effect is not as likely to occur as a result of breakwater development. Many HCP species depend on floodplain habitats during one or more life-history stages, or depend on host species with these requirements. Loss of access to these habitat types represents take.

9.6.3.2 Loss of LWD Recruitment

Placement of shoreline modification structures in riverine environments, particularly groins and bank barbs, can alter the transport of LWD, limiting recruitment of this important habitat component to downstream environments. The magnitude of this effect is expected to be less pronounced with breakwaters due to their orientation parallel to flow in riverine environments, as well as their typical location in higher order mainstem reaches. The risk of take associated with this submechanism is effectively the same as the risk of take from the loss of LWD recruitment directly from the riparian zone (versus downstream transport), as discussed above in Section 9.3 (*Riparian Vegetation Modifications*) under *Altered Habitat Complexity*.

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Table 9-1. Species- and habitat-specific risk of take for mechanisms of impacts associated with jetties.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in marine and lacustrine habitats suitable for jetty development and are thereby potentially exposed to stressors resulting from related impact mechanisms. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Coho salmon	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species has a complex and variable life history depending on race. In general, coho salmon occur in lacustrine and nearshore marine habitats suitable for jetty development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Chum salmon	N	H	I	N	H	I	N	L	I	N	H	I	N	H	I	N	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for jetty development. Therefore, likelihood of stressor exposure in lacustrine environments is considered discountable. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from jetty development in these environments. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Pink salmon	N	H	I	N	H	I	N	L	I	N	H	I	N	H	I	N	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, likelihood of stressor exposure in lacustrine environments is considered discountable. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for jetty development. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Sockeye salmon	N	H	H	N	H	H	N	L	L	N	H	H	N	H	H	N	H	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for jetty development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Steelhead	N	H	H	N	?	H	N	?	L	N	?	L	N	?	H	N	?	H	Spawning activity typically occurs in habitats that are not suitable for jetty development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the level of take associated with activities in these habitat types is less certain but is conservatively presumed to occur. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Coastal cutthroat trout	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species is prevalent in estuaries and large rivers (although it also occurs in Lake Washington) and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for jetty development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migration between marine or lacustrine and riverine habitats and adult foraging in the marine and estuarine environment. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Westslope cutthroat trout	N	N	H	N	N	H	N	NA	L	N	NA	H	N	NA	H	N	NA	H	These species occur primarily in coldwater streams and small to medium-sized rivers unsuitable for jetty development. Rearing juveniles and adults do occur in lacustrine environments, creating some potential for stressor exposure. As a consequence, there is effectively no risk of take in riverine environment types, while exposure in lacustrine environments may result in a moderate (from project effects on habitat quality and quantity) to high (from project construction) risk of take.
Redband trout	N	N	H	N	N	H	N	NA	L	N	NA	H	N	NA	H	N	NA	H	

Table 9-1 (continued). **Species- and habitat-specific risk of take for mechanisms of impacts associated with jetties.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Bull trout	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment. Most effects would occur from development in nearshore marine migratory corridors, as well as lacustrine and marine foraging habitats used by mature juveniles and adults. However, bull trout in lakes are typically (but not exclusively) found in deeper water, limiting the potential for direct stressor exposure.
Dolly Varden	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	
Pygmy whitefish	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Stressor exposure will only occur in lacustrine environments. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in the marine environment or lakes suitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Mountain sucker	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species is commonly found in large lakes potentially suitable for jetty development. Stressor exposure is likely to occur in these environments during the juvenile and adult life-history stages. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties that are unsuitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Leopard dace	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in reservoir habitats potentially suitable for jetty development at sensitive life-history stages, including egg incubation. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Umatilla dace	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for jetty development at sensitive life-history stages, including egg incubation. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for jetty development. Therefore, this subactivity type will have no effect on this species and there is effectively no risk of take.
River lamprey	N	H	H	N	H	H	N	?	?	N	?	?	N	H	H	N	H	H	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. The ammocoete life-history stage is potentially exposed to a range of impact mechanisms resulting from jetty development in lacustrine environments. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Impact mechanism effects affecting the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	N	I	H	N	I	H	N	I	?	N	I	?	N	I	H	N	I	H	Pacific lamprey are anadromous with migratory corridors that cross estuaries and mainstems of larger river systems suitable for jetty development. Ammocoetes burrow into riverine sediments to rear for extended periods. The ammocoete life-history stage is more susceptible to a range of impact mechanisms resulting from jetty development in lacustrine environments. In the marine environment Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors. Therefore, the moderate to high risk of take associated with structure-related habitat alteration and construction activities, respectively, applies primarily to lacustrine habitat. Impact mechanisms in marine and lacustrine environments that affect the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This in turn equates to a moderate risk of take.

Table 9-1 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with jetties.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered to be anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore potentially exposed to jetty-related impact mechanisms in lacustrine environments. Their relative lack of mobility increases sensitivity to a range of impact mechanisms. Sturgeon are wide ranging in marine waters. Green sturgeon occur in Washington State only as adults in marine waters, with fisheries occurring in the Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	N	?	H	N	?	H	N	?	L	N	?	?	N	?	H	N	?	H	
Longfin smelt	N	H	H	N	H	H	N	L	?	N	I	?	N	H	H	N	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems. Planktonic larvae and juveniles of these species may also be vulnerable to jetty-related stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. Similar to other species' exposure profiles, life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. The Lake Washington population of longfin smelt rears and forages in the lacustrine environment throughout the larval, juvenile, and nonspawning adult portion of its life history and is subject to the effects of jetties in this water body.
Eulachon	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic and geomorphic, and aquatic vegetation modifications is high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to stressors resulting from jetty related impact mechanisms. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	

Table 9-1 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with jetties.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Brown rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile once they have settled, juveniles are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life history, it is vulnerable to both short-term construction and water quality related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	I	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from jetty development, including construction and water quality effects. Being planktonic spawners, the species' spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	I	N	N	M	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species. Therefore, the potential for exposure to most stressors from jetty-related impact mechanisms is minimal. Exceptions include alteration of riparian vegetation affecting this vegetation community. Risk of take for this species is similarly limited, with the exception of a moderate risk of take resulting from potential effects on marine littoral vegetation, and low risk of take associated with behavioral avoidance of water quality degradation. It is important to note, however, that suitable habitats for these species do not typically occur in locations suitable for jetty development; therefore, the likelihood of stressor exposure in general is considered to be limited.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for jetty development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are likewise not suitable for jetty development. As such, there is effectively no risk of take resulting from jetty-related stressor exposure.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers and the mainstems of these systems in flowing water environments unsuitable for jetty development. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes, the latter being suitable for jetties. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect these species; however, indirect effects could occur through direct effects on host fish.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-2. Species- and habitat-specific risk of take for mechanisms of impacts associated with breakwaters.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species has a complex and variable life history, depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for breakwater development and are thereby potentially exposed to stressors resulting from related impact mechanisms. Spawning activity typically occurs in habitats that are not suitable for breakwaters. Therefore, stressor exposure will only occur during migratory life-history stages in the lower reaches of large rivers, and lacustrine and marine environments. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Coho salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species has a complex and variable life history, depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for breakwater development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for breakwaters. Therefore, stressor exposure will only occur during migratory life-history stages in the lower reaches of large rivers, and lacustrine and marine environments. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Chum salmon	H	H	I	H	H	I	L	H	I	H	H	I	L	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for breakwater development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River). As such, in addition to migratory juveniles and adults, spawning habitats may be exposed to stressors resulting from breakwater-related impact mechanisms. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from breakwater development in these environments.
Pink salmon	H	H	I	H	H	I	L	H	I	H	H	I	L	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in this environment type. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river environments potentially suitable for breakwater development. As such, this species may potentially be exposed to stressors resulting from related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Sockeye salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for breakwater development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridors. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Steelhead	H	H	H	H	?	H	L	?	H	H	H	H	L	?	H	Spawning activity typically occurs in habitats that are not suitable for breakwater development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the level of take associated with activities in these habitat types is less certain, but is conservatively presumed to occur. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coastal cutthroat trout	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species is prevalent in estuaries and large rivers (although it also occurs in Lake Washington) and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for breakwater development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for breakwaters; therefore, stressor exposure will only occur during juvenile rearing adult foraging. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Westslope cutthroat trout	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in estuaries of larger rivers suitable for breakwater development is highly unlikely; therefore, the risk of take associated with these structures is considered discountable. Stressor exposure in lacustrine environments is possible, however. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Redband trout	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	

Table 9-2 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with breakwaters.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Bull trout	H	H	H	L	H	H	L	H	H	H	H	H	L	H	H	Spawning by these species occurs in habitats that are generally unsuitable for breakwater development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. However, bull trout in lakes are typically (but not exclusively) found in deeper water, limiting the potential for direct stressor exposure. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Dolly Varden	H	H	H	L	H	H	L	H	H	H	H	H	L	H	H	
Pygmy whitefish	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for breakwater development; therefore, stressor exposure will only occur in lacustrine environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for breakwater development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for breakwater development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Mountain sucker	H	N	H	H	N	H	H	N	H	H	N	H	L	N	H	This species is commonly found in large rivers and lakes suitable for breakwater development. Stressor exposure is likely to occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties that are unsuitable for breakwater development. Therefore, stressor exposure will not occur, and there is effectively no risk of take.
Leopard dace	H	N	H	H	N	H	H	N	H	H	N	H	L	N	L	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for breakwater development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Umatilla dace	H	N	H	H	N	H	H	N	H	H	N	H	L	N	L	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for breakwater development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for breakwater development. Therefore, this subactivity type will have no-effect on this species and there is effectively no risk of take.
River lamprey	H	H	H	H	H	H	?	?	?	H	H	H	L	H	L	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. The ammocoete life-history stage is more susceptible to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults, which in turn equates to a moderate risk of take.
Pacific lamprey	H	I	H	H	I	H	?	I	?	H	I	H	L	I	L	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for breakwater development. Ammocoetes burrow into riverine sediments to rear for extended periods. The ammocoete life-history stage is more susceptible to acute transient water quality impacts, such as reduced dissolved oxygen or altered pH. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This in turn equates to a moderate risk of take.

Table 9-2 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with breakwaters.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore potentially exposed to water quality related impact mechanisms from breakwaters. Their relative lack of mobility increases sensitivity to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Sturgeon are wide ranging in marine waters. Green sturgeon occur in Washington State only as adults in marine waters, with fisheries occurring in Willapa Bay and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats and the related risk of take.
White sturgeon	H	?	H	H	?	H	H	?	H	H	?	H	H	?	H	
Longfin smelt	H	H	H	H	H	H	I	I	H	H	H	H	H	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems potentially suitable for breakwater development. Demersal adhesive eggs are vulnerable to acute transient water quality impacts, such as reduced dissolved oxygen or altered pH. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. The Lake Washington population of longfin smelt rears and forages in the lacustrine environment throughout the larval, juvenile, and nonspawning adult portion of its life history and is subject to the effects of breakwaters in this water body.
Eulachon	H	H	N	H	H	N	I	I	N	H	H	N	H	H	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic/geomorphic and aquatic vegetation modifications are high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from breakwaters. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and, therefore, have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	

Table 9-2 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with breakwaters.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile once they have settled, juveniles are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life-history, it is vulnerable to both short-term construction and water quality related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from breakwater development, including construction and water quality effects. Being planktonic spawners, spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	N	N	N	H	N	N	N	N	N	L	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species. Therefore, the potential for exposure to most stressors from breakwater-related impact mechanisms is minimal, as these offshore structures have limited effects on littoral vegetation. This is particularly true for *Salicornia* marshes, which predominantly occur in low-energy environments less subject to the effects of breakwaters on wave energy. The only potential risk of take associated with breakwater development is from temporary water quality effects. This risk is rated as low.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The great Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for breakwater development. As such, there is essentially no likelihood of stressor exposure and, therefore, no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are generally not suitable for breakwater development. Therefore, there is no risk of take associated with this subactivity type for either species.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	H	N	H	H	N	H	I	N	I	H	N	H	L	N	L	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. Only the latter species occurs in habitats suitable for breakwater development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect this species; however, indirect effects could occur through direct effects on host-fish.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-3. Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species has a complex and variable life history, depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for groin or bank barb development and are thereby potentially exposed to stressors resulting from related impact mechanisms across all life-history stages. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coho salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species has a complex and variable life history, depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for groin or bank barb development and may experience exposure to related stressors across all life-history stages. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Chum salmon	H	H	I	H	H	I	L	L	I	L	H	I	H	H	I	H	H	I	Chum salmon in Washington State do not use lacustrine habitats in any significant fashion. Therefore, the likelihood of stressor exposure in lacustrine environments is considered discountable. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River). As such, in addition to migratory juveniles and adults, spawning habitats may be exposed to stressors resulting from groin or bank barb related impact mechanisms. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from groin or bank barb development in these environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pink salmon	H	H	I	H	H	I	L	L	I	L	H	I	H	H	I	H	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in this environment type and the likelihood of stressor exposure in lacustrine environments is considered discountable. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through and spawns in the mainstems and estuaries of larger river systems potentially suitable for groin or bank barb development. As such, this species may potentially be exposed to stressors resulting from related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Sockeye salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species is highly dependent on lacustrine environments for juvenile rearing, and most spawning behavior occurs in smaller rivers and streams that are also suitable for groin or bank barb development. Lake spawning populations also face risk of stressor exposure at sensitive egg and alevin life-history stages in lacustrine environments. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridors. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Steelhead	H	?	H	H	?	H	L	?	L	L	?	H	H	?	H	H	?	H	Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats; the level of take associated with activities in these habitat types is less certain, but is conservatively presumed to occur. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coastal cutthroat trout	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for groin or bank barb development. Migratory behavior and residence timing are variable. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Westslope cutthroat trout	N	N	H	H	N	H	N	N	L	N	N	H	N	N	H	N	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Bank barb development in smaller streams may affect spawning adults, eggs, alevins, and rearing juveniles. Groin development in moderate-sized rivers may have similar effects across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Redband trout	N	N	H	H	N	H	N	N	L	N	N	H	N	N	H	N	N	H	

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Bull trout	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults, which are all potential sites for groin or bank barb development. In lakes, however, char are typically found in deeper water, limiting the potential for direct stressor exposure. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Dolly Varden	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	
Pygmy whitefish	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. These environments are suitable for groin or bank barb development, meaning that this species may be exposed to stressors from related impact mechanisms across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. These environments are generally not suitable for groin or bank barb development. Therefore, there is essentially no likelihood of stressor exposure and effectively no risk of take.
Margined sculpin	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages. While generally remote, these streams are potentially suitable for groin or bank barb development. Therefore, stressor exposure may occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Mountain sucker	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species is commonly found in large rivers and lakes suitable for groin or bank barb development. Stressor exposure is likely to occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lake chub	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats may be suitable for groin or bank barb development, presenting the potential for stressor exposure across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Leopard dace	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for groin or bank barb development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Umatilla dace	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for groin or bank barb development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Western brook lamprey	H	N	I	H	N	I	L	N	I	H	N	I	H	N	I	H	N	I	This species is characterized by isolated breeding populations favoring small streams and brooks. This species is particularly vulnerable to impact mechanisms resulting from bank barb development, and experiences exposure to related stressors across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Occurrence in lacustrine habitats is extremely rare; therefore, the likelihood of stressor exposure and the related potential for take in this environment type are considered discountable.
River lamprey	H	H	H	H	H	H	L	?	?	?	?	?	H	H	H	H	H	H	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries, and lower reaches of larger rivers to rear for extended periods, potentially years. This nonmobile life-history stage is more susceptible to acute construction-related impacts and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications, as well as other changes in habitat complexity. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanism effects affecting the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This equates to a moderate risk of take.

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Pacific lamprey	H	I	H	H	I	H	L	I	?	?	I	?	H	I	H	H	I	H	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for groin or bank barb development. Ammocoetes burrow into riverine sediments to rear for extended periods. This nonmobile life-history stage is more susceptible to acute construction-related impacts, and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications and other changes in habitat complexity. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This equates to a moderate risk of take.
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. This less mobile life-history stage is more susceptible to acute construction-related impacts and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Sturgeon are wide ranging in marine waters. Green sturgeon fisheries occur in the Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
White sturgeon	H	?	H	H	?	H	L	?	L	H	?	H	H	?	H	H	?	H	
Longfin smelt	H	H	H	H	H	H	L	I	L	I	I	H	H	H	H	H	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems potentially suitable for groin or bank barb development. Spawning habitat suitability may be adversely affected by construction and longer term modifications of habitat suitability from hydraulic and geomorphic modifications or other changes in habitat complexity. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Eulachon	H	H	N	H	H	N	L	L	N	I	I	N	H	H	N	H	H	N	
Pacific sand lance	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic/geomorphic and aquatic vegetation modifications are high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from groins and bank barbs. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and, therefore, have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	L	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life history, it is vulnerable to both short-term construction-related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	L	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from development associated with groins and bank barbs, including construction and water quality effects. Being planktonic spawners, this species' spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species and the potential for exposure to most stressors from groin or bank barb related impact mechanisms is minimal. Exceptions include alteration of riparian vegetation affecting this vegetation community in the direct footprint of these structures, as well as hydraulic and geomorphic modifications. Life-history stages exposed to construction activities face a high risk of take (from direct mortality or injury), while the effects of the structure on the environment are likely to result in a moderate to low risk of take.

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Giant Columbia River limpet	H	N	N	H	M	N	L	N	N	L	N	N	H	N	N	?	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for development of groins and bank barbs. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are suitable for groin or bank barb development; therefore, this subactivity type presents the risk of stressor exposure. These species are dependent on flowing water and therefore will not experience stressor exposure from related impact mechanisms in lacustrine environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Great Columbia River spire snail	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N	?	N	N	
California floater (mussel)	H	N	H	H	M	H	L	N	L	L	N	L	H	N	H	H	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats potentially suitable for groin or bank barb development and have potential for stressor exposure from all related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect this species; however, indirect effects could occur through direct effects on host-fish. This equates to a moderate risk of take.
Western ridged mussel	H	N	N	H	M	N	L	N	N	L	N	N	H	N	N	H	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

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10.0 Data Gaps

This section identifies data gaps, as well as what information is needed to fill those gaps, for each of the impact mechanisms associated with the construction and repair of shoreline modification activities.

There are several data gaps, primarily related to the general lack of knowledge related to some of the more obscure HCP species. Most of these relate to the extent of utilization of nearshore areas by species like green and white sturgeon and steelhead. Also poorly understood are the dependence of some species on both riparian and aquatic vegetation, such as Pacific lamprey and sturgeon.

10.1 Jetties

10.1.1 Marine Environments

Significant work has been performed on the various ways that shoreline modifications have altered the nearshore ecosystem, although much of this applies to shoreline hardening and human activities in general, not specific actions. Nearly all of the modifications associated with shoreline development can be attributed to jetties. Within this body of work, three areas have been identified that are relevant to the HPA process where data gaps remain:

- *Cutting-edge technique effectiveness assessments.* Current practical knowledge exists to protect navigational channels through the implementation of innovative engineering alternatives to jetties (e.g., engineered wood placement). These alternatives can provide the desired degree of infrastructure protection while restoring physical processes, habitat features, and/or ecological functions. Unfortunately, while technically feasible, the effectiveness of these techniques has not been fully tested through the implementation of prototype projects.

- *Peer-reviewed monitoring of constructed restoration efforts.* Habitat restoration activities have been undertaken in many nearshore settings throughout western Washington over the last 30 years. Some have been large and have included monitoring (Carney et al. 2005; Cheney et al. 1994), but most have a tendency to be carried out on small, private properties where no consistent monitoring of the project objectives has been calculated (Gerstel and Brown 2006). Even when monitoring has been performed, it has generally been used more as a design tool, rather than investigating the efficacy of the activities to restore the targeted species (Carney et al. 2005).

- *Cumulative impact studies*. Many other regions around the U.S. have established large interdisciplinary studies to document the cumulative impacts on the nearshore environment [e.g., the Great Lakes: (Meadows et al. 2005)]. Impacts on numerous HCP species have been documented due to hydraulic and geomorphic modifications associated with shoreline hardening in general (as detailed in Section 7.2 [*Breakwaters*]).

10.1.2 Freshwater Environments

Because there are few jetties in fresh water, and likely few to be constructed due to the relatively weak demand for them, information regarding any environmental impacts on HCP species stands as a data gap.

10.2 Breakwaters

10.2.1 Marine Environments

There have been a large number of studies related to the hydrogeomorphic and ecologic impacts of breakwaters, mostly as a result of the Environmental Design of Low Crested Coastal Defense Structures (DELOS) program sponsored by the European Community (Losada et al. 2005). The data collected as a part of the hydrogeomorphic portion of the program are relevant to projects in Washington State. The results that document the ecological transition from soft-substrate to hard-substrate communities are particularly relevant to rockfish and sculpin species, as well as the invertebrates.

However, the results of these studies as they apply to some of the other HCP species are not as relevant. In particular, the relative lack of salmonids on the European continent makes it impossible to assess the impacts on these species. The loss of fish that serve as food for the salmonids have been studied, but relating the construction of a single breakwater or artificial reef to salmonid loss has not. In addition, identifying the role that invasive species infestations associated with the deployment offshore rocky structures can have on native species needs to be investigated (e.g., the abundance of urchins or macroalgae at the expense of salmonids).

10.2.2 Freshwater Environments

The only literature on freshwater breakwaters comes from the Great Lakes (Marsden and Chotkowski 2001; Meadows et al. 2005). Because there are few breakwaters in Washington in fresh water, and likely few to be constructed due to the relatively weak demand for them, information regarding any environmental impacts on HCP species stands as a data gap. It would be helpful to have a study like Marsden and Chotkowski (2001) to identify the ability of Washington freshwater breakwaters to encourage invasive species infestations.

