# WHITE PAPER

# Fish Screens

Prepared for

Washington Department of Fish and Wildlife

March 2008 Draft (Working Draft – Do Not Cite)

**WHITE PAPER**

---

# Fish Screens

Prepared for

Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington 98501-1091

Prepared by

Herrera Environmental Consultants, Inc.
2200 Sixth Avenue, Suite 1100
Seattle, Washington 98121
Telephone: 206/441-9080

In collaboration with

Kozmo Ken Bates

March 2008 Draft (Working Draft – Do Not Cite)

# Contents

# Tables

# Figures

# 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 (ITP) 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 ensure that activities conducted under an HCP 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 white paper for a definition of "take"), and possible management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. Because 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 to 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 fish screen structures and operations which, for the purpose of this effort, include in-channel and off-channel screen designs.

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, operation, and maintenance of fish screens.

- Use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely to result from the construction, operation, and maintenance of fish screens.

- 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.

For the purpose of this white paper, the effects of fish screens are considered limited to those effects imposed by the screen only. The effects of flow control structures and/or channel modifications associated with the diversion or intake system requiring the screen are not considered in this analysis. The effects of these actions or activities have been addressed in companion white papers. The effects of water withdrawals are also not considered. The Washington State Department of Ecology (Ecology) has lead responsibility for water rights and withdrawal determinations, and is therefore responsible for identifying related effects. On this basis, the literature review conducted for this white paper identified five impact mechanisms that could potentially affect 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
- Operations
- Water quality modifications
- Hydraulic and geomorphic modifications
- Ecosystem fragmentation.

This white paper presents an overview of what is known about the potential impact mechanisms in relation to the 52 HCP species. Based on a separate analysis conducted using exposure-response matrices for each species, the risks of direct and indirect impacts on these species and their habitats are identified and described. This white paper also reviews data gaps and estimates the risk of take. In addition, habitat protection, conservation, mitigation, and management strategies that could avoid, minimize, or mitigate the identified potential impacts are also provided. The key goals of this white paper are to:

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

- Identify the risk of take associated with each of the identified impact mechanisms based on the distribution information.

- Identify cumulative impacts.

- Identify data gaps.

- Identify habitat protection, conservation, and mitigation strategies for each species.

# 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 white paper 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 an effort to compile the best available scientific information to protect these species during the construction, maintenance, and operation of fish screen structures. To accomplish this, WDFW is identifying management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. Because 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 to 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, maintenance, and operation of temporary and permanent fish screens. For the purpose of this white paper, fish screens are considered to fall into two basic categories:

- In-channel or “end-of-pipe” screens on intakes or outfalls
- Off-channel screens.

The off-channel screens are typically associated with diversion canals or similar structures downstream of a separate flow control structure.

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 (and each group is separated by a gray-shaded line in Table 1-1).

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* | SPHS/FSC/FT | Freshwater, Estuarine, Marine |
| White sturgeon | *Acipenser transmontanus* | SPHS | Freshwater, Estuarine, Marine |
| Longfin smelt | *Spirinchus thaleichthys* | SPHS | Freshwater, Estuarine, Marine |
| Eulachon | *Thaleichthys pacificus* | FC/SC | 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 cod | *Gadus macrocephalus* | FSC/SC | Marine (occ. Estuarine) |
| Pacific hake | *Merluccius productus* | FSC/SC | Marine & Estuarine |
| Walleye pollock | *Theragra chalcogramma* | FSC/SC | Marine (occ. Estuarine) |

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

| Common Name | Scientific Name | Status[a] | Habitat |
|---|---|---|---|
| Black rockfish | *Sebastes melanops* | SC | Marine & Estuarine |
| Bocaccio rockfish | *Sebastes paucispinis* | SC | Marine & Estuarine |
| 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-shaded 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 construction, maintenance, replacement, modification, and removal (hereafter collectively referred to as construction and maintenance) of fish screens and, where pertinent, their operation.

- Use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely resulting from the construction, maintenance, and operation of fish screens.

- 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 include 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, while the literature acquisition and review process is described further below.

Literature supporting the best available scientific information was acquired by conducting an extensive search of the available literature using the Thomson Scientific Web of Science (Thomson Scientific Web of Science 2007). This resource provides 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 was also searched (this database includes more than 500 reports pertaining to research conducted by Fisheries Research Institute personnel from its inception to the present). 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 (e.g., consultant reports and textbooks) were found in the personal collections of Herrera staff (the consulting firm working with WDFW to prepare this white paper).

To obtain as much relevant species-specific information as possible, a literature review using the Thompson Scientific Web of Science was conducted to collect information related to the individual stressors for the 52 HCP species. A keyword search of the scientific name and/or common name for each species in Table 1-1 was conducted. For those species where the search returned more than 1,000 references, a few recent citations were selected for inclusion. Species in this category were the five salmon species (sockeye, chum, pink, coho, and Chinook), steelhead, and coastal cutthroat trout. For the remaining species, every reference in the search result was reviewed for the relevance of species-specific information to be included in this white paper. For several species, searches for scientific names and common names returned no references. These species included the margined sculpin, giant Columbia River limpet, great Columbia spire snail, western ridged mussel, river lamprey, longfin smelt, Newcomb's littorine snail, and many of the rockfish species.

To identify data gaps and evaluate the state of scientific knowledge applicable to the potential impacts of fish screens on HCP species and their habitats, the acquired literature was examined to assess the broader issue of how these species use aquatic habitats and how fish screens and their construction, maintenance, and operation may alter species behavior and 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 each fish screen subactivity type. Through this process, a collection of information was assembled on the life history, habitat uses, and the potential impacts that fish screens pose to HCP species.

Reference material from each of the above databases was compiled in an Endnote personal reference database (i.e., 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, and publisher. 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 white paper (Appendix B).

# 4.0 Hydraulic Project Description

The fish screen activity type includes a broad array of possible structural designs intended for use in a variety of applications. These range from small, temporary structures used on a seasonal basis to large, permanent structures associated with agricultural diversions or industrial or municipal water intakes. For the purpose of this white paper, this variety of design types is divided into two distinct subactivity types: in-channel screens, and off-channel screens. Current WDFW and National Marine Fisheries Service (NMFS) design guidelines for fish screens are used to define the range of design types falling under each of these categories (NMFS 2004; WDFW 2001a). The white paper focuses specifically on the impact mechanisms and related stressors caused by the construction, maintenance, and operation of these subactivity types, the potential for exposure to these stressors and resulting effects on the 52 HCP species, and the related risk of take resulting from stressor exposure. This includes the impacts of construction, maintenance, and operation on the environment. Consistent with the other white papers in this series, this assessment considers the worst-case scenario for potential effects resulting from each subactivity type, qualifying the range of effects that are likely to occur for each design type relative to this standard.

It is recognized that fish screens are intended to address some of the environmental deficiencies caused by water intake and diversion systems. However, to fully assess the effects of fish screens, a comparison is made between a stream with a flow control structure and unaltered channel conditions. To assess impact mechanisms, resulting stressors, and biological responses to those stressors, the environmental baseline is considered the unaltered channel condition prior to installation of the structure (i.e., the channel prior to diversion or in-channel system development). This analysis does not address the effects of outfalls, diversion structures, canals, diversion dams, or other related channel modifications or flow control structures commonly associated with in-channel screens and off-channel screens. These sources of environmental impacts are addressed in other white papers, which are incorporated by reference as appropriate. Moreover, this white paper does not address the environmental effects of water withdrawals that these structures may permit.

A description of the two subactivity types and the elements of the existing Hydraulic Code applicable to permitting are provided in the following sections.

## 4.1 Characteristics, Applications, and Descriptions of Fish Screen Subactivity Types

The two fish screen subactivity types as defined for this white paper, in-channel and off-channel screens, include a variety of potential screen design types. Some of these screen designs can be used in either configuration. For the purpose of this analysis, the in-channel screen subactivity type includes the construction, maintenance, and operation of permanent, seasonal, and temporary screens in rivers, lakes, reservoirs, estuaries, and marine waters of the state. These include the typical "end-of-pipe" style screen systems, as well as bankline screen designs. Off-channel screens include both temporary and permanent screen systems located off of the main stream channel, adjacent to or downstream of the flow control structure providing the diversion, within artificially constructed canals. Typical configurations for these two subactivity types are shown in Figure 4-1.

The following sections describe the in-channel and off-channel screen subactivity types, including a description of the typical screen designs currently or potentially in use in the future. Screen designs used in both in-channel and off-channel settings are described.

### 4.1.1 In-Channel Screens

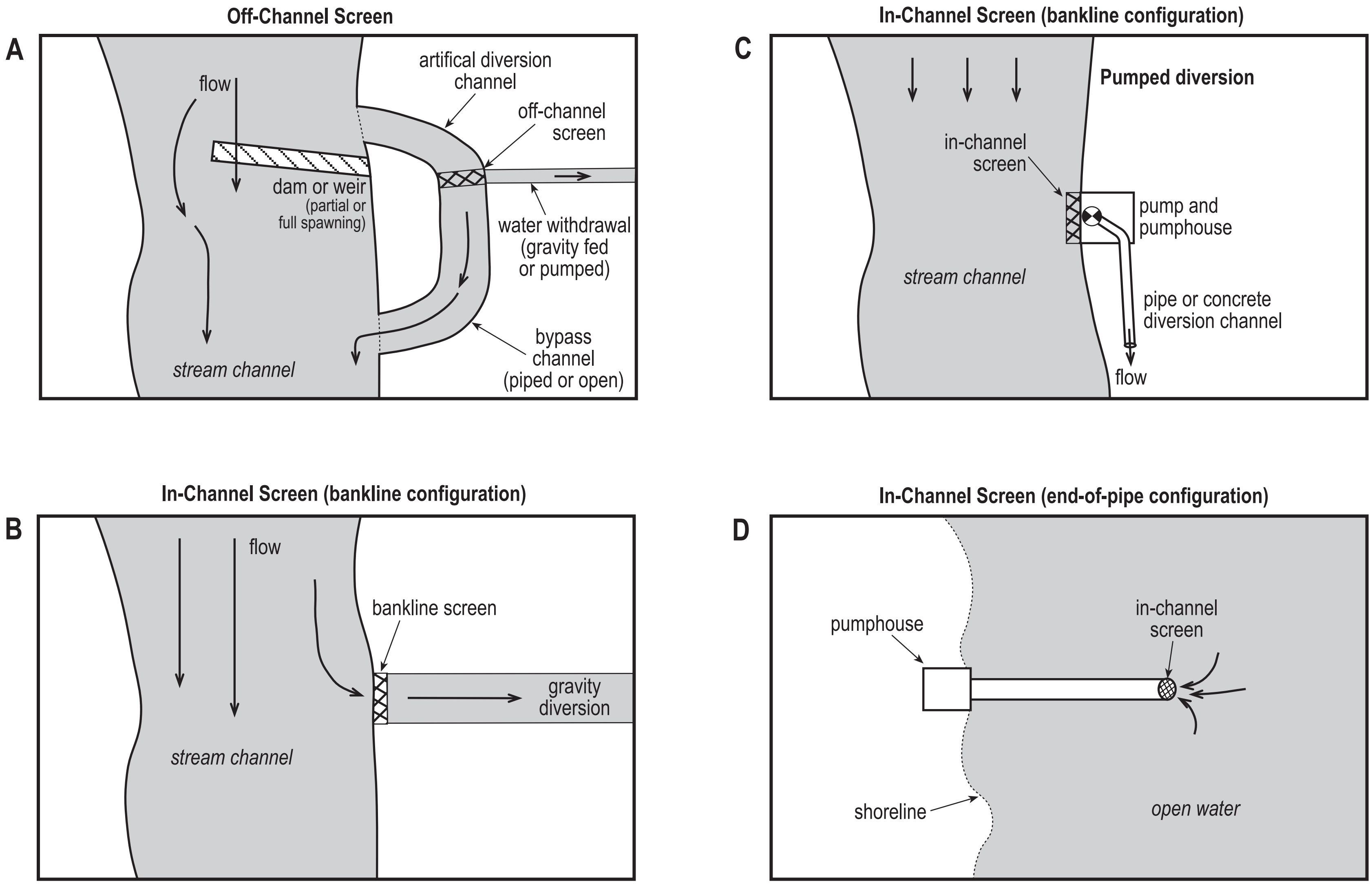
The term "in-channel screens" is a broad subactivity type that best describes both "end-of-pipe" type structures and bankline screens. End-of-pipe structures refer to screens that do not have a flow control device or structure between the screen and the source body. These screens are placed at the mouth of an intake pipe or outfall outlet and prevent the movement or entrainment of fish into or out of the intake or outfall. The scale of this type of screen can vary broadly, ranging from wire mesh screens on small temporary or permanent intake pipes for private water systems, to elaborate screen systems on large water diversion structures such as hydropower penstocks, industrial water intakes, or spillway outlets. Bankline screens refer to fixed or moving plate type screen designs installed flush with the stream bank. This type of structure is employed in a variety of settings. It differs from off-channel screens in that it lies between the source body and the diversion, without an intervening flow control structure such as a dam, weir, or artificial diversion channel.

In-channel screens are used in various environment types, ranging from small lakes and streams to off-shore marine, riverine, or lacustrine environments.

#### *4.1.1.1 End-of-pipe Configurations*

In-channel screens are used in various environment types, ranging from small lakes and streams to offshore marine, riverine, or lacustrine environments. Regardless of the nature of the screened structure, the predominant in-channel end-of-pipe screen design form is a barrier composed of an intake covered by perforated metal, wire or concrete, mesh, or some other permeable material. The designs are typically intended to limit organism entrainment and to diffuse the intake flow velocity to reduce the potential for impingement. However, there are some exceptions to this general rule. In certain cases, screens that incorporate sharp metal grids, grinders, or similar features that are purposefully designed to entrain and kill fish or other organisms may be employed in the outlet structures of flow-controlled lakes or reservoirs to prevent the downstream dispersal of undesirable exotic species.

Summary descriptions of these types of screen designs are provided in Section 4.1.3 (*Typical Screen Designs*), based on information provided in the most recent design guidance from WDFW and NMFS (NMFS 2004; WDFW 2001a).




*Note: In riverine environments, fish screens are employed on both gravity-fed (A & B) and pumped (C) diversion systems. In marine environments, intake systems are commonly constructed on the shore and pipelines extend out into the open water (D). In lacustrine environments, several configurations are possible including those shown in C and D, with type D being favored in most circumstances.*

HERRERA ENVIRONMENTAL CONSULTANTS

**Figure 4-1. Typical in-channel and off-channel screen configurations.**

#### *4.1.1.2 Bankline Configurations*

Bankline screens are screen systems that, as the term implies, are installed flush with the bank, providing a barrier between the diversion canal or gallery and the aquatic environment. This type of screen system commonly does not require an associated bypass channel (although it often includes some form of bypass system). Because of its location in the channel, construction and maintenance of this type of screen structure will impose a greater range of effects on the aquatic environment than a comparable off-channel screen system. Typical screen designs used in bankline configurations include:

- Fixed vertical or inclined plate screens
- Vertical or inclined traveling screens.

Summary descriptions of these screen designs are provided in Section 4.1.3 (*Typical Screen Designs*), based on information provided in the most recent design guidance from WDFW and NMFS (NMFS 2004; WDFW 2001a).

In certain circumstances, particularly when bankline screens are placed in sheltered embayments off the main channel, successful fish exclusion may require incorporation of pumped bypass systems. Such settings lack the necessary head loss to drive flow through a bypass, even if it is provided. In such cases, fish have no guidance away from the face of the screen, can become trapped within the screen chamber, and must be pumped or lifted into bypass systems and returned to the aquatic environment. Bypass systems are more commonly associated with off-channel screen designs. For the purpose of this white paper, the effects of bypass systems are addressed under Section 4.1.2 (*Off-Channel Screens*), with references provided for bankline screens as appropriate.



### 4.1.2 Off-Channel Screens

This subactivity type includes both modular temporary and permanent fish screen designs that are typically used in irrigation canals or similar off-channel diversions. As the description implies, these structures are typically integrated into or directly associated with a flow control structure such as a dam or a weir, as well as an artificial bypass system, either a channel or a pipe, designed to return aquatic organisms and debris back to the main channel.

Off-channel screens are typically constructed in artificial diversions off the main stream channel. Because these locations can be isolated, the risk and extent of construction-related effects on HCP species are low. As noted, bypass systems may be constructed using pipe, or artificial trenches or channels. Bypass systems must be carefully designed to function as intended. Specifically, they must provide adequate sweeping flows to draw organisms safely past the screen and into the bypass and then discharge them safely downstream. Bypass systems must also pass debris without jamming, or it could fail, leading to adverse ecological consequences.

In Washington State, the most common screen designs used in off-channel configurations include the following (WDFW 2001a, Schille 2008):

- Rotary drum screens
- Fixed plate screens (vertical and inclined designs)
- Vertical traveling screens (panel and belt types)
- Modular screens (rotating drum or vertical fixed plate).

Descriptions of these types of screen designs are provided in Section 4.1.3 (*Typical Screen Designs*), based on information provided in the most recent design guidance from WDFW and NMFS (NMFS 2004; WDFW 2001a).

### 4.1.3 Typical Screen Designs

This section provides a description of the typical screen designs used in Washington State.

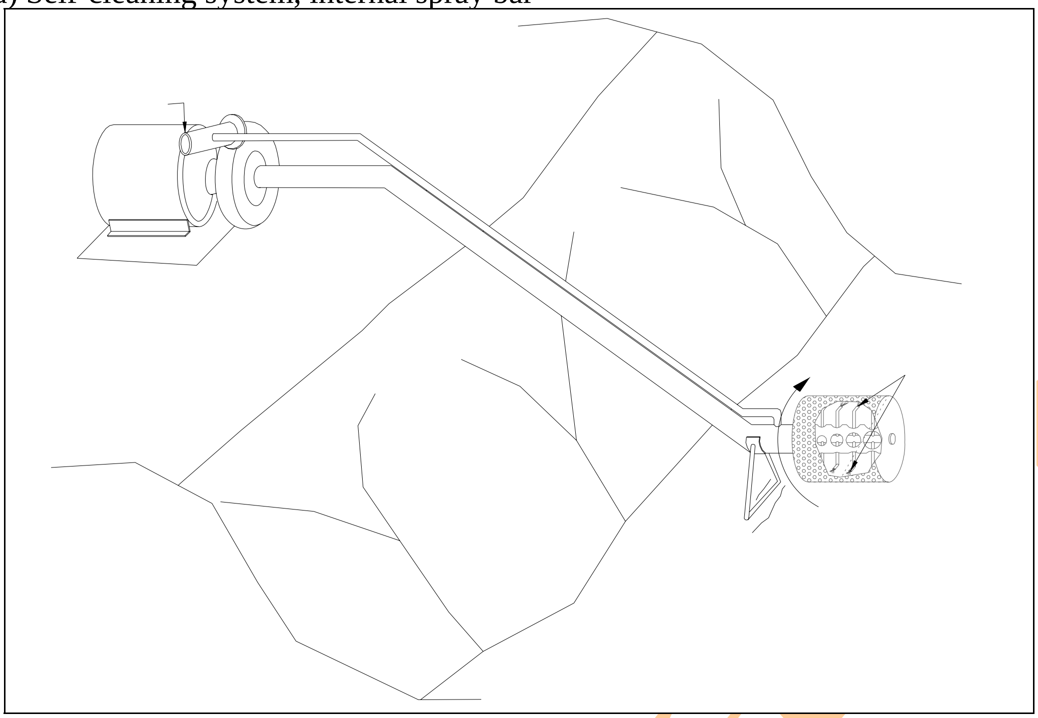
#### *4.1.3.1 End-of-Pipe Screens*

End-of-pipe style screens, also referred to as pump screens or intake screens, are used in a variety of applications. The nature and scale of this type of screen design range from small, relatively simple structures on temporary diversion pumps used for small seasonal water withdrawals, to large, permanent structures associated with large agricultural, industrial, or municipal water intake systems. Many different screen configurations are commercially available that are consistent with current screen guidance (WDFW 2000, 2001a). Example schematics of end-of-pipe fish screens are shown in Figure 4-2.

End-of-pipe screens are typically built in a chamber configuration, typically in a box or cylindrical shape, and attached to the end of a pipe. Smaller designs range in capacity from less than 1 cubic foot per second (cfs) intake capacity for small irrigation pumps, to larger tee-screen designs for intakes with a capacity of 50 cfs or greater. Screen configurations in this category vary depending on the application, with fixed drum and tee-screen designs being common. Removable pump screens associated with temporary diversion systems also fall into this category. Some models for small screens (up to 5 cfs) have extremely efficient water jet clearing systems. Small end-of-pipe screens are commonly used in conjunction with temporary diversion pumps. This type of system is in widespread use in Washington State.

Industrial or municipal water intake systems and power plant cooling water intakes commonly employ large end-of-pipe style screen systems. This type of system is used in riverine, marine, and lacustrine environments in association with high-capacity intake systems. They commonly incorporate an air-burst or water jet debris-clearing mechanism. Large end-of-pipe style screens are typically integrated into the mouth of the intake structure. This type of screen system is commonly used in lacustrine and marine environments. In these settings, the intake and screen system are usually located in deeper water away from nearshore areas used by sensitive organisms.

a) Self-cleaning system, internal spray bar



b) Passive debris clearing system, T-screen

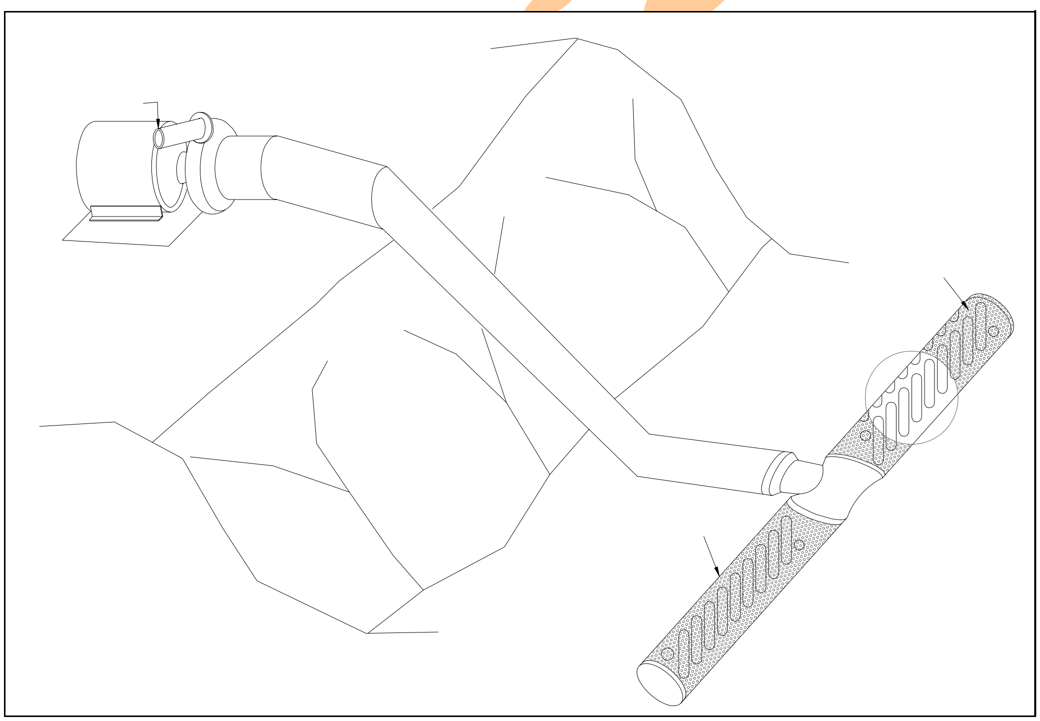


**Figure 4-2. Typical end-of-pipe style fish screens with (a) self-cleaning and (b) passive debris clearing systems.**

The advantages provided by end-of-pipe screens are that they are functional for both deep and shallow water intake systems. The disadvantages of this type of screen are primarily associated with the clearing of debris. This type of screen system requires sufficient ambient water velocity to carry debris away from the screen facility. Air burst clearing systems, the most common system used with in-channel screens, may not adequately remove debris accumulations, especially from the bottom of the screen. For HCP species, this is only problematic when debris accumulation decreases intake diffusion to the point that risk of impingement results. Otherwise, debris accumulation is only a problem for the water user. In-channel screens only operate effectively when fully submerged and intake flow is distributed over the entire surface, meaning that debris accumulation or partial exposure will reduce screen effectiveness. In smaller streams, lack of water depth necessary to fully submerge the screen and the intake system may also limit the effectiveness of the screen.

#### *4.1.3.2 Rotary Drum Screens*

The rotary drum screen is a common type of fish screen used in the Pacific Northwest. The design of the screen is effective because it incorporates both screening and debris removal in a relatively simple configuration. Drum screens can be scaled to accommodate a variety of flows, and they are effective at avoiding impingement and entrainment of juvenile fish. These attributes allow the drum screen to be us

The rotary drum screen removes debris collected on its face through rotation, and the debris is washed off the screen on the downstream side. Screen rotation is achieved by an electric motor, paddle wheel, solar drive, or hydraulic motor. Its most common application is in open channel flow situations, such as irrigation ditches. Using single or multiple drum configurations, rotary drum screens can accommodate a range of diversion rates. In Washington State, they have been used to screen flows ranging from as low as a few cfs up to 3,000 cfs. Drum screens are typically used in conjunction with gravity diversion canals but can also be used to screen water drawn into a pumping gallery. A schematic of a typical rotary drum screen is shown in Figure 4-3.

#### *4.1.3.3 Fixed Plate Screens*

Fixed plate screens include a variety of design types that are distinguished primarily by their orientation to flow. To suit site-specific design requirements, the screen can be oriented with a vertical, upward sloping, or downward sloping aspect relative to the direction of flow. This design is typically employed with gravity diversions, but it can also be used with pump intakes in certain configurations.

The vertical fixed plate screen, which is characterized by intake flow passing perpendicularly through a vertical screen surface, is commonly used for industrial, municipal, and agricultural water supply systems in the Pacific Northwest. This style can be used in either pump or gravity diversion intake configurations. The plate is commonly composed of punched metal or a profile bar, in either aluminum or stainless steel. Woven wire mesh is also used but is less typical, due to its tendency to accumulate debris that is difficult to clear. This design is relatively simple and tends to require less frequent maintenance because there are no moving parts or wear surfaces between the screen mesh and the structural frame. A major disadvantage to the design is that it does not passively clear accumulated debris readily. Typically, the design integrates a mechanical brush, hydraulic backspray, or some other type of debris-clearing system to overcome this limitation. A schematic of a typical vertical plate screen with a mechanical brush system is shown in Figure 4-4.

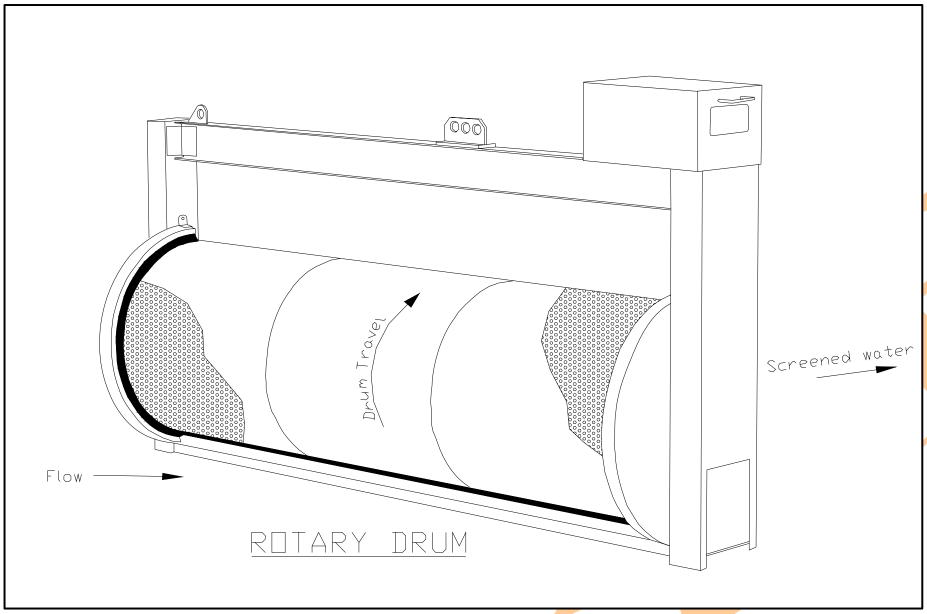



**Figure 4-3. Typical rotary drum screen (Source: Schille 2008).**

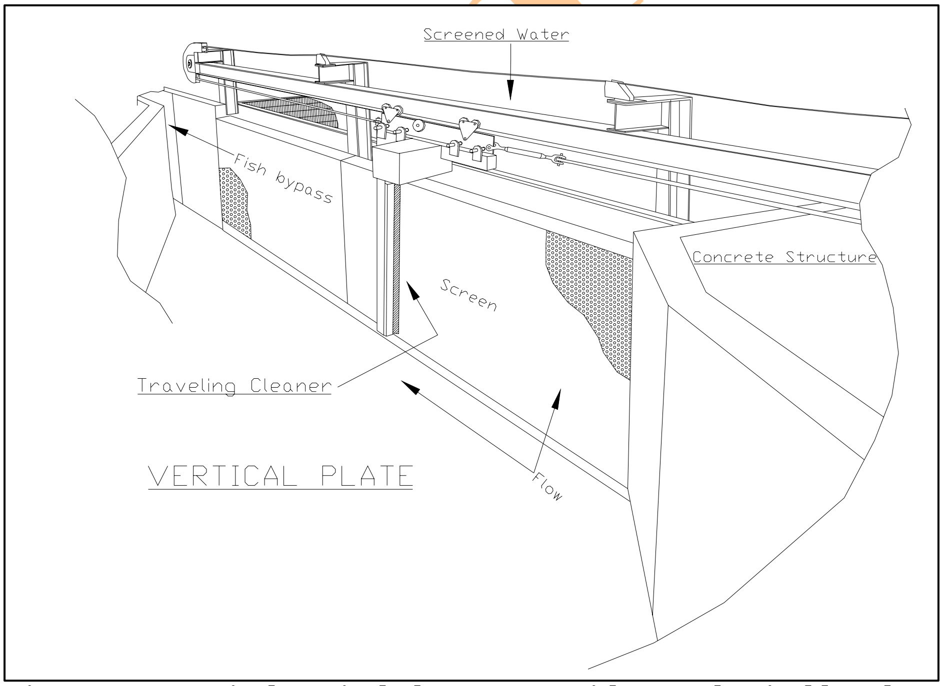



**Figure 4-4. Typical vertical plate screen with a mechanical brush system (Source: Schille 2008).**

Upward and downward sloping screens, much less commonly used, are characterized by diverted flow passing vertically through the inclined surface of the screen. They rely on large quantities of bypass flow to provide passive debris cleaning and avoid fish impingement. For this reason, they are typically used only where the diversion rate is small, relative to total flow. Continuous streamflow across the surface of the screen sweeps fish and debris off the surface of the structure. These designs are typically used in conjunction with a gravity diversion. However, specific sloping screen designs may be paired with pumping galleries and incorporate a bypass channel to return water and fish back to the mainstem channel. Certain sloping screen designs, such as Eicher screens, are used in hydropower systems to direct fish away from turbine intake systems.

Downward sloping screens can either be flat plate or contoured plate style designs (e.g., the Coanda screen). Water is directed from an impoundment created by a flow control structure (e.g., a small dam or weir) over the surface of the screen and into a bypass channel returning to the main channel. A portion of this flow passes through the screen and into the pump or gravity diversion. This type of design relies on the continuous movement of flowing water over the surface of the screen to clear debris and avoid fish impingement. These designs are occasionally used in in-channel settings, but are most commonly used in off-channel configurations. An example schematic of a typical inclined plate screen is provided in Figure 4-5.

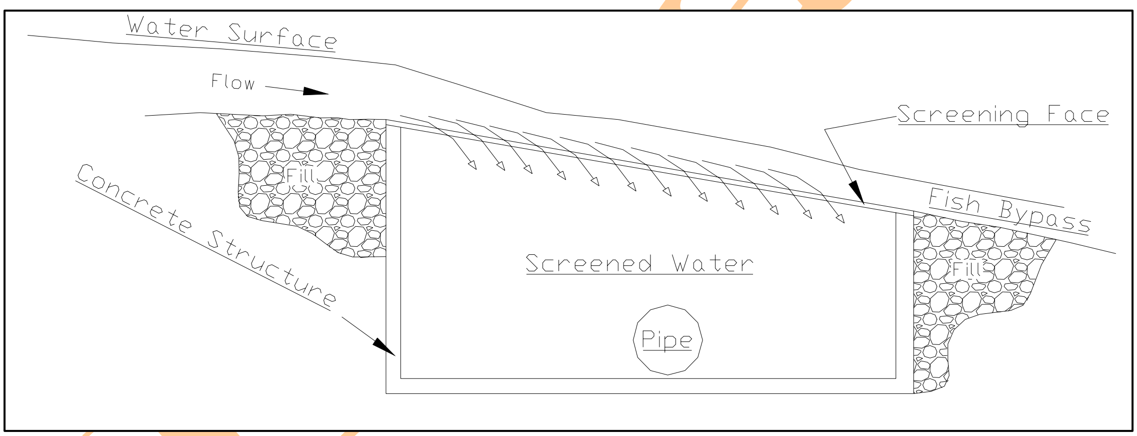



**Figure 4-5. Typical inclined plate screen (Source: Schille 2008).**

Upward sloping screens are quite similar in design except that their profile rises in the direction of the water flow. Excess water flowing over the top of the screen provides fish and debris bypass. Upward sloping screen designs do provide some degree of reliable passive debris clearance; however, they can become overwhelmed by large debris loads. This presents some risk of structural failure as the combined weight of water and debris may overcome the structural strength of the screen support frame. Active clearing systems are sometimes incorporated with these designs to reduce this risk.

The advantage of inclined plate screens is that there are no moving parts and they require no additional in-river diversion structures. Because this screen relies on passive hydraulics to clear debris and provide fish passage, it provides a reliability advantage over mechanical clearing systems. However, screen performance and reliability are highly dependent on precise flow control. Flow rates must be carefully balanced between the required rate of diversion and providing sufficient flow to clear debris and avoid fish impingement. Due to their sensitivity to debris and the need for consistent flow control to maintain performance, upward facing screen designs are typically not permitted in Washington State (Schille 2008); however, the WAC does not specifically preclude their use, meaning there is some potential for such designs to be permitted in the future, and a number of legacy structures are in operation. Examples include Eicher screens integrated into hydropower dams and hatchery water system intakes. Because these structures may be maintained under existing or new HPAs, they are considered in this analysis. Inclined plate screen performance is sensitive to flow control, but they are less prone to debris accumulation and structural failure. Some newer downward facing screen designs, such as the contoured Coanda screen, are considered experimental and may be permitted in certain circumstances.

#### *4.1.3.4 Vertical Traveling Screens*

Vertical traveling screens are similar in concept to rotary drum screens in that the mesh of the screen cycles continuously to remove debris collecting on its face. Two design configurations are commonly used: panel-type screens, with individual mesh panels; and belt-type vertical traveling screens with a continuous mesh belt. Both types of screens are usually driven by electric motors and are commonly used in conjunction with pump diversions. A schematic of a typical vertical traveling screen design is shown in Figure 4-6.

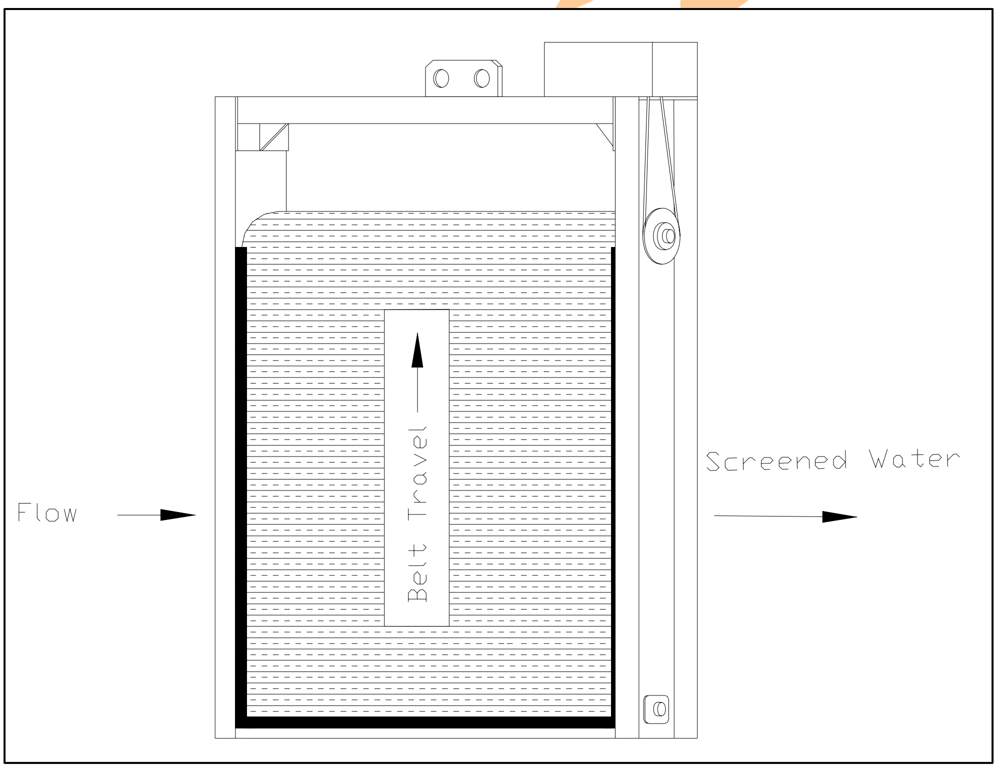



**Figure 4-6. Typical vertical traveling screen (Source: Schille 2008).**

The primary advantage of belt screens is they can be installed in deep water. The screen can be built to any length within the structural capacity of the frame and drive shafts. This design provides effective debris clearing across a range of water depths.

Other advantages of the vertical traveling screens are that they can be installed in bankline configuration (thereby requiring no bypass system); the associated foundation and frame are relatively compact; and they are self-clearing. Additional debris-clearing capacity can be added using jet spray or brush systems if needed.

Vertical traveling screens have historically been constructed with horizontal troughs, or ledges, built onto the face of the screen. The purpose of the troughs is to lift debris and fish with the screen as it rotates. A high-pressure spray bar, near the drive shaft, washes the debris and fish into a stationary trough on the deck of the structure. The debris can then be collected for removal. However, the troughs are problematic for fish protection. Fish entangled with debris and exposed to the spray bar prior to being deposited in the troughs may be injured or killed in the process. Once in the troughs, capture and removal may be difficult, increasing risk of stress and injury. Like upward facing plate screens, this type of screen system would typically not be permitted in Washington State today. However, the WAC currently does not preclude their use, meaning future permitted structures are possible, and a small number of legacy structures are in existence that may require permitting for future maintenance.

#### *4.1.3.5 Modular Screen Systems*

Modular screens are a recent addition to the suite of available screen design options (Schille 2008). Developed in the early 1990s by WDFW at their Yakima Screen Shop, various forms of modular screens are currently in wide use throughout the Pacific Northwest. The modular rotating drum and modular fixed plate systems are the most common forms. Originally designed for remote sites where conventional concrete construction was not feasible, modular screens can be assembled on site and installed in 1 or 2 days. They have proven to be an effective and inexpensive means for addressing numerous small, unscreened diversions. Schematics of the modular drum screen and the modular fixed plate screen are shown in Figures 4-7 and 4-8 respectively.

The modular drum screen is designed for diversions in the 2 to 6 cfs range. This type of system is typically employed in off-channel settings using a piped bypass system to channel fish back to their habitat. They are paddle wheel driven and can be fabricated to provide an angled orientation to flow. The plate screens were developed for diversions in the ½ to 3 cfs range and are used in both in-channel (i.e., bankline) and off-channel settings. The off-channel version uses rotating brushes driven by a paddle wheel to clear debris.

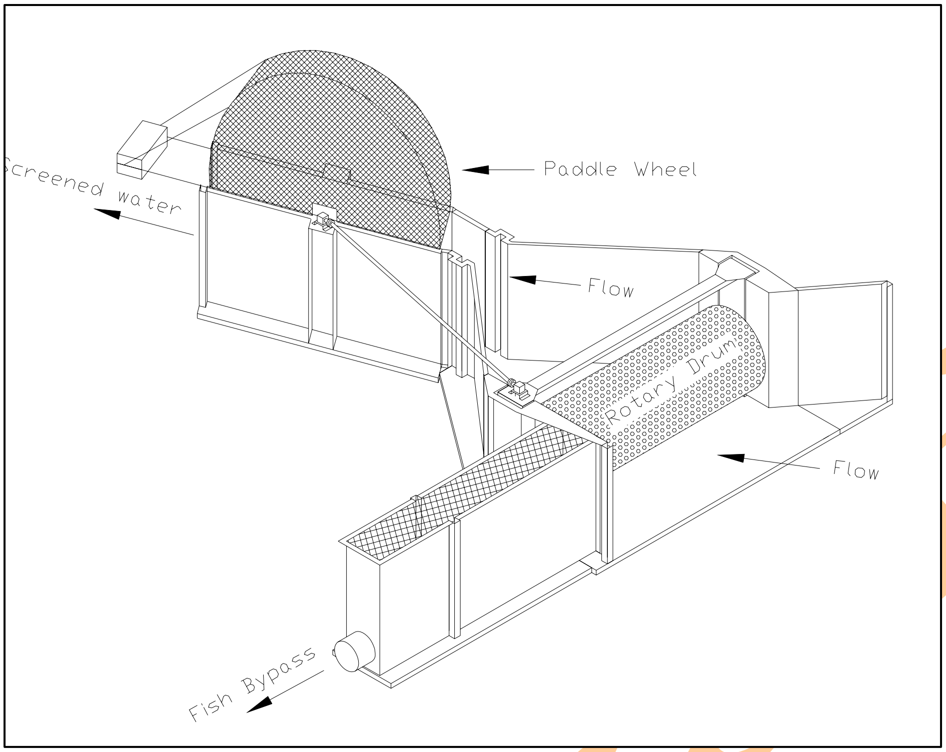



**Figure 4-7. Typical modular drum screen (Source: Schille 2008).**

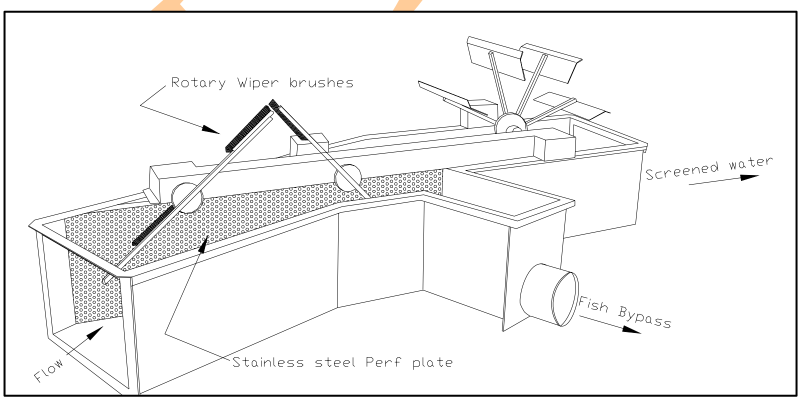



**Figure 4-8. Typical modular fixed plate screen (Source: Schille 2008).**

### 4.1.4 Screening Systems Not Considered in this Analysis

This white paper focuses on structural fish screen systems that are employed in in-channel and off-channel configurations. Other fish screen technologies are in use in Washington State that are not considered in this analysis. These include:

- Behavioral modification using environmental stimuli: Flashing strobe lights, underwater noise, or other forms of disturbance intended to induce avoidance of hazardous areas that are impractical to screen using traditional methods.

- Infiltration galleries: Intake pipes buried within the active channel that use the overlying alluvial bed material as a screen.

These types of screen designs represent a small proportion of the number of fish screen proposals submitted for approval under the HPA program. Behavioral modification using environmental stimuli is considered an experimental approach that is still in development. Infiltration galleries are infrequently used designs that represent a form of channel modification. The effects of this type of structure and the related risk of take are considered similar to those imposed by dredging, which is addressed in the Channel Modifications white paper (Herrera 2007b).

## 4.2 Statutes and Rules Regulating Fish Screens

The Revised Code of Washington (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.” Fish screens are by definition in-water structures that affect the natural flow and other aspects of aquatic ecosystem function and thus meet the definition of hydraulic projects. Fish screens are relatively specialized structures and are typically incorporated with larger hydraulic projects, such as flow control structures or channel modifications.

The mechanisms of impact on HCP species associated with fish screen projects include the long duration impacts associated with structure placement and operational activities, as well as the effects of construction activities that could result in short- to long-term modifications of physical and biological processes. These include modifications to hydraulic and geomorphic characteristics, aquatic and riparian vegetation, changes in water quality that could result in direct and indirect effects on HCP species, and the effects of ecological fragmentation imposed by changes in habitat access for the range of species affected. They also include the effects of fish (and invertebrate) handling, relocation, and exclusion associated with such activities.

The Washington Administrative Code (WAC) sections listed in Table 4-1 are applicable to the listed subactivity types.

**Table 4-1. Hydraulic Code sections potentially applicable to the permitting of fish screen construction, maintenance, and operation.**

| Activity Type | Freshwater WACs | Marine WACs |
|---|---|---|
| In-channel Screens | 220-110-050 (FW banks)<br>220-110-070 (water crossings)<br>220-110-080 (channel change)<br>220-110-120 (temporary bypass)<br>220-110-130 (dredging)<br>220-110-140 (gravel removal)<br>220-110-150 (LWD)<br>220-110-190 (diversions)<br>220-110-223 (lake banks) | No specific existing WACs for fish protection screens<br><br>220-110-250 (habitats of concern)<br>220-110-270 (common)<br>220-110-271 (prohibited work windows)<br>220-110-280 (nonSFRM bank)<br>220-110-285 (SFRM bank)<br>220-110-320 (dredging) |
| Off-channel screens | 220-110-050 (FW banks)<br>220-110-070 (water crossings)<br>220-110-080 (channel change)<br>220-110-120 (temporary bypass)<br>220-110-130 (dredging)<br>220-110-140 (gravel removal)<br>220-110-150 (LWD)<br>220-110-190 (diversions)<br>220-110-223 (lake banks) | Not applicable. |

FW = freshwater; LWD = large woody debris; SFRM = single-family residential marine.

# 5.0 Potentially Covered Species and Habitat Use

This white paper identifies what is known about the effects resulting from construction, maintenance, and operation of fish screen subactivity types on the environment, and the resulting risk of take these effects pose for the 52 HCP species. To understand species-specific impacts, it is necessary to understand the geographic distribution, general life history, and habitat preferences of these species and how these characteristics relate to the subactivity type in question. Table 5-1 provides a general summary of these characteristics and lists the scientific name, Water Resource Inventory Area (WRIA) of occurrence, tidal reference area, and the reproductive patterns and habitat requirements of each of the HCP species.