10.3 Groins and Bank Barbs

10.3.1 Marine and Lacustrine Environments

There have been no systematic studies of the cumulative impact of groins, as they have not been built in large number for some time because of other legal restrictions (e.g., Clean Water Act, Coastal Zone Management Act, Shoreline Management Act, and city and county critical areas regulations).

Also absent are studies of the differing degree of impacts from different types of common modifications that would be equivalent to a groin or barb. Construction or repair of access stairways would clearly be less disruptive than construction of a groin that extended from above the ordinary high water mark (OHWM) to subtidal (in marine environments) or limnetic (in lacustrine environments) depths. However, there is essentially no peer-reviewed literature on the impact of these modifications on nearshore geomorphology, let alone nearshore ecology.

Finally, there is no literature that describes innovative engineering techniques related to alternative structures typically installed as a part of this HPA on the shoreline, most notably structural access to intertidal or lacustrine littoral areas of the shoreline.

10.3.2 Riverine Environments

In a literature review of the effects of riprap, by far the most common material used in groins and barbs, on salmonids in streams and rivers of the Western Unites States, Schmetterling et al. (2001) noted four areas where research on this subject is lacking: (1) quantifying the habitat availability and quality of riprap; (2) correlating the effects of riprap banks on salmonid density in the absence of other dependent variables such as diking, channelization, and watershed land use; (3) comparative studies on the use of riprap and alternative “soft” techniques such as the integration of natural materials; and (4) the cumulative and synergistic effects of numerous bank-hardening projects at the watershed level.

11.0 Habitat Protection, Conservation, Mitigation, and Management Strategies

The Endangered Species Act requires that impacts on listed species or designated critical habitat be avoided or, if unavoidable, minimized to the maximum extent practicable. This section provides recommendations of strategies for the protection, conservation, mitigation, and management of HCP species based on a review of the scientific literature of the subactivity types addressed in this white paper. Because of the nature of the scientific literature (i.e., papers are written years after an action has been taken), some of these recommendations may already be commonplace. However, it is important to document support for those activities that have a basis in empirical science. Where citations are not provided, it should be assumed that no direct evidence for that recommendation exists, but the recommendation is based on a reasonable conclusion from the collective information surveyed in preparing this white paper.

11.1 Jetties

Jetties cause more damage to nearshore ecosystems than any other single shoreline modification measure discussed in this white paper. They intercept littoral transport, cut off groundwater supply, and disturb natural nearshore circulation. Finally, they are built for navigational purposes, so they also encourage vessel traffic, the impacts of which are summarized in the Marinas and Shipping/Ferry Terminals white paper (Herrera 2007a). However, there are two primary ways to reduce major littoral and nearshore circulation impacts from jetties.

One of the easiest means for reducing the impact of littoral drift disruptions is to develop a sediment bypass strategy (see Figure 11-1). The strategy typically consists of collecting sediment on the updrift side of a set of jetties (in the deposition basin in the figure). This sediment is then dredged and piped, trucked, or barged to the downdrift side of the other jetty. Sediment bypass is a common practice along the Gulf Coast (Seabergh and Kraus 2003) but has seen limited application in sheltered settings (NRC 2007) like Puget Sound. Although large tidal fluctuations can complicate the design, a large number of installed systems have indicated that the mean tide level is a reasonable crest elevation (Seabergh and Kraus 2003). In Puget Sound, for example, this would allow much of the sediment to bypass the jetty in the active sediment transport corridor on the upper foreshore (Finlayson 2006).

The alterations of the water column (tidal prism, nearshore circulation, stratification, etc.) in the vicinity of the jetty are more difficult to mitigate. However, the possibility exists to use engineered logjams (ELJs) or other secured (untreated) woody debris to provide the same function as a riprap or walled structure. These types of structures have been used successfully as groins on riverine shorelines (see Section 11.3 [*Groins and Bank Barbs*], below), even in locations where critical infrastructure is meant to be protected (Herrera 2004). To prevent the isolation of channel waters, these structures can be built to be semi-permeable to allow for the passage of water (and fish) with differing salinities to pass through them, thus minimizing the impacts on nearshore circulation.

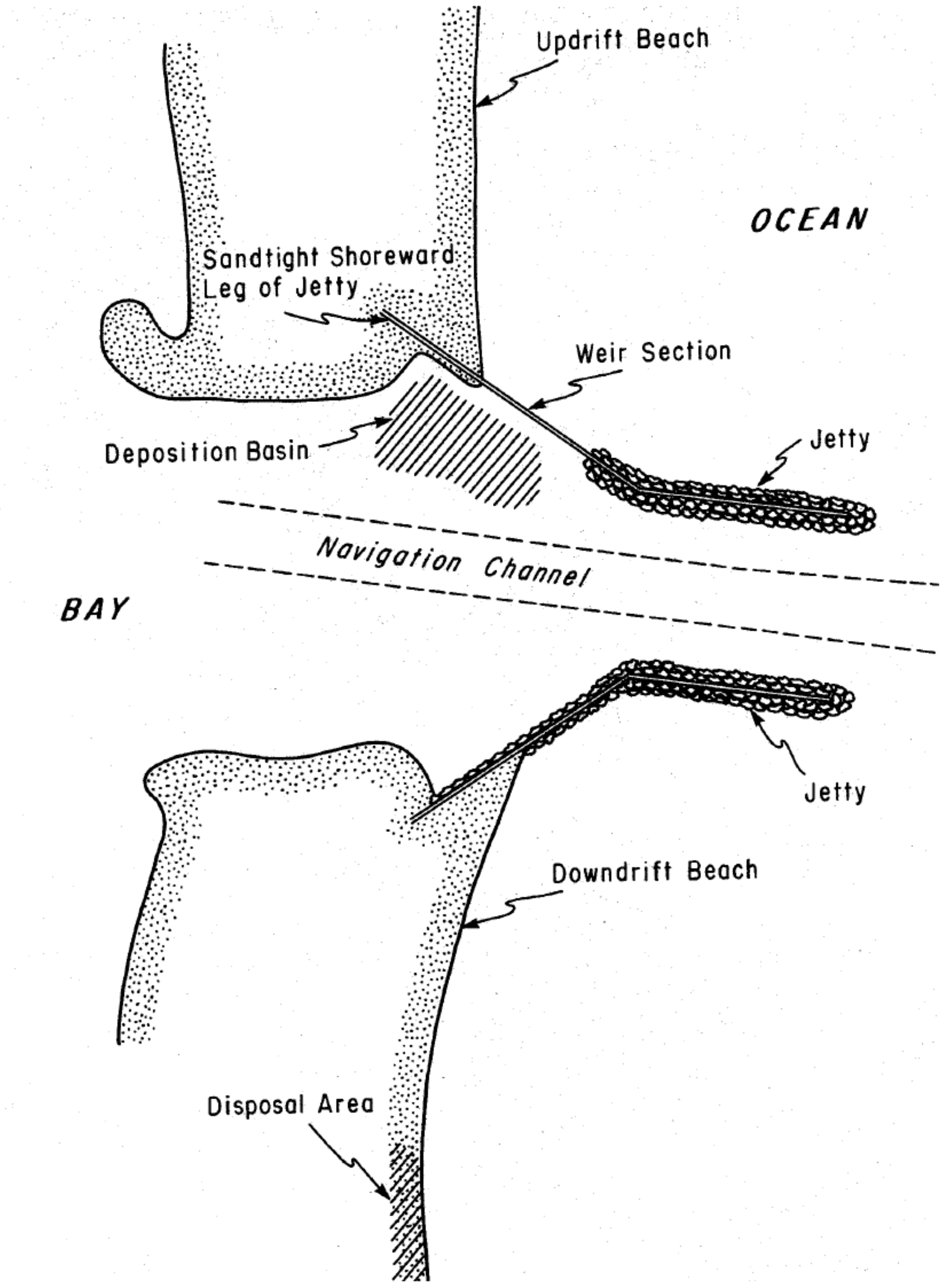

Figure 11-1. Schematic of sediment by-pass system (Seabergh and Kraus 2003).

11.2 Breakwaters

Thirty years of research on breakwaters and artificial reefs has led to a rich data set that has identified pitfalls in offshore, shore-parallel, hard-substrate deployment (Thompson et al. 2002). The following suggestions are taken directly from this work:

- Where possible, use temporary, removable shore-protection measures, such as moored floats.
- If a permanent breakwater is necessary, locate the breakwater(s) to best connect the activity site to other areas of hard-rock habitat in order to reduce the probability of an invasive species infestation (Thompson et al. 2002).
- Use clean earthen materials where possible (i.e., no materials that would leech metals or other exotic organic compounds [e.g., creosote-treated wood]).
- Avoid the use of vertical walls (Bulleri and Chapman 2004). Where possible, mimic the slope of predevelopment shoreline, which in most cases in Puget Sound is between 6:1 and 10:1 (Finlayson 2006).
- Submerge the breakwaters where possible (i.e., in areas of small tides and large waves, the outer coast).
- Where possible, use removable, temporary floating breakwaters in place of permanent, continuous breakwater walls.
- Avoid simple geometric designs. A complex landscape has been shown to be more productive for wide variety of fishes than simple geometries (Moschella et al. 2005; West et al. 1994).
- Provide a rough, complex surface on which a variety of organisms can colonize. Gullies and small caves can be especially fruitful (Moschella et al. 2005), particularly if they are large enough to allow sand to accumulate (Fabi et al. 2006).

For more details on how habitat can be maximized, and impacts on pre-existing communities minimized, see the *Artificial Reef* section of the Habitat Modifications white paper (Herrera 2007c).

11.3 Groins and Bank Barbs

On marine shorelines, the impacts of groins or groin-like structures can be minimized by following these guidelines:

- Minimize the structure's cross-section in the shore-parallel direction.
- Minimize groin wetted length.
- Use earthen materials or untreated wood where possible.
- Avoid the use of structural members that would interrupt groundwater exchange between the sea and the shore.
- Avoid the use of vertical walls (Bulleri and Chapman 2004).
- Allow wrack to accumulate in and around the structure.
- Reduce the protrusion of the structure into the flow as much as possible.

On riverine shorelines, modifications to stabilize banks should mimic natural geomorphic and riparian conditions to the extent possible to limit incidental take. Along riverine shorelines, this would include the placement of engineered logjams (see Figure 11-2), the reconnection of floodplains, and the restoration of riparian forests (Collins et al. 2003). In general, groins and bank barbs provide greater habitat diversity than simple rock revetments (Hjort et al. 1983; Li et al. 1984), and thus are preferred over the construction of rock revetments. Because rivers are dynamic systems, localized bank stabilization efforts can shift the ongoing channel response to an adjacent river segment (Leopold et al. 1964). Bank hardening projects should therefore consider such impacts and take appropriate measures to mitigate these effects.

Figure 11-2. Example of bank protection before (left) and after (right) removal of the rock revetment and installation of engineered logjams in the Mashel River near Eatonville, Washington.

To repair an existing groin, a licensed engineering geologist with experience evaluating projects should determine if removing the structure will cause more damage to the shoreline than letting it remain, or if significant impacts will occur to life or property if the groin is removed. Erosion occurring along adjacent beaches as a result of pre-existing geomorphic conditions near the property should not be considered a significant impact. In addition, two recommendations are made:

- The replacement structure should be designed to allow uninhibited passage of alongshore sediment movement.
- The footprint along the shoreline should be minimized to the greatest extent possible.

11.4 General Construction Best Management Practices (BMPs) in Shoreline Modification Projects

Construction phase recommendations have been thoroughly addressed in a recent USEPA publication (USEPA 2007). The report summarized BMPs that should be applied to hydromodification projects to reduce nonpoint source pollution. The recommendations relevant to construction phase activities include:

- Stockpile fertile topsoil for later use for plants.
- Use hand equipment rather than heavy equipment.
- If using heavy equipment, use wide-track or rubberized tires.
- Avoid instream work, except as authorized by the local fish and wildlife authority.
- Stay 100 feet away from water when refueling or adding oil.
- Avoid using wood treated with creosote or copper compounds.
- Protect areas exposed during construction.

Other nonconstruction-related recommendations in USEPA (2007) include:

- Incorporating monitoring and maintenance of structures
- Using adaptive management
- Conducting a watershed assessment to determine project fate and effects
- Focusing on prevention rather than mitigation
- Emphasizing simple, low-tech, and low-cost methods.

In the construction of jetties, breakwaters, groins, and bank barbs, avoidance or minimization of impacts can be accomplished through proper site selection. For construction and maintenance activities, management strategies can be implemented to minimize underwater noise, project area dewatering, and navigational dredging impacts. The following strategies can be used to avoid and, if avoidance is not possible, to minimize and mitigate negative impacts on habitats and HCP species associated with construction and maintenance activities.

11.4.1 Pile Driving

The intensity of underwater noise produced by pile driving varies considerably depending on site characteristics and the type of materials and methods employed. A desirable approach for avoiding underwater noise impacts from pile driving is to conduct this activity within a dewatered exclusion area. This measure may not be practicable in many circumstances, however. In such cases, a number of BMPs can be used to limit underwater noise impacts.

Where site conditions and design parameters permit flexibility, the following BMPs should be considered to minimize underwater noise related effects on HCP species:

- Use vibratory hammers; the low rise in sound over a longer period of time (see Section 7.1, Figure 7-1) is less stressful to aquatic animals, and the sound is typically 10 to 20 dB lower than impact hammer pile driving (WSDOT 2006).

- Use helical piles where possible. These piles do not require vibration or hammering. The only noise produced is from the screwing action of the driller.

- For projects with pile sizes less than 24 inches in diameter, use the smallest piling size practicable to lower sound pressure levels.

- Where practicable, consider using untreated wood or concrete piling materials as these induce lower sound pressure levels. Even though these materials are less strong, increasing the size of the structure would be considered less impactful as long as the structure does not become so large as to produce other hydrogeomorphic impacts (e.g., if the additional wood piles inhibit transport of sediment, water, or groundwater).

Certain projects will unavoidably produce high-intensity underwater noise conditions (e.g., projects requiring the use of large-diameter steel piles placed using an impact hammer in water depths greater than 3 feet). In such cases, the following BMPs should be employed to limit adverse effects on HCP species:

- Employ a dual-layer bubble curtain or similar noise abatement technology.

- Maintain the integrity of the air bubble curtain; no barges, boat traffic, or other structure or equipment should be allowed to penetrate the air curtain during pile driving activities.

- Use pile caps (wood blocks), if feasible and safe, to reduce the sound of pile driving below injury level (Laughlin 2006).

- Limit pile driving to daylight hours (to avoid attraction effects from construction lighting).

11.4.2 Noise

Elevated ambient noise levels are produced when construction vessels are operated continuously around a project site. To protect HCP species from the resulting stressors, operation of vessel engines and motorized equipment should be limited to the extent necessary to support construction work and the working environment. Where available and practicable, vessels with noise-deadening technology should be employed to reduce underwater noise levels produced.

11.4.3 Project Area Dewatering

For activities that require dewatering, impacts can be minimized by performing work during low-flow or dry conditions and by pumping sediment-laden water from the work area to an infiltration treatment site. Disturbed areas within the project area should be stabilized with a layer of sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site. Science-based protocols for fish removal and exclusion activities should be adopted to track and report the number and species of fish captured, injured, or killed. Projects should also require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols. During passive fish removal, fish removal by seining is recommended before resorting to electrofishing, which carries a greater risk of mortality (NMFS 2006).

Dewatering impacts on HCP species can be further minimized by taking the following precautions:

- Perform work during low-flow or dry conditions, and/or during dry weather.

- Pump sediment-laden water (from the work area that has been isolated from surrounding water) to an infiltration treatment site.

- Dispose of debris or sediment outside of the floodplain.

- Stabilize disturbed areas at the work site with sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site.
- Adopt science-based protocols for fish removal and exclusion activities, including tracking and reporting of number and species of fish captured, fish injured, and mortality.
- Define and require qualifications for personnel performing fish capture and handling; maintain a list of qualified personnel.
- Require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols.
- During passive fish removal, seining is recommended before resorting to electrofishing, which carries a greater risk of mortality.

Electrofishing Guidelines

- Require adherence to NOAA Fisheries electrofishing guidelines.
- Use lowest power output for effective electrofishing.
- Use least damaging direct current (not alternating current).
- Watch for burns or brands or muscle spasms as these indicate harm to the fish.
- Use spherical electrodes appropriate to the water conductivity and the desired size and intensity of the field (Snyder 2003).
- Minimize fish exposure to handling by netting rapidly.
- Frequently change holding water to ensure adequate dissolved oxygen levels and avoid excessive temperature rises.
- Avoid crowding of fish in holding areas.

11.4.4 Navigational Channel and Maintenance Dredging

General recommendations to avoid and minimize the impacts of dredging are provided in the Dredging: Marine Issues white paper (Nightingale and Simenstad 2001a) and include: (1) more extensive use of multiseason pre- and postdredge project biological surveys to assess animal community impacts; (2) incorporation of cumulative effects analysis into all dredging project plans; (3) increased use of landscape-scale planning concepts to plan for beneficial use projects most suitable to the area's landscape ecology and biotic community and food web relationships; (4) further identification of turbidity and noise thresholds to assess fish injury risks; and (5) further analysis and synthesis of the state of knowledge on what is known about the spatial and temporal distribution of fish and shellfish spawning, migration behaviors, and juvenile rearing to evaluate environmental windows for dredging on a site-specific basis.

The following recommendations are intended to reduce the effects of dredging on HCP species:

- For new marine, riverine, and lacustrine projects and significant expansions beyond general maintenance dredging, thoroughly assess the large-scale, cumulative impacts of the resulting changes in bathymetry, habitat loss, and change to estuarine/nearshore marine ecosystem dynamics (e.g., salinity intrusion).
- Require hopper dredges, scows, and barges or any other equipment used to transport dredged materials to the disposal or transfer sites to completely contain the dredged material.
- For long-term projects where continuous dredging and onloading to barges occurs, require periodic movement of the barge to reduce unnecessary shading (for more information regarding shading impacts, see the white paper on *Marine Overwater Structures* [Jones & Stokes et al. 2006]).
- Modify in-water work windows to take into consideration what is known about site-specific spatial and temporal distribution of fish and shellfish eggs, larvae, and juveniles.
- Evaluate the application of in-water work windows on a site-specific basis based on the location and features of the site, such as sediment composition, plant and animal assemblages, and timing of seasonal and migration patterns.
- Use presampling bathymetric surveys, records from previous dredging events, and best professional judgment to estimate the volume of sediments likely to be dredged; base sampling and testing requirements on this estimated volume.
- Avoid projects and expansions that convert intertidal to subtidal habitat. If such conversion is unavoidable, employ comprehensive, large-scale risk assessment to identify the cumulative effects of site-specific changes on ecosystem dynamics.
- Select dredging equipment types according to project-specific conditions, such as sediment characteristics.

- Base turbidity threshold testing for dredging operations on background site turbidity levels.

- In areas where dredging is proximal to sensitive habitats (or in projects where sediments both suitable and unsuitable for unconfined open water disposal will be dredged adjacent to each other), use the “Silent Inspector” (a computerized electronic sensor system) to monitor dredging operations. This tool can assist in operational documentation and regulatory compliance by providing record accessibility and clarity. It also offers advantages for planning, estimating, and managing dredging activities.

- Increase the use of multiseason, preproject surveys of benthos to compare with postproject surveys to understand dredging impacts.

- Where applicable and involving uncontaminated sediments, consider beneficial use of dredged materials that can contribute to habitat restoration, rehabilitation, and enhancement, particularly for projects that incorporate a landscape ecology approach.

- Avoid beneficial use projects that impose unnatural habitats and features on estuarine, marine, and riverine landscapes.

- Dredging should be conducted to a depth not greater than a navigation channel depth at the seaward end. If necessary, authorize dredging to depths greater than the navigation channel at the seaward end only in berthing areas and turning basins for commercial shipping purposes.

- Use hydrodynamic models to predict system-wide changes in salinity, turbidity, and other physicochemical regimes for project assessment planning that avoids or minimizes impacts on aquatic habitat.

12.0 References

Abbe, T.B., and D.R. Montgomery. 1996. Large woody debris jams, channel hydra and habitat formation in large rivers. *Regulated Rivers-Research & Management* 12(2-3): 201-221.

Abdelrhman, M.A. 2003. Effect of eelgrass *Zostera marina* canopies on flow and transport. *Marine Ecology-Progress Series* 248: 67-83.

Adams, P.B., and J.E. Hardwick. 1992. Lingcod. In *California's Living Marine Resources and Their Utilization*, edited by W.S. Leet, C.M. Dewees, and C.W. Haugen. Davis, California: California Sea Grant College Program.

Adams, P.B., C.B. Grimes, J.E. Hightower, S.T. Lindley, and M.L. Moser. 2002. Status Review for North American Green Sturgeon, *Acipenser mediorstris*. National Marine Fisheries Service, Southwest Fisheries Science Center.

AFS and SER. 2000. Review of the 29 April 1999 "Forests and Fish Report" and of associated "Draft Emergency Forest Practice Rules." Northwest Chapter of the Society for Ecological Restoration.

Ahearn, D.S., J.H. Viers, J.F. Mount, and R.A. Dahlgren. 2006. Priming the productivity pump: flood pulse driven trends in suspended algal biomass distribution across a restored floodplain. *Freshwater Biology* 51: 1417-1433.

Ainslie, B.J., J.R. Post, and A.J. Paul. 1998. Effects of Pulsed and Continuous DC Electrofishing on Juvenile Rainbow Trout. *North American Journal of Fisheries Management* 18(4): 905-918.