Knowledge of species-specific habitat needs facilitates the risk of take assessment because the timing, frequency, duration, and magnitude of stressor exposure can be rated against the sensitivity of the species' life-history stages that rely on the affected habitat (see Section 9 [*Potential Risk of Take*] and the exposure-response matrices for each of these species as presented in Appendix A). Once the risk of take has been identified, this information facilitates the identification of measures and guidance that can be used to avoid or minimize risk of take (see Section 11 [*Habitat Protection, Conservation, Mitigation, and Management Strategies*]).

**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)**<br>NOAA Fisheries recognizes eight 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.<br>**Reproduction/Life History**<br>Chinook runs are designated on the basis of adult migration timing:<br>• Spring-run Chinook: Tend to enter fresh water as immature fish, migrate far upriver, and finally spawn in the late summer and early autumn.<br>• 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.<br>• 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.<br>• Fall Chinook: Spawning occurs from late October to early December, with incubation occurring for 1–6 months. Emergence follows, approximately 6 months after fertilization.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction**<br>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.<br>(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)**<br>NOAA Fisheries has recognized three evolutionarily significant units (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 DPSs are listed under the ESA in Washington. Coastal cutthroat trout exhibit varied life-history forms including:<br>• Resident (stays in streams after rearing in their natal streams) – Resident coastal cutthroat trout utilize small headwater streams for all of their lifestages.<br>• Fluvial (migrates to larger rivers after rearing in their natal streams).<br>• Adfluvial (migrates to lakes after rearing in their natal streams).<br>• Anadromous (utilizes estuaries and nearshore habitat but has been caught offshore).<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Spawn in streams with clean, small gravel from March through May. Incubation takes approximately 1–3 months, with emergence occurring between June and July.<br>(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)**<br>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:<br>• Adfluvial (migrates to lakes)<br>• Fluvial (migrates to larger rivers)<br>• Resident (stays in streams).<br>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.<br>**Reproduction/Life History**<br>Spawning: all three life forms spawn in small gravel substrates of tributary streams in the spring (March to July) when water temperature is about $50^{\circ}$F ($10^{\circ}$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.<br>(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)**<br>Widely distributed in Washington; exhibit four life-history types:<br>• Resident (stays in streams after rearing in their natal streams)<br>• Fluvial (migrates to larger rivers after rearing in their natal streams)<br>• Adfluvial (migrates to lakes after rearing in their natal streams)<br>• Anadromous (bull trout in the nearshore ecosystem rely on estuarine wetlands and favor irregular shorelines with unconsolidated substrates).<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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:<br>• Resident (stays in streams after rearing in their natal streams)<br>• Fluvial (migrates to larger rivers after rearing in their natal streams)<br>• Adfluvial (migrates to lakes after rearing in their natal streams)<br>• Anadromous (migrates to marine waters after rearing in their natal streams).<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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 hubbs* | 08–24 | NA | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Harris 1974; Mongillo and Hallock 1999; WDNR 2005, 2006a) |
| Lake chub | *Couesius plumbeus* | 48, 61; other locations unknown | NA | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(WDNR 2005; Wydoski and Whitney 2003) |
| Leopard dace | *Rhinichthys falcatus* | 25–31, 37–41, 44–50 | NA | **General Information (Habitats and Feeding)**<br>In Washington, leopard dace inhabit the bottoms of streams and small to mid-sized rivers, specifically the Columbia, Snake, Yakima, and Simikameen 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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Wydoski and Whitney 2003) |
| Umatilla dace | *Rhinichthys umatilla* | 31, 36–41, 44–50, 59–61 | NA | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>Spawning behaviors are similar to those described for leopard dace, with spawning primarily occurring from early to mid-July.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Wydoski and Whitney 2003) |
| Green sturgeon | *Acipenser medirostris* | 22, 24, 28 | All | **General Information (Habitats and Feeding)**<br>NOAA Fisheries recognizes two DPSs (Distinct Population Segments) 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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Howell et al. 2001; Langer et al. 1977; Lewis et al. 2002; WDFW 2001b; 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)**<br>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.<br>**Reproduction**<br>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.<br>(Gotthardt 2006; WDNR 2005; Wydoski and Whitney 2003) |
| Pacific sand lance | *Ammodytes hexapterus* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Garrison and Miller 1982; Nightingale and Simenstad 2001; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997a) |
| Surf smelt | *Hypomesus pretiosus* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Nightingale and Simenstad 2001; 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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Nightingale and Simenstad 2001; Penttila 2000; Simenstad et al. 1979; WDFW 1997b) |
| Lingcod | *Ophiodon elongatus* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Albers and Anderson 1985; Bargmann 1980; Dunn and Matarese 1987; Garrison and Miller 1982; Hart 1973; Nightingale and Simenstad 2001; NMFS 1990; NRC 2001) |
| Pacific hake | *Merluccius productus* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Bailey 1982; McFarlane and Beamish 1986; NMFS 1990; NRC 2001; Quirollo 1992) |
| Walleye pollock | *Theragra chalcogramma* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Kramer and O'Connell 1995; WDNR 2006a) |
| Bocaccio rockfish | *Sebastes paucispinis* | NA | All | **General Information (Habitats and Feeding)**<br>Adults semidemersal in shallow water over rocks with algae, eelgrass, and floating kelp. Larvae feed on diatoms; juveniles feed on copepods and euphausiids.<br>**Reproduction/Life History**<br>Ovoviviparous spawning occurs year-round, with incubation lasting 40–50 days. Larvae and juveniles are pelagic.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Spawning is ovoviviparous and occurs from January through March. Larvae and juveniles are pelagic.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Ovoviviparous spawning from April through July, with larval release from May to July.<br>(Kramer and O'Connell 1995; WDNR 2006a) |
| Redstripe rockfish | *Sebastes proriger* | NA | All | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>Spawning is ovoviviparous, occurring from January through March. Larvae and juveniles are pelagic.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Ovoviviparous spawning peaks in May and June. Juveniles are pelagic.<br>(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)**<br>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.<br>**Reproduction /Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Ovoviviparous spawning in late fall or early winter, with the larvae released from May to July.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Ovoviviparous spawning from October through December. Incubation is between January and March. Larvae and juveniles are pelagic swimmers.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Baker 1995; Couch and Hassler 1990; West 1997) |
| Northern abalone | *Haliotis kamtschatkana* | NA | 10 | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(Gardner 1981; NMFS 2007a; WDNR 2006b; West 1997) |
| Newcomb's littorine snail | *Algamorda subrotundata* | NA | 14–17 | **General Information (Habitats and Feeding)**<br>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.<br>**Reproduction/Life History**<br>Broadcast spawning in salt marshes. Other reproductive information unknown.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>Broadcast external fertilization. Reproduction timing is unknown.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>They are short-lived, usually reaching sexual maturity within a year, at which time they breed and die. Unknown reproduction timing.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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)**<br>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.<br>**Reproduction/Life History**<br>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.<br>(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 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

The fish screen activity type occurs throughout Washington State in both freshwater and marine/estuarine environments, implying that the geographic extent of potential impacts and the range of HCP species exposed to this activity type are broad. In this white paper, an **impact** is defined as an unnatural disturbance to habitat-controlling factors. Habitat-controlling factors include a variety of ecosystem parameters such as light penetration, stream energy, substrate composition and stability, water quality parameters, littoral drift, or channel geomorphology. These controlling factors determine various aspects of the habitat structure (e.g., sand or cobble substrates, wide or shallow channels). Figure 6-1 illustrates the conceptual framework used in this white paper to identify impacts on HCP species and their habitats from fish screen construction, maintenance, and operation.

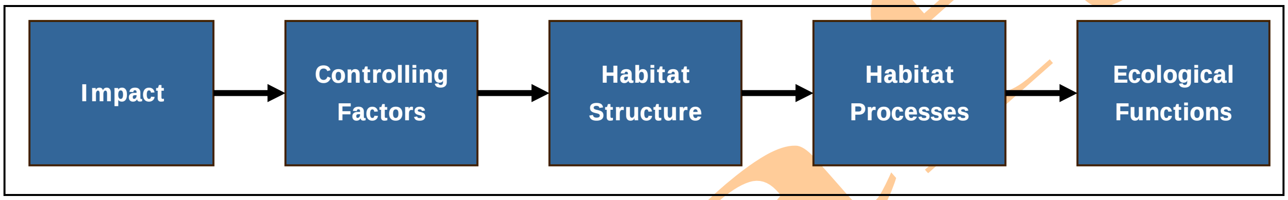



**Figure 6-1. Conceptual framework for assessing impacts (Williams and Thom 2001).**

For the purpose of this white paper, the range of habitat-controlling factors potentially impacted by the fish screen activity type is limited in comparison to other activity types (e.g., marinas, flow control structures, channel modifications). While acknowledging that the effects of flow control structures and channel modifications typically associated with fish screens are broad, this assessment focuses specifically on the effects of screen construction, maintenance, and operation. The broader effects of the flow control and channel modification subactivity types are addressed in companion white papers in this series (Herrera 2007a, 2007b).

Table 6-1 identifies the common **mechanisms of impact** that are known to be associated with the in-channel and off-channel screen subactivity types that are covered in this white paper. This white paper presents what is known about the effects of these mechanisms on HCP species. By identifying the nature and extent of these impacts and the ecological stressors these impacts impose on HCP species, measures can be implemented to avoid and, if avoidance is not possible, to minimize harmful impacts on these species and the habitats that support their growth and survival.

The range of impact mechanisms resulting from fish screen construction, maintenance, and operation, and the number of submechanisms under the impact mechanisms are generally considered more limited than they are for other activity types. The rationale for a more constrained range of impact mechanisms is that the effects of many screen designs are expected to occur as a result of the construction, maintenance, and operation of the flow control structures and/or channel modifications that are associated with fish screen construction. The intent is to avoid overstating the range of environmental effects expected to result from the fish screen activity type, as well as to avoid a duplication of the discussion of impact mechanisms provided in the companion Flow Control Structures and Channel Modifications white papers (Herrera 2007a, 2007b).

**Table 6-1. Impact mechanisms and submechanisms associated with fish screen subactivity types.**

| Subactivity Type(s) | Impact Mechanism | Submechanisms |
|---|---|---|
| In-Channel Screens and Off-Channel Screens | Construction and Maintenance Activities | ▪ Equipment operation and materials placement<br>▪ Dewatering and handling<br>▪ Dredging and fill |
| | Operations | ▪ Visual, physical, and noise-related disturbance<br>▪ Entrainment and impingement |
| | Water Quality Modifications | ▪ Elevated suspended sediments<br>▪ Altered pH<br>▪ Introduction of toxic substances |
| | Hydraulic and Geomorphic Modifications | ▪ Altered flow conditions<br>▪ Altered channel geometry<br>▪ Altered substrate composition and stability |
| Off-Channel Screens | Riparian Vegetation Modifications | ▪ Altered shading and an altered ambient air temperature regime<br>▪ Altered stream bank stability<br>▪ Altered allochthonous inputs<br>▪ Altered habitat complexity<br>▪ Altered groundwater-surface water interactions |
| | Ecosystem Fragmentation | ▪ Passage barriers (e.g., delayed migration)<br>▪ Modified upstream transport of allochthonous nutrients<br>▪ Modified downstream transport of LWD and organic material |

LWD = large woody debris.

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 (as described in Section 3 [*Methods*]). The goals of this framework are to:

- Elucidate impacts associated with each HPA subactivity.

- Determine how those impacts manifest in effects on habitat and habitat functions utilized by the HCP species.

- Develop recommendations for impact avoidance, minimization, and mitigation measures that target the identified impacts.

The analysis process begins with an impact that, in this case, would consist of activities authorized under an HPA for a fish screen 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, sediment transport) 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(s) and therefore on HCP species.

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

This impact analysis serves to identify the direct and indirect impacts that could potentially affect HCP species. 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 impact mechanism and submechanism was initially examined and evaluated to:

- Identify and characterize specific impacts or stressors (i.e., nature and magnitude)

- Evaluate the potential for exposure (potential for species to be exposed = stressor timing/duration/frequency, coincident with habitat use by the various life-history forms of the species in question)

- Identify the anticipated exposure response based on the exposure parameters and life-history specific sensitivity

- Identify measures that could reduce 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 the terms used in this exposure-response 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 life-history stages 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 *[Cumulative Effects]*).

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 and includes 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 on 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, and BMPs) for minimizing or mitigating project impacts or risk of take were developed. (These are presented in Section 11 [*Habitat Protection, Conservation, Mitigation, and Management Strategies*].)

**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 operations or maintenance) | — | 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 5 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 approximately 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 an extended period 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 an 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 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 identifies the range of ecological stressors caused by each impact mechanism, the exposure pathway, and species response to stressor exposure (i.e., the direct and indirect effects of the stressor). This discussion covers the state of knowledge about these issues as reflected in the best available science.

This section also summarizes the available information on each impact mechanism category and impact submechanism where available, and provides specific examples pertinent to the 52 HCP species addressed in this white paper. Note that specific information is not provided for each species. Instead, relevant information on species groupings or species with similar life-history characteristics is used to provide examples of the likely forms of direct and indirect effects that will result from stressor exposure. This section references the specific information provided for each species and species grouping in the exposure-response matrices in Appendix A. The matrices elaborate on the direct and indirect effects caused by stressor exposure and response.

This section is organized by subactivity type, impact mechanism, and submechanism. The distinctions between the direct and indirect effects of these subactivity types in the three environmental settings (i.e., riverine, marine, and lacustrine) are addressed in this text, supported by the assessment of impact mechanism related stressor exposure and response for the 52 HCP species as explicitly discussed in the exposure-response matrices. The matrices, presented in Appendix A, explicitly address the differences in effects on each species and species grouping by habitat type.

Several of the impact mechanisms imposed by in-channel and off-channel screen subactivity types are similar in terms of potential direct and indirect effects under the worst-case scenario. For this reason, this effects analysis incorporates a common discussion of the effects of each impact mechanism and submechanism on HCP species (see Section 7.3 [*Effects of Common Impact Mechanisms and Stressors*]). The effects of each subactivity type are summarized based on their relative magnitude in comparison to the description of common effects. Where the effects of a specific impact submechanism are sufficiently unique, additional discussion of these effects is provided at the subactivity level as appropriate.

As discussed in Section 4 (*Hydraulic Project Description*) and reiterated in Section 6 (*Conceptual Framework for Assessing Impacts*), this white paper considers only the effects of screen construction, maintenance, and operation. Flow control structures or channel modifications (e.g., diversion canals) are not explicitly addressed in this white paper, as the effects of these structures on HCP species have been discussed in detail in the Flow Control Structures and Channel Modifications white papers, respectively (Herrera 2007a, 2007b). Where appropriate, however, information from these white papers is referenced to enhance the information presented here.

# 7.1 In-Channel Screens

As discussed in Section 4.1 (*Characteristics, Applications, and Descriptions of Fish Screen Subactivity Types*), in-channel screen designs predominantly include end-of-pipe structures on water intake systems. These types of structures are typically employed in riverine, marine, and lacustrine environments and are intended to reside below the water surface during operation. The scale of this type of structure ranges from relatively simple designs used on small, private water intake pumps to large, complex screen systems for industrial or power plant water intakes that incorporate air or hydraulic burst systems to clear debris.

## 7.1.1 Construction and Maintenance

The effects of the construction and maintenance of fish screens are expected to be generally similar to those caused by the placement of other types of hard structures in flowing water systems. Specifically, these activities are expected to impose a number of construction-related impact submechanisms, including noise, visual and physical disturbance, stressors associated with dredging and fill and dewatering and handling, and construction-related water quality effects. These construction-related impact submechanisms are expected to be common across both fish screen subactivity types. Therefore, the discussion of common impact mechanisms and stressor response provided in Section 7.3.1 (*Construction and Maintenance)* under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)* is incorporated by reference. The impact submechanisms associated with fish screen construction and maintenance include the following:

- Elevated underwater noise and visual and physical disturbance: Caused by equipment operation and materials placement, and occasional debris removal (see Section 7.3.1.1.1 [*Equipment Operation and Materials Placement*]).

- Dewatering and handling: Capture and relocation of fish (and/or invertebrates) as necessary for construction-related dewatering (see Section 7.3.1.1.2 [*Dewatering and Handling*]).

- Dredging and fill: Associated with construction, maintenance, and/or replacement of the structure (see Section 7.3.1.1.3 [*Dredging and Fill*]).

## 7.1.2 Operations

The effects of fish screen operation are relatively unique in comparison to other HCP activity types. Fish screens are specifically designed to minimize the entrainment of fish into water intakes or diversions at the point where these structures transfer water out of the aquatic environment. These structures operate continuously at this transition point to avoid or minimize the extent to which organisms are adversely affected by their inadvertent removal from the aquatic habitat. While these operational effects are somewhat unique to the fish screen activity type, the impact submechanisms are expected to be common across both fish screen subactivity types. Therefore, the discussion of common impact mechanisms and stressor response provided in Section 7.3.2 (*Operations)* under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)* is incorporated by reference. The impact submechanisms associated with fish screen operations include:

- Visual, physical, and noise-related disturbance: Caused by mechanical operation of the screen unit and related debris-clearing systems (see Section 7.3.2.1.1 [*Visual, Physical, and Noise Related Disturbance*]).

- Entrainment and impingement: Poor screen performance resulting in the removal of organisms from aquatic habitat into the intake; or stress, physical injury, or mortality from entrainment through bypass mechanisms, or contact with the screen surface, debris, or debris-clearing mechanisms (see Section 7.3.2.1.2 [*Entrainment and Impingement*]).

Screen performance is a function of the efficiency of debris-clearing mechanisms and screen maintenance. In general, poor screen performance is typically more of a problem for the water user as it affects the efficiency of water withdrawal. However, impingement risk may increase in circumstances where debris accumulations concentrate inflow to a smaller area of the screen.

### 7.1.3 Water Quality Modifications

Several water quality related impact submechanisms may occur as a result of fish screen development and operation. While both screen subactivity types have the potential to produce similar types of stressors, the likelihood and intensity of these stressors vary between the two. In-channel screens inherently involve more in-channel work, and therefore greater potential for specific water quality related stressors. The construction and maintenance of off-channel screens by definition occurs largely outside of the aquatic environment, and therefore presents less potential for these stressors to occur. However, off-channel screens typically require construction and maintenance of bypass channels and outfall structures, which present the potential for a similar range of water quality related effects, albeit of lesser intensity.

With this understanding, the discussion of common impact mechanisms and stressor response provided in Section 7.3.3 (*Water Quality Modifications*) under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)* is incorporated by reference as appropriate. See the discussion of ecological stressors and related effects on HCP species in the referenced subsections for each of the impact submechanisms presented below.

- Elevated suspended sediments: Temporary or short-term sediment pulses associated with construction, maintenance, and operations (e.g., debris clearing) (see Section 7.3.3.1.2 [*Elevated Suspended Sediments*]).

- Altered pH: Short-term episodes may occur during construction and maintenance as a result of leakage of concrete leachate to surface waters (see Section 7.3.3.1.4 [*Altered pH*]).

- Introduction of toxic substances: Temporary episodes may occur as a result of accidental spills during construction and maintenance, or from failure of mechanical systems during operations (see Section 7.3.3.1.5 [*Introduction of Toxic Substances*]).

### 7.1.4 Riparian Vegetation Modifications

Certain types of in-channel fish screens, specifically bankline screens employing bypass systems, can result in riparian vegetation modifications. Bypass systems commonly take the form of constructed channels or pipe systems that return bypassed organisms and debris to the aquatic environment. The potential effects of this impact mechanism are discussed in Section 7.3.4 (*Riparian Vegetation Modifications*), under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)*.

### 7.1.5 Hydraulic and Geomorphic Modifications

In-channel fish screens are expected to have relatively moderate and localized effects on hydraulic and geomorphic conditions relative to the flow control structures that they are associated with. This type of structure typically has a relatively limited footprint in comparison to the intake pipe or diversion channel.

Hydraulic and geomorphic modification related impact submechanisms associated with in-channel fish screens include:

- Altered flow conditions: Alteration of local hydraulic conditions within the affected reach caused by the physical effects of the structure on flow or current conditions. These effects are expected to be modest.

- Altered channel geometry: Caused by bank or channel bed hardening to accommodate the structure.

- Altered substrate composition and stability: Potential change in depositional conditions adjacent to the structure due to effects on channel geometry and flow conditions.

For the purpose of this white paper, the hydraulic and geomorphic impact submechanisms imposed by in-channel screens (and related effects on HCP species) are expected to be integral to those described for the intake subactivity type in the Flow Control Structures white paper (Herrera 2007a). The reader is referred to that white paper for discussion of potential effects on HCP species. The nature of these effects will vary depending on the scale of the intake and screen structure in question. Large, permanent screen structures associated with large intake systems will by nature impose effects on the higher end of the range described in comparison to small structures associated with temporary seasonal diversions.

### 7.1.6 Ecosystem Fragmentation

Fish screens can be associated with a number of forms of ecosystem fragmentation. Many of these are inextricably linked to water withdrawal, and are therefore not discussed in further detail here. Water withdrawals are regulated by Ecology and are not subject to approval under the HPA program. Depending on its configuration, the intake or diversion structure employing the in-channel screen may cause some form of ecosystem fragmentation. The extent of the related ecological stressors and the resulting effects on HCP species caused by intake and diversion structures are addressed in the Flow Control Structures white paper (Herrera 2007a) and are not discussed in detail here. Within this narrower context, the degree to which in-channel fish screens can cause ecosystem fragmentation becomes more limited. Essentially, in-channel fish screens are not expected to impose any incremental ecosystem fragmentation beyond what is imposed by the withdrawal of water or the intake or diversion structure for most in-channel screen types. Certain types of bankline screens are an exception, as described below.

#### *7.1.6.1 Impact Submechanisms*

Intake systems employing bankline screens in sheltered alcoves or embayments in riverine, marine, or lacustrine environments may be associated with certain ecosystem fragmentation effects. Specifically, these types of screen systems commonly employ pumped bypass systems because there is insufficient hydraulic head to operate a gravity-driven bypass. Planktonic eggs and larvae, or weak swimming or behaviorally driven fish species, may be drawn into these embayments by the inflow and become trapped (Bates 2008). Depending on the location and effectiveness of the bypass system, organisms may be drawn repeatedly into the embayment area. This may result in delayed migration or hinder dispersal to suitable rearing habitats with the potential to adversely affect HCP species.

#### *7.1.6.2 Effects on Fish and Invertebrates*

Many HCP 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 productive rearing areas. Similarly, HCP species such as eulachon and longfin smelt have larval life-history stages that are dependent on current-driven transport to estuarine rearing areas. Highly fecund species that produce spatially variable planktonic spawn rely on these transport and retention mechanisms for reproductive productivity (Hernandez-Miranda et al. 2003; Rooper et al. 2006; Sinclair 1992). Intake systems in marine, lacustrine, and riverine environments have the potential to trap and retain planktonic eggs and larvae in less desirable rearing areas, contributing to elevated mortality. Bankline and other in-channel screen systems (e.g., Gunderboom screens) attempt to limit this mortality by reducing entrainment-related mortality by bypassing organisms back to the aquatic environment. This requires entrainment through and/or impingement on bypass systems, the effects of which are described in Section 7.3.2.2 (*Effects on Fish and Invertebrates*).

Ecosystem fragmentation occurs when organisms are trapped in embayments by ineffective screening and bypass systems, limiting dispersal to favorable rearing areas. Similarly, screen systems may bypass planktonic or weak-swimming organisms but return them to locations where prevailing currents draw them back into the intake embayment. This is likely to lead to elevated mortality through predation, starvation, unfavorable water quality conditions, or a combination of these effects (Sinclair 1992). Fish species that migrate along nearshore marine and lacustrine environments, such as juvenile anadromous salmonids, that are drawn repeatedly into bypass systems may experience delayed migration, with attendant effects on survival, growth, and fitness.

## 7.2 Off-Channel Screens

Off-channel screens are typically associated with water diversions in riverine settings. This subactivity type includes a diversity of design types, which can be employed in all environment types (marine, lacustrine, and riverine habitats). However, this type of design is almost exclusively associated with agricultural water diversions in riverine environments. This is due to the fact that off-channel screen designs use bypass systems that require dedicated streamflows to return organisms and debris to the aquatic ecosystem. As the name implies, off-channel screens are constructed and operated largely outside of the aquatic environment (i.e., within an artificial diversion system). Because of these characteristics, the impact mechanisms they impose and the severity of related stressors differ from those imposed by in-channel screens.

### 7.2.1 Construction and Maintenance

Construction of off-channel screens takes place within the artificial diversion system, outside of the aquatic environment. As such, screen construction typically takes place “in the dry,” resulting in few construction-related impacts. Screen maintenance activities may require dewatering of the diversion system. Because the dewatered area is limited primarily to the artificial diversion, potential effects on HCP species are limited (but not entirely negated). Some in-channel construction may be required for certain elements of the screen system (e.g., erosion protection at the bypass system discharge point). Where in-channel construction and maintenance activities occur, the related impact submechanisms are expected to be generally similar to those associated with in-channel screens. Therefore, the discussion of common impact mechanisms and stressor response provided in Section 7.3.1 (*Construction and Maintenance)* under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)* is incorporated by reference. See the discussion of ecological stressors and related effects on HCP species in the referenced subsections for each of the impact submechanisms presented below.

- Elevated underwater noise and visual and physical disturbance: Caused by equipment operation and materials placement, as well as occasional debris removal (see Section 7.3.1.1.1 [*Equipment Operation and Materials Placement*).

- Dewatering and handling: Capture and relocation of fish (and/or invertebrates) as necessary for construction and during routine maintenance (see Section 7.3.1.1.2 [*Dewatering and Handling*]).

- Dredging and fill: Associated with construction and/or replacement of the structure (see Section 7.3.1.1.3 [*Dredging and Fill*]).

When interpreting this information, it is important to note that the likely extent of construction and maintenance related impacts for off-channel screens will be at the lower ends of the ranges discussed for the reasons identified above. Furthermore, in the case of off-channel screens, these effects are limited to the freshwater HCP species.

### 7.2.2 Operations

As discussed in Section 7.1.2 (*Operations*), the impact submechanisms associated with fish screen operation are expected to be generally similar across both fish screen subactivity types. Therefore, the discussion of common impact mechanisms and stressor response provided in Section 7.3.2 (*Operations)* under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors)* is incorporated here by reference. The impact submechanisms associated with fish screen operations include:

- Visual, physical, and noise-related disturbance: Caused by mechanical operation of the screen unit and related debris-clearing systems (see Section 7.3.2.1.1 [*Visual, Physical, and Noise Related Disturbance*]).

- Entrainment and impingement: Poor screen performance resulting in the removal of organisms from aquatic habitat into the diversion or intake or trash collection channels; or stress, physical injury, or mortality from contact with the screen surface, debris, or debris-clearing mechanisms, or travel through bypass channels (see Section 7.3.2.1.2 [*Entrainment and Impingement*]).

There are, however, some important distinctions with regard to the function of off-channel screens that require further discussion. Specifically, some off-channel screen designs incorporate bypass systems with the potential for operational effects that extend beyond those associated with in-channel screen designs. These distinctions are discussed in Section 7.3.2 (*Operations*). Furthermore, the operational effects of off-channel screens are limited to freshwater HCP species and anadromous HCP species during freshwater life-history stages.

### 7.2.3 Water Quality Modifications

Water quality related impact submechanisms are generally similar across both fish screen subactivity types. Therefore, the discussion of common impact mechanisms and stressor response provided in Section 7.3.3 (*Water Quality Modifications*) under Section 7.3 (*Effects of Common Impact Mechanisms and Stressor)* is incorporated by reference as appropriate. See the discussion of ecological stressors and related effects on HCP species in the referenced subsections for each of the impact submechanisms presented below.

- Elevated stream temperatures: May occur when bypass system operation affects low flow conditions in source body. Rapid dewatering of bypass channels occupied by HCP species when diversions are shut down may cause exposure to stranding and elevated water temperatures.

- Decreased dissolved oxygen: May occur when bypass channels are dewatered, as described above.

- Elevated suspended sediments: Temporary or short-term sediment pulses associated with construction, maintenance, and operations (e.g., debris clearing), or screen and/or bypass failure (see Section 7.3.3.1.2 [*Elevated Suspended Sediments*]).

- Altered pH: Short-term episodes may occur during construction and maintenance as a result of leakage of concrete leachate to surface waters (see Section 7.3.3.1.4 [*Altered pH*]).

- Introduction of toxic substances: Temporary episodes may occur as a result of accidental spills during construction and maintenance, or from failure of mechanical systems during operations (see Section 7.3.3.1.5 [*Introduction of Toxic Substances*]).

When interpreting this information, it is important to note that the magnitude and occurrence of water quality effects potentially caused by off-channel screens differ from in-channel screens. Specifically, because they are constructed within artificial diversions, commonly in the dry, off-channel screens are less likely to contribute to construction-related water quality impacts in comparison to in-channel screens. In contrast, unlike in-channel screens, off-channel screens are constructed in artificial channels, and the performance of the entire system is dependent on effective flow control and debris clearance. If a screen clogs with debris and is overtopped, or high flows overwhelm channel and screen capacity, the entire system could fail, leading to extensive upland and bank erosion with direct delivery of elevated suspended sediments to the stream channel. This could occur at levels consistent with the higher end of the range of effects discussed in Section 7.3.3.1.2 (*Elevated Suspended Sediments*). This potential should be considered when interpreting the potential for effects on HCP species.

### 7.2.4 Riparian Vegetation Modifications

By design, off-channel screen systems require the use of bypass systems, commonly in the form of constructed channels or pump or gravity-fed pipes that return bypassed organisms and debris to the aquatic environment. The construction of these bypass systems requires work in the riparian zone, leading to potential modification of riparian vegetation. The potential effects of this impact mechanism are discussed in Section 7.3.4 (*Riparian Vegetation Modifications*), under Section 7.3 (*Effects of Common Impact Mechanisms and Stressors).*

### 7.2.5 Hydraulic and Geomorphic Modifications

*Note to reviewers of this March 2008 working draft version of the white paper: WDFW and Herrera staff disagree whether there is adequate evidence to support that the types of geomorphic modifications discussed in this section could actually result from a fish screen bypass. WDFW will have this white paper independently peer reviewed and direct those reviewers to specifically state their opinions on the validity of this discussion in relation to fish screens. This section will then be finalized based upon the input of the peer reviewers. For this reason, this section has been formatted with highlighted text.*

Off-channel fish screens are used almost exclusively in riverine environments. Riverine hydraulic and geomorphic processes distribute water, sediment, and organic material along a linear path toward lower elevations. Fishes and invertebrates depend upon the diversity of habitats created by hydraulic and geomorphic forces that scour, transport, and deposit diverse sediments, large woody debris (LWD), nutrients, and organic material along the river profile (Montgomery et al. 1999). HCP species, such as sturgeon, char, bull trout, salmonids, and freshwater mussels, depend on particular riverine sediment types and habitats. In short, the reproduction, growth, and survival of these HCP species depend on particular hydraulic and geomorphic regimes to maintain suitable habitats. Alterations to river form that change the flow of water and the ability of the water to move sediments, LWD, and organic material can have direct and indirect effects on HCP species.

Fish screens are generally expected to have relatively minor effects on hydraulic and geomorphic processes relative to other HCP-permitted activities, such as the flow control structures and channel modifications they are typically associated with. This is particularly true in the case of off-channel screens, as these structures are, as the name implies, located out of the aquatic environment within artificially constructed diversion features. This orientation limits the extent to which off-channel screens can result in hydraulic and geomorphic modifications; however, there are pathways through which potential effects can occur. This section describes the impact submechanisms and ecological stressors that could result through these pathways, applying a worst-case scenario perspective.

#### *7.2.5.1 Impact Submechanisms*

Due to the fact that off-channel screens are constructed and operated within artificial diversions away from the natural aquatic environment, the extent of potential hydraulic and geomorphic modifications caused by the screen structure itself is limited. As such, the resulting range of stressors is similarly limited. However, off-channel screens by necessity incorporate bypass systems, which can impose ecologically meaningful hydraulic and geomorphic effects in specific circumstances. Operation of the bypass system requires the diversion of an additional increment of flow above and beyond the amount diverted from the system for consumptive use, meaning that that flow is unavailable to the aquatic ecosystem until it is discharged.

In most cases, the length of channel affected will be relatively small, and these effects would be considered insignificant. Moreover, because fish screens are designed and sized to specific flow requirements, they serve to limit accidental or intentional diversions in excess of permitted uses. This may mitigate any hydraulic and geomorphic effects resulting from bypass system operation. However, in a worst-case scenario, the discharge point may be a considerable distance downstream from the diversion, meaning that some length of instream habitat is affected by an incremental loss in flow. Because there is an operational element to this effect, its timing and duration may be variable as well. These effects can alter flow conditions within the affected reach, and the range of flow variability that occurs during active water diversion.

The following hydraulic and geomorphic modification impact submechanisms may result from implementation of a fish screen subactivity type:

- Altered flow conditions
- Altered channel geometry
- Altered substrate composition and stability
- Altered habitat complexity.

These impact submechanisms and the ecological stressors they can impose are described in the following sections. When interpreting this discussion, it is important to recognize that these impact submechanisms are likely to occur only in certain circumstances and through the specific pathways described. Discussion of the potential effects of resulting stressor exposure on HCP fish and invertebrate species follows.

#### *7.2.5.1.1 Altered Flow Conditions*

As noted, off-channel screens incorporate flow bypass systems that require an additional increment of streamflow for operation. Screens having longer bypass systems will make this additional diverted flow unavailable to the aquatic environment throughout the bypassed reach. In some cases, the length of stream affected can be significant. Bypass channels can range from tens to hundreds or even thousands of feet in length. In smaller stream systems, bypass channels may require a relatively larger proportion of the natural streamflow for effective operation, on the order of 1 to 10 percent of the baseflow (Schille 2008), denying these flows to the active channel. This has the potential to impose a range of flow-related effects, with increased stream temperatures being a primary concern in temperature-limited systems.

Altered flow conditions could also affect flow velocity, when the diversion is a significant proportion of the flow. However, because the proportion of streamflow used as bypass flow is relatively small, on the order of 5 to 10 percent of the diverted flow (Schille 2008), the potential for measurable operational effects on flow velocity are negligible and insignificant from an ecological perspective. However, under certain circumstances, flow reductions of this magnitude could result in persistent changes in channel configuration within the bypassed segment and the channel downstream of the bypass outfall. These changes may alter flow conditions in the affected reach, especially during higher flow conditions. These flow alterations could have ecologically meaningful effects on HCP species. The mechanisms through which this can occur are discussed in Section 7.2.5.1.2 (*Altered Channel Geometry*).

It is important to place these potential effects into context with the broader effects of flow diversions on hydraulic and geomorphic conditions, and the benefits provided by fish screens. In flowing water systems, diversion of water for consumptive uses is expected to have a much larger effect than the relatively small incremental effect of flow diversion for bypass system operation. In addition, because screen systems are designed to operate under a controlled range of diversion flows, they tend to limit withdrawals that exceed (intentionally or unintentionally) the water right at the diversion point. As such, they provide a mechanism for maintaining base flows that may outweigh any incremental effect from bypass system operation.

In addition, bypass channels may have intended or unintended beneficial effects on flow conditions in certain circumstances. In many stream environments, hydromodification and water withdrawals have limited the quality and quantity of off-channel habitats. Depending on their configuration and accessibility, constructed bypass channels may function similarly to watered side channels, providing increased habitat area. Bypass systems may also discharge to existing side channels, supplementing streamflows in these habitats. Juvenile salmonids and other fish species have been observed rearing in active bypass channels (Schille 2008). However, this beneficial condition may impose an “attractive nuisance” problem. Bypass flows in many cases represent the majority or even the entirety of streamflow in bypass channels. When the diversion is shut down streamflows in the bypass can decrease rapidly, creating a stranding hazard (Bates 2008). This potential problem is well recognized, and dewatering of bypass channels that function as rearing habitat during normal screen operations is not permitted under the HPA program. However, fish screen operation presents at least some potential for stranding. Stranding in turn presents the potential for exposure to elevated water temperatures and decreased DO concentrations, and/or dewatering. The effects of exposure to elevated water temperatures and decreased DO are discussed in Section 7.3.3 (*Water Quality Modifications*). The effects of stranding and dewatering on HCP species are discussed in Section 7.3.1.1.2 (*Dewatering and Handling*).

##### *7.2.5.1.2 Altered Channel Geometry*

Depending on size, configuration, and ecological setting, the diversion of water by off-channel fish screens has the potential to influence channel geometry. The circumstances where this effect can occur are characterized as follows: (1) a bypass system diverts an additional increment of streamflow (in addition to the consumptive use) for operation; (2) the bypass system extends over a significant length of the stream channel; and (3) the bypass flow represents a measurable proportion of the remaining baseflow in the stream. The mechanism through which altered channel geometry can occur is flow-induced changes in vegetation encroachment.

Vegetation plays a key role in the channel form of natural streams. A number of studies have investigated the relationship between flow conditions, riparian vegetation establishment, and channel form (Francis 2006; Gran and Paola 2001; Paola et al. 2006; Rowntree and Dollar 1999). Collectively, these studies have shown that vegetation tends to increase stream bank cohesion, enhance deposition in the vegetated areas, and decrease the width-to-depth ratio of channels between vegetated patches (Francis 2006; Gran and Paola 2001). Vegetation encroachment into the active channel of small streams has been observed in association with water withdrawals at small diversions (Poff et al. 1997; Bohn and King 2000; Stamp and Schmidt 2006). Invasive species, such as reed canarygrass, are particularly capable of exploiting opportunities to encroach upon the stream channel (Richards et al. 2002). Once established, they trap sediment and encourage further expansion of vegetation, leading to measurable effects on the evolution of channel form. The dominant geomorphic effect of reed canarygrass is to add bank cohesion through the growth of an extensive root mat. As a result of the increased bank cohesion, width-to-depth ratios decrease, and bank erosion and habitat-forming processes such as LWD recruitment and channel migration occur less frequently. The encroachment of aquatic vegetation (such as watercress) can increase in-channel roughness and promote local sediment deposition within the active channel. The mechanisms by which these processes occur are generally associated with the interplay between soil cohesion, flow resistance, sediment supply (both rate and grain-size distribution), and the tractive forces available for sediment transport. Sediment deposition in turn can be expected to notably change the dynamics of community succession. This occurs principally through the ability of vegetation to trap sediment and form an in-channel bar (Clary et al. 1996). Indeed, any change in vegetation encroachment may have a positive feedback effect, in that sediment deposition leads to greater plant growth, leading to greater flow resistance and bar stability, leading to more sedimentation, and so on (Francis 2006).

Once these positive feedback processes are initiated, the channel morphology is forced to evolve along a new trajectory. As riparian and aquatic vegetation becomes more established, the additional root cohesion can result in channel narrowing and an increase in flow velocities. The increased roughness provided by bank vegetation and channel narrowing can decrease the conveyance of the channel and affectively reduce the bankfull discharge, thereby decreasing the erosive power of flood flows and their effectiveness at eradicating the vegetation.

Channel responses to flow reductions and vegetation encroachment are complex and variable depending on a range of site-specific factors such as the degree of peak flow reduction due to flow control, reach location in the watershed, the extent of vegetation encroachment, the plant species involved, substrate and bank materials, and the threshold flows necessary for bed and bank erosion following vegetation establishment (Bohn and King 2000; Cluett 2005). The channel response may range from reduced channel capacity and sedimentation to narrowing and incision. The effect of vegetation encroachment on sediment transport has potentially the greatest rammifications for channel evolution. Increased sediment deposition driven by vegetation encroachment can lead to changes in channel configuration downstream of the area of effect. In extreme cases, increased sediment accumulation in the vegetated channel could lead to starvation and downcutting in downstream areas. This in turn would have the effect of increasing channel gradient between the upstream and downstream limits of vegetation encroachment, causing concomitant changes in channel form.

In extrapolating observations of channel change at diversions from small streams to the potential for a similar geomorphic response as a result of bypass operation, a key conclusion can be collectively drawn from these studies. The process of flow-mediated vegetation encroachment is nonlinear. Small changes in base flow conditions can have a marked influence on the process of vegetation establishment, particularly in circumstances when the baseflow is the threshold necessary to inhibit vegetation encroachment. Once vegetation encroachment occurs, it can in specific circumstances initiate positive feedback mechanisms that encourage additional vegetation establishment sufficient to alter the trajectory of channel evolution. In addition, once established, vegetation encroachment can be persistent, resisting erosion and removal by "channel resetting" flood flows (Cluett 2005), meaning that these changes in channel morphology can be persistent.

While vegetation encroachments effects are expected to be driven primarily by the consumptive use of water, these findings suggest that flow reductions for bypass system operation could cause similar effects in bypassed reaches in cases where operational flows represent a measurable component of total streamflow. When considered relative to the potential for bypass flows to represent 5 to 10 percent of baseflow, this suggests at least the possibility for bypass system operation to produce conditions that could exacerbate vegetation encroachment, and perhaps causing it to occur.

Therefore, off-channel screens must be considered to have at least some potential to induce localized changes in channel geometry in circumstances where bypass flows are sufficiently large over a sufficient length of stream to allow for vegetation encroachment. Under these types of circumstances, changes in flow velocity, sediment transport, and habitat complexity may occur that are large enough to have ecologically meaningful effects. While it is understood that relatively frequent, high-flow events (such as the bankfull flow) are the dominant control on channel geometry, the influence of these channel-forming processes on channel form are affected by antecedent conditions (such as vegetation encroachment) that prevail during low-flow conditions.

##### *7.2.5.1.3 Altered Substrate Composition and Stability*

The effects of fish screens on sediment composition and stability are expected to be generally minimal in most cases due to the limited physical footprint of these structures. Again, off-channel screen designs with bypass channels are an exception under the specific circumstances discussed in the previous section. Vegetation-induced alterations in channel morphology may in turn result in localized changes in sediment transport capacity within the bypassed reach. Vegetation may encourage increased deposition in the affected reach, while causing bed degradation in downstream reaches due to reduced sediment transport. This can cause changes in channel gradient and, in extreme cases, eventual headcut migration, with subsequent effects on floodplain connectivity.  This can lead to changes in substrate conditions throughout the affected reach as channel morphology changes.

Because HCP species depend on the presence or absence of particular substrate types to support important life-history functions, changes in substrate composition can have direct and indirect effects on those species.

##### *7.2.5.1.4 Altered Habitat Complexity*

Alteration of hydraulic and geomorphic processes caused by vegetation encroachment (as well as modifications of riparian and aquatic vegetation) associated with fish screen development can lead to changes in habitat complexity.  As with all fish screen related impact mechanisms, the magnitude of effects imposed by fish screens on this ecological parameter must be considered relative to the effects of the diversion or intake structure, and the withdrawal of water for consumptive uses.  Furthermore, the incremental effect of a screen on habitat area and complexity is relatively small in comparison, limited in most cases to the reach between the diversion and the bypass discharge point.  However, under a worst-case scenario, these effects could be significant within the bypassed reach, and may be more broadly significant when the cumulative effects of multiple screens are considered.  The effects of flow control structures (i.e., diversions or intake systems) on habitat complexity are discussed in the Flow Control Structures white paper (Herrera 2007b).

#### ***7.2.5.2 Effects on Fish and Invertebrates***

Fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, and foraging.  For example, Chinook salmon tolerate velocities up to 49.9 ft/sec (15.2 m/sec) (Johnson et al. 2003) during migration, whereas Pacific lamprey seek out slower velocities (0–0.33 ft/sec) for rearing (Stone and Barndt 2005).  Optimal velocities for spawning habitat for mountain suckers in Lost Creek, Utah, are 2.4–7.9 in/sec (0.06–0.2 m/sec) (Wydoski and Wydoski 2002).  Spawning velocities for Columbia River white sturgeon are similarly low (~2.6 ft/sec [0.8 m/sec]) (Paragamian et al. 2001), although this species spawns successfully in areas with higher average velocities by using riverbed dunes and similar features for hydraulic refuge (Young and Scarnecchia 2005).

A principal concern associated with the streamflow-related effects of off-channel screens is the adoption of bypass channels by HCP species when the bypass is in operation.  As noted, in certain cases bypass channels can serve functionally as side-channel habitat, particularly in hydromodified systems where these features have been lost.  Anecdotal observations confirm the use of bypass channels by rearing juvenile salmonids and other aquatic species (Schille 2008; Bates 2008).  Rapid dewatering of these channels when the diversion is shut down can lead to direct mortality from stranding, elevated water temperatures, and asphyxiation (Bates 2008).  The recognition that bypass channels can provide valuable habitat in some environments has resulted in operational changes, in circumstances where bypass channels are recognized as rearing habitat, flow through these channels are maintained after diversions are shut off.

Increases in flow velocities at bypass channel discharge points could present potential barriers to fish migration or could exceed thresholds for various life-history stages of some HCP species. Direct effects from altered velocities include stress to migrating species through increased activity, exhaustion, and delayed migration. Indirect effects include changes in habitat accessibility, habitat quality, and increased predation. For instance, leopard and Umatilla dace inhabit riverine environments where the velocities are less than 1.6 ft/sec (Wydoski and Whitney 2003). Bypass return flows that result in channel velocities exceeding this limit would render upstream habitats unavailable for these species.

Flow velocities also influence swimming activity and respiration in fish species. Increased flow velocities can also force fish species to rest in areas of slower-moving water to recover from increased activity. This behavior can result in unsuccessful recruitment from delayed migration upstream for anadromous species (e.g., salmonids, sturgeon, lamprey), or increased predation from remaining longer in slow pools downstream of weirs and high-velocity reaches.

Direct and indirect effects of altered flow velocities on invertebrates are not well understood and represent an area for further research. However, for the HCP invertebrate species that are filter feeders (e.g., California floater and western ridged mussel) or rely on stable substrate for habitat structure, altered sediment transport is likely more important than changes in flow velocities.

Alteration of channel geometry has both direct and indirect effects on fish and invertebrates. Fish and invertebrates require certain widths and depths for habitat, spawning, and cover. For example, mountain suckers in Lost Creek, Utah, showed a preference for spawning depths of 4.3–11.8 inches (11–30 cm) (Wydoski and Wydoski 2002). Indirect impacts arising from the alteration of channel geometry include the modification of natural sediment transport, a reduction in habitat connectivity, and a reduction in habitat complexity. The effects of altered substrate composition and stability on HCP species are described below. The effects of reduced habitat connectivity are discussed in Section 7.2.6 (*Ecosystem Fragmentation*).

Alteration of the substrate composition through coarsening or fining of the bed materials can have direct and indirect effects on HCP species. The ecological effects of substrate coarsening and fining on salmonids 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 invertebrate species. Altered substrate composition and stability can affect habitat suitability for spawning by salmonids and other fish species. Salmon require a range of sediment sizes, and spawning success depends on how well they can move sediment to create a redd with their tail. As a result, different species use gravels of different size and can effectively move only certain size classes of sediment (Kondolf 1997; Kondolf and Wolman 1993). Large substrates, exceeding the maximum size mobilized by spawning salmonids, are avoided during redd building (Kondolf and Wolman 1993), including areas where erosion to bedrock has occurred. 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 7-1.