Aksnes, D.L., and A.C.W. Utne. 1997. A revised model of visual range in fish. *Sarsia* 82(2): 137-147.

Albers, W.D., and P.J. Anderson. 1985. Diet of Pacific cod, *Gadus macrocephalus*, and predation on the northern pink shrimp, *Pandalus borealis*, in Pavlof Bay, Alaska. *Fishery Bulletin* 83: 601-10.

Altayaran, A.M., and I.M. Madany. 1992. Impact of Desalination Plant on the Physical and Chemical Properties of Seawater, Bahrain. *Water Research* 26(4): 435-441.

Amoser, S., and F. Ladich. 2005. Are Hearing Sensitivities of Freshwater Fish Adapted to the Ambient Noise in Their Habitats? *The Journal of Experimental Biology* 208: 3533-3542.

Angradi, T.R., E.W. Schweiger, D.W. Bolgrien, P. Ismert, and T. Selle. 2004. Bank stabilization, riparian land use and the distribution of large woody debris in a regulated reach of the upper Missouri River, North Dakota, USA. *River Research and Applications* 20(7): 829-846.

Armstrong, D.A., B.G. Stevens, and J.E. Hoeman. 1982. Distribution and Abundance of Dungeness Crab and Crangon Shrimp and Dredging-related Mortality of Invertebrates and Fish in Grays Harbor, Washington. DACW67-80-C-0086. Seattle, Washington: School of Fisheries, University of Washington.

Au, D.W.T., C.A. Pollino, R.S.S. Wu, P.K.S. Shin, S.T.F. Lau, and J.Y.M. Tang. 2004. Chronic effects of suspended solids on gill structure, osmoregulation, growth, and triiodothyronine in juvenile green grouper *Epinephelus coioides*. *Marine Ecology-Progress Series* 266: 255-264.

Babanin, A.V. 2006. On a wave-induced turbulence and a wave-mixed upper ocean layer. *Geophysical Research Letters* 33(20).

Bacchiocchi, F., and L. Airoldi. 2003. Distribution and dynamics of epibiota on hard structures for coastal protection. *Estuarine Coastal and Shelf Science* 56(5-6): 1157-1166.

Bailey, K.M. 1982. The early life history of the Pacific hake, *Merluccius productus*. *Fishery Bulletin* 80: 589-598.

Bailey, K.M., T.J. Quinn, P. Bentzen, and W.S. Grant. 1999. Population structure and dynamics of walleye pollock, *Theragra chalcogramma*. *Advances in Marine Biology* 37: 179-255.

Baker, P. 1995. Review of ecology and fishery of the Olympia oyster, *Ostrea lurida* with annotated bibliography. *Journal of Shellfish Research* 14(2): 501-518.

Baldwin, D.H., J.F. Sandahl, J.S. Labenia, and N.L. Scholz. 2003. Sublethal effects of copper on coho salmon: Impacts on nonoverlapping receptor pathways in the peripheral olfactory nervous system. *Environmental Toxicology and Chemistry* 22(10): 2266-2274.

Ban, M. 2006. Rearing conditions to develop seawater tolerance in underyearling sockeye salmon smolt. *Fisheries Science* 72(1): 128-135.

Banner, A., and M. Hyatt. 1973. Effects of noise on eggs and larvae of two estuarine fishes. *Transaction of the American Fisheries Society* 102: 134-136.

Bargmann, G.C. 1980. Studies on Pacific cod in Agate Pass, Washington. Washington Department of Fish and Wildlife.

Bartholow, J.M. 2002. Estimating cumulative effects of clearcutting on stream temperatures. Available at: http://smig.usgs.gov/SMIG/features_0902/clearcut.html.

Barton, D.R., W.D. Taylor, and R.M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. *North American Journal of Fisheries Management* 5: 364-378.

Bash, J., C. Berman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. Washington State Transportation Center.

Baxter, R.M. 1977. Environmental Effects of Dams and Impoundments. *Annual Review of Ecology and Systematics* 8: 255-283.

Beacham, T.D., and C.B. Murray. 1990. Temperature, Egg Size, and Development of Embryos and Alevins of 5 Species of Pacific Salmon - a Comparative-Analysis. *Transactions of the American Fisheries Society* 119(6): 927-945.

Beamer, E., Aundrea McBride, Correigh Greene, Rich Henderson, Greg Hood, Karen Wolf, Kim Larsen, Casey Rice, and Kurt Fresh. 2005. Delta and Nearshore Restoration for the Recovery of Wild Skagit River Chinook Salmon: Linking Estuary Restoration to Wild Chinook Salmon Populations. LaConner, Washington: Skagit River System Cooperative.

Bearzi, G., D. Holcer, and G.N. Di Sciara. 2004. The Role of Historical Dolphin Takes and Habitat Degradation in Shaping the Present Status of Northern Adriatic Cetaceans. *Aquatic Conservation: Marine and Freshwater Ecosystems* 14(4): 363-379.

Beauchamp, D.A., E.R. Byron, and W.A. Wurtsbaugh. 1994. Summer habitat use by littoral-zone fishes in Lake Tahoe and the effects of shoreline structures. *North American Journal of Fisheries Management* 14: 385-394.

Beckett, D.C., T.P. Aartila, and A.C. Miller. 1992. Contrasts in Density of Benthic Invertebrates between Macrophyte Beds and Open Littoral Patches in Eau-Galle Lake, Wisconsin. *American Midland Naturalist* 127(1): 77-90.

Beechie, T., E. Beamer, and L. Wasserman. 1994. Estimating salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. *North American Journal of Fisheries Management* 14: 797-811.

Beeman, J.W., D.W. Rondorf, and M.E. Tilson. 1994. Assessing Smoltification of Juvenile Spring Chinook Salmon (*Oncorhynchus-Tshawytscha*) Using Changes in Body Morphology. *Canadian Journal of Fisheries and Aquatic Sciences* 51(4): 836-844.

Bellotti, G. 2004. A simplified model of rip currents systems around discontinuous submerged barriers. *Coastal Engineering* 51(4): 323-335.

Bennett, D.H., W.P. Connor, and C.A. Eaton. 2003. Substrate composition and emergence success of fall chinook salmon in the Snake River. *Northwest Science* 77(2): 93-99.

Berg, L., and T.G. Northcote. 1985. Changes in Territorial, Gill-Flaring, and Feeding-Behavior in Juvenile Coho Salmon (*Oncorhynchus-Kisutch*) Following Short-Term Pulses of Suspended Sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 42(8): 1410-1417.

Bernhardt, E.S., and M.A. Palmer. 2007. Restoring Streams in an Urbanizing World. *Freshwater Biology* 52: 738-751.

Berry, W., N. Rubenstein, B. Melzian, and B. Hill. 2003. The Biological Effects of Suspended and Bedded Sediments (SABS) in Aquatic Systems: A Review. Narragansett, Rhode Island: U.S. Environmental Protection Agency, Office of Research and Development.

Beschta, R.L. 1991. Stream habitat management for fish in the northwestern United States: the role of riparian vegetation. *American Fisheries Society Symposium* 10: 53-58.

Beschta, R.L. 1997. Riparian shade and stream temperature: An alternative perspective. *Rangelands* 19(2): 25-28.

Beschta, R.L., and R.L. Taylor. 1988. Stream Temperature Increases and Land-Use in a Forested Oregon Watershed. *Water Resources Bulletin* 24(1): 19-25.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1988. Stream temperature and aquatic habitat: Fishery and forestry interactions. In *Streamside Management: Forestry and Fishery Interactions: Contribution No. 57.* Seattle, Washington: University of Washington, Institute of Forest Resources. pp. 191-232.

Bilby, R.E. 1984. Post-logging removal of woody debris may affect stream channel stability. *Journal of Forestry* 82(10): 609-613.

Bilby, R.E., and P.A. Bisson. 1992. Allochthonous Versus Autochthonous Organic-Matter Contributions to the Trophic Support of Fish Populations in Clear-Cut and Old-Growth Forested Streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49(3): 540-551.

Bilby, R.E., and P.A. Bisson. 1998. Function and distribution of large woody debris. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. Springer-Verlag, New York. pp. 324-347.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat requirements of salmonids in streams. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 83-138.

Blaber, S.J.M., D.P. Cyrus, J.J. Albaret, C.V. Ching, J.W. Day, M. Elliott, M.S. Fonseca, D.E. Hoss, J. Orensanz, I.C. Potter, and W. Silvert. 2000. Effects of fishing on the structure and functioning of estuarine and nearshore ecosystems. *ICES Journal of Marine Science* 57(3): 590-602.

Black, J.A. and W.J. Birge. 1980. An Avoidance Response Bioassay for Aquatic Pollutants. Res. Report No. 123, U.S. NTIS PB80-180490. Water Resources Research Institute, University of Kentucky, Lexington. (As cited in Stratus 2005b).

Black, K.P., and G.D. Parry. 1999. Entrainment, Dispersal, and Settlement of Scallop Dredge Sediment Plumes: Field Measurements and Numerical Modeling. *Canadian Journal of Fisheries and Aquatic Sciences* 56(12): 2271-2281.

Black, M.C., D.S. Millsap, and J.F. Mccarthy. 1991. Effects of Acute Temperature-Change on Respiration and Toxicant Uptake by Rainbow-Trout, *Salmo-Gairdneri* (*Richardson*). *Physiological Zoology* 64(1): 145-168.

Blaxter, J.H.S., J.A.B. Gray, and E.J. Denton. 1981. Sound and startle responses in herring shoals. *Journal of Marine Biology Associated with U.K.* 6: 851-869.

Block, D.G. 1955. Trout migration and spawning studies on the North Fork drainage of the Flathead River. University of Montana, Missoula, Montana.

Blomqvist, S., A. Gunnars, and R. Elmgren. 2004. Why the limiting nutrient differs between temperate coastal seas and freshwater lakes: A matter of salt. *Limnology and Oceanography* 49(6): 2236-2241.

Boehlert, G.W. 1980. Size composition, age composition, and growth of canary rockfish, *Sebastes pinniger*, and splitnose rockfish, *S. diploproa*, from the 1977 rockfish survey. *Marine Fisheries Review* 42: 57-63.

Boehlert, G.W., and R.F. Kappenman. 1980. Variation of growth with latitude in two species of rockfish (*Sebastes pinniger* and *S. diploproa*) from the northeast Pacific Ocean. *Marine Ecology-Progress Series* 3: 1-10.

Boehlert, G.W., M.M. Yoklavich, and D.B. Chelton. 1989. Time series of growth in the genus *Sebastes* from the northeast Pacific Ocean. *Fishery Bulletin* 87: 791-806.

Boese, B.L., B.D. Robbins, and G. Thursby. 2005. Desiccation is a limiting factor for eelgrass (*Zostera marina L.*) distribution in the intertidal zone of a northeastern Pacific (USA) estuary. *Botanica Marina* 48(4): 274-283.

Bolton, S., and J. Shellberg. 2001. Ecological issues in floodplains and riparian corridors. Available at: http://www.wsdot.wa.gov/research/reports/fullreports/524.1.pdf (accessed October 4, 2006).

Booth, D.B. 1990. Stream Channel Incision Following Drainage Basin Urbanization. *Water Resources Bulletin* 26(3): 407-417.

Booth, D.B., and P.C. Henshaw. 2001. Rates of Channel Erosion in Small Urban Streams. In *Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas*, edited by M. Wigmosta and S. Burges. AGU Monograph Series, Water Science and Application Volume 2. pp. 17-38.

Bordalo, A.A. 2003. Microbiological water quality in urban coastal beaches: the influence of water dynamics and optimization of the sampling strategy. *Water Research* 37(13): 3233-3241.

Bostrom, C., and J. Mattila. 1999. The relative importance of food and shelter for seagrass-associated invertebrates: a latitudinal comparison of habitat choice by isopod grazers. *Oecologia* 120(1): 162-170.

Botton, M.L., R.E. Loveland, and T.R. Jacobsen. 1994. Site Selection by Migratory Shorebirds in Delaware Bay, and Its Relationship to Beach Characteristics and Abundance of Horseshoe-Crab (*Limulus-Polyphemus*) Eggs. *Auk* 111(3): 605-616.

Bowen, K.L., N.K. Kaushik, and A.M. Gordon. 1998. Macroinvertebrate communities and biofilm chlorophyll on woody debris in two Canadian oligotrophic lakes. *Archiv Fur Hydrobiologie* 141(3): 257-281.

Bowman, D., and E. Pranzini. 2003. Reversed responses within a segmented detached breakwater, the Tuscany coast Italy - a case study. *Coastal Engineering* 49(4): 263-274.

Bowman, M.F., and R.C. Bailey. 1998. Upper pH Tolerance Limit of the Zebra Mussel (*Dreissena Polymorpha*). *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 76(11): 2119-2123.

Box, J.B., D. Wolf, J. Howard, C. O'Brien, D. Nez, and D. Close. 2003. The distribution and status of freshwater mussels in the Umatilla River system. Portland, Oregon: Bonneville Power Administration.

Brannon, E.L., M.S. Powell, T.P. Quinn, and A. Talbot. 2004. Population structure of Columbia River Basin chinook salmon and steelhead trout. *Reviews in Fisheries Science* 12(2-3): 99-232.

Brennan, J. 2004. Riparian functions and the development of management actions in marine nearshore ecosystems. In *Proceedings of the DFO/PSAT Sponsored Marine Riparian Experts Workshop, Tsawwassen, BC, February 17-18, 2004*, edited by J.P. Lemieux, J.S. Brennan, M. Farrell, C.D. Levings and D. Myers. Canadian Manuscript Report of the Fisheries and Aquatic Sciences No. 2680.

Brennan, J.S., and H. Culverwell. 2004. Marine Riparian: An assessment of riparian functions in marine ecosystems. Seattle, Washington: Washington Sea Grant Program.

Brennan, J.S., K.F. Higgins, J.R. Cordell, and V.A. Stamatiou. 2004. Juvenile salmon composition, timing distribution, and diet in marine nearshore waters of central Puget Sound in 2001-2002. Seattle, Washington: King County Dept. of Natural Resources and Parks.

Bricelj, V.M., and R.E. Malouf. 1984. Influence of Algal and Suspended Sediment Concentrations on the Feeding Physiology of the Hard Clam Mercenaria-Mercenaria. *Marine Biology* 84(2): 155-165.

Bricelj, V.M., R.E. Malouf, and C. Dequillfeldt. 1984. Growth of Juvenile Mercenaria-Mercenaria and the Effect of Resuspended Bottom Sediments. *Marine Biology* 84(2): 167-173.

Brooks, K.M. 2004. Polycyclic Aromatic Hydrocarbon Migration from Creosote-Treated Railway Ties into Ballast and Adjacent Wetlands. Madison, Wisconsin: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory.

Brosofske, K.D., J. Chen, R.J. Naiman, and J.F. Franklin. 1997. Harvesting effects on microclimatic gradients from small streams to uplands in Western Washington. *Ecological Applications* 7(4): 1188-1200.

Brown, G.W. 1970. Predicting Effect of Clearcutting on Stream Temperature. *Journal of Soil and Water Conservation* 25(1): 11-13.

Bryan, M.D., and D.L. Scarnecchia. 1992. Species Richness, Composition, and Abundance of Fish Larvae and Juveniles Inhabiting Natural and Developed Shorelines of a Glacial Iowa Lake. *Environmental Biology of Fishes* 35(4): 329-341.

Buck, E.H. 1995. Acoustic Thermometry of Ocean Climate, Marine Mammal Issues. Report Number 95-603 ENR. Washington DC: Congressional Research Service, Library of Congress.

Buckley, R.M., and G.J. Hueckel. 1985. Biological Processes and Ecological Development on an Artificial Reef in Puget-Sound, Washington. *Bulletin of Marine Science* 37(1): 50-69.

Buer, K.Y., J.N. Eaves, R.G. Scott, and J.R. McMillan. 1984. Basin changes affecting salmon habitat in the Sacramento River. *Pacific Northwest stream habitat management workshop*, Arcata, California.

Bulleri, F., and M.G. Chapman. 2004. Intertidal assemblages on artificial and natural habitats in marinas on the north-west coast of Italy. *Marine Biology* 145(2): 381-391.

Bulleri, F., M. Abbiati, and L. Airoldi. 2006. The colonization of human-made structures by the invasive alga *Codium fragile* ssp *tomentosoides* in the north Adriatic Sea (NE Mediterranean). *Hydrobiologia* 555: 263-269.

Burgess, W.C., and S.B. Blackwell. 2003. Acoustic Monitoring of Barrier Wall Installation at the Former Rhône-Poulenc Site, Tukwila, Washington. Tukwila, Washington: Greenridge Sciences, Inc.

Burgner, R.L. 1991. Life History of Sockeye Salmon *(Oncorhynchus nerka)*. Pacific Salmon Life Histories. Edited by C. Groot and L. Margolis. Vancouver, British Columbia: UBC Press.

Busby, M.S., and R.A. Barnhart. 1995. Potential food sources and feeding ecology of juvenile fall chinook salmon in California's Mattole river lagoon. *California Fish and Game* 81(4): 133-146.

Busby, P.J., T.C. Wainwright, G.J. Bryant, L.J. Lierheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino. 1996. Status Review of West Coast Steelhead from Washington, Idaho, Oregon, and California. Seattle, Washington: National Oceanic and Atmospheric Administration.

Byrnes, M.R., and M.W. Hiland. 1995. Large-Scale Sediment Transport Patterns on the Continental-Shelf and Influence on Shoreline Response - St-Andrew-Sound, Georgia to Nassau-Sound, Florida, USA. *Marine Geology* 126(1-4): 19-43

Caceres, I., A. Sanchez-Arcilla, B. Zanuttigh, A. Lamberti, and L. Franco. 2005. Wave overtopping and induced currents at emergent low crested structures. *Coastal Engineering* 52(10-11): 931-947.

Cake, E.W.J. 1983. Habitat suitability index models: Gulf of Mexico American oyster. U.S. Department of Interior, Fish and Wildlife Service, Washington D.C.

Cardoso, P.G., D. Raffaelli, and M.A. Pardal. 2007. Seagrass beds and intertidal invertebrates: an experimental test of the role of habitat structure. *Hydrobiologia* 575: 221-230.

Carlson, T.J., D.A. Woodruff, G.E. Johnson, N.P. Kohn, G.R. Plosky, M.A. Weiland, J.A. Southard, and S.L. Southard. 2005. Hydroacoustic Measurements During Pile Driving at the Hood Canal Bridge, September through November. 2004. Sequim, Washington: Battelle Marine Sciences Laboratory.

Carney, L.T., J.R. Waaland, T. Klinger, and K. Ewing. 2005. Restoration of the bull kelp *Nereocystis luetkeana* in nearshore rocky habitats. *Marine Ecology-Progress Series* 302: 49-61.

Carrasquero, J. 2001. Over-Water Structures: Freshwater Issues. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Herrera Environmental Consultants, Seattle, Washington. April 2001.

Cederholm, C., R.E. Bilby, P. Bisson, T. Bumstead, B. Fransen, W. Scarlett, and J. Ward. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. *North American Journal of Fisheries Management* 17: 947-963.

CEQ. 1997. Considering cumulative effects under the National Environmental Policy Act. Council on Environmental Quality.

Chambers, P.A., R.E. DeWreede, E.A. Irlandi, and H. Vandermeule. 1999. Management issues in aquatic macrophyte ecology: A Canadian perspective. *Canadian Journal of Botany* 77(4): 471.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. *Transactions of the American Fisheries Society* 117: 1-21.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1992. Vegetation Responses to Edge Environments in Old-Growth Douglas-Fir Forests. *Ecological Applications* 2(4): 387-396.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1993. Contrasting Microclimates among Clear-Cut, Edge, and Interior of Old-Growth Douglas-Fir Forest. *Agricultural and Forest Meteorology* 63(3-4): 219-237.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1995. Growing-Season Microclimatic Gradients from Clear-Cut Edges into Old-Growth Douglas-Fir Forests. *Ecological Applications* 5(1): 74-86.

Chen, J.Q., S.C. Saunders, T.R. Crow, R.J. Naiman, K.D. Brosofske, G.D. Mroz, B.L. Brookshire, and J.F. Franklin. 1999. Microclimate in forest ecosystem and landscape ecology - Variations in local climate can be used to monitor and compare the effects of different management regimes. *Bioscience* 49(4): 288-297.

Cheney, D., R. Oestman, G. Volkhardt, and J. Getz. 1994. Creation of Rocky Intertidal and Shallow Subtidal Habitats to Mitigate for the Construction of a Large Marina in Puget-Sound, Washington. *Bulletin of Marine Science* 55(2-3): 772-782.

Cheng, W., and J.C. Chen. 2000. Effects of pH, temperature and salinity on immune parameters of the freshwater prawn *Macrobrachium rosenbergii*. *Fish & Shellfish Immunology* 10(4): 387-391.

Chisholm, I., M. E. Hensler, B. Hansen, and D., Skaar. 1989. Quantification of Libby Reservoir levels needed to maintain or enhance reservoir fisheries: summary report 1983-1985. Portland, Oregon: U.S. Department of Energy, Bonneville Power Administration, Oregon Division of Fish and Wildlife.

Chow, V.T. 1959. *Open-Channel Hydraulics*. San Francisco: McGraw-Hill.

Chrzastowski, M.J., and T.A. Thompson. 1994. Late Wisconsin and Holocene Geologic History of the Illinois-Indiana Coast of Lake-Michigan. *Journal of Great Lakes Research* 20(1): 9-26.