**Table 7-1. 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 $m^2$) |

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

Gravel and cobble substrate is preferred by spawning white sturgeon because their adhesive eggs are susceptible to burial by sand and silt-sized substrate (Paragamian et al. 2001). Gravel substrate is also preferred spawning habitat for Dolly Varden (Kitano and Shimazaki 1995). Changes in substrate composition, both coarsening or increased deposition of fines, can alter spawning habitat suitability. Mobilization and redeposition of fines can affect incubation success. Excessive deposition of fines can lead to substrate embeddedness, reducing the water circulation necessary to oxygenate the eggs and remove metabolic wastes (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 (Bennett et al. 2003; Chapman 1988; Cooper 1965; 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.

With regard to effects on invertebrates, burial and entrainment in mobilized sediments and habitat modification are primary stressors resulting from hydraulic and geomorphic modifications. The effects of these on HCP invertebrate species are discussed in greater detail in Section 7.3.1.1.3 (*Dredging and Fill*).

The deposition of fine sediment can also adversely affect 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). This can lead to changes in community composition that affect food web productivity and prey availability for HCP species.

As noted above, substrate embeddedness may increase in reaches affected by bypass channels due to changes in sediment transport capacity. This may in turn affect hyporheic exchange. Hyporheic exchange, characterized by the exchange between surface water and subsurface water in streams and rivers, is extremely important for the health of riverine systems (Jones et al. 1995; Mulholland et al. 1997; Sheibley, Duff et al. 2003; Triska et al. 1989). Increased hyporheic exchange between surface and subsurface waters will benefit aquatic biota by increasing benthic dissolved oxygen levels and promoting solute uptake, filtration, and transformation. Studies have shown that the availability of dissolved oxygen to incubating salmonid embryos is dependent on hyporheic exchange (Geist 2000; Greig et al. 2007) and that the occlusion of this exchange through siltation can lead to hypoxia within redds and decreased embryo survival.

With regard to habitat complexity, numerous studies have indicated that decreased complexity negatively affects the survival and growth of aquatic organisms. Reduced shelter availability will increase predation and is not energetically favorable for fishes. In a recent study by Finstad et al. (2007), it was found that juvenile Atlantic salmon exhibit accelerated mass loss rates with decreasing access to shelter, indicating that the juvenile fish had to expend greater energy when there was no available shelter. In another study by Babbitt and Tanner (1998), tadpole survival was 32 percent greater under high cover compared to low cover, suggesting that increased cover decreased predator foraging efficiency. Although the prey in this study were not HCP species, the effect of cover on predation rates can be extrapolated to HCP species that utilize vegetated cover during early life stages. Finally, limited habitat availability will lead to density-dependent mortality for those species that cannot find unoccupied cover and may be exposed to increased predation or high-energy environments (Forrester and Steele 2004).

Riparian vegetation and LWD are also important for bank-side habitat and cover from predation and temperature. For example, radio-tagged cutthroat trout were observed using pools associated with LWD for cover (Harvey et al. 1999). In addition, undercut banks provide shade, lower temperatures, and cover from predation.

Fish rely on habitat complexity 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, Washington, found that increasing habitat complexity by adding LWD caused an increase in winter populations of juvenile coho salmon and age-0 steelhead. 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. Nonetheless, several studies have documented fish species utilizing complex habitats with LWD (Bryant et al. 2007).

Freshwater macrophytes are also known to contribute to habitat complexity by changing surface water patterns, 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 matter content and capturing smaller grain size sediment than normally occurs in uncolonized areas (Carrasquero 2001). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange, which in turn will promote nutrient cycling. For example, White (1990) found that dense vegetation hummocks promote upwelling of porewater into the rootmass, which provides nutrients that encourage and sustain vegetation growth. In these ways, aquatic vegetation can contribute to habitat complexity and food web productivity.

An indirect impact from the loss of decreased habitat complexity is an increase in nutrient loading to downstream receiving waters. Channel complexity promotes the retention of water and organic material. This retention 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. Decreased nutrient retention affects both local waterways and downstream receiving waters. Local waterways are affected through the associated reduction in primary production, and receiving waters (which are primarily located in more nutrient-impacted lowland areas) are affected through additional nutrient loading, which may lead to eutrophication. Alteration of nutrient cycling is likely to affect food web complexity, which can have a range of effects on HCP fish and invertebrate species limiting to survival, growth, and fitness.

Collectively these studies demonstrate the importance of habitat complexity to the function and productivity of aquatic habitats. It follows that loss of habitat complexity can contribute directly to decreased growth, survival, and population productivity of HCP species.

### 7.2.6 Ecosystem Fragmentation

Like the in-channel screen subactivity type, the predominant forms of ecosystem fragmentation that can be associated with off-channel screens occur predominantly as a result of water withdrawals and the effects of the flow control or diversion structure on the environment. Therefore, these effects are not addressed explicitly here. However, the off-channel screen subactivity type includes some unique design characteristics having their own potential to impose ecosystem fragmentation. Specifically, the additional increment of streamflows required for bypass system operation can modify channel and flow conditions in ways that fragment off-channel habitat. Moreover, bypass systems that discharge into blind side channels may create flow conditions that confuse migratory pathways.

As with the other impact mechanisms associated with fish screens, the extent and magnitude of these submechanisms are expected to be minor in comparison to those imposed by the diversion system and water withdrawal the fish screen is inherently associated with. However, the potential effects of the off-channel screen subactivity type are sufficiently distinct to warrant a separate discussion here.

#### *7.2.6.1 Impact Submechanisms and Stressors*

The potential ecosystem fragmentation impact submechanisms potentially imposed by off-channel fish screens include the following:

- Passage and dispersal barriers: Bypass channel flows may attract upstream migrants, causing an unintentional migration delay. Sweeping flows in diversion channels may not be sufficient to draw downstream migrants into the bypass system, leading to unintentional delays in downstream migration or dispersal.

- Modified downstream transport of woody debris and organic material: Woody debris and organic material cleared from screen surfaces may not be returned to the aquatic ecosystem.

- Altered lateral habitat connectivity: Decreased flows within the bypassed reach may alter the connectivity to and availability of side-channel and off-channel habitats under lower flow conditions.

Additional detail on these impact submechanisms, the ecological stressors they impose, and the effects of stressor exposure on HCP species is provided in the following sections.

##### *7.2.6.1.1 Passage and Dispersal Barriers*

Fish screens are intended to block the movement of fish and other organisms out of their habitat with water withdrawn from the system. In this sense, they impose a passage barrier that should be considered beneficial. However, in certain worst-case scenarios, bypass systems may create flow conditions that can lead to migration delay.

The following ecological stressors can result from this impact submechanism:

- Delayed migration or dispersal (upstream and downstream, depending on life-history stage)

- Injury and energy expenditure

- Increased predation exposure

- Phenotypic and lifehistory selectivity

- Species selectivity.

While the probability of these ecological stressors occurring is low in the case of most fish screens, they nonetheless can have a broad range of effects on HCP species should they occur.

The worst-case scenario range of effects is described below in Section 7.2.6.2 (*Effects on Fish and Invertebrates*).

#### *7.2.6.1.2 Altered Lateral and Longitudinal Connectivity*

In certain circumstances, vegetation encroachment induced by bypass system operation may result in changes in channel form that can in turn fragment lateral habitat connectivity (see Section 7.2.5 [*Hydraulic and Geomorphic Modifications*]). Decreased lateral connectivity with side-channel, slough, and floodplain ponds can have a range of effects on HCP species. Side channels create refugia for juvenile fish (Jungwirth et al. 1993), while floodplain ponds and backwater sloughs create zones of high retention and productivity that provide vital rearing habitat (Hall and Wissmar 2004; Sommer et al. 2005) and important sources of organic material for the channel (Tockner et al. 1999). The loss of connectivity between the river and these habitats can result in a decrease in organic matter recruitment (Tockner et al. 1999; Valett et al. 2005) and reduced access to valuable foraging and rearing habitat (Henning et al. 2006). 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 in restored channels and engages habitat that would otherwise be inaccessible.

As noted in Section 7.2.5.1.1 (*Altered Flow Conditions*), however, bypass systems that increase flow into existing natural side channels, or effectively create an artificial off-channel environment can mitigate effects on habitat fragmentation in some cases. In highly hydromodified environments, this effect could be beneficial, increasing available habitat area and complexity.



#### *7.2.6.1.3 Modified Downstream Transport of Woody Debris and Organic Material*

While these effects are more commonly associated with structures that impose barriers to fish passage, fish screen structures may alter the downstream transport of wood and organic materials when measured against the natural stream condition or the environmental baseline. Certain off-channel screen designs (e.g., traveling drum screens) incorporate debris collection trays that isolate wood and organic material from the stream channel. Other off-channel screens may be prone to debris jams on the screen or in bypass systems that require manual clearing. These materials may be returned to the channel as an operational practice, or may be disposed of upland. In the latter case, there would be an incremental decrease in the amount of wood and organic material available to downstream reaches. However, on the whole, this effect is expected to be small relative to those imposed by the diversion of water and the presence of flow control structures.

### ***7.2.6.2 Effects on Fish and Invertebrates***

The fish screen subactivity types addressed in this white paper are intended to minimize entrainment losses of juvenile and adult fish associated with diversions. Therefore, habitat fragmentation effects discussed in this section should be viewed in that context. In certain cases, however, fish screens may inadvertently result in passage selectivity, migration delays, or other factors that diminish population productivity and diversity relative to the baseline of the natural stream.

Even in the absence of a well-defined migratory behavior, the ability to move between different habitat types is nonetheless important for many resident fish species (Rodriguez 2002). The ecological implications of decreased habitat access are potentially significant. For example, the effects of restricted access caused by dams and weirs have been broadly implicated in population declines of freshwater fish species from around the world (Northcote 1998). Fish screens are specifically intended to rectify fish entrainment and loss caused by unscreened intakes and diversions associated with flow control structures. As such, they effectively minimize potentially significant ecological effects. However, due to design limitations or improper maintenance and operations, fish screens can nonetheless produce unintended adverse effects on fish passage. Specifically, fish screens can delay migration, exposing fish to increased stress and predation-related mortality; or may impose timing- or size-specific selection pressures on affected fish species.

For example, in Washington State a concerted effort to design and broadly implement diversion screening requirements has been in place since the 1980s (McMichael et al. 2004). This cooperative program has promulgated research-based design guidance and monitoring criteria that are in broad use. While this program has produced fish screens that have undoubtedly reduced entrainment-related losses of anadromous and resident fishes, some of these screens have imposed unintentional barriers to fish passage (Carter et al. 2003; McMichael and Chamness 2001; Vucelick and McMichael 2003; Vucelick et al. 2004). These barriers can take the form of physical conditions that delay upstream or downstream migration, potentially coupled with conditions that increase predation risk, or that impede migration entirely during certain flow rates.

In the case of juvenile salmonids, downstream migration delays can occur when improperly designed screens may fail to provide sweeping flows adequate to draw fish into the bypass channel. For adult fish, upstream migration delay can be caused by false attraction to bypass outfalls or by locating the bypass discharge point in proximity to the diversion intake, causing fish disoriented by exiting the bypass system to enter and fall back through the bypass. Migration delays and nonlethal stressors may also increase predation exposure, resulting in increased mortality rates. For example, shear stresses associated with passage through dam bypass channels have been associated with temporary disorientation that leads to increased mortality rates (Cada et al. 1999; Mesa 1994). While it is unclear whether stresses occurring in bypass channels reach levels sufficient to increase predation vulnerability (Cada et al. 2003), WDFW guidance cites this potential as an important consideration in bypass channel design, noting that outlets should be located where conditions are unfavorable for predators to loiter (WDFW 2001a). Such steps may help to mitigate predation losses. For example, Mesa and Olson (1993) found that flow velocities in excess of 39–51 inches/second (100–130 centimeters per second [cm/s]) were likely to exceed the sustained swimming speed of predatory northern pikeminnow (referred to as squawfish by the authors), and cited this range of flow rates as useful guidance for locating bypass channel discharge points.

Fish screens may also impose unintended selection pressures by only providing effective downstream passage of juveniles during part of the downstream migration (Kiefer and Lockhart 1995). For example, Kemp et al. (2006) found that flow velocity and depth strongly influenced the behavior of juvenile fish entering bypass systems at Snake River hydroelectric facilities. While this study did not explicitly evaluate the effects of screens, it nonetheless demonstrates the sensitivity of certain HCP species to parameters that are important in screen design. The potential for these parameters to influence species-specific screen performance is also a concern. Sweeping flows that function well for salmonids may not be suitable for other fish species such as dace (Cyprinidae) or lamprey (Close et al. 1998).

Fish screens can also unintentionally affect passage success based on life-history stage (i.e., by size). Because juvenile salmonids migrate downstream in many river systems, they must travel past numerous diversions with screens of various designs. Off-channel screens must provide sufficient sweeping flows to draw fish into bypass channels without significant migration delays. Suitable sweeping flows may vary by species and by size. For example, juvenile salmonids have been shown to respond preferentially to different velocity conditions when traveling downstream through weirs (Kemp et al. 2006). Attraction velocities must be balanced against other factors such as avoiding impingement while achieving the desired diversion rate. Design guidance focused on achieving this balance for juvenile salmon may be suitable for some other species, such as bull trout (Zydlewski and Johnson 2002), but may or may not provide adequate protection for other fish species with different swimming or biological requirements (Bestgen et al. 2004; Blackley 2004; Close et al. 1998; Moyle and Israel 2005; Peake 2004). A basic premise of screen and bypass design is that fish are actively migrating and seeking a downstream migration path (Bates 2008). This premise may be inappropriate for fish that are passively dispersing rather than migrating.

Upstream migration and other movements within freshwater rearing habitats are also recognized as important factors to consider when designing fish screens. Direct study and review of available research have demonstrated that juvenile salmonids (both anadromous and resident species) are seasonally migratory, moving between refuge and rearing habitats (Bolton et al. 2002; Kahler and Quinn 1998; Kahler et al. 2001). While fish screens are less of a factor than the flow control structures they are typically associated with, certain designs may nonetheless have undesirable effects on upstream passage. Specifically, dedicated bypass channel flows may unintentionally attract upstream migrants into impassable side channels (WDFW 2001a). For example, this may delay dispersal to habitats suitable for summer rearing. Proper design guidance may avoid this unintended impact.

Juvenile salmonids are known to migrate seasonally between rearing freshwater habitats, regardless of species (Kahler and Quinn 1998; Kahler et al. 2001). Juveniles may cover considerable distances to occupy available rearing habitats, indicating that this dispersal mechanism is important to survival (Bolton et al. 2002). Therefore, fish screen structures that unintentionally block access to key summer and winter rearing habitats may be key factors limiting juvenile survival, growth, and fitness.

The preponderance of evidence indicates that reductions in the delivery of marine-derived nutrients can affect food web productivity in ways that are detrimental to the growth, fitness, and productivity of juvenile salmonids as well as other native fish species (Bilby et al. 1998; Heintz et al. 2004; MacAvoy et al. 2000). The effects of fish screens on the upstream transport of marine-derived and other allochthonous nutrient sources are generally expected to be limited, relative to the effects of related flow control structures (see the Flow Control Structures white paper [Herrera 2007a]). Therefore, while fish screens may impose some effects through this submechanism, the incremental magnitude of these effects is expected to be small.

To the extent that they affect upstream fish passage, fish screens may also have the unintended effect of restricting the dispersal of HCP freshwater invertebrate species. This can occur in two ways. First, the structure may restrict the distribution of host fish, affecting the dispersal of parasitic larvae (Vaughan 2002; Watters 1996). Second, attraction flows may draw mussels and snails capable of crawling along the stream bottom to bypass channel outlets that restrict their further upstream movement. For example, certain freshwater mussel species are known to move at least some distance upstream, using their muscular foot and byssal threads (Vaughan 2002). Species such as California floater mussels may face demographic risks from the unintentional limitations on distribution because the effects of fish screens on their host-fish species (minnows and other cyprinids) are less well understood.

Lateral and longitudinal habitat connectivity provides a range of important habitat functions for HCP species. Through inference or direct evidence, it can be shown that fragmentation of this connectivity is likely to have a range of detrimental 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, a sensitive cyprinid species (Ribeiro et al. 2004), fishes rearing in floodplain habitat were healthier and larger than fish from the same cohort that did not rear in this type of environment. Swales and Levings (1989) found that off-channel habitat in the Coldwater River, British Columbia, were vital rearing areas for coho, while juvenile Chinook, steelhead, and Dolly Varden were most abundant in floodplain ponds. Larval white sturgeon have been shown to disperse to flooded riparian habitats for early rearing. Fragmentation of these habitats may be a factor in the decreased productivity of this species (Coutant 2004).

## 7.3 Effects of Common Impact Mechanisms and Stressors

This section provides a discussion of the ecological stressors imposed by impact mechanisms that are common among the two fish screen subactivity types examined in this white paper, as well as the effects of stressor exposure on HCP species. The intent of providing a single, organized discussion of these effects is to reduce redundancy and promote readability. The following sections provide a detailed description of each impact mechanism by component submechanism, the ecological stressors they impose, and the effects of stressor exposure on HCP species. The discussion of effects on HCP species is organized somewhat differently between impact mechanisms, again to promote readability. In most cases, the effects discussion is combined at the impact mechanism level because the stressors imposed by each component submechanism are fundamentally interrelated. In specific cases, the stressors imposed by each impact submechanism are sufficiently unique that a separate discussion of their effects is useful.

The discussion provided in this section presents a worst-case scenario evaluation of the effects of stressor exposure resulting from these common impact mechanisms. For many of the subactivity types in question, the magnitude of stressor exposure may be less than what is presented here. Therefore, when interpreting the potential effects on HCP species, the anticipated magnitude of these impact mechanisms and resulting level of effects for each subactivity type must be considered, as described in Sections 7.1 (*In-Channel Screens*) and 7.2 (*Off-Channel Screens*).

The following are common impact mechanisms associated with both fish screen subactivity types:

- Construction and maintenance
- Operations
- Water quality modifications
- Hydraulic and geomorphic modifications.

### 7.3.1 Construction and Maintenance

In most cases, the development of any type of fish screen will require the construction and maintenance of an in-water structure. Applying the worst-case scenario perspective employed throughout this white paper, construction and maintenance activities involve forms of disturbance that are generally similar regardless of the type of structure being developed. Common impact submechanisms resulting from construction and maintenance activities and their related ecological stressors are described below.

#### *7.3.1.1 Impact Submechanisms*

Submechanisms of impact associated with construction and maintenance activities include equipment operation and materials placement, dewatering and handling, and dredging and fill, as described below. Direct and indirect effects on fish and invertebrates are summarized below following each submechanism.

##### *7.3.1.1.1 Equipment Operation and Materials Placement*

Equipment operation and materials placement associated with fish screen construction have the potential to produce various forms of disturbance. Specifically, these activities cause physical and noise-related disturbance in the aquatic environment, both of which are stressors that can produce effects on HCP species. Stressors produced by visual and physical disturbance are well represented by the effects of dredging, as discussed in detail in Section 7.3.1.1.3 (*Dredging and*

*Fill*). Therefore, the discussion presented in this section focuses on the effects of underwater noise produced during construction.

Equipment operation and materials placement during the construction and maintenance of fish screens can produce underwater noise of varying duration and intensity, depending on the source. In general, noise produced by impulsive sources (i.e., short duration, high-intensity noise from sources such as pile driving or materials placement) is likely to produce different effects than noise produced by a more continuous source (e.g., continuous operation of flow bypass pumps). 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).

<u>Characterization of Underwater Noise</u>

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 $dB_{RMS}$ 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 $dB_{peak}$ 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, or the sound pressure level at which barotrauma injury is likely to 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).

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. 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.

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 does not fully 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 great deal 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 because the mathematical relationships it is based upon are derived from site-specific measurements of sound propagation. 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 in the specific environments they investigated, most likely because it does not account for a number of site-specific factors that affect sound propagation. They developed an alternative model based on a 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 under site-specific circumstances, but does not explicitly identify each factor or its specific effects. Because the resulting formulae are highly site specific, they are impractical for generalized use in ESA consultation. Therefore, USFWS and NOAA Fisheries rely on the more conservative practical spreading loss model (Stadler 2007; Teachout 2007).

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.

The fish screen activity type is most likely to produce underwater noise caused by equipment operation and materials placement during construction. Once operational, fish screens may also alter ambient noise levels by changing local hydraulic conditions; however, these effects are expected to be within the natural range of ambient noise levels found in the affected environment.

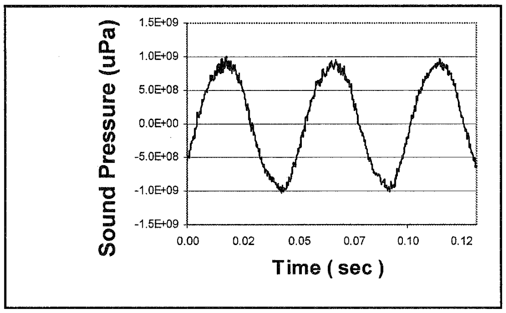
Materials Placement

Underwater noise caused by materials placement is a subject that has 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. Practical experience indicates that the vast majority of fish screen structures do not incorporate structural piles (Schille 2007). However, installation and maintenance of permanent in-channel screen structures are likely to require the development of a temporarily dewatered work area. This practice commonly involves the placement of sheet pile cofferdams, which involves pile driving (Schille 2008). As such, it is appropriate to consider underwater noise impacts related to pile driving, with the understanding that these impacts are likely to occur in a small minority of projects typically involving larger permanent structures.

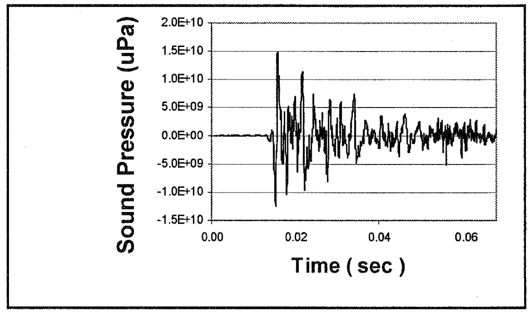
Two major types of pile driving hammers are in common use, vibratory hammers and impact hammers. There are four kinds of impact hammers: diesel, air or steam driven, hydraulic, and drop hammer (typically used for smaller timber piles). Vibratory hammers produce a more rounded sound pressure wave with 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.

Site-specific conditions may dictate the type of pile driving methods with greater noise impacts; this effects analysis addresses the full extent of these potential effects. 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 waveform generated by these different types of hammers are pictured in Figure 7-1.

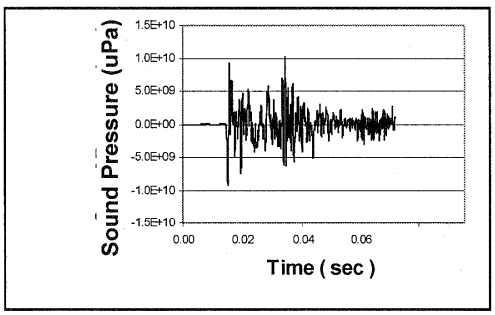
Piling composition also influences the nature and magnitude of underwater noise produced during pile driving. Driven piles are typically composed of one of three basic material types: timber, concrete, or steel (although other specialized materials such as plastic may be used). Steel piles are often used as casings for pouring concrete piles. Noise levels associated with each of these types of piles are summarized in Table 7-2. Reference noise levels are denoted in both $dB_{peak}$ and $dB_{RMS}$ values, at the specified measurement reference distance.



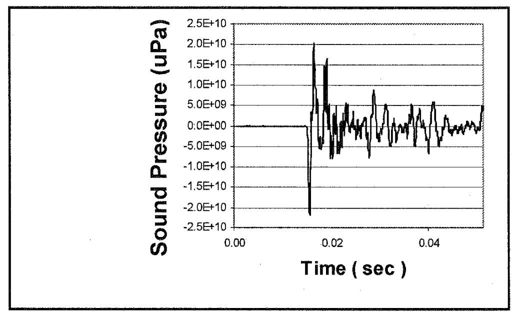

**Vibratory Hammer wave form—set duration**




**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 different pile driving hammer types (WSDOT 2006).**

As noted, data on noise levels produced by placement of other construction-related materials (e.g., the dumping of structural rock, placement of wood or concrete elements) are limited. For example, 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 routine vessel traffic (WSDOT 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 $dB_{peak}$. These data suggest that noise associated with these activities, such as tool use, placement of large rock, steel or other construction materials, 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, such as the continuous operation of flow bypass pumps, may generate continuous noise for longer periods. This would have 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.

**Table 7-2. Reference noise levels, by pile structural material 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 literature source; 9 m = 29.5 ft; 10 m = 32.8 ft; 11 m = 36 ft; 30 m = 98 ft; 100 m = 328 ft.

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. Therefore, these noise measurements, particularly those recorded in the vicinity of ferry terminals and other high-activity locations, are indicative of the noise that could be produced by project construction, maintenance, 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 $dB_{peak}$ (WSDOT 2005). Carlson et al. (2005) measured the underwater baseline for 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 $dB_{peak}$ 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 $dB_{peak}$ 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 the available information. For example, high-gradient rivers, fast-flowing streams, 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 pressure sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dB re: $1\mu Pa^2$/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 $dB_{peak}$ 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 $dB_{peak}$ across the frequency range (again with lower frequency sounds being loudest).

<u>Effects on Fish and Invertebrates</u>

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 and sensory 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 swimbladder 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 Jr. 1988). In contrast, the hearing capacity of salmonids and other hearing generalist species is limited in bandwidth and intensity threshold by the less sophisticated nature of their hearing organs. The Atlantic salmon, for example, is functionally deaf at sound pressure wavelengths above 380 hertz (Hz) (Hawkins and Johnstone 1978). In these fish, the swimbladder does not likely enhance hearing.

Water is a dense medium relative to air that effectively transmits changes in pressure. Because the bodies of aquatic organisms have comparable density to the medium they inhabit, their bodies are highly sensitive to pressure changes. Noise sources such as pile driving can produce high-intensity sound pressure waves that produce direct effects ranging from temporary stress and behavioral avoidance, to temporary impairment of sensory organs, to injury to pressure sensitive organ systems (e.g., hearing, heart, kidney, swim bladder, and other vascular tissue) sufficient to limit survival or even cause 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 sensitivity 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 and impulsive 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 and 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; Scholik and Yan 2002; Wisby et al. 1964). Prolonged exposure to underwater noise can reduce the sensitivity of fish to underwater noise stimuli, altering these key behaviors.

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 hearing sensitivity, increasing the noise 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. Both of these species belong to the family Cyprinidae, which as a group are considered hearing specialist species. HCP fish species in this group include mountain sucker, leopard and Umatilla dace, and lake chub.

Amoser and Ladich (2005) 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.

With regard to invertebrates, information on the effects of elevated underwater noise is generally lacking, 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 impulsive sound at 217 $dB_{peak}$, suggesting the potential for injury or mortality when underwater noise exceeds these levels.

No research has been identified regarding the effects of lower intensity continuous underwater noise on invertebrates. However, continuous construction and 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 nonmotile, 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.

#### *7.3.1.1.2 Dewatering and Handling*

In many cases, construction of HPA-permitted projects may require the exclusion of streamflows or even the dewatering of the work area to protect aquatic life and/or provide a suitable environment for construction. This practice is commonly conducted using cofferdams, which are constructed using a variety of practices, including:

- Pile driving to place sheet piles (see the effects described in Section 7.3.1.1.1 [*Equipment Operation and Materials Placement*])

- Dredging and fill to create temporary earthen cofferdams, or placement of sandbags, ecology blocks, or other materials (see effects described in Section 7.3.1.1.3 [*Dredging and Fill*])

- Placement of sandbags, ecology blocks, or other materials, causing effects similar to those described for dredging and filling (see effects described in Section 7.3.1.1.3 [*Dredging and Fill*]).

Cofferdams are commonly used to place in-channel screens and may also be used in certain circumstances to place bank armoring or other erosion protection around bypass channel outlets for off-channel screens. The effects of materials placement and related activities on HCP species are described in the referenced sections.

The effects of dewatering and organism removal and relocation are associated with additional direct and indirect effects on HCP species. Fish exclusion and dewatering involve the placement of barriers (e.g., block nets, temporary berms, cofferdams) around a work area and the capture and removal of fish and other aquatic life within the work area. Electrofishing is a common practice used for fish capture in freshwater environments, as is the use of minnow traps, hand nets, beach seines, and other net-based capture methods. Because electrofishing is ineffective in brackish or salt water, net-based capture methods are used in these environment types.

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
- Behavioral modification caused by slow dewatering for passive relocation, increased competition, and stress
- 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 species.

<u>Effects on Fish and Invertebrates</u>

Of the various methods used for dewatering and 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, McMichael et al. (1998) observed a 5.1 percent injury rate for juvenile Yakima River steelhead captured using 30 Hz pulsed DC equipment. Ainslie et al. (1998 ) reported injury rates of 15–39 percent in juvenile rainbow trout using continuous and pulsed DC equipment, and found that while pulsed DC equipment producing injury more frequently, these injuries were less severe in nature.

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 completely 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) exposed 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 between 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 handling stress impaired predator evasion in salmonids 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. Based on interviews with state fisheries agency staff, 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 the competition there 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 handling associated with stream crossing projects. One factor used in calculating incidental take from these activities was an estimated stranding rate of 8 percent for ESA-listed salmonids (which equates to 8 percent mortality) (NMFS 2006), which was based on expected 45 percent capture efficiency using three-pass electrofishing (Peterson et al. 2004). The assumed electrofishing injury rate for this type of activity was 25 percent (NMFS 2006).

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.

Installation, operation, and removal of a stream bypass system to rewater a channel can increase 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 during rewatering are discussed in Section 7.3.3.1.2 (*Elevated Suspended Sediments*).

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 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. 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).

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

#### *7.3.1.1.3 Dredging and Fill*

Construction of fish screens may require excavation dredging and placement of fill within the ordinary high water mark (OHWM). A common practice through which this occurs is the development of temporary earthen cofferdams. For example, a bulldozer or excavator may be used to create temporary pushup berms from native alluvium to create an exclusion area. Similarly, sand bulk bags with a liner may be placed in the channel to create the exclusion area. Additional dredging and fill may be required to configure the channel bed or banks to support the screen and related erosion protection. While most often required for placement of in-channel screens, some in-channel dredging and fill may be required for off-channel screen construction where bypass systems discharge back to the source body. In-channel screens commonly require additional dredging to remove sediment accumulation that adversely affects screen and intake performance, meaning that this subactivity type is likely to impose this impact mechanism at greater frequency than off-channel screens.

This impact submechanism can impose direct stressors on aquatic organisms in the form of dredge entrainment and burial, as well as indirect stressors through alteration of hydraulic and geomorphic conditions. Entrainment in this context refers to incidental capture in dredged materials. These types of effects are most often mitigated by conducting work within cofferdams, meaning that the placement of the dam is the primary source of dredging and fill-related impacts. However, some in-channel dredging may be necessary to maintain in-channel screens as discussed. Entrainment within and/o burial by unstable eroding bed materials caused by screen construction can also occur. This form of entrainment is viewed separately from entrainment into the intake or diversion system, which is discussed in Section 7.3.2.1.2 (*Entrainment and Impingement*).

Regardless of cause, these stressors can adversely affect HCP fish and invertebrate species. The nature and magnitude of these effects are discussed in the following sections.

Effects on Fish and Invertebrates

Dredging-related burial and 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.

Because they are nonmotile, HCP invertebrate species are less able to avoid exposure to burial and entrainment-related stressors. Although some specifics on the effects of burial are known for marine invertebrate species (Hinchey et al. 2006), data on the tolerance limits of HCP freshwater mollusks with respect to burial are more limited. However, sufficient data are available on both marine and freshwater species to draw some conclusions about the effects of burial resulting from impact submechanisms such as elevated suspended sediment levels and other sources such as construction-related dredging and fill.

Stress or mortality resulting from partial and complete burial of various mollusk species has 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. Mechanical and physiological adaptations contribute to this tolerance. Olympia oysters have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b). Thus, it can be inferred that burial of these organisms would lead to mortality. Increased fine sediment deposition has been shown to adversely affect estuarine mollusk species with low motility (Hinchey et al. 2006). Limpets in intertidal habitat are affected by burial and interference with feeding activity. In a field study in the United Kingdom, grazing by limpets was decreased by 35 percent after the addition of fine sediments, to as little as 0.04 in (1 mm) thick (equivalent to 1.02 x $10^{-5}$ lb/ft$^2$ [50 mg/m$^2$]), with mortality and inhibition of feeding at higher levels of fine sediment (4.09 x $10^{-5}$ lb/ft$^2$ [200 mg/m$^2$]) (Airoldi and Hawkins 2007). The mechanism of effect is postulated to be the clogging of the organism's filtering organs by fine sediments.

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 mussel 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 and other effects.

Krueger et al. (2007) also studied the effects of suction dredge entrainment on these two species of 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 sensitive to burial and are assumed to suffer high mortality from mechanical injury, smothering, anoxia, starvation, or desiccation associated with entrainment. 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 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 resides in the gills of host-fish where stressor exposure is less likely to occur.

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

### 7.3.2 Operations

Fish screens are continuously operating structures so long as the water intakes or diversions they are associated with are withdrawing water. Because fish screens are intended to mitigate significant environmental problems associated with water withdrawals, operational effectiveness is a principal means through which effects on HCP species may occur.

#### *7.3.2.1 Impact Submechanisms*

The principal impact submechanisms associated with fish screen operations are visual, physical, and noise-related disturbance associated with moving mechanical parts and debris-clearing systems; and risk of entrainment or impingement of organisms due to limitations of screen design or maintenance related failures. Direct and indirect effects on fish and invertebrates are summarized together for these two submechanisms.

##### *7.3.2.1.1 Visual, Physical, and Noise Related Disturbance*

Many fish screen designs incorporate moving mechanical elements or features capable of agitating the water column in ways sufficient to create disturbance. For example, rotating drum or panel screens are in continuous movement, creating both mechanical noise and water disturbance. Most permanent fish screen designs also incorporate debris-clearing systems involving mechanical brushes, jets of pressurized air or water, or other types of systems that create short-term impulsive disturbance (Blackley 2004; WDFW 2001a).

As discussed in Section 4.1.4 (*Screening Systems Not Considered in this Analysis*), the intentional use of noise and light systems to produce a screening effect by modifying fish behavior is not given extensive treatment in this white paper. However, artificial lighting and/or noise may be used in conjunction with screens to improve the function of bypass systems, so some discussion is useful. Popper and Carlson (1998) conducted a broad review of available research on the use of these stimuli to affect fish behavior. Their conclusion was that, with a few interesting exceptions, the literature provides no clear consensus on consistently effective behavioral methods or systems with a track record of clear operational success. The exceptions they identified include the use of sound to deter clupeids (e.g., herring) away from intake systems, and the use of strobe or mercury lights in specific situations to attract or repel certain fish species (e.g., the use of strobe lights to repel Chinook and coho salmon away from hydropower intakes). The research studies reviewed demonstrated that avoidance or attraction responses to a variety of stimuli at both laboratory and field scales were observed across a range of species. However, the utility of these stimuli is limited because the responses were highly variable depending on the species in question, site-specific factors, and variable parameters such as water clarity.

##### *7.3.2.1.2 Entrainment and Impingement*

Entrainment of fish and invertebrates into water intake and diversion systems has long been recognized for potentially significant adverse effects on the productivity and abundance of a broad range of aquatic species (Close et al. 1998; Moyle and Israel 2005; Moyle and White 2002; NRC 1996; Stevens et al. 1985; Taft and Mussalli 1978; Travnichek et al. 1993;

Zydlewski and Johnson 2002). Entrained organisms are killed when they are removed from the aquatic environment, or may suffer significant injury. Fish screens are by design intended to avoid and minimize entrainment leading to injury and mortality to the greatest extent possible. From this standpoint, this activity type should be considered beneficial. However, in some cases entrainment of certain species may continue to occur due to design limitations, improper maintenance, or structural failure. For example, large debris or ice can damage screens, leading to increased entrainment. All off-channel screens and certain in-channel screen designs intentionally entrain fish in bypass systems that return fish to their environment. This is a necessary function of the screen, but the physical disturbance imposed upon the bypassed organisms and the configuration may lead to additional effects such as predation risk.

In the context of fish screens, the term impingement broadly refers to both the occurrence and consequences of an organism being drawn into contact with the surface of the fish screen, its debris-clearing mechanisms, or trapped debris. Depending on contact duration and suction force, the effects of impingement can range from few if any detectable changes in behavior (Danley et al. 2002; White et al. 2007; Zydlewski and Johnson 2002) to significant behavioral effects (Peake 2004), mechanical injury, and mortality (White et al. 2007). The hydraulic force of water drawn through the fish screen and/or the mechanical workings of the screen and debris-clearing systems are the mechanisms through which this occurs. Impingement is a consequence of screen installation that is avoided or minimized by proper design and maintenance. Specifically, flow rates through screens and across the screen surface must be balanced so that weak-swimming organisms are able to avoid becoming trapped on the surface of the screen. Effective debris clearance must also be provided to maintain flow performance. The screen system must also be designed to avoid pinch points that can trap or injure aquatic organisms. The avoidance and minimization of impingement is both a focus of design guidance and a topic of considerable research.

#### *7.3.2.2 Effects on Fish and Invertebrates*

With regard to noise, visual, and physical disturbance resulting from fish screen operations, three general categories of disturbance-related stressors are expected to occur: continuous disturbance associated with screen or debris-clearing system operation; periodic impulsive disturbance associated with air burst or hydraulic jet type debris clearing; and the intentional use of noise and light to attract organisms toward bypasses or repel them away from intake systems.

Motorized fish screens (e.g., rotating drum screens) and mechanized debris-clearing systems will produce continuous underwater noise, a stressor with potentially undesirable effects. If the stressors are of sufficient intensity, fish may modify their behavior to avoid the affected habitats. Noise, bubbles, flashing strobe lights, and other forms of disturbance demonstrably cause avoidance behavior in fishes, to the point that they have been evaluated for application as screening devices (Johnson et al. 2005; Popper and Carlson 1998; Welton et al. 2002). This suggests that continual water disturbance, bubble creation, or operational noise may also cause habitat avoidance in certain circumstances. In the absence of behavioral avoidance, organisms that habituate to habitats where auditory masking effects occur may experience increased predation exposure or may not be able to forage as effectively (see Section 7.3.1.1.1 [*Equipment Operation and Materials Placement*]). These effects may be compounded by fish screen systems that unintentionally hinder the downstream passage of migratory species (e.g., juvenile salmonids), forcing fish to delay in locations where masking effects are most pronounced.

With regard to periodic impulsive disturbance, short-term bursts of prolonged noise can produce behavioral responses with important sublethal effects. The effects of this type of impulsive sound are expected to be similar to the lower end of the range of effects for impulsive sound described in Section 7.3.1.1.1 (*Equipment Operation and Materials Placement*). These effects range from behavioral alteration (e.g., "startle" responses) to avoidance, interruption of feeding behavior, and increased stress. Stress and behavioral avoidance of otherwise suitable habitats may in turn lead to effects on survival, growth, and fitness.

Information on the effects of elevated underwater noise and visual disturbance on invertebrate species is more limited, indicating that additional research on the subject is needed. Some studies have demonstrated that increased background noise over intermediate-term periods (e.g., 3 or more months) may affect at least some invertebrate species. Lagardère and Régnault (1980) found that sand shrimp (*Crangon crangon*) exposure to noise about 30 dB above ambient levels for 3 months showed decreases in both growth and reproductive rates, and Lagardère (1982) and Régnault and Lagardère (1983) found that increased noise levels changed sand shrimp metabolism and physiology. In the case of growth rate and reproduction, the effects were evident for up to 1 month following the termination of the signal (Lagardère and Régnault 1980). While instructive, it must be noted that these findings may not apply to HCP invertebrate species, as none of these species are arthropods like the test subjects.

As discussed in Section 4.1.3 (*Typical Screen Designs*), this white paper does not focus extensively on the use of artificial lighting and noise as fish screen systems. Artificial noise systems used to repel fish are expected to emit noise levels sufficient to cause behavioral responses, but not to cause physical injury. Commonly, however, these systems may not provide effective deterrence for all HCP species (Popper and Carlson 1998). In the absence of the desired behavioral avoidance effect, organisms that habituate to habitats where auditory masking effects occur may experience increased predation exposure or may not be able to forage as effectively (see Section 7.3.1.1.1 [*Equipment Operation and Materials Placement*]).

Relative to the effects of behavioral light modification, Popper and Carlson (1998) found little literature available regarding adverse effects on vertebrates. They nonetheless suggested that some adverse effects from illumination are possible if not probable. Changes in ambient light conditions, both at higher-than-normal intensities and outside the normal 24-hour light–dark cycle, have been shown to adversely affect organisms. In particular, organisms living near the extra illumination may alter their natural circadian rhythms, with potentially deleterious effects on feeding, migration and dispersal, and reproductive potential.

Entrainment into water intakes or diversion systems is commonly associated with fish mortality, and the large numbers of unscreened diversions distributed ubiquitously across the landscape have been broadly implicated in the declines of a number of species, including salmonids, lamprey, and other HCP species (Bestgen et al. 2004; Hadderingh 1979; Hadderingh and Jager 2002; Moyle and White 2002; NRC 1996; Spence et al. 1996; Stevens et al. 1985; Travnichek et al. 1993). In general, unscreened intakes or diversions are likely to result in entrainment of HCP species during weak-swimming migratory or dispersal stages. While even properly screened diversions pose some risk of entrainment, screens have been shown to be effective in reducing the mortality of salmonids. For example, Gale and Zale (2005) studied entrainment of juvenile westslope cutthroat and bull trout in screened diversions on Skalkaho Creek, a tributary to Montana’s Bitterroot River. They found that while juveniles were drawn into the head ditches of the diversions under certain flow conditions, the majority were effectively screened and directed into the bypass system for return to the creek, with only small numbers entrained into the diversion.

The term “entrainment” is also used to describe passage through screens or pumps and into bypass systems that return organisms to the aquatic environment. Entrainment into and through bypass systems is a necessary function of certain in-channel screen types (i.e., bankline screens) and all forms of off-channel screens. Bypass systems are designed to quickly sweep organisms away from the screen face and discharge them back to the aquatic environment. Discharge from bypass systems may cause temporary disorientation, leading to the potential for increased predation risk. As discussed in Section 7.2.6.2 (*Effects on Fish and Invertebrates*), WDFW guidance cites this potential as an important consideration in bypass channel design, noting that outlets should be located where conditions are unfavorable for predators to loiter (WDFW 2001a). Such steps may help to limit predation losses.

The direct effects of entrainment through pumped bypass systems have received broad study. Helfrich et al. (2001) studied survival and injury rates for juvenile Chinook salmon and splittail entrained through large Hidrostal pump bypass systems in central California water diversions. Also in California, McNabb et al. (2003) studied the survival and injury rates from bypass of juvenile Chinook salmon and other fish species in the Sacramento River drainage through Hidrostal pumps and Archimedes lifts associated with diversion systems. Rodgers and Patrick (1985) and Patrick and McKinley (1987) conducted similar studies on injury and mortality rates in rainbow trout, yellow perch, alewives, and American eels from passage through Hidrostal pump bypass systems. Collectively, these studies indicate that pumped bypass systems are generally effective at transporting fish with low rates of injury and mortality, with results varying by species and factors such as pump speed. Potentially debilitating injury and mortality do occur, however, albeit at rates ranging from less than one to as high as 10 percent of bypassed individuals. In the absence of injury or mortality, studies of stress induced by passage through these systems indicate that these effects are minimal. Weber et al. (2002) studied the effects of passage through Hidrostal pumps and Archimedes lifts on the cortisol stress response of Chinook salmon and found no significant indication of physiological effects.

While fish screens are broadly recognized as an important tool for reducing entrainment mortality, their effectiveness at protecting a broad range of species is surprisingly poorly understood (Moyle and Israel 2005; Moyle and White 2002). Screen design criteria used in Washington State are directed at protecting the “smallest and weakest” swimming fish under the worst-case conditions, and were developed using juvenile salmonids. However, the effectiveness of screens at protecting age 0+ juveniles (i.e., smaller fish) was uncertain (Gale and Zale 2005). In addition, designs that adequately protect species of interest such as salmonids may not recognize the needs of other HCP species. For example, weak-swimming juvenile lamprey may experience high rates of impingement on fish screens and bypass systems at dams and water diversions designed to pass juvenile salmonids (Close 2000; Close et al. 1998).

Fish screens employed in marine and estuarine systems must be designed to protect a broader range of species and life-history patterns. Of particular interest, in-channel screen systems employed in this type of environment may not be able to provide full protection for planktonic eggs and/or larvae of HCP fish and invertebrate species (Henderson and Seaby 2000). Experimental technologies (e.g., Gunderboom screens) designed to allow the intentional impingement of larval life forms for removal from water intake systems are in development. However, even short-term impingement of larval life forms has been associated with high mortality rates of larval fish and invertebrates in experimental settings, suggesting that these technologies may not be adequately protective. For example, Tenera Environmental (2005) studied the survival of planktonic eggs and larvae of several gobies, California anchovy, and several invertebrate species following entrainment into a desalinization plant intake and observed high mortality rates across all species. Furthermore, maintenance requirements may limit the effectiveness of this technology, particularly in marine environments prone to biofouling (Henderson and Seaby 2000).

Inadequate screen maintenance can produce conditions that lead to increased risk of entrainment or impingement. For example, debris accumulations, failure of hydraulic spray systems, and faulty seals have been identified as sources of increased impingement and entrainment in diversions on Columbia River tributaries (Cameron et al. 1997; Carter et al. 2003; Knapp 1992; McMichael and Chamness 2001; Neitzel et al. 1990; Vucelick and McMichael 2003; Vucelick et al. 2004). Gale and Zale (2005) observed increased entrainment of larger juvenile cutthroat and bull trout into irrigation ditches when high water conditions and sediment and debris accumulations temporarily affected screen performance.

Sensitivity to injury and stress from contact with screen surfaces may also vary by species, the duration of impingement, and impact velocity with the screen surface. For example, Zydlewski and Johnson (2002) observed that juvenile bull trout were regularly impinged on irrigation diversion screens designed to protect trout and salmon (*Oncorhynchus* spp.), but were able to escape without any apparent injury. Danley et al. (2002) found that Sacramento splittail could experience limited contact with screen surfaces and show no significant indications of physiological stress at low inflow velocities. White et al. (2007) examined the responses of the same species at higher across-screen flow rates and observed increasing stress, injury, and mortality with increasing impact velocity. Peake (2004) found that impingement-related injury and mortality of juvenile northern pike were dependent on the flow velocity through the screen.

Fish size and condition factors can also influence impingement risk. For example, Weisberg et al. (1987) found that fish size was a primary determinant of the effectiveness of power plant intake screens, with screen effectiveness decreasing rapidly for individuals less than 0.2 inches (5 millimeters) in length. Dorn et al. (1979) found that the swimming performance of gravid female surfperch (*Hypsurus caryi*) decreased significantly relative to nongravid individuals, increasing the risk of intake screen impingement.