Church, M., M.A. Hassan, and J.F. Wolcott. 1998. Stabilizing self-organized structures in gravel-bed stream channels: Field and experimental observations. *Water Resources Research* 34(11): 3169-3179.

Clarke, D.G., and D.H. Wilber. 2000. Assessment of Potential Impacts of Dredging Operations Due to Sediment Resuspension. DOER-E9.

Clarke, S.J. 2002. Vegetation growth in rivers: influences upon sediment and nutrient dynamics. *Progress in Physical Geography* 26(2): 159-172.

Collins, B. 2000. Mid-19th Century Stream Channels and Wetlands Interpreted from Archival Sources for Three North Puget Sound Estuaries. LaConner, Washington: Skagit System Cooperative.

Collins, B.D., D.R. Montgomery, and A.J. Sheikh. 2003. Reconstructing the Historical Riverine Landscape of the Puget Lowland. In *Restoration of Puget Sound Rivers*, edited by D.R. Montgomery, S. Bolton, D.B. Booth and L. Wall. Seattle: University of Washington Press. pp. 79-128.

Cooper, A.C. 1965. The Effects of Transported Stream Sediments on the Survival of Sockeye and Pink Salmon Eggs and Alevins. International Pacific Salmon Fisheries Commission Bulletin 18. 71p.

Cordell, J.R. 1986. Structure and dynamics of an epibenthic harpacticoid assemblage and the role of predation by juvenile salmon. University of Washington, Seattle, Washington.

COSEWIC. 2003. Rocky Mountain Ridge Mussel, *Gonidea angulata*. Status Report. Committee on the Status of Endangered Wildlife in Canada. Ottawa, Ontario: Canadian Wildlife Service, Environment Canada.

Couch, D., and T.J. Hassler. 1990. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Olympia Oyster. PBS Record: 115470. U.S. Army Corps of Engineers and U.S. Fish and Wildlife Service.

Cox, M., P.H. Rogers, A.N. Popper, and W.M. Saidel. 1987. Anatomical effects of intense tone stimulation in the goldfish ear: Dependence on sound-pressure level and frequency. *Journal of Acoust. Soc. Am.*

Creque, S.M., M.J. Raffenberg, W.A. Brofka, and J.M. Dettmers. 2006. If you build it, will they come? Fish and angler use at a freshwater artificial reef. *North American Journal of Fisheries Management* 26(3): 702-713.

da Silva, J.F., and R.W. Duck. 2001. Historical changes of bottom topography and tidal amplitude in the Ria de Aveiro, Portugal - trends for future evolution. *Climate Research* 18(1-2): 17-24.

Dalbey, S.R., T.E. McMahon, and W. Fredenberg. 1996. Effect of Electrofishing Pulse Shape and Electrofishing-Induced Spinal Injury to Long-Term Growth and Survival of Wild Rainbow Trout. *North American Journal of Fisheries Management* 16: 560-569.

Dale, R.K., and D.C. Miller. 2007. Spatial and temporal patterns of salinity and temperature at an intertidal groundwater seep. *Estuarine Coastal and Shelf Science* 72(1-2): 283-298.

de Croux, P., M. Julieta, and A. Loteste. 2004. Lethal effects of elevated pH and ammonia on juveniles of Neotropical fish *Colosoma macropomum* (Pisces, Caracidae). *Journal of Environmental Biology* 25(1): 7-10.

Dean, R.G., and R.A. Dalrymple. 2002. *Coastal Processes with Engineering Applications*. Cambridge, UK: Cambridge University Press.

Desbonnet, A., P. Pogue, V. Lee, and N. Wolff. 1994. Vegetated buffers in the coastal zone - A summary review and bibliography. University of Rhode Island Graduate School of Oceanography.

DesJardin 2003. Personal communication, as cited in WSDOT (2006). No additional information provided in WSDOT (2006).

DFO. 2007. Concrete Wash Water: Characteristics. Available at: http://www-heb.pac.dfo-mpo.gc.ca/water_quality/fish_and_pollution/conc_char_e.htm (accessed June 3, 2007).

Dias, J.M.A., and W.J. Neal. 1992. Sea Cliff Retreat in Southern Portugal - Profiles, Processes, and Problems. *Journal of Coastal Research* 8(3): 641-654.

Dietrich, W.E., J.W. Kirchner, H. Ikeda, and F. Iseya. 1989. Sediment Supply and the Development of Coarse Surface Layer in Gravel Bedded Rivers. *Nature* 340(6230): 215-217.

Dill, L.M., R.C. Ydenberg, and A.H.G. Fraser. 1981. Food abundance and territory size in juvenile coho salmon (*Oncorhynchus kisutch*). *Canadian Journal Of Zoology* 59: 1801-1809.

Dooley, K.M., C.F. Knopf, and R.P. Gambrell. 1999. Final Report: pH-Neutral Concrete for Attached Microalgae and Enhanced Carbon Dioxide Fixation - Phase I. Louisiana State University, Baton Rouge, Louisiana.

Downing, J. 1983. *The coast of Puget Sound: Its processes and development*. Seattle, Washington: Washington SeaGrant Program. University of Washington Press.

Downing, J.A., Y. Rochon, M. Perusse, and H. Harvey. 1993. Spatial Aggregation, Body Size, and Reproductive Success in the Freshwater Mussel Elliptio Complanata. Journal of the North American Benthological Society 12: 148-156.

Dudley, S.J., J.C. Fischenich, and S.R. Abt. 1998. Effect of woody debris entrapment on flow resistance. *Journal of the American Water Resources Association* 34(5): 1189-1197.

Dunn, J.R., and A.C. Matarese. 1987. A review of early life history of northeast Pacific gadoid fishes. *Fisheries Research* 5: 163-184.

Dwyer, W.P., and R.G. White. 1997. Effect of Electroshock on Juvenile Arctic Grayling and Yellowstone Cutthroat Trout Growth 100 Days after Treatment. *North American Journal of Fisheries Management* 17: 174-177.

Eby, L.A., L.B. Crowder, C.M. McClellan, C.H. Peterson, and M.J. Powers. 2005. Habitat degradation from intermittent hypoxia: impacts on demersal fishes. *Marine Ecology-Progress Series* 291: 249-261.

Ecology. 1999. Working in the Water. Washington State Department of Ecology.

Ecology. 2002. Evaluating Criteria for the Protection of Freshwater Aquatic Life in Washington's Surface Water Quality Standards, Dissolved Oxygen: Draft Discussion Paper and Literature Summary. Publication Number 00-10-071. Olympia, Washington: Washington State Department of Ecology.

Edwards, R.T. 1998. The hyporheic zone. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Eilers, H.P.I. 1975. Plants, plant communities, net production and tide levels: the ecological biogeography of the Nehalem salt marshes, Tillamook County, Oregon. Oregon State University, Corvallis, Oregon, 368 pp.

Ellis, M.M. 1942. Fresh-Water Impoundments. *Transactions of the American Fisheries Society*, 71st Annual Meeting, pp. 80-93.

Embrey, S.S., and P.W. Moran. 2006. Quality of streamwater in the Puget Sound Basin–A decade of study and beyond [poster]. *Toxics in Puget Sound: Connecting Marine Environment to Human Health and the Economy, Puget Sound Action Team Forum*, Seattle, Washington, April 5, 2006.

Emmett, R.L., S.L. Stone, S.A. Hinton, and M.E. Monaco. 1991. Distribution and Abundance of Fishes and Invertebrates in West Coast Estuaries, Volume II: Species Life History Summaries. Rockville, Maryland: NOAA/NOA Strategic Environmental Assessments Division.

Enger, P.S. 1981. Frequency Discrimination in Teleosts - Central or Peripheral? In *Hearing and Sound Communication in Fishes*, edited by W.N. Tavolga, A.N. Popper and R.R. Fay. New York: Springer-Verlag. pp. 243-255.

Ensign, S.H., and M.W. Doyle. 2005. In-channel transient storage and associated nutrient retention: Evidence from experimental manipulations. *Limnology and Oceanography* 50(6): 1740-1751.

Environment Canada. 1992. Interim Criteria for Quality Assessment of St. Lawrence River Sediment. St. Lawrence Action Plan.

Erftemeijer, P.L.A., and R.R.R. Lewis. 2006. Environmental impacts of dredging on seagrasses: A review. *Marine Pollution Bulletin* 52(12): 1553-1572.

Eschmeyer, W.N., E.S. Herald, and H. Hammon. 1983. *A Field Guide to Pacific Coast Fishes of North America*. Boston, Massachusetts: Houghton Mifflin.

Everett, R.A., and G.M. Ruiz. 1993. Coarse woody debris as a refuge from predation in aquatic communities. *Oecologia* 93(4): 475-486.

Fabi, G., S. Manoukian, and A. Spagnolo. 2006. Feeding behavior of three common fishes at an artificial reef in the northern Adriatic Sea. *Bulletin of Marine Science* 78(1): 39-56.

Fausch, K.D., C. Gowan, A.D. Richmond, and S.C. Riley. 1995. The role of dispersal in trout population response to habitat formed by large woody debris in Colorado mountain streams. *Bulletin Francais De La Peche Et De La Pisciculture*(337-9): 179-190.

Fay, R.R. 1988. Peripheral Adaptations for Spatial Hearing in Fish. In *Sensory Biology of Aquatic Animals*, New York: Springer-Verlag. pp. 711-731.

Feist, B.E., J. Anderson, and R. Miyamoto. 1992. Potential Impacts of Pile Driving on Juvenile Pink (*Oncorhynchus Gorbuscha*) and Chum (*O. Keta*) Salmon Behavior and Distribution. Seattle, Washington: University of Washington.

Feller, M.C. 2005. Forest Harvesting and Streamwater Inorganic Chemistry in Western North America: A Review. *Journal of the American Water Resources Association* 41(4): 785-811.

Ferraro, S.P., and F.A. Cole. 2007. Benthic macrofauna-habitat associations in Willapa Bay, Washington, USA. *Estuarine Coastal and Shelf Science* 71(3-4): 491-507.

Feyrer, F., T. Sommer, and W. Harrell. 2006. Importance of flood dynamics versus intrinsic physical habitat in structuring fish communities: Evidence from two adjacent engineered floodplains on the Sacramento River, California. *North American Journal of Fisheries Management* 26(2): 408-417.

Finlayson, D. 2006. The Geomorphology of Puget Sound Beaches. University of Washington, Seattle, 216 pp.

Fischer, H.B., List, E. J., Koh, R. C. Y., Imberger, J., Brooks, N. H. 1979. *Mixing in Inland and Coastal Waters*. San Diego, California: Academic Press.

Fitzsimons, J.D. 1996. The significance of man-made structures for lake trout spawning in the Great Lakes: Are they a viable alternative to natural reefs? *Canadian Journal of Fisheries and Aquatic Sciences* 53: 142-151.

Flett, P.A., K.R. Munkittrick, G. VanDerKraak, and J.F. Leatherland. 1996. Overripening as the cause of low survival to hatch in Lake Erie coho salmon (*Oncorhynchus kisutch*) embryos. *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 74(5): 851-857.

Fonseca, M.S., and S.S. Bell. 1998. Influence of physical setting on seagrass landscapes near Beaufort, North Carolina, USA. *Marine Ecology-Progress Series* 171: 109-121.

Fonseca, M.S., J.C. Zieman, G.W. Thayer, and J.S. Fisher. 1983. The Role of Current Velocity in Structuring Eelgrass (*Zostera-Marina L*) Meadows. *Estuarine Coastal and Shelf Science* 17(4): 367-380.

Francis, T.B., and D.E. Schindler. 2006. Degradation of littoral habitats by residential development: Woody debris in lakes of the Pacific Northwest and Midwest, United States. *Ambio* 35(6): 274-280.

Fredenberg, W.A. 1992. Evaluation of Electrofishing-Induced Spinal Injuries Resulting from Field Electrofishing Surveys in Montana. Montana Department of Fish, Wildlife and Parks. Helena, Montana.

Freeman, P.L., and M.S. Schorr. 2004. Influence of Watershed Urbanization on Fine Sediment and Macroinvertebrate Assemblage Characteristics in Tennessee Ridge and Valley Streams. *Journal of Freshwater Ecology* 19(3): 353-362.

Fresh, K. 2006. Juvenile Pacific Salmon and the Nearshore Ecosystem of Puget Sound. Seattle: Puget Sound Nearshore Ecosystem Restoration Program.

Frest, T.J., and E.J. Johannes. 1995. Freshwater Molluscs of the Upper Sacramento System, California with Particular Reference to the Cantara Spill. Sacramento, California: State of California, Department of Fish and Game.

Frisch, A.J., and T.A. Anderson. 2000. The Response of Coral Trout (*Plectropomus Leopardus*) to Capture, Handling and Transport and Shallow Water Stress. *Fish Physiology and Biochemistry* 23(1): 23-24.

Fritioff, A., and M. Greger. 2003. Aquatic and terrestrial plant species with potential to remove heavy metals from stormwater. *International Journal of Phytoremediation* 5(3): 211-224.

Fuchs, S.A., S.G. Hinch, and E. Mellina. 2003. Effects of streamside logging on stream macroinvertebrate communities and habitat in the sub-boreal forests of British Columbia, Canada. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 33(8): 1408-1415.

Gagnaire, B., H. Frouin, K. Moreau, H. Thomas-Guyon, and T. Renault. 2006. Effects of temperature and salinity on haemocyte activities of the Pacific oyster, *Crassostrea gigas* (Thunberg). *Fish & Shellfish Immunology* 20(4): 536-547.

Gallardo, A.H., and A. Marui. 2006. Submarine groundwater discharge: an outlook of recent advances and current knowledge. *Geo-Marine Letters* 26(2): 102-113.

Gardner, F. 1981. Washington Coastal Areas of Major Biological Significance. Olympia, Washington: Washington State Department of Ecology, Baseline Studies Program.

Gardner, S.C., C.E. Grue, W.W. Major, and L.L. Conquest. 2001. Aquatic invertebrate communities associated with purple loosestrife (*Lythrum salicaria*), cattail (*Typha latifolia*), and bulrush (*Scirpus acutus*) in central Washington, USA. *Wetlands* 21(4): 593-601.

Garrison, K.J., and B.S. Miller. 1982. Review of the Early Life History of Puget Sound Fishes. Seattle, Washington: University of Washington, Fish. Res. Inst.

Gay, P., G. Vellidis, and J.J. Delfino. 2006. The Attenuation of Atrazine and Its Major Degradation Products in a Restored Riparian Buffer. *Transactions of the ASABE* 49(5): 1323-1339.

Gelfenbaum, G., T. Mumford, J. Brennan, H. Case, M. Dethier, K. Fresh, F. Goetz, M. van Heeswijk, T.M. Leschine, M. Logsdon, D. Myers, J. Newton, H. Shipman, C.A. Simenstad, C. Tanner, and D. Woodson. 2006. Coastal Habitats in Puget Sound: A Research Plan in Support of the Puget Sound Nearshore Partnership. Available at: http://www.pugetsoundnearshore.org/techical_papers/coastal_habitats.pdf.

Gendron, A. 2005. Water table dynamics at Lowell Point, Camano Island, Washington: a designer study for Puget Sound. University of Washington, Seattle. 30 pp.

Gerstel, W.J., and J.F. Brown. 2006. Alternative Shoreline Stabilization Evaluation Project; Final Report September 15, 2006. Olympia, Washington: Puget Sound Action Team.

Giorgi, A.E. 1981. The Environmental Biology of the Embryos, Egg Masses and Nesting Sites of the Lingcod, *Ophiodon Elongatus*. Seattle, Washington: National Marine Fisheries Service, NWAFC Proc. Rept.

Gipson, R.D., and W.A. Hubert. 1993. Spawning-site selection by kokanee along the shoreline of Flaming Gorge Reservoir, Wyoming-Utah. *North American Journal of Fisheries Management* 13(3): 475-482.

Goetz, F.A., E. Jeanes, E. Beamer, G. Hart, C. Morello, M. Camby, C. Ebel, E. Conner, and H. Berge. 2004. Bull Trout in the Nearshore (Preliminary Draft). U.S. Army Corps of Engineers, Seattle District.

Gonor, J., J. Sedell, and P. Benner. 1988. What we know about large trees in estuaries, in the sea, and on coastal beaches. In *From the forest to the sea: a story of fallen trees*, edited by C. Maser, R. Tarrant, J. Trappe and J. Franklin. Portland, Oregon.: U.S.D.A, Forest Service, Pacific Northwest Research Station. pp. 83-112.

Gotthardt, T. 2006. Longfin Smelt. Anchorage, Alaska: Alaska Natural Heritage Program.

Grant, J., and B. Thorpe. 1991. Effects of Suspended Sediment on Growth, Respiration, and Excretion of the Soft-Shell Clam (*Mya-Arenaria*). *Canadian Journal of Fisheries and Aquatic Sciences* 48(7): 1285-1292.

Grant, J., C.T. Enright, and A. Griswold. 1990. Resuspension and Growth of *Ostrea-Edulis* - a Field Experiment. *Marine Biology* 104(1): 51-59.

Gray, J.S., R.S.S. Wu, and Y.Y. Or. 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology-Progress Series* 238: 249-279.

Gregory, R.S. 1993. Effect of Turbidity on the Predator Avoidance-Behavior of Juvenile Chinook Salmon (*Oncorhynchus-Tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 50(2): 241-246.

Gregory, R.S., and C.D. Levings. 1998. Turbidity reduces predation on migrating juvenile Pacific salmon. *Transactions of the American Fisheries Society* 127(2): 275-285.

Groberg, W.J., R.H. Mccoy, K.S. Pilcher, and J.L. Fryer. 1978. Relation of Water Temperature to Infections of Coho Salmon (*Oncorhynchus-Kisutch*), Chinook Salmon (*O-Tshawytscha*), and Steelhead Trout (*Salmo-Gairdneri*) with *Aeromonas-Salmonicida* and *Aeromanas-Hydrophila*. *Journal of the Fisheries Research Board of Canada* 35(1): 1-7.

Groot, C., and L. Margolis. 1991. Pacific Salmon Life Histories. Vancouver, British Columbia: University of British Columbia Press.

Guidetti, P. 2004. Fish assemblages associated with coastal defense structures in south-western Italy (Mediterranean Sea). *Journal of the Marine Biological Association of the United Kingdom* 84(3): 669-670.

Guidetti, P., L. Verginella, C. Viva, R. Odorico, and F. Boero. 2005. Protection Effects on Fish Assemblages, and Comparison of Two Visual-Census Techniques in Shallow Artificial Rocky Habitats in the Northern Adriatic Sea. Journal of the Marine Biological Association of the United Kingdom 85(2): 247-255.

Guidetti, P., S. Bussotti, and F. Boero. 2005. Evaluating the effects of protection on fish predators and sea urchins in shallow artificial rocky habitats: a case study in the northern Adriatic Sea. *Marine Environmental Research* 59(4): 333-348.

Gustafson, R.G., T.C. Wainwright, G.A. Winans, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1997. Status Review of Sockeye Salmon from Washington and Oregon, NOAA Technical Memorandum. Seattle, Washington: Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Haas, M.E., C.A. Simenstad, J.R. Cordell, D.A. Beauchamp, and B.S. Miller. 2002. Effects of large overwater structures on epibenthic juvenile salmon prey assemblages in Puget. Puget Sound, Washington.

Hagerthey, S.E., and W.C. Kerfoot. 2005. Spatial variation in groundwater-related resource supply influences freshwater benthic algal assemblage composition. *Journal of the North American Benthological Society* 24(4): 807-819.

Haldorson, L., and L.J. Richards. 1986. Post-larval copper rockfish in the Strait of Georgia: Habitat use, feeding, and growth in the first year. In *Proceedings International Rockfish Symposium*, Anchorage, Alaska: Alaska Sea Grant College Program. Cited in NRC 2001. pp. 129-141.

Hall, M.O., M.J. Durako, J.W. Fourqurean, and J.C. Zieman. 1999. Decadal changes in seagrass distribution and abundance in Florida Bay. *Estuaries* 22(2B): 445-459.

Hallock, M., and P.E. Mongillo. 1998. Washington State Status Report for the Pygmy Whitefish. Olympia, Washington: Washington Department of Fish and Wildlife, Fish Management Program, Freshwater Resource Division.

Hammer, T.R. 1972. Stream Channel Enlargement Due to Urbanization. *Water Resources Research* 8: 1530-1541.

Hansen, J.A., J.C.A. Marr, J. Lipton, D. Cacela, and H.L. Bergman. 1999a. Differences in neurobehavioral responses of chinook salmon (*Oncorhynchus tshawytascha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper and cobalt: behavioral avoidance. *Environ. Toxicol. Chem.* 18:1972-1978. (As cited in Stratus 2005b).

Hansen, J.A., J.D. Rose, R.A. Jenkins, K.G. Gerow, and H.L. Bergman. 1999b. Chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper: neurophysiological and histological effects on the olfactory system. *Environ. Toxicol. and Chem.* 18:1979-1991. (As cited in Stratus 2005b).

Hansen, J.A., P.G. Welsh, J. Lipton, D. Cacela, and A.D. Dailey. 2002. Relative sensitivity of bull trout (*Salvelinus confluentus*) and rainbow trout (*Oncorhynchus mykiss*) to acute exposures of cadmium and zinc. *Environ. Toxicol. Chem.* 21:67-75. (As cited in Stratus 2005b).

Hard, J.J., R.G. Kope, W.S. Grant, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1996. Status Review of Pink Salmon from Washington, Oregon, California. Seattle, Washington: Northwest Fisheries Science Center.

Hardyniec, S., and S. Skeen. 2005. Pile Driving and Barotraumas Effects. No. 1941. *Journal of Transportation Research Board*.

Harrahy, L.N.M., C.B. Schreck, and A.G. Maule. 2001. Antibody-producing cells correlated to body weight in juvenile chinook salmon (*Oncorhynchus tshawytscha*) acclimated to optimal and elevated temperatures. *Fish & Shellfish Immunology* 11(8): 653-659.

Harris, C. 1974. The Geographical Distribution and Habitat of the Olympic Mudminnow, (*Novumbra hubbsi*). Master's Thesis, University of Washington, Seattle, Washington.

Hart, C.W., and S.L.H. Fuller. 1974. *Pollution ecology of freshwater invertebrates*. New York: Academic Press.

Hart, J.L. 1973. Pacific Fishes of Canada. *Fish. Res. Board Can. Bull* 180: 730.

Harvey, B.C., R.J. Nakamoto, and J.L. White. 1999. Influence of large woody debris and a bankfull flood on movement of adult resident coastal cutthroat trout (*Oncorhynchus clarki*) during fall and winter. *Canadian Journal of Fisheries and Aquatic Sciences* 56(11): 2161-2166.

Hasler, A.D., A.T. Scholz, and R.M. Horrall. 1978. Olfactory Imprinting and Homing in Salmon. *American Scientist* 66(3): 347-355.

Hasler, A.D., and A.T. Scholz. 1983. Olfactory imprinting and homing in salmon. In *Zoo-physiol. 14*, New York: Springer-Verlag. pp. 3-38.

Hastings, M.C. 1995. Physical Effects of Noise on Fishes. *Proceedings of INTER-NOISE 95, The 1995 International Congress on Noise Control Engineering*. pp. 979–984.

Hastings, M.C., A.N. Popper, J.J. Finneran, and P.J. Lanford. 1996. Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish *Astronotus ocellatus*. *Journal of the Acoustical Society of America* 99(3): 1759-1766.

Hastings, M.C., and A.N. Popper. 2005. Effects of Sound of Fish. Prepared for California Department of Transportation by Jones and Stokes, Sacramento, California.

Hawkins, A.D. 1986. Underwater Sound and Fish Behavior. In *The Behavior of Teleost Fishes*, edited by T.J. Pitcher. Maryland: Johns Hopkins University Press. pp. 114-151.

Hawkins, A.D., and A.D.F. Johnstone. 1978. The Hearing of the Atlantic Salmon, *Salmo Salar*. *Journal of Fish Biology* 13: 655-673.

Hawkins, C.P., and J.R. Sedell. 1981. *Longitudinal and Seasonal Changes in Functional Organization of Macroinvertebrate Communities in Four Oregon Streams. Ecology* 62(2): 387-397.

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of Fish and Salamanders in Relation to Riparian Canopy and Physical Habitat in Streams of the Northwestern United-States. *Canadian Journal of Fisheries and Aquatic Sciences* 40(8): 1173-1185.

Hazelwood, R.A., and J. Connelly. 2005. Estimation of Underwater Noise - a Simplified Method. *International Journal of the Society of Underwater Technology* 26(3): 51-57.

Healey, M.C. 1982. Juvenile Pacific Salmon in Estuaries: The Life Support System. In *Estuarine Comparisons*, edited by V.S. Kennedy. New York, New York: Academic Press. pp. 315-341.

Healey, M.C. 1991. Life History of Chinook Salmon (*Oncorhynchus tshawytscha*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British: University of British Columbia Press. pp. 311-394230.

Heard, W.R. 1991. Life History of Pink Salmon (*Oncorhynchus gorbuscha*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 120-230.

Heath, A.G., and G.M. Hughes. 1973. Cardiovascular and Respiratory Changes During Heat Stress in Rainbow-Trout (*Salmo-Gairdneri*). *Journal of Experimental Biology* 59(2): 323-338.

Heathershaw, A.D., P.D. Ward, and A.M. David. 2001. The Environmental Impact of Underwater Sound. *Proceedings of the Institute of Acoustics*, pp. 1-13.

Henning, J. 2004. An evaluation of fish and amphibian use of restored and natural floodplain wetlands. Olympia, Washington: Washington Department of Fish and Wildlife.

Henning, J.A., R.E. Gresswell, and I.A. Fleming. 2006. Juvenile salmonid use of freshwater emergent wetlands in the floodplain and its implications for conservation management. *North American Journal of Fisheries Management*. 26 (2): 367-376. May 2006.

Hernandez-Miranda, E., A.T. Palma, and F.P. Ojeda. 2003. Larval fish assemblages in nearshore coastal waters off central Chile: temporal and spatial patterns. *Estuarine Coastal and Shelf Science* 56(5-6): 1075-1092.

Herrera. 2004. As-Built Considerations and Monitoring Recommendations: Hoh River Erosion Control Project U.S. Highway 101 at Milepost 174.4. Seattle: Washington State Department of Transportation.

Herrera. 2005. Marine shoreline sediment survey and assessment, Thurston County, Washington. Prepared for Thurston Regional Planning Council, Olympia, Washington, by Herrera Environmental Consultants, Inc., Seattle, Washington.

Herrera. 2006. Landslide Disposal Site Identification: Sound Transit Everett-to-Seattle Commuter Rail. Seattle, Washington: Sound Transit.

Herrera. 2007a. Marinas and Shipping/Ferry Terminals White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. July 2007 Draft.

Herrera. 2007b. Water Quality Statistical and Pollutant Loading Analysis: Green-Duwamish Watershed Water Quality Assessment. Prepared for King County Department of Natural Resources and Parks by Herrera Environmental Consultants, Inc. (Herrera), Seattle, Washington.

Herrera. 2007c. Habitat Modifications white paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. August 2007 Draft.

Hershey, A.E., and G.A. Lamberti. 1992. Stream macroinvertebrate communities. In *Watershed Management – Balancing sustainability and environmental change*, edited by R.J. Naiman. New York: Springer-Verlag.

Hetrick, N.J., M.A. Brusven, T.C. Bjornn, R.M. Keith, and W.R. Meehan. 1998. Effects of canopy removal on invertebrates and diet of juvenile coho salmon in a small stream in southeast Alaska. *Transactions of the American Fisheries Society* 127(6): 876-888.

Higgins, K., P. Schlenger, J. Small, D. Hennessy, and J. Hall. 2005. Spatial relationships between beneficial and detrimental nearshore habitat parameters in WRIA 9 and the City of Seattle. *Proceedings of the 2005 Puget Sound Georgia Basin Research Conference*.

Hilderbrand, R.H., A.D. Lemly, C.A. Dolloff, and K.L. Harpster. 1997. Effects of large woody debris placement on stream channels and benthic macroinvertebrates. *Canadian Journal of Fisheries and Aquatic Sciences* 54(4): 931-939.

Hinchey, E.K., L.C. Schaffner, C.C. Hoar, B.W. Vogt, and L.P. Batte. 2006. Responses of estuarine benthic invertebrates to sediment burial: the importance of mobility and adaptation. *Hydrobiologia* 556: 85-98.

Hinton, S.A., R.L. Emmett, and G.T. McCabe. 1992. Benthic invertebrates, demersal fishes and sediment characteristics at and adjacent to ocean dredge material Disposal Site F, offshore from the Columbia River, June 1989-1990.

Hirose, T., and K. Kawaguchi. 1998. Sediment size composition as an important factor in the selection of spawning site by the Japanese surf smelt Hypomesus japonicus. *Fisheries Science* 64(6): 995-996.

Hjort, R.C., P.L. Hulett, L.D. LaBolle, and H.W. Li. 1983. Fish and invertebrates of revetments and other habitats in the Willamette River, Oregon. Vicksburg, Mississippi: Waterways Experiment Station, Army Corps of Engineers.

Hoffmann, A. 2000. The association of the stream caddisfly Lasiocephala basalis (KOL.) (*Trichoptera : Lepidostomatidae*) with wood. *International Review of Hydrobiology* 85(1): 79-93.

Holliman, F.M., J.B. Reynolds, and T.J. Kwak. 2003. Electroshock-Induced Injury and Mortality in the Spotfin Chub, a Threatened Minnow. *North American Journal of Fish Management* 23(3): 962-966.

Hood, W.G. 2006. A conceptual model of depositional, rather than erosional, tidal channel development in the rapidly prograding Skagit River Delta (Washington, USA). *Earth Surface Processes and Landforms* 31(14): 1824-1838.

Horner, R.A. 1998. Harmful Algal Blooms in Puget Sound: General Perspective. Puget Sound Research: From Basic Science to Resource Management, Seattle, Washington, March 12 and 13, 1998, pp. 809-811.

Hoshikawa, H., Y. Sakai, and A. Kijima. 1998. Growth Characteristics of the Hybrid between Pinto Abalone, *Haliotis Kamtschatkana* Jonas, and Exo Abalone, *H-Discus* Hannai Ino, under High and Low Temperature. *Journal of Shellfish Research* 17(3): 673-677.

Howell, M.D., M.D. Romano, and T.A. Rien. 2001. Draft Outmigration Timing and Distribution of Larval Eulachon, *Thaleichthys Pacificus*, in the Lower Columbia River, Spring 2001. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

Hughes, G.W., and A.E. Peden. 1989. Status of the Umatilla Dace, *Rhinichthys Umatilla*, in Canada. *Canadian Field-Naturalist* 103(2): 193-200.

Hunt, R.J., M. Strand, and J.F. Walker. 2006. Measuring groundwater-surface water interaction and its effect on wetland stream benthic productivity, Trout Lake watershed, northern Wisconsin, USA. *Journal of Hydrology* 320(3-4): 370-384.

Illingworth and Rodkin, I. 2001. Noise and Vibration Measurements Associated with the Pile Installation Demonstration Project for the San Francisco-Oakland Bay Bridge East Span, Final Data Report.

Ingersoll, C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyuer, C.E. Henke, N.E. Kemble, D.R. Mount, and R.G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the maphipod *Hyalella axteca* and midge *Chrionomus riparius*. *Journal of Great Lakes Research* 22(3): 602-623.

Ingram, J. 1982. Migration of creosote and its components from treated piling sections in a marine environment. *Proceedings of the Annual Meeting of American Wood-Preservers' Association* 78: 120-128.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber Harvest Impacts on Small Headwater Stream Channels in the Coast Ranges of Washington. *Journal of the American Water Resources Association* 37(6): 1533-1549.

Jennings, M.J., M.A. Bozek, G.R. Hatzenbeler, E.E. Emmons, and M.D. Staggs. 1999. Cumulative effects of incremental shoreline habitat modifications on fish assemblages in north temperate lakes. *North American Journal of Fisheries Management* 19: 18-27.

Johannes, R.E. 1980. The Ecological Significance of the Submarine Discharge of Groundwater. *Marine Ecology-Progress Series* 3(4): 365-373.

Johnson, L.L., T.K. Collier, and J.E. Stein. 2002. An Analysis in Support of Sediment Quality Thresholds for Polycyclic Aromatic Hydrocarbons (PAHs) to Protect Estuarine Fish. *Aquatic Conservation: Marine and Freshwater Ecosystems* 12: 517-538.

Johnson, O.W., M.H. Ruckelshaus, W.S. Grant, F.W. Waknitz, A.M. Garrett, G.J. Bryant, K. Neely, and J.J. Hard. 1999. Status Review of Coastal Cutthroat Trout from Washington, Oregon, and California. Seattle, Washington: Northwest Fisheries Science Center.

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California, NOAA Technical Memorandum Seattle, Washington: U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Johnson, S.L., and J.A. Jones. 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* 57(S2): 30-39.

Jones & Stokes. 2006. Overwater Structures and Non Structural Piling (White Paper). Prepared by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C., and R2 Consultants for the Washington Department of Fish and Wildlife, Olympia, Washington.

Kahler, T.H., M. Grassley, and D. Beauchamp. 2000. A summary of the effects of bulkheads, piers, and other artificial structures and Shorezone Development on ESA-listed salmonids in lakes. Prepared for the City of Bellevue by the Watershed Company, Kirkland, Washington. July 2000.

Kaller, M.D., and W.E. Kelso. 2007. Association of macroinvertebrate assemblages with dissolved oxygen concentration and wood surface area in selected subtropical streams of the southeastern USA. *Aquatic Ecology* V41(1): 95-110.

Kalmijn, A.J. 1988. Hydrodynamic and Acoustic Field Detection. In *Sensory Biology of Aquatic Animals*, edited by J. Atema, R.R. Fay, A.N. Popper and W.N. Tavolga. New York: Springer-Verlag. pp. 83-130.

Kang, S.M., J.J. Morrell, J. Simonsen, and S.T. Lebow. 2003. Creosote Movement from Treated Wood Immersed in Freshwater: Initial PAH Migration. *International Research Group on Wood Preservation 34th Annual Meeting*, Brisbane, Australia, May 18-25, 2003.

Karp, C.A., and G. Mueller. 2002. Razorback sucker movements and habitat use in the San Juan River inflow, Lake Powell, Utah, 1995-1997. *Western North American Naturalist* 62(1): 106-111.

Karr, J.R. 1991. Biological Integrity - a Long-Neglected Aspect of Water-Resource Management. *Ecological Applications* 1(1): 66-84.

Kawamata, S. 2001. Adaptive mechanical tolerance and dislodgement velocity of the kelp *Laminaria japonica* in wave-induced water motion. *Marine Ecology-Progress Series* 211: 89-104.

Kelley, D.S., J.A. Baross, and J.R. Delaney. 2002. Volcanoes, fluids, and life at mid-ocean ridge spreading centers. *Annual Review of Earth and Planetary Sciences* 30: 385-491.

Kelsey, K.A., and S.D. West. 1998. Riparian wildlife. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Kendall, A.W., and W.H. Lenarz. 1986. Status of Early Life History Studies of Northeast Pacific Rockfishes. In *Proceedings International Rockfish Symposium*, Anchorage, Alaska: Alaska Sea Grant College Program. pp. 99-128.

Kiel, S. 2006. New records and species of molluscs from tertiary cold-seep carbonates in Washington State, USA. *Journal of Paleontology* 80(1): 121-137.

King, D.M., N.J. Cooper, J.C. Morfett, and D.J. Pope. 2000. Application of offshore breakwaters to the UK: A case study at Elmer Beach. *Journal of Coastal Research* 16(1): 172-187.

Kiorboe, T., F. Mohlenberg, and O. Nohr. 1981. Effect of Suspended Bottom Material on Growth and Energetics in Mytilus-Edulis. *Marine Biology* 61(4): 283-288.

Knudsen, F.R., P.S. Enger, and O. Sand. 1992. Awareness reactions and avoidance responses to sound in juvenile Atlantic salmon, *Salmo salar*. *Journal of Fish Biology* 40: 523-534.

Knutson, K.L., and V.L. Naef. 1997. Management Recommendations for Washington's Priority Habitats. *Riparian*. Olympia, Washington: Washington State Department of Fish and Wildlife.

Koch, E.M. 2001. Beyond light: Physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries* 24(1): 1-17.

Komar, P.D. 1998. *Beach Processes and Sedimentation*. Princeton Prentice Hall.

Kondolf, G.M., and M.G. Wolman. 1993. The Sizes of Salmonid Spawning Gravels. *Water Resources Research* 29(7): 2275-2285.

Kondolf, G.M., and R.R. Curry. 1986. Channel erosion along the Carmel River, Monterey County, California. *Earth Surface Processes and Landforms* 11: 307-319.

Konrad, C.P. 2000. The frequency and extent of hydrologic disturbances in streams in the Puget Lowland, Washington. Ph.D. dissertation Thesis, University of Washington, Seattle, Washington, 212 pp.

Kramer, D.E., and V.M. O’Connell. 1995. Guide to Northeast Pacific Rockfishes. *Alaska Sea Grant Marine Advisory Bulletin* No. 25.

Krueger, K., P. Chapman, M. Hallock, and T. Quinn. 2007. Some Effects of Suction Dredge Placer Mining on the Short-Term Survival of Freshwater Mussels in Washington Final Report (Draft) USFSW HPA/HCP Grant E-29-HP. Olympia, Washington: Washington Department of Fish and Wildlife Habitat and Fisheries Programs.

Kynard, B., E. Parker, and T. Parker. 2005. Behavior of early life intervals of Klamath River green sturgeon, *Acipenser medirostris* with a note on body color. *Environmental Biology of Fishes* 72(1): 85-97.

Lagasse, P.F., L.W. Zevenbergen, J.D. Schall, and P.E. Clopper. 2001. Bridge scour and stream instability countermeasures. Washington DC: Federal Highway Administration, U.S. Department of Transportation.

Lake, R.G., and S.G. Hinch. 1999. Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 56(5): 862-867.

Lamb, M.P., E. D'Asaro, and J.D. Parsons. 2004. Turbulent structure of high-density suspensions formed under waves. *Journal of Geophysical Research-Oceans* 109(C12).

Lamberti, G.A., S.V. Gregory, L.R. Ashkenas, A.D. Steinman, and C.D. McIntire. 1989. Productive Capacity of Periphyton as a Determinant of Plant-Herbivore Interactions in Streams. *Ecology* 70(6): 1840-1856.

Lane, E.W. 1955. The importance of fluvial morphology in hydraulic engineering. *Proceedings of the American Society of Civil Engineers* 81: 745-761.

Langer, O.E., B.G. Shepherd, and P.R. Vroom. 1977. Biology of the Nass River Eulachon (*Thaleichthys Pacificus)*. Department of Fisheries and Environment Canada, Fisheries and Marine Service.

Laroche, W.A., and S.L. Richardson. 1981. Development of larvae and juveniles of the rockfishes *Sebastes entomelas* and *S. zacentrus* (Family Scorpaenidae) and occurrence off Oregon, with notes on head spines of *S. mystinus*, *S. flavidus*, and *S. melanops*. *Fishery Bulletin* 79: 231-256.

Larsen, E.M., E. Rodrick, and R. Milner. 1995. Management Recommendations for Washington's Priority Species. Volume I: Invertebrates. Olympia, Washington: Washington Department of Wildlife.

Laughlin, J. 2004. Underwater Sound Levels Associated with the Construction of the SR 240 Bridge on the Yakima River at Richland. Seattle, Washington: Washington State Department of Transportation, Office of Air Quality and Noise.

Laughlin, J. 2005. Underwater Sound Levels Associated with the Restoration of the Friday Harbor Ferry Terminal. Seattle, Washington: Washington State Department of Transportation.

Laughlin, J. 2006. Underwater Sound Levels Associated with Pile Driving at the Cape Disappointment Boat Launch Facility, Wave Barrier Project. Olympia, Washington: Washington State Department of Transportation.

Lawler, D.M. 2005. Turbidity and Nephelometry. In *Encyclopedia of Analytical Science*, edited by P.J. Worsfold, A. Townshend and C.F. Poole. Elsevier. pp. 343-351.

Leary, R.F., and F.W. Allendorf. 1997. Genetic confirmation of sympatric bull trout and Dolly Varden in western Washington. *Transactions of the American Fisheries Society* 126: 715-20.

Lebow, S.T., and M. Tippie. 2001. Guide for Minimizing the Effect of Preservative-Treated Wood on Sensitive Environments. Madison, Wisconsin: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory.

Lemieux, J.P. 2004. Proceedings of the DFO/PSAT sponsored marine riparian experts workshop. Tsawwassen, British Columbia: Canadian Manuscript Report of Fisheries and Aquatic Sciences.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. *Fluvial Processes in Geomorphology*. New York: Freeman.

Leopold, L.B., R. Huppman, and A. Miller. 2005. Geomorphic Effects of Urbanization in Forty-One Years of Observation. *Proceedings of the American Philosophical Society* 149: 349-371.

Levings, C.D., and G. Jamieson. 2001. Marine and estuarine riparian habitats and their role in coastal ecosystems, Pacific region. Canadian Science Advisory Secretariat.

Lewis, A.F.J., M.D. McGurk, and M.G. Galesloot. 2002. Alcan's Kemano River Eulachon (*Thaleichthys Pacificus*) Monitoring Program 1988-1998. Kitimat, British Columbia: Consultant's report prepared by Ecofish Research Ltd. for Alcan Primary Metal Ltd.

Li, H.W., C.B. Schreck, and R.A. Tubb. 1984. Comparison of Habitat near Spur Dikes, Continuous Revetments, and Natural Banks for Larval, Juvenile, and Adult fishers of the Willamette River. Corvallis, Oregon: USGS: Water Resources Research Institute

Liang, H.Z., M.P. Lamb, and J.D. Parsons. 2007. Formation of a sandy near-bed transport layer from a fine-grained bed under oscillatory flow. *Journal of Geophysical Research-Oceans* 112(C2).

Liknes, G.A., and P.J. Graham. 1988. Westslope Cutthroat Trout in Montana: Life History, Status and Management. In *Status and Management of Cutthroat Trout*, edited by R.E. Gresswell. Bethesda, Maryland: American Fisheries Society. pp. 53-60.

Lisle, T.E., and J. Lewis. 1992. Effects of Sediment Transport on Survival of Salmonid Embryos in a Natural Stream - a Simulation Approach. *Canadian Journal of Fisheries and Aquatic Sciences* 49(11): 2337-2344.

Livingston, P.A. 1991. Food Habits and Population Level Consumption of Groundfish. In *Groundfish Food Habits and Predation on Commercially Important Prey Species in the Eastern Bering Sea from 1984 to 1986*, edited by P.A. Livingston. Seattle, Washington: U.S. Dept. Commerce, NOAA Tech. Memo., NMFS F/NWC-207. pp. 9-88.

Lloyd, D.S. 1987. Turbidity as a Water Quality Standard for Salmonid Habitats in Alaska. *North American Journal of Fisheries Management* 7: 34-45.

Long, E.R., and L.G. Morgan. 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. Seattle, Washington: National Oceanic and Atmospheric Administration.