The extent and nature of impingement risk and the resultant effects of stressor exposure vary depending on the screen design and performance at different flow conditions. For example, downward sloping plate screens (e.g., Coanda screens) with proper flow control are effective at avoiding impingement (Bestgen et al. 2004; Vucelick and McMichael 2003; Vucelick et al. 2004; Wahl 1995, 2003; Wahl and Einhellig 2000). However, the practical use of these designs has been limited because it is difficult to maintain the required flow conditions in real world settings (Bates 2008; Schille 2008). In contrast, vertical traveling screens have historically been built with horizontal troughs, or ledges, built onto the face of the screen. While no longer permitted, legacy structures are still in use in Washington State (Bates 2008). The purpose of the troughs is to lift debris and fish with the screen as it rotates to a point where a high-pressure spray bar washes the debris and fish into a stationary trough on the deck of the structure for removal. These designs pose risk of mechanical injury for fish entrained in debris and exposed to the spray wash. Even if they are deposited uninjured in the debris trough, the fish are effectively trapped in an enclosure where capture and removal may be difficult (WDFW 2001a). This would have the added consequence of exposing fish to capture- and handling-related stressors (see Section 7.3.1.1.2 [*Dewatering and Handling*]).

Several HCP species occurring in marine and lacustrine environments have planktonic egg or larval life-history stages. Certain species occurring in riverine environments have similar life histories. For example, white sturgeon, eulachon, and longfin smelt all have adhesive eggs that attach to bottom substrates for incubation, hatching planktonic larvae that are dispersed to and retained in favorable rearing habitats by circulation and/or current conditions. These ichthyoplankton (i.e., free-floating eggs and larvae) are susceptible to entrainment regardless of screen design (Goodyear 1977; Travnichek et al. 1993), and this effect can only be avoided effectively through careful placement of the intake or diversion structure and/or operational modifications. For example, Edinger and Kolluru (2000) modeled the potential entrainment of fish eggs and larvae into power plant cooling water intakes in the Delaware River estuary and determined that larval entrainment would vary considerably depending on tidal and flow conditions and the size of the intake system. Hadderingh and Zager (2002) found that the relocation of a power plant intake to an offshore location resulted in a net increase in the number of larval fish entrained in comparison to the original intake location in a sheltered nearshore environment. The species composition of entrained larvae changed as well. This demonstrates the need to consider the life history and habitat use of species of concern when determining suitable intake locations and selecting an appropriate screen design to minimize impacts on sensitive species.

Even when impingement and entrainment-related mortality of planktonic eggs and larvae occurs, these effects may or may not be significant at the population level. As discussed in Section 7.3.1.1.3 (*Dredging and Fill*), the natural mortality rate of shellfish species is sufficiently high that in many cases entrainment-related mortality would not have a noticeable population-level effect. This is expected to be generally true for highly fecund marine HCP fish and invertebrate species such as oysters, pollock, and hake. However, research has demonstrated that intake structures located in critical spawning and nursery areas can result in larval mortality rates high enough to overwhelm compensatory mechanisms and affect population abundance of highly fecund species like striped bass (Goodyear 1977). While the resulting mortality is more accurately attributed to the intake system and water withdrawal, the limitations of screening technology to prevent this mortality are also a factor. On this basis, HCP species with planktonic life-history stages are considered susceptible to risk of entrainment-related mortality associated with intake screens.

However, where many fish and invertebrate species are concerned, stressor exposure would have to be quite extensive to result in significant population-level effects because these species have larval mortality rates that are naturally high (e.g., commonly exceeding 90 percent) (Lunz 1985; McGurk 1986). For HCP species with planktonic life-history stages in marine and lacustrine environments, it is likely that larval mortality from entrainment into water diversions is relatively insignificant when viewed from the perspective of natural population dynamics. In freshwater environments, the potential extent of this effect is unknown. Diversions in these systems may appropriate a significant component of flow, suggesting greater potential for effects. However, riverine HCP invertebrate species have adapted specific life-history strategies for flowing water environments that limit these effects. The great Columbia River spire snail and giant Columbia River limpet have sessile young. In the case of freshwater mussels, adults have evolved to expel parasitic glochidia life-history stage in the presence of host fish so they can rapidly attach to the gills. This strategy tends to limit stressor exposure where screening technology is appropriately designed for the host fish species.

Entrainment of HCP invertebrate species during sessile life-history stages is considered an issue of lesser concern. These organisms are less likely to be drawn into contact with fish screens, so the potential for entrainment is limited. Should this occur, however, they are unlikely to be able to escape, increasing the risk of impingement-related mortality. Available data on mollusk impingement on fish screens is generally focused on problems related to biofouling and not useful for assessing potential effects on HCP species. However, it is reasonable to speculate that fish screens may pose some risk of impingement and/or mechanical injury for sessile organisms that crawl along the bottom (e.g., northern abalone and the great Columbia River spire snail) under the presumption that these organisms could come into contact with screen components.

### 7.3.3 Water Quality Modifications

HCP-permitted projects under the fish screen activity type are likely to result in varying degrees of water quality modifications. However, the impact submechanisms and related stressors are generally expected to be temporary to short-term perturbations associated with construction and maintenance activities, such as construction-related suspended sediments, or accidental releases of toxic substances from construction-related spills or operational equipment failures.

The discussion of the effects of each water quality-related stressor presented in the following section represents a worst-case scenario perspective. When interpreting the effects of stressor exposure, the magnitude of impact submechanisms and stressors anticipated to result from each subactivity type must be considered. This discussion is provided in the *Water Quality Modifications* discussion for each type of fish screen project (see Sections 7.1.3 and 7.2.3 [both *Water Quality Modifications*]).

#### *7.3.3.1 Impact Submechanisms*

Impact submechanisms associated with water quality modifications include elevated suspended sediments, altered pH, and the introduction of toxic substances, as described below. Direct and indirect effects on fish and invertebrates are summarized following each submechanism.

##### *7.3.3.1.1 Elevated Water Temperatures*

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.

A substantial amount of information is available regarding tolerances of HCP species (particularly salmonids) to thermal stress. For instance, it has been found that the development of coho egg, alevin, and fry is most rapid at 39°F (4°C), while alevin and fry of pink and chum salmon develop fastest at 46°F (8°C) (Beacham and Murray 1990).

Elevated water temperatures can also impair adult migration and spawning. Adult migration blockages occur consistently when temperatures exceed 70–72°F (21–22°C) (Poole and Berman 2001a; 2001b). 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°F (14°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°F (7–14°C) (Brannon et al. 2004; McCullough et al. 2001).

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 7-3; as shown, 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 7-4 presents highest 7-day average maximum thresholds as promulgated in the state standards.

**Table 7-3. 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.**

| 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.
Note: 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).

**Table 7-4. Aquatic life temperature criteria in fresh water.**

| 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) |
| Non-anadromous 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).
Note: Aquatic life temperature criteria. Except where noted, 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.

Table 7-3 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. These effects begin to occur at temperatures ranging from 52 to 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 (Babanin 2006; Marine and Cech 2004; McCormick et al. 1999) and constitutes a significant impairment to juvenile survival.

Table 7-3 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. These effects begin to occur at temperatures ranging from 52 to 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 (Babanin 2006; Marine and Cech 2004; McCormick et al. 1999) and constitutes a significant impairment to juvenile survival.

Additional studies, mainly in the laboratory, have developed limits for other HCP species. Wagner et al. (1997) showed that rainbow trout mortaility occurred at temperatures of 67.8 to 73.0°F (19.9 to 22.8°C). Temperatures above 71.6°F (22°C) can cause deformities in developing white sturgeon, with best performance between 59 and 66°F (15 and 19°C) (Mayfield and Cech 2004). Furthermore, elevated temperatures can make white sturgeon more susceptible to infection from viruses (Watson et al. 1998). Temperatures between 73 and 79°F (23 and 26°C) can cause complete mortality in developing green sturgeon embryos, with upper limits for survival around 63–64°F (17–18°C) (Van Eenennaam et al. 2005). Dolly Varden show decreased appetite above 61°F (16°C), and lethal temperatures are observed above 68°F (20°C) (Takami et al. 1997). A laboratory study of the early life stages of Pacific and western brook lamprey showed that the temperature for zero development for Pacific lamprey was 40.7°F (4.85°C), and for western brook lamprey it was 40.9°F (4.97°C), with survival greatest for both at 64°F (18°C) and lowest at 71.6°F (22°C), and abnormalities in the larval stage greatest in the 71.6°F (22°C) treatment (Meeuwig et al. 2005).

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). Thermal barriers to migration can isolate extensive areas of potentially suitable spawning habitat and contribute to prespawning mortality. 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, the susceptibility of aquatic organisms 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 that results from increased gill ventilation at increased temperature resulted in the rapid uptake of toxicants, including metals and organic chemicals, via the gills. Salmonids also become more susceptible to infectious diseases 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.

Considerably less research exists defining thermal criteria for freshwater HCP invertebrates. It is unclear what sublethal effect(s) may be a significant factor with invertebrate populations.

#### *7.3.3.1.2 Elevated Suspended Sediments*

Elevated suspended sediments and turbidity can occur as a result of fish screen construction, maintenance, and operation. For operations and maintenance, an effective screen necessitates a low water velocity, which causes deposition of material that might otherwise remain suspended or moving as bedload (Bates 2008). Once the material is deposited, it has to be dealt with as a maintenance issue. It might be dredged from the flowing water (in-channel screen) or from a dry canal (off-channel screen). It might also be sluiced downstream in either design. Material might accumulate during a high flow, but then be sluiced during a normal maintenance operation during a period of lower flow. This presents the potential to produce elevated suspended sediment levels during flow periods when the transport capacity is low, meaning that the effects are occurring during periods when suspended sediment levels are low under natural conditions.

Elevated suspended sediments and turbidity have a number of effects on HCP species. In general, the response of aquatic biota to elevated suspended solids concentrations is highly variable and dependent on life-history stage, species, background suspended solids concentrations, and ambient water quality. The following sections summarize pertinent research on the effects of stressor exposure on HCP fish and invertebrate species.

When interpreting this information, it is important to consider that fish screens and related installation requirements vary considerably in scale, and the intensity of suspended sediment stressors will vary in conjunction with these requirements. For example, large, permanent in-channel screens will require in-water construction, potentially including the placement of cofferdams to create in-water exclusion areas. The bed and bank disturbance associated with these activities will produce far more suspended sediments than placement of a temporary pump intake and in-channel screen assembly associated with a temporary water withdrawal. In contrast, off-channel screens are commonly constructed in the dry (Bates 2008; Schille 2008), presenting less potential for suspended sediment impacts. However, connection and watering of off-channel screen bypass channels and/or placement of erosion protection around outlets may present some potential for sediment impacts. However, these impacts are likely to be of lower intensity than those produced by construction of larger in-channel structures.

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 this 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 range of suspended particles of varying origin and composition. These include inorganic materials like silt and clay, as well as organic materials like tannins, algae, plankton, microorganisms, and detritus. 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.

<u>Effects on Fish and Invertebrates</u>

A broad range of research has demonstrated that suspended sediment and elevated turbidity can have a wide range of adverse effects on aquatic organisms, ranging from minor, short-term behavioral alterations, to effects on food web productivity and forage success that influence survival, growth, and fitness, to direct injury and mortality (Henley et al. 2000). As would be expected, these effects are complex and variable depending on the magnitude of the sediment impact in question relative to natural background conditions and the specific sensitivity of the organisms exposed to the stressor. For example, juveniles of many fish species (such as salmonids) thrive in rivers and estuaries with naturally high concentrations of suspended solids. However, studies have shown that the suspended solids concentration (as well as the duration of exposure) can be an important factor in assessing risks posed to salmonid populations (McLeay et al. 1987; Newcombe and MacDonald 1991; Servizi and Martens 1987). Given this complexity, some understanding of the level of exposure associated with a given HCP activity type is necessary to understand the range of likely effects. Suspended sediment levels associated with injury or mortality are typically quite high. Lake and Hinch (1999) found suspended solids concentrations in excess of 40,000 parts per million (ppm) to elicit stress responses in juvenile coho salmon. Suspended solids concentrations this high would likely only be associated with the most extreme construction-related impacts. However, other studies have shown lethal effects at much lower concentrations in salmonids, indicating that the issue is complex, and a precautionary approach to assessing sediment impacts is desirable to limit the potential for adverse effects.

For example, 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 a 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).

For white sturgeon, laboratory studies have shown that the survival of developing embryos was reduced to 5 percent in the presence of 0.19–0.8 in (5–20 mm) thick layers of sediment compared to more than 80 percent survival in controls (Kock et al. 2006).

#### 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 the observed chronic and acute impacts from high suspended solids to a 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.

Indirect effects on fish through alteration of their food source have been documented. Suttle et al. (2004) observed that steelhead trout were affected by an increase in sediments because it caused a shift to burrowing macroinvertebrate taxa that then became unavailable to them as a food source.

The nonlethal effects of elevated suspended sediment levels are not uniformly negative. Experiments have shown that predation on white sturgeon larvae by prickly sculpin increased in the presence of low-turbidity water (Gadomski and Parsley 2005). This suggests that some species rely on turbidity to some extent as cover.

#### Behavioral Effects

Aksnes and Utne (1997), Mazur and Beauchamp (2003), and Vogel and Beauchamp (1999) all report that suspended solids at sublethal concentrations 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. The general sensitivity of different fish species to suspended sediments is illustrative of potential effects on species for which data are lacking. However, it is important to note that under certain circumstances, elevated suspended solids may actually benefit certain species, such as salmonids, by providing cover (Gregory and Levings 1998) or triggering a sense of refuge from predation (Gregory 1993). The studies of Gregory and

Northcote (1993) indicated that when suspended solids concentrations exceeded 200 ppm, juvenile salmon increased their feeding rates while demonstrating pronounced behavioral changes in prey reaction and predator avoidance. Observed prey reaction distance decreased log-linearly with turbidity, yet feeding behavior peaked at moderate turbidity levels. They hypothesized that increased feeding behavior at higher suspended sediment concentrations may be due to contrast enhancement (caused by the light scattering effect of suspended particles) that makes prey more visible, and a perceived reduction in predation risk. These “tradeoff” factors compensated for an observed reduction in the ability to detect prey at distance as suspended sediment concentrations increased.

In studies of coho behavior in the presence of short-term pulses of suspended solids, Berg and Northcote (1985) found that territorial, gill flaring, and feeding behaviors were disrupted. At turbidity levels of between 30 and 60 nephelometric turbidity units (NTUs), social organization broke down, gill flaring occurred more frequently, and only after a return to a turbidity of 1–20 NTUs was the social organization re-established. Similarly, feeding success was also found to be linked to turbidity levels, with higher turbidity levels reducing prey capture success. In a study of dredging impacts on juvenile chum in Hood Canal, Salo et al. (1980) found that juvenile chum salmon showed avoidance reactions to high suspended sediment concentrations.

#### Effects on Invertebrates

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.

When suspended solids concentrations rise above the filtering capacities of bivalves, 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 the 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).

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.

#### *7.3.3.1.3 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).

In the context of fish screens, decreased DO levels are likely to occur only in specific cases where the operation of bypass channels associated with off-channel screens and specific types of in-channel screens results in rapid dewatering and stranding of organisms in the channel. Permitting of fish screens under the HPA program takes this potential into consideration, and this type of rapid dewatering is not allowed. However, there is nonetheless some potential for this impact submechanism to occur, leading to the related potential for decreased DO effects on HCP species.

Effects on Fish and Invertebrates

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 1-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. Tolerance for low oxygen levels varies across other species as well. For example, pygmy whitefish can withstand dissolved oxygen conditions below 5 ppm (Zemlak and McPhail 2006). Table 7-5 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 7-5. 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)*<br>and<br>No measurable change when waters are above 52°F (11°C) (weekly average) during incubation. | Applies throughout the period from spawning through emergence<br>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)*<br>and<br>≥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)*<br>≥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-, 90-DADMin = lowest 7-, 30- or 90-day average of daily minimum concentrations during incubation period.

It should be noted that recommendations are presented in Table 7-5 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 spawning, 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.

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). For example, in Chen et al. (2001), freshwater mussels (Unionidae) showed a wide range of tolerance for low DO levels depending on the types of habitats they inhabit. As would be expected, they found that species inhabiting slack water and warm water environments show greater tolerance for low DO levels, while species found in flowing water and cold water environments were far more sensitive.

Depleted DO levels can affect other invertebrate species as well, with implications for food web productivity. However, the range of sensitivity varies significantly across taxa. For example, leaches (Hirudinea), crustaceans (Decapoda), and many species of aquatic insects tolerate DO levels below 1.0 ppm (Hart and Fuller 1974; Nebeker et al. 1992), while other aquatic invertebrate species (e.g., Ephemeroptera, Plecoptera, Trichoptera) show variable sensitivity depending on the environments to which they are adapted. In general, organisms adapted to colder flowing water environments where DO levels are naturally high are expected to have lower tolerance for DO depletion (Nebeker 1972).

Kaller and Kelso (2007) found benthic macroinvertebrate density, including mollusks, to be greatest in low dissolved oxygen areas of a Louisiana wetland, while a literature review by Gray et al. (2002) noted 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 biochemical oxygen demand (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 on-site wastewater systems. These conditions in Puget Sound highlight the importance of reducing anthropogenically generated BOD.

##### *7.3.3.1.4 Altered pH*

When concrete is used in the construction of fish screens, discharge of concrete leachate or curing of concrete in contact with surface waters can drastically alter the pH of the receiving body. The pH of fresh and salt water normally ranges from 6.5–8.5 (Schlesinger 1997). When fish screens are constructed using concrete, the pH of surrounding waters can be affected if the uncured concrete or leachate is allowed to contact the receiving water body. Uncured concrete can dissolve in water and, depending on the temperature, 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), but once construction or maintenance 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. It is also important to note that these effects will typically be less pronounced in marine waters, which have a greater capacity to buffer perturbations in pH.

<u>Effects on Fish and Invertebrates</u>

Fish have adapted to the ambient pH levels of their particular habitat and tend to have narrow ranges of pH tolerance. The effects of high pH levels outside of their tolerance range can include death; damage to gills, eyes, and skin; and an inability to excrete metabolic wastes (DFO 2007). When ambient conditions are characterized by elevated ammonia and pH, ammonia toxicity in fish can occur because the organisms have difficulty excreting ammonia waste through their gills. At ambient ammonia concentrations of 5 ppm, the mortality of tambaqui (*Colosoma macropomum*; also known as pacu), a neotropical fish, 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.

pH alone can affect fish exposed to alkaline conditions. In a toxicity study of rainbow trout, 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). In white sturgeon, decreased sperm motility was observed when fish were exposed to pH levels below 7.5 (Ingermann et al. 2002).

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.

#### *7.3.3.1.5 Introduction of Toxic Substances*

Fish screens can result in the introduction of toxic substances to the aquatic environment through two primary pathways: accidental spills during construction, and operational failure. Heavy equipment used during the construction of fish screens requires fuel and lubricants. Even when managed with appropriate BMPs, the accidental introduction of these substances to the environment may occur. Fish screens also employ motorized mechanical and hydraulic systems for debris clearing and flow control, elements of which operate underwater. Equipment failure presents the risk of accidental introductions of toxic substances to the aquatic environment. Heavy equipment use may also be a pathway for the introduction of metals (e.g., copper and zinc from brake pad wear), which have the potential for toxic effects. The effects of exposure to these types of pollutants are described in the following section.

Effects on Fish and Invertebrates

The introduction of toxic substances to the water column can injure or kill aquatic organisms (NMFS 2005). Petroleum-based contaminants, such as fuel, oil, and some hydraulic fluids, contain polycyclic aromatic hydrocarbons (PAHs) that could be acutely toxic to salmonids at high levels of exposure and could also cause chronic lethal, and acute and chronic sublethal effects on aquatic organisms (Hatch and Burton 1999). Misitano et al. (1994) exposed larval surf smelt to Puget Sound (Eagle Harbor) sediments with high concentrations of PAHs and found 100 percent mortality after 96 hours of exposure. After diluting the sediments and repeating the experiments, they found that those larvae that did not expire within 96 hours suffered from decreased growth rates. Table 7-6, adapted from Jones & Stokes (2006) and Stratus (2005), depicts effects thresholds for PAHs in surface water for Pacific herring, zooplankton, mysids and marine amphipods, and trout.

**Table 7-6. Organism effects thresholds for PAHs in surface water.**

| Organism | Exposure Source | Toxicity | Concentration (parts per billion) | Citation |
|---|---|---|---|---|
| Mysid (*Mysidopsis bahia*) | Elizabeth River, Virginia, sediment extracts | 24-hr lethal concentration of a chemical within a medium that kills 50% of sample population | 180 | Padma et al. 1999 |
| Amphipod (*Rhepoxynius abronius*) | Eagle Harbor, Washington sediment extracts | 96-hour and 24-hr lethal concentration of a chemical within a medium that kills 50% of sample population | 1,800 | Swartz et al. 1989 |
| Pacific herring | PAHs leaching from 40-year old pilings | 24-hr lethal concentration of a chemical within a medium that kills 50% of sample population | 50 | Vines et al. 2000 |
| | PAHs leaching from 40-year old pilings | Significant reduction in hatching success and increased abnormalities in surviving larvae | 3 | Vines et al. 2000 |
| Zooplankton | PAHs leaching from pilings placed in microcosms | No observable effects concentration | 11.1 | Sibley et al. 2004 |
| | Commercial creosote added to microcosms | No observable effects concentration | 3.7 | Sibley et al. 2001 |
| Trout | Commercial creosote added to microcosms | Lowest observable effects concentration for immune effects | 0.6 | Karrow et al. 1999 |

Sources: Jones & Stokes 2006 and Stratus 2005.

Organic chemical contaminants can also affect prey production by limiting the suitability of substrates in the impacted area. Fish eggs can be particularly vulnerable to chemical contaminant exposure due to their inability to move out of the impacted area. Invertebrates can be similarly vulnerable due to the inability to move (or move quickly) out of the impacted area.

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 State 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, are listed in Table 7-7. Freshwater toxicity thresholds are hardness-dependent and can vary widely depending on calcium and magnesium carbonate concentrations. The standards presented here are based on median hardness concentrations estimated from an extensive 3-year data set (2001–2003) from the Green River watershed (Herrera 2007c).

**Table 7-7. Water quality criteria for metals in marine and freshwaters of the state of Washington.**

| | Freshwater [a] | | Marine | |
|---|---|---|---|---|
| Constituent | Acute | Chronic | Acute | Chronic |
| Arsenic | 360 | 190 | 69 | 36 |
| Copper | 7 | 7.5 | 4.8 | 3.1 |
| Lead | 22.9 | 1.5 | 210 | 8.1 |
| Nickel | 640 | 104 | 74 | 8.2 |
| Zinc | 51.6 | 69.2 | 90 | 81 |

Units: ports per billion (ppb).
Adapted from: WAC 173-201A.
[a] Freshwater toxicity thresholds are hardness-dependent.

### 7.3.4 Riparian Vegetation Modifications

Bankline in-channel screens and all off-channel fish screens pose some potential for riparian vegetation modification due to the fact that these types of screens require a bypass system. Bypass systems typically take the form of a pipe or a constructed channel to return water to the stream system. These elements are integral parts of the screen system; therefore, any effects they impose on riparian vegetation should be considered a function of this subactivity type.

The extent of riparian modification associated with bypass systems varies widely depending on the type of system and its extent. Some screen systems may employ a piped bypass that returns flow to the stream almost directly downstream of the diversion, requiring little additional riparian modification. Even longer bypass pipe systems installed by hand labor may have only minor effects on riparian vegetation. In contrast, due to the size of the diversion and local topography, some screen systems may employ constructed bypass channels of considerable length. In such cases, development of these channels may require extensive riparian modification. Assessing the extent of likely effects requires consideration of the scale and design of the screen system in question.

Using the worst-case scenario perspective, screen systems employing bypass channels that parallel the stream system for a significant length have the greatest potential to produce adverse impacts. Under such circumstances, the impact submechanisms resulting from the related riparian modification would be expected to be similar to those caused by channel creation and realignment, which are discussed in the Channel Modifications white paper (Herrera 2007b). These include:

- Altered shading and altered ambient air temperature regime
- Altered stream bank stability
- Altered allochthonous input
- Altered habitat complexity
- Altered groundwater-surface water interactions.