Lorang, M.S., J.A. Stanford, F.R. Hauer, and J.H. Jourdonnais. 1993. Dissipative and Reflective Beaches in a Large Lake and the Physical Effects of Lake Level Regulation. *Ocean & Coastal Management* 19(3): 263-287.

Losada, I.J., J.L. Lara, E.D. Christensen, and N. Garcia. 2005. Modeling of velocity and turbulence fields around and within low-crested rubble-mound breakwaters. *Coastal Engineering* 52(10-11): 887-913.

Love, M.S., and A. York. 2006. The relationships between fish assemblages and the amount of bottom horizontal beam exposed at California oil platforms: fish habitat preferences at man-made platforms and (by inference) at natural reefs. *Fishery Bulletin* 104(4): 542-549.

Love, M.S., D.M. Schroeder, W. Lenarz, A. MacCall, A.S. Bull, and L. Thorsteinson. 2006. Potential use of offshore marine structures in rebuilding an overfished rockfish species, bocaccio (*Sebastes paucispinis*). *Fishery Bulletin* 104(3): 383-390.

Love, M.S., P. Morris, M. McCrae, and R. Collins. 1990. Life History Aspects of 19 Rockfish Species (Scorpaenidae: Sebastes) from the Southern California Bight. National Oceanic and Atmospheric Administration.

Lunz, J. 1985. An analysis of available information concerning the entrainment of oyster larvae during hydraulic cutterhead dredging operations with commentary on the reasonableness of seasonally restricting dredging windows. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

Lussier, S.M., J.H. Gentile, and J. Walker. 1985. Acute and chronic effects of heavy metals and cyanide on *Mysidopsis bahia* (Crustacea:Mysidacea). *Aquat. Toxicol.* 7(1-2):25-35. (As cited in Stratus 2005b).

MacBroom, J.G. 1998. *The River Book–The Nature of Streams in Glaciated Terrains*. Hartford, Conn.: Connecticut Department of Environmental Protection.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 39(1): 20-31.

MacDonald, J.S., E.A. MacIsaac, and H.E. Herunter. 2003. The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 33(8): 1371-1382.

MacDonald, K.B., D. Simpson, B. Paulsen, J. Cox, and J. Gendron. 1994. Shoreline armoring effects on the physical coastal processes in Puget Sound, Washington. Olympia, Washington: Shorelands and Coastal Zone Management Program, Washington Department of Ecology.

Marine, K.R., and J.J. Cech. 2004. Effects of high water temperature on growth, smoltification, and predator avoidance in Juvenile Sacramento River Chinook salmon. *North American Journal of Fisheries Management* 24(1): 198-210.

Marsden, J.E., and M.A. Chotkowski. 2001. Lake trout spawning on artificial reefs and the effect of zebra mussels: Fatal attraction? *Journal of Great Lakes Research* 27(1): 33-43.

Martens, D.W., and J.A. Servizi. 1993. Suspended sediment particles inside gills and spleens of juvenile Pacific salmon (*Oncorhynchus* spp.) *Canadian Journal of Fisheries and Aquatic Sciences* 50: 586-590.

Maser, C., and J. Sedell. 1994. *From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans*. Delray Beach, Florida: St. Lucie Press.

Matthews, K.R. 1990. An experimental study of the habitat preferences and movement patterns of copper, quillback, and brown rockfishes (*Sebastes* spp.). *Environmental Biology of Fishes* 29: 161-178.

May, C.W. 1998. Assessment of the Cumulative Effects of Urbanization on Small Streams in the Puget Sound Lowland Ecoregion: Implications for Salmonid Resource Management. Seattle, Washington: University of Washington.

May, C.W. 2003. Stream-Riparian Ecosystems in the Puget Sound Lowland Eco-Region: A Review of Best Available Science. Watershed Ecology LLC.

Mayer, P., S. Reynolds, T.J. Canfield, and M.D. McCutchen. 2005. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. EPA/600/R-05/118. Ada, Oklahoma: U.S. Environmental Protection Agency.

Mazur, M.M., and D.A. Beauchamp. 2003. A comparison of visual prey detection among species of piscivorous salmonids: effects of light and low turbidities. *Environmental Biology of Fishes* 67(4): 397-405.

MBC Applied Environmental Sciences. 1987. Ecology of Important Fisheries Species Offshore California. MMS 86-0093. Washington, D.C.: Pacific Outer Continental Shelf Region.

McCool, W.W., and J.D. Parsons. 2004. Sedimentation from buoyant fine-grained suspensions. *Continental Shelf Research* 24(10): 1129-1142.

McCormick, S.D., R.A. Cunjak, B. Dempson, M.F. O'Dea, and J.B. Carey. 1999. Temperature-related loss of smolt characteristics in Atlantic salmon (*Salmo salar*) in the wild. *Canadian Journal of Fisheries and Aquatic Sciences* 56(9): 1649-1658.

McCullough, M.C., S. Spalding, and D. Sturdevant. 2001. Summary of Technical Literature Examining the Physiological Effects of Temperature on Salmonids. EPA-910-D-01-005. Washington, D.C.: U.S. Environmental Protection Agency.

McFarlane, G.A., and R.J. Beamish. 1986. Biology and fishery of Pacific hake *Merluccius productus* in the Strait of Georgia. *Int. N. Pac. Fish. Comm. Bull.* 50: 365-392.

McKinnell, S., J.J. Pella, and M.L. Dahlberg. 1997. Population-specific aggregations of steelhead trout (*Oncorhynchus mykiss*) in the North Pacific Ocean. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 2368-2376.

McLeay, D.L., I. Birtwell, G. Hartman, and G. Ennis. 1987. Response of arctic grayling (*Thymallus arcticus*) to acute and prolonged exposure to Yukon placer mining sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 44: 658-673.

McMichael, G.A., L. Fritts, and T.N. Pearsons. 1998. Electrofishing Injury to Stream Salmonids; Injury Assessment at the Sample, Reach, and Stream Scales. *North American Journal of Fisheries Management* 18: 894-904.

Meador, J.P., J.E. Stein, W.L. Reichert, and U. Varansi. 1995. Bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms. *Reviews of Environmental Contamination Toxicology* 143: 79-165.

Meadows, G.A., S.D. Mackey, R.R. Goforth, D.M. Mickelson, T.B. Edil, J. Fuller, D.E. Guy, L.A. Meadows, and E. Brown. 2005. Cumulative habitat impacts of nearshore engineering. *Journal of Great Lakes Research* 31: 90-112.

Meals, K.O., and L.E. Miranda. 1991. Variability in abundance of age-0 centrarchids among littoral habitats of flood control reservoirs in Mississippi. *North American Journal of Fisheries Management* 11(3): 298-304.

Meier, A.H., and N.D. Horseman. 1977. Stimulation and depression of growth, fat storage, and gonad weight by daily stimulus in the teolost fish, *Tilapia aurea*. *Eighth Annual Meeting* 1977.

Melo, E., and R.T. Guza. 1991. Wave-Propagation in Jettied Entrance Channels. II: Observations. *Journal of Waterway Port Coastal and Ocean Engineering-ASCE* 117(5): 493-510.

Michael, H.A., A.E. Mulligan, and C.F. Harvey. 2005. Seasonal oscillations in water exchange between aquifers and the coastal ocean. *Nature* 436(7054): 1145-1148.

Michny, F., and R. Deibel. 1986. Sacramento River Chico Landing to Red Bluff Project 1985 juvenile salmon study. Sacramento, California: U.S. Army Corps of Engineers.

Millar, R.G., and M.C. Quick. 1998. Stable width and depth of gravel-bed rivers with cohesive banks. *Journal of Hydraulic Engineering-Asce* 124(10): 1005-1013.

Miller, B.S., C.A. Simenstad, and L.R. Moulton. 1976. Puget Sound Baseline Program: Nearshore Fish Survey, Annual Report July 1974-September 1975. Seattle, Washington: Fisheries Research Institute, University of Washington.

Miller, D.R., R.K.L. Emmett, and S.A. Hinton. 1990. A preliminary survey of benthic invertebrates in the vicinity of the Coos Bay Oregon, Navigation Channel: Coastal Zone and Estuarine Studies. Prepared for U.S. Army Corps of Engineers by Northwest Fisheries Science Center, National Marine Fisheries Center, Seattle, Washington.

Miller, M.C., I.N. McCave, and P.D. Komar. 1977. Threshold of Sediment Motion under Unidirectional Currents. *Sedimentology* 24(4): 507-527.

Miller, M.C., R.M. Thom, G.D. Williams, J.A. Southard, S.L. Blanton, and L.K. O'Rourke. 2001. Effects of Shoreline Hardening and Shoreline Protection Features on Fish Utilization and Behavior, Washaway Beach, Washington. PNNL-13635-(2). Richland, Washington: Pacific Northwest National Lab.

Mills, K.E., and M.S. Fonseca. 2003. Mortality and productivity of eelgrass *Zostera marina* under conditions of experimental burial with two sediment types. *Marine Ecology-Progress Series* 255: 127-134.

Minakawa, N., and G.F. Kraft. 2005. Homing behavior of juvenile coho salmon (*Oncorhynchus kisutch*) within an off-channel habitat. *Ecology of Freshwater Fish* 14(2): 197-201.

Mitson, R.B., and H.P. Knudsen. 2003. Causes and effects of underwater noise on fish abundance estimation. Aquatic Living Resources. 16 (3): 255-263. July 2003.

Mohlenberg, F., and T. Kiorboe. 1981. Growth and Energetics in Spisula-Subtruncata (Da Costa) and the Effect of Suspended Bottom Material. *Ophelia* 20(1): 79-90.

Mongillo, P.E., and M. Hallock. 1998. Washington State Status Report for the Margined Sculpin. Washington Department of Fish and Wildlife.

Mongillo, P.E., and M. Hallock. 1999. Washington State Status Report for the Olympic Mudminnow. Olympia, Washington: Washington Department of Fish and Wildlife.

Montgomery, D.R., E.M. Beamer, G.R. Pess, and T.P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. *Canadian Journal of Fisheries and Aquatic Science* 56: 377-387.

Montgomery, D.R., J.M. Buffington, N.P. Peterson, D. Schuett-Hames, and T.P. Quinn. 1996. Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. *Canadian Journal of Fisheries and Aquatic Sciences* 53: 1061-1070.

Moore, H.L., and H.W. Newman. 1956. Effects of Sound Waves on Young Salmon. Special Scientific Report Fisheries 172. U.S. Fish and Wildlife Service

Moore, K.A. 2004. Influence of seagrasses on water quality in shallow regions of the lower Chesapeake Bay. *Journal of Coastal Research* 20: 162-178.

Morley, S.A., P.S. Garcia, T.R. Bennett, and P. Roni. 2005. Juvenile Salmonid (*Oncorhynchus* Spp.) Use of Constructed and Natural Side Channels in Pacific Northwest Rivers. *Canadian Journal of Fishery Aquatic Science* 62(12): 2811-2821.

Moschella, P.S., M. Abbiati, P. Aberg, L. Airoldi, J.M. Anderson, F. Bacchiocchi, F. Bulleri, G.E. Dinesen, M. Frost, E. Gacia, L. Granhag, P.R. Jonsson, M.P. Satta, A. Sundelof, R.C. Thompson, and S.J. Hawkins. 2005. Low-crested coastal defense structures as artificial habitats for marine life: Using ecological criteria in design. *Coastal Engineering* 52(10-11): 1053-1071.

Moulton, L.L. 1977. Ecological Analysis of Fishes Inhabiting the Rocky Nearshore Regions of Northern Puget Sound. Ph.D. Dissertation Thesis, University of Washington.

Moyle, P.B., and J.J. Cech. 1988. *Fishes: An Introduction to Ichthyology*. Second Edition. New Jersey: Prentice Hall Publishing.

Mueller, G., P.C. Marsh, G. Knowles, and T. Wolters. 2000. Distribution, movements, and habitat use of razorback sucker (*Xyrauchen texanus*) in a lower Colorado River reservoir, Arizona-Nevada. *Western North American Naturalist* 60(2): 180-187.

Mulholland, P.J., J.D. Newbold, J.W. Elwood, L.A. Ferren, and J.R. Webster. 1985. Phosphorus Spiraling in a Woodland Stream - Seasonal-Variations.

Mulholland, R. 1984. Habitat suitability index models: hard clam. National Coastal Ecosystems Team. Division of Biological Services Research and Development, Fish and Wildlife Service, U.S. Dept. of the Interior.

Murphy, M. 1998. Primary productivity. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag. pp. 144-168.

Murphy, M.L., and W.R. Meehan. 1991. Stream Ecosystems. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 17-46.

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of Canopy Modification and Accumulated Sediment on Stream Communities. *Transactions of the American Fisheries Society* 110(4): 469-478.

Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, W.S. Grand, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California. Seattle, Washington: Northwest Fisheries Science Center.

Myers, R.D. 1993. Slope stabilization and erosion control using vegetation: A manual of practice for coastal property owners. Olympia, Washington: Shorelands and Coastal Zone Management program, Washington Department of Ecology.

Myrberg, A.A. 1972. Using Sound to Influence the Behavior of Free-Ranging Marine Animals. *Plenum* 2: 435-368.

Myrberg, A.A., and R.J. Riggio. 1985. Acoustically Mediated Individual Recognition by a Coral Reef Fish (*Pomacentrus Partitus*). *Animal Behavior* 33: 411-416.

Naiman, R.J., E.V. Balian, K.K. Bartz, R.E. Bilby, and J.J. Latterell. 2002. Dead wood dynamics in stream ecosystems. USDA Forest Service.

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian Ecology and Management in the Pacific Coastal Rain Forest. *Bioscience* 50(11): 996-1011.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Connor, P.L. Olson, and E.A. Steel. 1992. Fundamental Elements of Ecologically Healthy Watersheds in the Pacific Northwest Coastal Ecoregion. In *Watershed Management – Balancing sustainability and environmental change*, edited by R.J. Naiman. New York: Springer Verlag.

Nakamoto, R.J., and T.T. Kisanuki. 1995. Age and growth of Klamath River green sturgeon (*Acipenser medirostris*). Project #93-FP-13. Arcata, California: U.S. Forest Service.

Nakayama, T., M. Watanabe, K. Tanji, and T. Morioka. 2007. Effect of underground urban structures on eutrophic coastal environment. *Science of the Total Environment* 373(1): 270-288.

National Conservation Training Center. 2004. The Analytical Approach to Consultation. Lacey, Washington: Advanced Interagency Consultation - Regional Training curriculum.

Nebeker, A.V., S.T. Onjukka, D.G. Stevens, G.A. Chapman, and S.E. Dominguez. 1992. Effects of Low Dissolved-Oxygen on Survival, Growth and Reproduction of *Daphnia*, *Hyalella* and *Gammarus*. *Environmental Toxicology and Chemistry* 11(3): 373-379.

Nedeau, E., A.K. Smith, and J. Stone. 2005. Freshwater Mussels of the Pacific Northwest. Vancouver, Washington: U.S. Fish and Wildlife Service.

Nedwell, J., A. Martin, and N. Mansfield. 1993. Underwater tool noise: implications for hearing loss. In *Advances in Underwater Technology, Ocean Science and Offshore Engineering,* edited by Subtech '93. Dordrecht, the Netherlands: Kluwer Academic Publishers. pp. 267-275.

Nedwell, J., A. Turnpenny, J. Langworthy, and B. Edwards. 2003. Measurements of Underwater Noise During Piling at the Red Funnel Terminal, Southampton, and Observations of Its Effect on Caged Fish. Prepared by Subacoustics LTD, Hampshire, UK.

Nedwell, J., and B. Edwards. 2002. Measurements of Underwater Noise in the Arun River During Piling at County Wharf, Littlehampton. Prepared by Subacoustech, Hants, England.

Neitzel, D.A., and T.J. Frest. 1989. Survey of Columbia River Basin Streams for Giant Columbia River Spire Snail, *Fluminicola columbiana* and Great Columbia River Limpet, *Fisherola nuttalli*. Richland, Washington: Pacific Northwest Laboratory.

Neitzel, D.A., and T.J. Frest. 1990. Survey of Columbia River Basin Streams for Columbia Pebblesnail and Shortface Lanx. *Fisheries* 15(2): 2-3.

Nelson, D.R. 1965. Hearing and Acoustic Orientation in the Lemon Shark *Negaprion Brevirostris* (Poey), and Other Large Sharks. *Bulletin of Southern Californian Academic Sciences* 68(3): 131-137.

Nelson, D.R., R.H. Johnson, and L.G. Waldrop. 1969. Responses in Bahamian Sharks and Groupers to Low-Frequency, Pulsed Sounds. *Bulletin of Southern Californian Academic Sciences* 38: 131-137.

Nelson, T.A., A.V. Nelson, and M. Tjoelker. 2003. Seasonal and spatial patterns of "Green tides" (Ulvoid algal blooms) and related water quality parameters in the coastal waters of Washington state, USA. *Botanica Marina* 46(3): 263-275.

Newcomb, T.W., and T.A. Flagg. 1983. Some Effects of Mount St. Helens Volcanic Ash on Juvenile Salmon Smolts. *Marine Fisheries Review* 45(2): 8-12.

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of suspended sediment on aquatic ecosystems. *North American Journal of Fisheries Management* 11(1): 72-82.

Newcombe, C.P., and J.O.T. Jensen. 1996. Channel suspended sediment and fisheries: A synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(693-727).

Nightingale, B., and C. Simenstad. 2001a. Dredging Activities: Marine Issues. Prepared for the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Wetland Ecosystem Team, University of Washington, Seattle, Washington.

Nightingale, B., and C. Simenstad. 2001b. Marine Overwater Structures: Marine Issues. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Wetland Ecosystem Team, University of Washington, Seattle, Washington.

Nittrouer, C.A., and R.W. Sternberg. 1975. Fate of a Fine-Grained Dredge Spoils Deposit in a Tidal Channel of Puget-Sound, Washington. *Journal of Sedimentary Petrology* 45(1): 160-170.

NMFS. 1990. West Coast of North America Coastal and Ocean Zones Strategic Assessment: Data Atlas. Washington, District of Columbia: NOAA's National Marine Fisheries Service.

NMFS. 1998. Draft document - Non-fishing threats and water quality: A reference for EFH consultation. Seattle, Washington: National Marine Fisheries Service.

NMFS. 2004. Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation: SR 104 Edmonds Crossing Ferry Terminal Project, Snohomish County Prepared for U.S. Department of Transportation, Federal Highway Administration by NOAA's National Marine Fisheries Service March 25, 2004.

NMFS. 2006. Endangered Species Act - Section 7 Formal Consultation Biological and Conference Opinion and Magnuson Stevens Fishery Management Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho.

NMFS. 2007a. Species of Concern and Candidate Species: Pinto Abalone. Office of Protected Covered Species Paper - Invertebrates 5-10. National Marine Fisheries Service.

NMFS. 2007b. Rationale for the Use of 187 dB Sound Exposure Level for Pile Driving Impacts Threshold. Unpublished memorandum. Seattle, Washington: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Noggle, C.C. 1978. Behavioral, physiological and lethal effects of suspended sediment on juvenile salmonids. Master's Thesis, University of Washington, Seattle, Washington.

Nordstrom, K.F. 1992. *Estuarine Beaches: An Introduction of the Physical and Human Factors Affecting Use and Management of Beaches in Estuaries, Lagoons, Bays and Fjords*. New York: Elsevier Applied Science.

Northcote, T.G., and P.A. Larkin. 1989. The Fraser River: a major salmonine production system. *Canadian Special Publication of Fisheries and Aquatic Sciences* 106: 172-204.

NRC. 2001. Final Species Memorandum and Habitat Assessment in the King County HCP Planning Area. Volume 2: Marine Fish. Prepared for King County Wastewater Treatment Division by Natural Resources Consultants, Inc., Seattle, Washington. May 2001.

NRC. 2007. *Mitigating Shore Erosion along Sheltered Coasts*. National Research Council of the National Academies. Washington, DC: National Academies Press.

Nybakken, J.W., and M.D. Bertness. 2005. *Marine Biology: An Ecological Approach*. San Francisco: Pearson Benjamin Cummings.

O'Connell, V.M., and D.W. Carlile. 1993. Habitat-specific density of adult yelloweye rockfish *Sebastes ruberrimus* in the eastern Gulf of Alaska. *Fishery Bulletin* 91: 304-309.

Olla, B.L., M.W. Davis, and C.B. Schreck. 1995. Stress-Induced Impairment of Predator Evasion and Non-Predator Mortality in Pacific Salmon. *Aquaculture Research* 26(6): 393-398.

Olyphant, G.A., and S.W. Bennett. 1994. Contemporary and Historical Rates of Eolian Sand Transport in the Indiana Dunes Area of Southern Lake-Michigan. *Journal of Great Lakes Research* 20(1): 153-162.

Ona E., O.R. Godo, N.O Handegard, V. Hjellvik, R. Patel, and G. Pedersen. 2007. Silent research vessels are not quiet. *Journal of the Acoustical Society of America.* 121 (4): EL145-EL150. April 2007.

Pacific Biodiversity Institute. 2007. Homepage.   Available at: http://www.pacificbio.org/ (accessed July 2007).

Parametrix. 1996. Anacortes Ferry Terminal eelgrass, macroalgae, and macrofauna habitat survey report. Prepared for Sverdrup Civil, Inc. and the Washington State Department of Transportation by Parametrix and Battelle, Seattle, Washington.

Parkhill, K.L., and J.S. Gulliver. 2002. Effect of inorganic sediment on whole-stream productivity. *Hydrobiologia* 472(1-3): 5-17.

Paul, A.J., and J.M. Paul. 1998. Respiration Rate and Thermal Tolerances of Pinto Abalone *Haliotis Kamtschatkana*. *Journal of Shellfish Research* 17(3): 743-745.

Pauley, G.B., K.L. Oshima, and G.L. Thomas. 1988. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest)–Sea-run Cutthroat trout. U.S. Fish and Wildlife Service Biological Report.

Penland, S., P.F. Connor, A. Beall, S. Fearnley, and S.J. Williams. 2005. Changes in Louisiana's shoreline: 1855-2002. *Journal of Coastal Research*: 7-39.

Penttila, D. 1978. Studies of the surf smelt (*Hypomesus pretiosus*) in Puget Sound. Washington Department of Fish. Tech. Rep.

Penttila, D.E. 1995. Investigations of the spawning habitat of the Pacific Sand Lance (*Ammodytes hexapterus*) in Puget Sound. *Proceedings of Puget Sound Research 1995*, Seattle, Washington. pp. 855-859.

Penttila, D.E. 2000. Forage fishes of the Puget Sound region. *NWSC/PSAMP Data Conference*, LaConner, Washington.

Penttila, D.E. 2001. Effects of Overhanging Shading Vegetation on Egg Survival for Summer-Spawning Surf Smelt on Upper Intertidal Beaches in Northern Puget Sound, Washington, Draft. WDFW, Marine Resources Division.

Perez-Ruzafa, A., J.A. Grarcia-Charton, E. Barcala, and C. Marcos. 2006. Changes in benthic fish assemblages as a consequence of coastal works in a coastal lagoon: The Mar Menor (Spain, Western Mediterranean). *Marine Pollution Bulletin* 53(1-4): 107-120.

Persaud, D., R. Jaagumagi, and A. Hayton. 1991. The Provincial Sediment Quality Guidelines (Draft). Toronto, Canada: Water Resources Branch, Ontario Ministry of the Environment.

Peterson, H.W.U., and L. Amiotte. 2006. Decline of Skokomish nation spot shrimp catch in low dissolved oxygen waters of the hood Hood Canal, Puget Sound, State of Washington. *Ethnicity & Disease* 16(4): 17-17.

Peterson, J.T., N.P. Banish, and R.F. Thurow. 2005. Are Block Nets Necessary? Movement of Stream-Dwelling Salmonids in Response to Three Common Survey Methods. *North American Journal of Fisheries Management* 25: 732-743.

Peterson, J.T., R.F. Thurow, and J.W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. *Transactions of the American Fisheries Society* 133(2): 462-475.

Phillips, R.C. 1984. Ecology of Eelgrass Meadows in the Pacific Northwest: A Community Profile. FWS/OBS-84/24. Portland, Oregon: U.S. Fish and Wildlife Service.

Pickering, H., and D. Whitmarsh. 1997. Artificial reefs and fisheries exploitation: A review of the 'attraction versus production' debate, the influence of design and its significance for policy. *Fisheries Research* 31(1-2): 39-59.

Pickett, P.J. 1997. Pollutant loading capacity for the Black River, Chehalis River system, Washington. *Journal of the American Water Resources Association* 33(2): 465-480.

Pihl, L., S. Baden, N. Kautsky, P. Ronnback, T. Soderqvist, M. Troell, and H. Wennhage. 2006. Shift in fish assemblage structure due to loss of seagrass *Zostera marina* habitats in Sweden. *Estuarine Coastal and Shelf Science* 67(1-2): 123-132.

Pondella, D.J., and J.S. Stephens. 1994. Factors Affecting the Abundance of Juvenile Fish Species on a Temperate Artificial Reef. *Bulletin of Marine Science* 55(2-3): 1216-1223.

Pondella, D.J., J.S. Stephens, and M.T. Craig. 2002. Fish production of a temperate artificial reef based on the density of embiotocids (Teleostei : Perciformes). *ICES Journal of Marine Science* 59: S88-S93.

Poole, G., J. Dunham, M. Hicks, D. Keenan, J. Lockwood, E. Materna, D. McCullough, C. Mebane, J. Risley, S. Sauter, S. Spaulding, and D. Sturdevant. 2001. Technical Synthesis Scientific Issues Relating to Temperature Criteria for Salmon, Trout, and Char Native to the Pacific. EPA 910-R-01-007. Environmental Protection Agency, Region 10.

Poole, G.C., and C.H. Berman. 2001a. An ecological perspective on in-stream temperature: Natural heat dynamics and mechanisms of human-caused thermal degradation. *Environmental Management* 27(6): 787-802.

Poole, G.C., and C.H. Berman. 2001b. Pathways of Human Influence on Water Temperature Dynamics in Stream Channels. *Environmental Management* 27: 787-802.

Popper, A.N., and N.L. Clarke. 1976. The auditory system of the goldfish (*Carassius auratus*): effects of intense acoustic stimulation. *Comparative Biochemistry and Physiology* 53: 11-18.

Popper, A.N., and R.R. Fay. 1973. Sound Detection and Processing by Teleost Fishes - Critical Review. *Journal of the Acoustical Society of America* 53(6): 1515-1529.

Popper, A.N., and R.R. Fay. 1993. Sound Detection and Processing by Fish - Critical-Review and Major Research Questions. *Brain Behavior and Evolution* 41(1): 14-38.

Popper, A.N., M.E. Smith, P.A. Cott, B.W. Hanna, A.O. MacGillivray, M.E. Austin, and D.A. Mann. 2005. Effects of exposure to seismic airgun use on hearing of three fish species. *Journal of the Acoustical Society of America* 117(6): 3958-3971.

Poston, T. 2001. Treated Wood Issues Associated with Overwater Structures in Marine and Freshwater Environments. White Paper. Olympia, Washington: Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington State Department of Transportation.

Poston, T.M., K.M. Krupka, and M.C. Richmond. 1996. Estimation of Treated Piling Emplacement and Piling Leachate Concentrations in the Columbia River. Richland, Washington: Pacific Northwest National Laboratory.

Pratt, T.C., and K.E. Smokorowski. 2003. Fish habitat management implications of the summer habitat use by littoral fishes in a north temperate, mesotrophic lake. *Canadian Journal of Fisheries and Aquatic Sciences* 60(3): 286-300.

Prince, E.D., and O.E. Maughan. 1978. Freshwater Artificial Reefs: Biology and Economics. *Fisheries* 3(1): 5-9.

Pusch, M., D. Fiebig, I. Brettar, H. Eisenmann, B.K. Ellis, L.A. Kaplan, M.A. Lock, M.W. Naegeli, and W. Traunspurger. 1998. The role of micro-organisms in the ecological connectivity of running waters. *Freshwater Biology* 40(3): 453-495.

Qiao, F.L., J. Ma, C.S. Xia, Y.Z. Yang, and Y.L. Yuan. 2006. Influences of the surface wave-induced mixing and tidal mixing on the vertical temperature structure of the Yellow and East China Seas in summer. *Progress in Natural Science* 16(7): 739-746.

Quinn, T.P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout*. Seattle, Washington: University of Washington Press.

Quirollo, L.F. 1992. Pacific Hake. In *California's Living Marine Resources and Their Utilization. California Sea Grant College Program*, edited by W.S. Leet, C.M. Dewees and C.W. Haugen. Davis, California. pp. 129.

Ranasinghe, R., and I.L. Turner. 2006. Shoreline response to submerged structures: A review. *Coastal Engineering* 53(1): 65-79.

Ranasinghe, R., I.L. Turner, and G. Symonds. 2006. Shoreline response to multi-functional artificial surfing reefs: A numerical and physical modeling study. *Coastal Engineering* 53(7): 589-611.

Rao, Y.R., and D.J. Schwab. 2007. Transport and mixing between the coastal and offshore waters in the great lakes: a review. *Journal of Great Lakes Research* 33(1): 202-218.

Reilly, C.A., T.W. Wyllie-Echeverria, and S. Ralston. 1992. Interannual variation and overlap in the diets of pelagic juvenile rockfish (Genus: *Sebastes*) off central California. *Fishery Bulletin* 90: 505-515.

Reiman, B.E., and J.D. McIntyre. 1993. Demographic and Habitat Requirements for the Conservation of Bull Trout. Ogden, Utah: USDA Forest Service Intermountain Research Station.

Reine, K., and D. Clarke. 1998. Entrainment by hydraulic dredges - A review of potential impacts. Technical Note. DOER-EI. U.S. Army Engineer Research and Development Center.

Reyff, J., P. Donavan, and C.R. Greene Jr. 2003. Underwater Sound Levels Associated with Seismic Retrofit Construction of the Richmond-San Rafael Bridge. Prepared for California Department of Transportation by Illingworth & Rodkin, Inc. and Greeneridge Sciences Sacramento, California.

Ribeiro, F., P.K. Crain, and P.B. Moyle. 2004. Variation in condition factor and growth in young-of-year fishes in floodplain and riverine habitats of the Cosumnes River, California. *Hydrobiologia* 527(1): 77-84.

Rice, C.A. 2006. Effects of shoreline modification on a northern Puget Sound beach: microclimate and embryo mortality in surf smelt. *Estuaries and Coasts* 29(1): 63-71.

Richard, J.D. 1968. Fish Attraction with Pulsed Low Frequency Sound. *Journal of Fisheries Research Board of Canada* 25(7): 1441-1452.

Richardson, E.V., and S.R. Davies. 2001. Evaluating Scour at Bridges. Fourth Edition. FHWA NHI 01-001 HEC-18, Hydraulic Engineering Circular No. 19. Washington D.C: Federal Highway Administration, U.S. Department of Transportation.

Richardson, M.D., A.G. Carey, and W.A. Colgate. 1977. Aquatic disposal field investigations - Columbia River disposal site, Oregon. Appendix C: the effects of dredged material disposal on benthic assemblages. Vicksburg, Mississippi: Report to U.S. Army Corps of Engineers Waterways Expt. Station.

Richter, A., and S.A. Kolmes. 2005. Maximum temperature limits for chinook, coho, and chum salmon, and steelhead trout in the Pacific Northwest. *Reviews in Fisheries Science* 13(1): 23-49.

Rinne, J.N., W.L. Minckley, and P.O. Bersell. 1981. Factors Influencing Fish Distribution in Two Desert Reservoirs, Central Arizona. *Hydrobiologia* 80(1): 31-42.

Roby, D.D., K. Collis, D.E. Lyons, D.P. Craig, J.Y. Adkins, A.M. Myers, and R.M. Suryan. 2002. Effects of colony relocation on diet and productivity of Caspian terns. *Journal of Wildlife Management* 66(3): 662-673.

Rodway, M.S., H.M. Regehr, J. Ashley, P.V. Clarkson, R.I. Goudie, D.E. Hay, C.M. Smith, and K.G. Wright. 2003. Aggregative response of Harlequin Ducks to herring spawning in the Strait of Georgia, British Columbia. *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 81(3): 504-514.

Roegner, G.C., B.M. Hickey, J.A. Newton, A.L. Shanks, and D.A. Armstrong. 2002. Wind-induced plume and bloom intrusions into Willapa Bay, Washington. *Limnology and Oceanography* 47(4): 1033-1042.

Rolauffs, P., D. Hering, and S. Lohse. 2001. Composition, invertebrate community and productivity of a beaver dam in comparison to other stream habitat types. *Hydrobiologia* 459: 201-212.

Rolletschek, H., and H. Kühl. 1997. Die Auswirkungen von Röhrichtschutzbauwerken auf die Gewässerufer (The impacts of reed protecting structures on lakesides). *Limnologica* 27(3-4): 365-380.

Rooper, C.N., D.R. Gunderson, and B.M. Hickey. 2006. An examination of the feasibility of passive transport from coastal spawning grounds to estuarine nursery areas for English sole. *Estuarine Coastal and Shelf Science* 68(3-4): 609-618.

Rosenthal, R.J., V. Moran-O'Connell, and M.C. Murphy. 1988. Feeding ecology of ten species of rockfishes (*Scorpaenidae*) from the Gulf of Alaska. *California Fish and Game* 74: 16-36.

Roth, B.M., I.C. Kaplan, G.G. Sass, P.T. Johnson, A.E. Marburg, A.C. Yannarell, T.D. Havlicek, T.V. Willis, M.G. Turner, and S.R. Carpenter. 2007. Linking terrestrial and aquatic ecosystems: The role of woody habitat in lake food webs. *Ecological Modeling* 203(3-4): 439-452.

Saiki, M.K., and B.A. Martin. 2001. Survey of fishes and environmental conditions in Abbotts Lagoon, Point Reyes National Seashore, California. *California Fish and Game* 87(4): 123-138.

Salo, E., N. Bax, T. Prinslow, C. Whitmus, B. Snyder, and C.A. Simenstad. 1980. The effects of construction of naval facilities on the outmigration of juvenile salmonids from Hood Canal, Washington. Final Report FRI-UW-8006. Seattle, Washington: University of Washington, Fisheries Research Institute.

Salo, E.O. 1991. Life History of Chum Salmon (*Oncorhynchus Keta*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 231-310.

Sampson, D.B. 1996. Stock Status of Canary Rockfish off Oregon and Washington in 1996: Appendix C in Pacific Fishery Management Council. Status of the Pacific Coast Groundfish Fishery through 1996 and Recommended Acceptable Biological Catches for 1997: Stock Assessment and Fishery Evaluation. Portland, Oregon: Pacific Fishery Management Council.

Sandahl, J.F., D.H. Baldwin, J.J. Jenkins, and N.L. Scholz. 2004. Odor-evoked field potentials as indicators of sublethal neurotoxicity in juvenile coho salmon (*Oncorhynchus kisutch*) exposed to copper, chlorpyrifos, or esfenvalerate. *Can. J. Fish. Aquat. Sci*. 61:404-413. (As cited in Stratus 2005b).

Sand-Jensen, K. 1998. Influence of submerged macrophytes on sediment composition and near-bed flow in lowland streams. *Freshwater Biology* 39(4): 663-679.

Sane, M., H. Yamagishi, M. Tatelshi, and T. Yamagishi. 2007. Environmental impacts of shore-parallel breakwaters along Nagahama and Ohgata, District of Joetsu, Japan. *Journal of Environmental Management* 82(4): 399-409.

Scavia, D., and S.B. Bricker. 2006. Coastal Eutrophication Assessment in the United States. *Biogeochemistry* 79(1-2): 187-208.

Schaffter, R.G., P.A. Jones, and J.G. Karlton. 1983. Sacramento River and tributaries bank protection and erosion control investigation–evaluation of impacts on fisheries. Sacramento, California: U.S. Army Corps of Engineers.

Scheuerell, M.D., and D.E. Schindler. 2004. Changes in the spatial distribution of fishes in lakes along a residential development gradient. *Ecosystems* 7(1): 98-106.

Schlesinger, W.H. 1997. *Biogeochemistry: An analysis of global change*. San Diego, California: Academic Press.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of riprap bank reinforcement on stream salmonids in the Western United States. *Fisheries* 26(7): 6-13.

Scholik, A.R., and H.Y. Yan. 2001. Effects of Underwater Noise on Auditory Sensitivity of a Cyprinid Fish. *Hearing Research* 152: 17-24.

Scholik, A.R., and H.Y. Yan. 2002. The Effects of Noise on the Auditory Sensitivity of the Bluegill Sunfish, *Lepomis Macrochirus*. *Comparative Biochemistry and Physiology* A133: 43-52.

Schoonover, J.E., and K.W.J. Williard. 2003. Ground water nitrate reduction in giant cane and forest riparian buffer zones. *Journal of the American Water Resources Association* 39(2): 347-354.

Schuett-Hames, D., B. Conrad, A. Pleus, and K. Lautz. 1996. Literature Review and Monitoring Recommendations for Salmonid Spawning Gravel Scour. TFW-AM9-96-001. Northwest Indian Fisheries Commission and Washington Department of Fish and Wildlife.

Schwarz, A.L., and G.L. Greer. 1984. Responses of Pacific herring, *Clupea barengus pallast*, to some underwater sounds. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1183-1192.

Seabergh, W.C., and N.C. Kraus. 2003. Progress in management of sediment bypassing at coastal inlets: Natural bypassing, weir jetties, jetty spurs, and engineering aids in design. *Coastal Engineering Journal* 45(4): 533-563.

Sedell, J.R., F.J. Swanson, and S.V. Gregory. 1985. Evaluating fish response to woody debris. In *Proceedings of the Pacific Northwest stream habitat workshop*, edited by T.J. Hassler. Arcata, California: California Cooperative Fishery Research Unit, Humboldt State University. pp. 222-245

Seitz, R.D., R.N. Lipcius, and M.S. Seebo. 2005. Food availability and growth of the blue crab in seagrass and unvegetated nurseries of Chesapeake Bay. *Journal of Experimental Marine Biology and Ecology* 319(1-2): 57-68.

Selong, J.H., T.E. McMahon, A.V. Zale, and F.T. Barrows. 2001. Effect of temperature on growth and survival of bull trout, with application of an improved method for determining thermal tolerance in fishes. *Transactions of the American Fisheries Society* 130(6): 1026-1037.

Sergeant, C.J., and D.A. Beauchamp. 2006. Effects of physical habitat and ontogeny on lentic habitat preferences of juvenile Chinook salmon. *Transactions of the American Fisheries Society* 135(5): 1191-1204.

Servizi, J.A., and D.W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon. *Canadian Special Publication of Fisheries and Aquatic Sciences* 96: 254-264.

Servizi, J.A., and D.W. Martens. 1991. Effect of Temperature, Season, and Fish Size on Acute Lethality of Suspended Sediments to Coho Salmon (*Oncorhynchus-Kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 48(3): 493-497.

Servizi, J.A., and D.W. Martens. 1992. Sublethal Responses of Coho Salmon (*Oncorhynchus-Kisutch*) to Suspended Sediments. *Canadian Journal of Fisheries and Aquatic Sciences* 49(7): 1389-1395.

Servizi, J.A., and R.W. Gordon. 1990. Acute lethal toxicity of ammonia and suspended sediment mixtures to chinook salmon (*Oncorhynchus tshawytscha*). *Bulletin of Environmental Contamination and Toxicology* 44(4): 650-656.

Sharber, N.G., and S.W. Carothers. 1988. Influence of Electrofishing Pulse Shape on Spinal Injuries in Adult Rainbow Trout. *North American Journal of Fisheries Management* 8: 117-122.

Shepard, B.B., K.L. Pratt, and P.J. Graham. 1984. Life Histories of Westslope Cutthroat Trout and Bull Trout in the Upper Flathead River Basin, Montana. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical Changes in the Columbia River Estuary. *Progress in Oceanography* 25(1-4): 299-352.

Shields, F.D. 1991. Woody Vegetation and Riprap Stability Along the Sacramento River Mile 84.5-119. *Water Resources Bulletin* 27(3): 527-536

Shields, F.D., and D.H. Gray. 1992. Effects of woody vegetation on levee integrity *Water Resources Bulletin* 28: 917-931.

Sigler, J.W., T.C. Bjornn, and F.H. Everest. 1984. Effects of Chronic Turbidity on Density and Growth of Steelheads and Coho Salmon. *Transactions of the American Fisheries Society* 113(2): 142-150.

Simenstad, C.A. 1990. Effects of Dredging on Anadromous Pacific Coast Fishes Summary and Conclusions from Workshop and Working Group Discussions. September 8-9, 1990. pp. 144-152.

Simenstad, C.A., B. Nightingale, R.A. Thom, and D.K. Shreffler. 1999. Impacts of ferry terminals on juvenile salmon migrating along Puget Sound shorelines: Phase I synthesis of state of knowledge. Research Project T9903 Task A2. Seattle, Washington: Washington State Transportation Center.

Simenstad, C.A., B.S. Miller, C.F. Nyblade, K. Thornburgh, and L.J. Bledsoe. 1979. Food Web Relationship of Northern Puget Sound and the Strait of Juan de Fuca. EPA Interagency Agreement No. D6-E693-EN. Office of Environmental Engineering and Technology, U.S. Environmental Protection Agency.

Simenstad, C.A., W.J. Kinney, S.S. Parker, E.O. Salo, J.R. Cordell, and H. Buechner. 1980. Prey community structure and trophic ecology of outmigrating juvenile chum and pink salmon in Hood Canal, Washington: A synthesis of three years' studies, 1977-1979. Seattle, Washington: University of Washington.

Simmons, G.M. 1992. Importance of Submarine Groundwater Discharge (SGWD) and Seawater Cycling to Material Flux across Sediment Water Interfaces in Marine Environments. *Marine Ecology-Progress Series* 84(2): 173-184.

Simon, A. 1994. Gradation processes and channel evolution in modified west Tennessee streams process, response, and form. Professional Paper 1470. Denver, Colorado: U.S. Geological Survey.

Simon, A., and C.R. Hupp. 1992. Geomorphic and Vegetative Recovery Processes along Modified Tennessee Streams: An Interdisciplinary Approach to Disturbed Fluvial Systems. Washington DC: International Association of Hydrological Sciences.

Sinclair, M. 1992. Marine Populations - an Essay on Population Regulation and Speciation. Books in Recruitment Fishery Oceanography. Seattle and London: Washington Sea Grant Program, University of Washington Press

Smith, S.L., D.D. MacDonald, K.A. Keenleyside, C.G. Ingersoll, and L.J. Field. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *Great Lakes Resources* 22(3): 624-638.

Smokorowski, K.E., T.C. Pratt, W.G. Cole, L.J. McEachern, and E.C. Mallory. 2006. Effects on periphyton and macroinvertebrates from removal of submerged wood in three Ontario lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 63(9): 2038-2049.

Snoeyink, V.L., and D. Jenkins. 1980. *Water chemistry*. New York: Wiley.

Snyder, D.E. 2003. Electrofishing and Its Harmful Effects on Fish. Information and Technology Report USGS/BRD/ITR--2003-0002. Denver, Colorado: U.S. Geological Survey.

Snyder, N.P., S.A. Wright, C.N. Alpers, L.E. Flint, C.W. Holmes, and D.M. Rubin. 2006. Reconstructing depositional processes and history from reservoir stratigraphy: Englebright Lake, Yuba River, northern California. *Journal of Geophysical Research-Earth Surface* 111(F4).

Sobocinski, K.L. 2003. The impact of shoreline on supratidal beach fauna of central Puget Sound University of Washington, Seattle, Washington, 83 pp.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham, and W.J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* 58(2): 325-333.

Soulsby, C., A. F. Youngson, H. J. Moir and I. A. Malcolm. 2001. Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: a preliminary assessment. *The Science of the Total Environment*. Volume 265, Issues 1-3, 29 January 2001, Pages 295-307.

Spanhoff, B., W. Riss, P. Jakel, N. Dakkak, and E.I. Meyer. 2006. Effects of an experimental enrichment of instream habitat heterogeneity on the stream bed morphology and chironomid community of a straightened section in a sandy lowland stream. *Environmental Management* 37(2): 247-257.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. Corvallis, Oregon: ManTech Environmental Research Services Corporation.

Sprague, J.B. 1964. Avoidance of copper-zinc solutions by young salmon in the laboratory. *J. Water Pollut. Control Fed.* 36:990-1004. (As cited in Stratus 2005b).

Sprague, J.B. 1968. Avoidance reactions of rainbow trout to zinc sulfate solutions. *Water Res.* 2:367-372. (As cited in Stratus 2005b).

Sridhar, V., A.L. Sansone, J. LaMarche, T. Dubin, and D.P. Lettenmaier. 2004. Prediction of stream temperature in forested watersheds. *Journal of the American Water Resources Association* 40(1): 197-213.

Stadler, J., NOAA Fisheries, Seattle, Washington. 2007. E-mail regarding NOAA Fisheries use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation with Eric Doyle of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Stanford, J.A., and J.V. Ward. 1992. Management of aquatic resources in large catchments: Recognizing interactions between ecosystem connectivity and environmental disturbance. In *Watershed Management – Balancing sustainability and environmental change*, edited by R.J. Naiman. New York: Springer-Verlag.

Stanford, J.A., and J.V. Ward. 1993. An Ecosystem Perspective of Alluvial Rivers - Connectivity and the Hyporheic Corridor. *Journal of the North American Benthological Society* 12(1): 48-60.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A general protocol for restoration of regulated rivers. *Regulated Rivers: Research & Management* 12(4-5): 391-413.

Stanley, D.R., and C.A. Wilson. 2004. Effect of hypoxia on the distribution of fishes associated with a petroleum platform off coastal Louisiana. *North American Journal of Fisheries Management* 24(2): 662-671.

Starr, R.M., D.S. Fox, M.A. Hixon, B.N. Tissot, G.E. Johnson, and W.H. Barss. 1996. Comparison of submersible-survey and hydroacoustic survey estimates of fish density on a rocky bank. *Fishery Bulletin* 94: 113-123.

Stein, D., and T.J. Hassler. 1989. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest): Brown Rockfish, Copper Rockfish, and Black Rockfish. U.S. Fish and Wildlife Service.

Stocker, M. 2002. Fish Mollusks and Other Sea Animals, and the Impact of Anthropogenic Noise in the Marine Acoustical Environment. Michael Stocker Associates for Earth Island Institute.

Storry, K.A., C.K. Weldrick, M. Mews, M. Zimmer, and D.E. Jelinski. 2006. Intertidal coarse woody debris: A spatial subsidy as shelter or feeding habitat for gastropods? *Estuarine Coastal and Shelf Science* 66(1-2): 197-203.

Stratus. 2005a. Creosote-Treated Wood in Aquatic Environments: Technical Review and Use Recommendations. Prepared for NOAA Fisheries Habitat Conservation Division Southwest Habitat Conservation Division by Stratus Consulting Santa Rosa, California.

Stratus. 2005b. Treated Wood in Aquatic Environments: Technical Review and Use Recommendations. Prepared for NOAA Fisheries Habitat Conservation Division Southwest Habitat Conservation Division by Stratus Consulting, Santa Rosa, California.

Sukhodolov, A., C. Engelhardt, A. Kruger, and H. Bungartz. 2004. Case study: Turbulent flow and sediment distributions in a groyne field. *Journal of Hydraulic Engineering-Asce* 130(1): 1-9.

Sullivan, K., D.J. Martin, R.D. Cardwell, J.E. Toll, and S. Duke. 2000. *An Analysis of the Effects of Temperature on Salmonids of the Pacific Northwest with Implications for Selecting Temperature Criteria*. Portland, Oregon: Sustainable Ecosystems Institute.

Sullivan, K., J. Tooley, K. Doughty, J.E. Caldwell, and P. Knudsen. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. TFW-WQ3-90-006. Olympia, Washington: Washington Department of Natural Resources.

Sumer, B.M., J. Fredsoe, A. Lamberti, B. Zanuttigh, M. Dixen, K. Gislason, and A.F. Di Penta. 2005. Local scour at roundhead and along the trunk of low crested structures. *Coastal Engineering* 52(10-11): 995-1025.

Sumida, B.Y., and H.G. Moser. 1984. Food and feeding of Bocaccio and comparison with Pacific hake larvae in the California current. *California Cooperation Oceanic Fishery Investigation Report* 25: 112-118.

Teachout, E., Fish and Wildlife Biologist with U.S. Fish and Wildlife Service, Lacey, Washington. 2007. E-mail regarding USFWS use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation with Julie Hampden of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Terich, T.A. 1987. *Living with the Shore of Puget Sound and the Georgia Strait*. Living with the Shore. Edited by O.H. Pilkey and W.J. Neal. Durham, NC: Duke University Press.

Terrados, J., C.M. Duarte, M.D. Fortes, J. Borum, N.S.R. Agawin, S. Bach, U. Thampanya, L. Kamp-Nielsen, W.J. Kenworthy, O. Geertz-Hansen, and J. Vermaat. 1998. Changes in community structure and biomass of seagrass communities along gradients of siltation in SE Asia. *Estuarine Coastal and Shelf Science* 46(5): 757-768.

The Watershed Company. 2006. Shoreline Analysis Report. Kirkland, Washington: City of Kirkland.

Theurer, F.D., K.A. Voos, and W.J. Miller. 1984. Instream Water Temperature Model. Instream Flow Informational Paper 16 FWS/OBS-84/15. U.S. Fish and Wildlife Service.

Thom, R.M., and D.K. Shreffler. 1996. Eelgrass meadows near ferry terminals in Puget Sound. Sequim, Washington: Battelle Pacific Northwest Laboratories.

Thom, R.M., C.A. Simenstad, J.R. Cordell, and E.O. Salo. 1989. Fish and their epibenthic prey in a marina and adjacent mudflats and eelgrass meadow in a small estuarine bay. Seattle, Washington: University of Washington.

Thom, R.M., D.K. Shreffler, and K. MacDonald. 1994. Shoreline Armoring Effects on Coastal Ecology and Biological Resources in Puget Sound, Washington. Shorelands and Environmental Assistance Program, Washington Department of Ecology.

Thomalla, F., and C.E. Vincent. 2003. Beach response to shore-parallel breakwaters at Sea Palling, Norfolk, UK. *Estuarine Coastal and Shelf Science* 56(2): 203-212.

Thompson, K.G., E.P. Bergersen, R.B. Nehring, and D.C. Bowden. 1997. Long-Term Effects of Electrofishing on Growth and Body Condition of Brown and Rainbow Trout. *North American Journal of Fisheries Management* 17: 154-159.

Thompson, R.C., T.P. Crowe, and S.J. Hawkins. 2002. Rocky intertidal communities: past environmental changes, present status and predictions for the next 25 years. *Environmental Conservation* 29(2): 168-191.

Thomson Scientific Web of Science. 2007. Available at: http://scientific.thomson.com/products/wos/ (accessed May–July 2007).

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer, and J.V. Ward. 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). *Freshwater Biology* 41(3): 521-535.

Toft, J., C. Simenstad, J. Cordell, and L. Stamatiou. 2004. Fish distribution, abundance, and behavior at nearshore habitats along City of Seattle marine shorelines, with an emphasis on juvenile salmonids. Prepared for Seattle Public Utilities by Wetland Ecosystem Team, University of Washington School of Aquatic and Fishery Sciences, Seattle, Washington. March 2004.

Toft, J., C. Simenstad, J. Cordell, and L. Stamatiou. 2007. Fish Distribution, Abundance, and Behavior along City Shoreline Types in Puget Sound. *North American Journal of Fisheries Management* 27: 465-480.

Tops, S., W. Lockwood, and B. Okamura. 2006. Temperature-driven proliferation of *Tetracapsuloides bryosalmonae* in bryozoan hosts portends salmonid declines. *Diseases of Aquatic Organisms* 70(3): 227-236.

Turnpenny, A.W.H., K.P. Thatcher, and J.R. Nedwell. 1994. The Effects on Fish and Other Marine Animals of High-Level Underwater Sound. Fawley Aquatic Research.

Urban, E.R., and C.J. Langdon. 1984. Reduction in Costs of Diets for the American Oyster, *Crassostrea-Virginica (Gmelin*), by the Use of Non-Algal Supplements. *Aquaculture* 38(4): 277-291.

Urban, E.R., and D.L. Kirchman. 1992. Effect of Kaolinite Clay on the Feeding-Activity of the Eastern Oyster *Crassostrea-Virginica (Gmelin*). *Journal of Experimental Marine Biology and Ecology* 160(1): 47-60.

Urick, R.J. 1983. Principles of Underwater Sound. Ch. 7 In *The Noise Background of the Sea*, Los Altos, California: Peninsula Publishing.

USACE. 1983. Engineering and Design - Dredging and Dredged Material Disposal. U.S. Army Corps of Engineers. Washington, DC: CECW-EH-D.

USACE. 2007. DOER Program. March 2007. U.S. Army Corps of Engineers. Available at: http://el.erdc.usace.army.mil/dots/doer/ (accessed September 3, 2007).

USEPA. 1996. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod *Hyalella azteca* and the Midge *Chironomus riparius*: Assessment and Remediation of Contaminated Sediments (ARCS) Program. EPA 905-R96-008. Washington, DC: U.S. Environmental Protection Agency.

USEPA. 2002. National Recommended Water Quality Criteria: 2002. EPA 822-R-02-047. Washington, DC: U.S. Environmental Protection Agency. November. (As cited in Stratus 2005b).

USEPA. 2007. National Management Measures to Control Nonpoint Source Pollution from Hydromodification. EPA 841-B-07-002. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

USFS, NMFS, USBLM, USFWS, USNPS, and USEPA. 1993. Forest Ecosystem Management: An Ecological, Economic, and Social Assessment. U.S. Forest Service, National Marine Fisheries Service, U.S. Bureau of Land Management, U.S. Fish and Wildlife Service, National Park Service, and U.S. Environmental Protection Agency. Portland, Oregon: USFS PNW Region.

USFS. 2007. Fish Resources. Available at U.S. Forest Service website: http://www.fs.fed.us/r6/fishing/forests/fishresources/win_coldwater.html#redband).

USFWS. 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Steelhead Trout. *Biological Report* 82 (11.82). Lafayette, Louisiana: U.S. Fish and Wildlife Service, Coastal Ecology Group.

USFWS. 2000. Impacts of riprapping to ecosystem functioning, Lower Sacramento River, California. Prepared for U.S. Army Corps of Engineers by U.S. Fish and Wildlife Service, Sacramento, California. June 2000.

Vagle, S. 2003. On the Impact of Underwater Pile Driving Noise on Marine Life. Government Report. Canada Department of Fisheries and Oceans, Institute of Ocean Sciences, Ocean Science and Productivity Division.

Valett, H.M., C.L. Crenshaw, and P.F. Wagner. 2002. Stream nutrient uptake, forest succession, and biogeochemical theory.

Valett, H.M., M.A. Baker, J.A. Morrice, C.S. Crawford, M.C. Molles, C.N. Dahm, D.L. Moyer, J.R. Thibault, and L.M. Ellis. 2005. Biogeochemical and metabolic responses to the flood pulse in a semiarid floodplain. *Ecology* 86(1): 220-234.

Vannote, R.L., and G.W. Minshall. 1982. Fluvial Processes and Local Lithology Controlling Abundance, Structure, and Composition of Mussel Beds. *Proceedings of the National Academy of Science* 79: 4103-4107.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Science* 37: 130-137.

Vellidis, G., R. Lowrance, P. Gay, and R.K. Hubbard. 2003. Nutrient Transport in a Restored Riparian Wetland. *Journal of Environmental Quality* 32(2): 711-726.

Vines, C.A., T. Robbins, F.J. Griffin, and G.N. Cherr. 2000. The Effects of Diffusible Creosote-Derived Compounds on Development in Pacific Herring (*Clupea Pallasi*). *Aquatic Toxicology* 51: 225-239.

Vogel, J.L., and D.A. Beauchamp. 1999. Effects of light, prey size, and turbidity on reaction distances of lake trout (*Salvelinus namaycush*) to salmonid prey. *Canadian Journal of Fisheries and Aquatic Sciences* 56(7): 1293-1297.

WAC 173-201A. November 20, 2006. Water quality standards for surface waters of the state of Washington. Washington Administrative Code.

Wagner, C.M., and H.M. Austin. 1999. Correspondence between environmental gradients and summer littoral fish assemblages in low salinity reaches of the Chesapeake Bay, USA. *Marine Ecology-Progress Series* 177: 197-212.

Wagner, E.J., T. Bosakowski, and S. Intelmann. 1997. Combined effects of temperature and high pH on mortality and the stress response of rainbow trout after stocking. *Transactions of the American Fisheries Society* 126(6): 985-998.

Wantzen, K.M. 2006. Physical pollution: effects of gully erosion on benthic invertebrates in a tropical clear-water stream. *Aquatic Conservation-Marine and Freshwater Ecosystems* 16(7): 733-749.

Warmke, S., and D. Hering. 2000. Composition, microdistribution and food of the macroinvertebrate fauna inhabiting wood in low-order mountain streams in Central Europe. *International Review of Hydrobiology* 85(1): 67-78.

Wasson, K., K. Fenn, and J.S. Pearse. 2005. Habitat differences in marine invasions of central California. *Biological Invasions* 7(6): 935-948.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects and Control. American Fisheries Society. Bethesda, Maryland.

Watters, G.T. 1999. Freshwater mussels and water quality: A review of the effects of hydrologic and instream habitat alterations. *First Freshwater Mollusk Conservation Society Symposium*, Columbus, Ohio.

WDFW. 1997a. Washington State Forage Fish: Sand Lance Webpage. Available at: http://wdfw.wa.gov/fish/forage/lance.htm (accessed July 2007).

WDFW. 1997b. Washington State Forage fish fact sheet: Puget Sound herring fact sheet. Washington Department of Fish and Wildlife. Available at: http://www.wdfw.wa.gov/fish/forage/herring.htm (accessed Nov 3, 2006).

WDFW. 1997c. Washington State Forage fish fact sheet: Washington State surf smelt fact sheet. Washington Department of Fish and Wildlife. Available at: http://www.wdfw.wa.gov/fish/forage/smelt.htm (accessed Nov. 3, 2006).

WDFW. 2001. Washington and Oregon Eulachon Management Plan. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW. 2003. Integrated Streambank Protection Guidelines. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW. 2004. Zebra mussels discovered at Washington-Idaho border. Washington Department of Fish and Wildlife. Available at: http://wdfw.wa.gov/do/newreal/release.php?id=may2504a (accessed June 22, 2007).

WDFW. 2007. Data provided by Washington Department of Fish and Wildlife, Olympia, Washington. MS Excel spreadsheet, WhitePaperdatatypes11_01_06_1.xls Provided to Herrera Environmental Consultants. May 9, 2007.

WDNR. 2005. Forest Practices - Habitat Conservation Plan. Olympia, Washington: Washington State Department of Natural Resources.

WDNR. 2006a. Draft Covered Species Paper - Fish. Olympia, Washington: Washington Department of Natural Resources.

WDNR. 2006b. Draft Covered Species Paper - Invertebrates. Olympia, Washington: Washington Department of Natural Resources.

Webster, M.T., and R.C. Loehr. 1996. Long-term leaching of metals from concrete products. *Journal of Environmental Engineering-Asce* 122(8): 714-721.

Wedemeyer, G.A., R.L. Saunders, and W.C. Clarke. 1980. Environmental Factors Affecting Smoltification and Early Marine Survival of Anadromous Salmonids. *Marine Fisheries Review* 42(6): 1-14.

Weis, J.S., and P. Weis. 1994. Effects of contaminants from chromated copper arsenate-treated lumber on benthos. *Arch. Environ. Contam. Toxicol.* 26: 103-109.

Weis, J.S., and P. Weis. 1996. Effects of using wood treated with chromated copper arsenate in shallow-water environments: a review. *Estuaries* 19(2A): 306-310.

Weis, P., J.S. Weis, and J. Couch. 1993. Histopathology and bioaccumulation in *Crassostrea virginica* living on wood preserved with chromated copper arsenate. *Diseases of Aquatic Organisms* 17: 41-46.

Welch, E.B., J.M. Jacoby, and C.W. May. 1998. Stream Quality. In *River Ecology and Management*, edited by R.J. Naiman and R.E. Bilby. New York: Springer. pp. 69-94.

Welker, T.L., S.T. McNulty, and P.H. Klesius. 2007. Effect of sublethal hypoxia on the immune response and susceptibility of channel catfish, *Ictalurus punctatus*, to enteric septicemia. *Journal of the World Aquaculture Society* 38(1): 12-23.

West, J. 1997. Protection and Restoration of Marine Life in the Inland Waters of Washington State. Puget Sound/Georgia Basin Environmental Report Series (23): Number 6. Olympia, Washington: Puget Sound Water Quality Action Team.

West, J.E., R.M. Buckley, and D.C. Doty. 1994. Ecology and Habitat Use of Juvenile Rockfishes (*Sebastes* Spp.) Associated with Artificial Reefs in Puget-Sound, Washington. *Bulletin of Marine Science* 55(2-3): 344-350.

Whitman, R.P., T.P. Quinn, and E.L. Brannon. 1982. Influence of suspended volcanic ash on homing behavior of adult Chinook salmon. *Transactions of the American Fisheries Society.*

Widdows, J., P. Fieth, and C.M. Worrall. 1979. Relationships between Seston, Available Food and Feeding-Activity in the Common Mussel *Mytilus-Edulis*. *Marine Biology* 50(3): 195-207.

Wijnberg, K.M. 2002. Environmental controls on decadal morphologic behavior of the Holland coast. *Marine Geology* 189(3-4): 227-247.

Wilber, D.H., and D.G. Clarke. 2001. Biological effects of suspended sediments: A review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. *North American Journal of Fisheries Management* 21(4): 855-875.

Wilcox, D.A., J.E. Meeker, P.L. Hudson, B.J. Armitage, M.G. Black, and D.G. Uzarski. 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: A Great Lakes evaluation. *Wetlands* 22(3): 588-615.

Wildish, D.J., and J. Power. 1985. Avoidance of suspended sediments by smelt as determined by a new "single fish" behavioral bioassay. *Bulletin of Environmental Contamination and Toxicology* 34: 770-774.

Williams, G.D., and R.M. Thom. 2001. Marine and Estuarine Shoreline Modification Issues White Paper. Olympia, Washington.

Williams, G.D., R.M. Thom, and J.E. Starkes. 2001. Reconnaissance Assessment of the State of the Nearshore Ecosystem: Eastern Shore of Central Puget Sound, Including Vashon and Maury Islands (WRIAs 8 and 9). Prepared for King County Department of Natural Resources, Seattle, Washington.

Williams, J.D., M.L. Warren, K.S. Cummings, J.L. Harris, and R.J. Neves. 1993. Conservation Status of Fresh-Water Mussels of the United-States and Canada. *Fisheries* 18(9): 6-22.

Willson, M.F., R.H. Armstrong, M.C. Hermans, and K. Koski. 2006. Eulachon: A Review of Biology and an Annotated Bibliography. Alaska Fisheries Science Center and NOAA Fisheries.

Wisby, W.J., J.D. Richard, D.R. Nelson, and S.H. Gruber. 1964. Sound Perception in Elasmobranchs. In *Marine Bio-Acoustics*, edited by W.N. Tavolga. New York: Pergamon Press. pp. 255-268.

Wohl, E.E. 2004. *Disconnected rivers: linking rivers to landscapes*. New Haven: Yale University Press.

WSDOT. 2006. Biological Assessment Preparation for Transportation Projects. Advanced Training Manual. Olympia, Washington: Washington State Department of Transportation, Environmental Affairs Office.

Wydoski, R.S., and R.R. Whitney. 2003. *Inland Fishes of Washington, Second Edition*. American Fisheries Society and University of Washington Press.

Xiao, Y., J. Simonsen, and J.J. Morrell. 2002. Effect of water flow rate and temperature on leaching from creosote-treated wood. Madison, Wisconsin: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory.

Yates, S. 2001. Effects of Swinomish Channel jetty and causeway on outmigrating chinook salmon (*Oncorhynchus tshawytscha*) from the Skagit River Washington. Bellingham, Washington: Western Washington University. 65 pp.

Zimmermann, A.E., and M. Lapointe. 2005. Intergranular flow velocity through salmonid redds: Sensitivity to fines infiltration from low intensity sediment transport events. *River Research and Applications* 21(8): 865-881.