These impact submechanisms, the stressors they impose, and the resulting effects on HCP species are discussed in detail in the Channel Modifications white paper (Herrera 2007b). A summary of these effects relative to the magnitude of stressors expected to result from off-channel screen bypass systems is provided below.

Stressors associated with altered shading and altered ambient air temperature regime primarily include undesirable changes in stream temperatures. In summer, for example, lack of stream shading caused by the removal of overhanging trees to construct a bypass channel can increase solar irradiation and raise ambient air temperatures, which in turn can lead to increased stream temperatures. These effects may exacerbate stressful temperature conditions in situations where water withdrawals are sufficiently large to have a significant effect on base flow conditions.

Alteration of riparian vegetation also affects bank stability by reducing root cohesion. In the context of constructed bypass channels, the erosion protection function supplied by riparian vegetation was likely replaced by shoreline armoring in the past, leading to further undesirable changes in habitat quality through decreased inputs of allochthonous nutrients, altered groundwater and surface water interactions, and loss of habitat complexity.

Stressors resulting from these impact submechanisms include altered water temperature regime, decreased food web productivity, and decreased habitat complexity. The effects of stressor exposure on HCP fish and invertebrate species are discussed in detail in the Channel Modifications white paper (Herrera 2007b). In general, each of these stressors has the potential to impact the survival, growth, and fitness of freshwater HCP species that occur in environments where bankline in-channel and off-channel screens are commonly employed.

# 8.0 Cumulative Effects

This section provides an assessment of the cumulative effects that the two fish screen subactivity types evaluated in this white paper may have on the HCP species. This assessment has three primary emphases: (1) the cumulative effect of all direct and indirect effects associated with all of the impact mechanisms associated with a given subactivity type; (2) the cumulative effects of multiple fish screen structures distributed throughout the landscape; and (3) sequential fish screen structures that have a cumulative effect on individual fish populations.

As frequently stated throughout this white paper, fish screens are intended to minimize certain adverse effects from water withdrawals on aquatic species. Screening of diversion and intake structures has been broadly imposed as a matter of management policy across the landscape. This policy decision represents a defensibly precautionary approach to water resources management. While fish screens in many cases demonstrably reduce entrainment mortality, they may also impose unforeseen or unavoidable effects that must be considered.

The majority of the negative effects associated with fish screens occur as a result of two discrete impact mechanisms: construction and maintenance, and operations. The effects of fish screens realized through other impact mechanisms, such as hydraulic and geomorphic modifications, are expected to be minor in comparison, with specific exceptions. When the cumulative effects of multiple screens are multiplied across the landscape, or when these effects are considered in the context of the broader effects of flow control structures and water withdrawals on the ecosystem, they could potentially be significant. Construction-related effects are temporary to short term in nature, while operational effects are long term in nature but less intensive on an individual screen basis. Consequently, the cumulative impacts associated with operational effects are unlikely to occur unless multiple projects are being constructed simultaneously and in proximity to each other.

Fish screens are a necessary impact minimization technology used to limit the effects of dams, diversions, and intake systems. When properly employed, they can reduce cumulative mortality effects caused by entrainment into intake and diversion systems that can have significant implications for the population productivity of many HCP species. From this standpoint, the positive impacts of fish screens outweigh the negatives. However, fish screens may impose some detrimental cumulative effects as a result of the broad application of numerous screens across the landscape. The extent of these effects is difficult to predict and/or assess. Examples of potential cumulative impacts are provided below.

- Delayed migration: Multiple off-channel screen systems arrayed along a stream corridor could conceivably significantly delay migration, presenting a number of adverse consequences. In the case of upstream migration, screens with accessible bypass channels and/or high-flow bypass discharges may cause confusion regarding the migratory corridor, slowing migration or attracting fish up blind channels. Upstream migrant juveniles may be repeatedly drawn into bypass systems and discharged downstream, slowing migration to desirable habitats. In the case of juvenile downstream migration, the bypass system must provide suitable sweeping flows to avoid fish avoidance of the bypass structure and loitering in the diversion.

- Delayed or modified dispersal: The dispersal of weak-swimming or planktonic fish and invertebrate larvae may be affected by the operation of fish screens. Organisms drawn into screen systems may be effectively bypassed and removed, but could be discharged to environments that are unfavorable for rearing, or dispersal to favorable habitats may be delayed by exposure to multiple screens.

- Nonlethal impingement, bypass entrainment: Juvenile fish may experience nonlethal impingement on in-channel and off-channel screen surfaces, followed by escape, or stress from entrainment through high velocity bypass systems and discharge to the stream channel. While the effects of temporary impingement or bypass entrainment from a single screen may be small, the combined effects of incremental migration delays, stress, and injuries may be cumulatively significant.

- Effects of multiple screens on channel geometry and habitat complexity: Certain fish screen designs, specifically off-channel screens incorporating bypass channels, have the potential to exacerbate vegetation encroachment induced by changes in base flow conditions. This can in turn result in changes in channel geometry, flow velocity, substrate conditions, and resulting effects on habitat complexity in the affected bypass reach. Multiple off-channel screens are distributed throughout a stream system present some potential for more extensive cumulative effects on channel form.

Cumulative effects on fish will manifest primarily on species that are either migratory or are dependent on dispersal throughout the affected habitat types. Anadromous and migrant resident salmonids are a prime example. The potential for entrainment-related losses of salmonids was a primary concern driving the widespread use of fish screens on agricultural diversions in the Columbia River basin and elsewhere. Most fish screens in Washington State are primarily focused on avoiding adverse effects on salmon. Because of their migratory nature, however, salmon have the potential to be exposed to many fish screens throughout their life history. As such, they are likely to be exposed to impingement, migration delay, entrainment through bypass systems, and other related stressors several times. Individually, these stressors may not impose noticeable effects on survival, growth, and fitness, but the cumulative effects of multiple exposures could be significant.

Other HCP species potentially affected by the cumulative effects of fish screens include white sturgeon, mountain suckers, lamprey, and the dace. Lamprey, suckers, and sturgeon are also migratory species and are therefore potentially exposed to multiple fish screens during their life history. For lamprey, many screens designed to protect salmonids may not be adequately protective of weak-swimming amocoetes. Similarly, sturgeon larvae may depend on dispersal to nearshore and inundated riparian habitats for successful recruitment, exposing them to screen-related stressors. Fish screens may not provide adequate protection for these life-history stages. Dace, while not explicitly migratory, may depend on dispersal between suitable habitats to maintain population diversity. The cumulative effects of multiple fish screens could potentially limit the effectiveness of these dispersal mechanisms, affecting gene flow between populations and colonization of suitable habitats. Freshwater mussel species may be subject to cumulative indirect effects from cumulative effects on host fish distribution and abundance.

The potential for cumulative effects from multiple off-channels screens on habitat conditions in smaller stream systems may also be of concern. Where bypass systems represent a significant component of stream length and bypass flows are sufficient to exacerbate vegetation encroachment, broad scale changes in channel geometry could occur. These could result in extensive changes in habitat complexity, with implications for the survival, growth, and fitness of HCP species.

In marine systems, fish screens may similarly help to limit entrainment-related losses. However, it is difficult to avoid entrainment of species with planktonic eggs and larvae, such as hake, cod, and Olympia oyster, when these life-history stages are present. These entrainment-related effects are more the result of intake operation than the effects of the screens, and better represent the cumulative effects of this type of flow control structure. However, these effects also reflect fish screen design limitations. Knowledge of planktonic egg and larval sensitivity to entrainment and technologies suitable for limiting adverse effects may not be available for all potentially affected HCP species. In addition, currently available technologies are sensitive to biofouling and require consistent maintenance to remain effective.

It is important to restate that this assessment of effects considers the effects of fish screens relative to a natural system baseline. The cumulative effects of fish screens are, on balance, likely to be of lesser magnitude than the impacts of multiple unscreened intakes and diversions. In a similar fashion, the cumulative effects of fish screens are likely to be small relative to the combined effects of multiple water withdrawals on habitat capacity and productivity.

# 9.0 Potential Risk of Take

This section provides an assessment of the risk of take resulting from the impact mechanisms associated with the fish screen activity type. In the current regulatory environment, fish screens are intended to protect against adverse effects on aquatic species caused by entrainment into or impingement on water intake or diversion systems. Current design guidance encourages the selection of screen designs that are appropriate for their ecological context. However, while it is acknowledged that fish screens provide an environmental benefit, for the purpose of assessing risk of take, the baseline condition for this analysis is the stream system in the absence of artificial structures.

Two broad categories of fish screens are considered in this white paper: in-channel screens, and off channel screens. The risk of take resulting from construction and/or operation of these two fish screen subactivity types will vary considerably, given the differences between them. Moreover, fish screen designs within these two categories range considerably in scale and application, meaning that the magnitude and/or intensity of ecological stressors resulting from each subactivity type will vary depending on the design in question. Finally, risk of take will also vary by HCP species, depending on the nature of the stressor, 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 scale of the fish screen structure and where it is located. Therefore, the assessment of risk of take associated with each impact mechanism addressed here is necessarily broad and applies a "worst-case scenario" standard, with discussion of how actual risks may vary depending on design.

For this assessment, the species occurrence and life-history specific uses of habitats where fish screens are typically developed must also be considered. The risk of take is rated by impact mechanism for each species using the criteria presented in Table 6-3 and defined as follows:

- **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 a Likely to Adversely Affect (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, and/or 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.

- **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 summary is organized by subactivity type and impact mechanism category. The risk of take associated with in-channel screens is discussed in Section 9.1 (*In-Channel Screens*), and off-channel screens are addressed in Section 9.2 (*Off-Channel Screens*). Several impact mechanisms and stressors are common across both subactivity types. To limit redundancy, the risk of take associated with these common impact mechanisms and stressors is provided in Section 9.3 (*Risk of Take Associated with Common Impact Mechanisms*). Where appropriate, risk of take ratings unique to a particular environment type are specifically identified.

The narrative summary is supported by risk of take assessment matrices for each subactivity type summarizing the overall risk of take for each of the 52 HCP species by impact mechanism category and environment (Tables 9-1 and 9-2) presented at the end of the narrative portion of Section 9). The summary risk of take presented in the narrative and the matrices for each impact mechanism category represents the greatest overall risk of take from all impact submechanisms in that category.



## 9.1 In-Channel Screens

In-channel screens vary widely in scale and application. As discussed in Section 4.1.1 (*In-Channel Screens*], this subactivity type includes a variety of design types ranging from simple temporary structures (e.g., simple T-screens used on temporary intake systems for seasonal irrigation diversion pumps), to large and complex systems (e.g., permanent screen systems on power plant cooling water intakes, or bankline screen systems on large diversions). Clearly, the impact mechanisms and resulting ecological stressors produced by small and/or temporary screen systems will be of lesser magnitude or intensity than those produced by large, permanent structures. As such, some qualification of the possible risk of take resulting from this subactivity type is required.

The impact mechanisms associated with in-channel screens produce a number of environmental stressors with the potential to impose risk of take of HCP species. The degree of risk associated within and between these impact mechanisms varies depending on a number of factors. First, in-channel screen designs vary broadly in scale and application, and the magnitude or intensity of related impact mechanisms and stressors will vary in kind. Second, some impact mechanisms are expected to produce stressors with a relatively low risk of take due to their limited extent and/or short-term nature, while others may result in stressors with the potential to produce direct mortality or injury, or long-term modifications in habitat conditions. Using the risk of take criteria presented above, any impact mechanism with the potential to cause direct injury or mortality, or long-term changes in habitat conditions detrimental to survival, growth, and fitness are associated with a high risk of take.

The effects of in-channel screens also vary by environment type, due to the fact that the nature and scale of in-channel screen designs vary significantly depending on application. In smaller streams and rivers, in-channel screens typically take the form of small, often temporary, end-of-pipe style structures. The risk of take associated with these types of structures will generally be quite low. In contrast, in-channel screen designs employed in larger rivers, estuaries, large lakes and reservoirs, and the marine environment are commonly larger, permanent structures with greater potential for adverse effects, and therefore a greater risk of take. Bankline screens employed in marine and lacustrine systems, as well as larger rivers, may be located in embayments where they can impose ecosystem fragmentation effects. Moreover, these types of screen systems may also employ pump or lift-driven bypass systems with additional potential for adverse effects.

Impact mechanisms associated with in-channel screens include the following:

- Construction and Maintenance: This impact mechanism is associated with a variable risk of take, depending on the nature and scale of the structure in question. In-channel screens associated with temporary pumped intake systems require little in the way of in-water construction, and would generally be associated with a low risk of take. Larger, permanent bankline or end-of-pipe intake screens may require extensive in-water construction. Construction and maintenance of these types of screens would be associated with a high risk of take, due to the potential for actions associated with direct injury or mortality of HCP species.

- Water Quality Modifications: As with construction and maintenance, this impact mechanism is associated with a variable risk of take depending on the nature and scale of the screen in question. Risk of take levels are expected to range from insignificant for temporary in-channel screens, to moderate or high for large, permanent structures.

- Riparian Vegetation Modifications: Some modifications of riparian vegetation may be required to install piped bypass systems. These effects are expected to be minor, and the risk of take associated with these effects low (see Section 9.3.4 [*Riparian Vegetation Modifications*]).

- Hydraulic and Geomorphic Modifications: This impact mechanism is associated with a low risk of take for the majority of in-channel screen designs because the physical extent of hydraulic and geomorphic effects is expected to be limited relative to the intake structure.

- Ecosystem Fragmentation: This impact mechanism is associated with a high risk of take for bankline screens employing bypass systems. Other in-channel screen systems are associated with a low risk of take.

The risk of take associated with this subactivity type is summarized by impact mechanism in the following sections, qualifying the range of likely effects and resulting risk of take for in-channel screens of various scales. More detailed discussion of the risk of take associated with these impact mechanisms and justification for associated risk of take ratings is provided in Section 9.3 (*Risk of Take Associated with Common Impact Mechanisms*). These ratings apply to species that occur in habitats suitable for this subactivity type. Species-specific risk of take ratings by impact mechanism are provided in Table 9-1 (presented at the end of the narrative portion of Section 9). The ratings shown in Table 9-1 reflect the typical types of in-channel screen designs likely to occur in habitats used by the species in question. The ratings discussed in Section 9.3 represent the worst-case scenario perspective for common stressors imposed by in-channel screens. The information presented in this section should be used to assess the risk of take associated with different in-channel screen applications relative to this standard.

### 9.1.1 Construction and Maintenance

Construction and maintenance requirements for in-channel screens vary widely. For example, temporary pump intake screens require little in the way of what would be considered construction. They are simply placed in the source body with the intake pipe and anchored in place using some type of anchoring mechanism. They are commonly placed by hand, resulting in little disturbance of the stream bank or substrate. They are removed at the end of the use period. Screen placement and removal would be expected to result in minor visual and noise-related disturbance and minor pulses of suspended sediments, resulting in temporary behavior modification. Screen maintenance involves removal, cleaning, and replacement, resulting in similarly limited effects. Using the criteria presented above, these effects would equate to a low risk of take.

The worst-case scenario for in-channel screen construction would be associated with large, permanent end-of-pipe intake screens or bankline screen structures. These screen designs would likely require extensive in-water construction activity, potentially including dewatering and fish handling, pile driving (for cofferdam placement), and in-water use of heavy equipment. This impact mechanism is associated with a high risk of take due to the potential for direct injury or mortality from multiple impact submechanisms.

### 9.1.2 Operations

As with construction and maintenance, the risk of take associated with in-channels screen operation is variable depending on the type of screen design in question, and the submechanisms associated with that risk vary as well. Small, temporary screen structures employing passive debris clearing (e.g., T-screens on temporary pump intakes) or continuous active debris clearing (e.g., low velocity water jets or mechanical brushes) will have minimal effect on the aquatic environment. Risk of take associated with these structures is primarily associated with impingement or entrainment risk resulting from inadequate maintenance. Operational risk of take for these types of screens is generally considered to be low, providing that the structures are adequately maintained.

Large in-channel screen structures pose similar impingement and entrainment risks; however, they pose additional risk of take from the operation of certain active debris-clearing and bypass systems. Air burst debris-clearing mechanisms can produce periodic visual and noise-related disturbance over the lifetime of the structure. Similarly, active debris-clearing mechanisms (e.g., rotating panel screens) can create continuous mechanical noise and disturbance. These types of screens may also employ bypass systems, which while necessary are also associated with some potential for injury or mortality of organisms entrained through the system. This amounts to a long-term alteration of the surrounding aquatic environment by periodic stressors capable of causing injury, and/or modifying the behavior or the physiology of HCP species. These types of stressors are equated with a high risk of take.

### 9.1.3 Water Quality Modifications

Water quality modifications associated with in-channel fish screens are primarily the result of construction and maintenance, and to a lesser extent with operations. As with the other impact mechanisms associated with this subactivity type, the magnitude of stressors and resulting risk of take will vary considerably depending on the type of screen in question. Small screens on temporary intake structures would be expected to produce minimal impacts on water quality, as they are commonly placed by hand and require little if any disturbance of the bed or banks of the source body. Water quality effects under these circumstances would most likely be limited to minor, short-term pulses of suspended sediments. This type of stressor would be associated with an insignificant to low risk of take.

In contrast, large bankline or permanent intake structures will require more extensive construction and maintenance activities with the potential for greater water quality impacts. The risk of take associated with this impact mechanism for these types of stressors is well represented by the worst-case scenario risk described in Section 9.3.3 (*Water Quality Modifications*).

### 9.1.4 Hydraulic and Geomorphic Modifications

The hydraulic and geomorphic effects of in-channel screens are expected to be relatively modest in comparison to the intake or diversion structure they are associated with, as well as the broader effects of water withdrawals. However, some level of effect may result and should be incorporated into the risk of take assessment. As with the other impact mechanisms associated with in-channel screens, the magnitude of hydraulic and geomorphic impact submechanisms, and resulting stressors and risk of take, will vary depending on the scale and placement of the screen in question.

Small end-of-pipe screens on temporary pump intakes are expected to have little if any measurable hydraulic and geomorphic effect in most settings. These types of screens are small in scale and in place on a temporary basis. They have little potential to alter flow conditions, channel geometry, or substrate composition (Schille 2008). The resulting risk of take associated with this type of structure is expected to be insignificant.

In contrast, large permanent bankline or end-of-pipe screens may require placement of significant structures, with shoreline armoring and other forms of erosion protection. This presents the potential for a broader range of hydraulic and geomorphic effects and a greater risk of take. However, for the purpose of this white paper, these requirements are considered to be components of the intake or diversion system with which the screen is associated. The related effects and resulting risk of take are therefore considered also to be the result of the intake or diversion. Risk of take ratings from these subactivity types are addressed in the Flow Control Structures white paper (Herrera 2007a).

### 9.1.5 Ecosystem Fragmentation

In-channel fish screens have the potential to produce ecosystem fragmentation effects in specific circumstances. Intakes employing bankline screens in marine and lacustrine environments are commonly located in embayments (bankline screens in large rivers may be similarly configured). Because there is little or no available hydraulic head to operate bypass systems in these environments, aquatic organisms drawn into the intake must be pumped or lifted into bypass systems. HCP species with planktonic eggs and larvae may be drawn into these embayments by the intake and either retained or bypassed by the screen. Bypass systems have their own inherent potential to cause injury and mortality (as described in Section 9.1.2 [*Operations*]). From a worst-case scenario perspective, this type of screen could also impose ecosystem fragmentation effects if organisms drawn into the embayment area cannot be effectively bypassed, or if they are repeatedly bypassed and drawn back into the intake system. These effects are associated with a high risk of take.

## 9.2 Off-Channel Screens

Like in-channel screens, off-channel screens vary widely in scale but less broadly in application. For all practical purposes, off-channel screens are employed solely in riverine environments. As discussed in Section 4.1.2 (*Off-Channel Screens*), this subactivity type includes a variety of design types ranging from small, modular structures (e.g., modular drum screens with piped bypass systems used on a seasonal basis), to large and complex systems (e.g., permanent screen systems used on large irrigation diversion canals). The impact mechanisms and resulting ecological stressors produced by small, modular screen systems installed by hand will be of lesser magnitude or intensity than those produced by large, permanent structures. As such, some qualification of the possible risk of take resulting from this subactivity type is required.

The risk of take for HCP species exposed to off-channel screens varies depending on a number of factors. First, like in-channel screens, off-channel screen designs vary broadly in scale and application, and the magnitude or intensity of related impact mechanisms and stressors will likewise vary. Second, some impact mechanisms are expected to produce stressors with a relatively low risk of take due to their limited extent and/or short-term nature, while others may result in stressors with the potential to produce direct mortality or injury, or long-term modifications in habitat conditions. Using the risk of take criteria presented in the introduction to this section, any impact mechanism with the potential to cause direct injury or mortality, or long-term changes in habitat conditions detrimental to survival, growth, and fitness is associated with a high risk of take.

Off-channel screens are associated with the following impact mechanisms:

- Construction and Maintenance: This impact mechanism is associated with a variable risk of take, depending on the nature and scale of the structure and the construction setting. Screen construction and maintenance constructed "in the dry" are associated with an insignificant risk of take. However, in a worst-case scenario, construction of in-water features (e.g., bypass outfalls) would be associated with a high risk of take due to the potential for actions that are associated with direct injury or mortality of HCP species.

- Water Quality Modifications: This impact mechanism is associated with a high risk of take due to the potential for short-term water quality impacts during construction and operations that can cause direct mortality or injury. In most cases, however, a moderate risk of take is more appropriate.

- Riparian Vegetation Modifications: Some modifications of riparian vegetation may be required to install piped bypass systems. These effects are expected to be minor, and the risk of take associated with these effects low (see the Section 9.3.4 [*Riparian Vegetation Modifications*]).

- Hydraulic and Geomorphic Modifications: This impact mechanism is associated with a low risk of take for the majority of off-channel screen designs because the physical extent of hydraulic and geomorphic effects is expected to be limited. However, screen designs with integrated bypass systems may impose more extensive effects.

- Ecosystem Fragmentation: This impact mechanism is associated with a low risk of take. While some of the off-channel screen designs pose long-term risk of ecosystem fragmentation in comparison to the natural system baseline, these effects are likely to be insignificant.

The risk of take associated with off-channel screens is summarized by impact mechanism in the following sections, with the range of likely effects and resulting risk of take qualified for off-channel screen designs of different scales. More detailed discussion of the risk of take associated with these impact mechanisms and justification for associated risk of take ratings is provided in Section 9.3 (*Risk of Take Associated with Common Impact Mechanisms and Stressors)*. These ratings apply to species that occur in habitats suitable for this subactivity type. Species-specific risk of take ratings by impact mechanism are provided in Table 9-2 (presented at the end of the narrative portion of Section 9). The ratings discussed in Section 9.3 represent the worst-case scenario perspective. The information presented in this section should be used to assess the risk of take associated with different off-channel screen applications relative to this standard.

### 9.2.1 Construction and Maintenance

A defining characteristic of off-channel screens is that, by definition, this type of structure is constructed outside of the aquatic environment. Off-channel screens are constructed either in an artificial diversion channel environment or entirely in the dry as the channel is being constructed. This configuration also allows for relatively simple isolation of the structure as required for maintenance purposes. This in turn limits the potential for construction and maintenance related impacts on HCP species, depending on a number of factors.

When screen systems are constructed "in the dry," the potential for construction-related disturbance and water quality impacts is considerably diminished. Even when placed in existing diversion channels, the structures can be placed behind splashboard dams or similar flow control structures avoiding the need for dewatering. In such cases, the need for in-water construction work in most circumstances would be limited to the connection of bypass channels to the aquatic ecosystem.

Construction and maintenance effects will also vary depending on the nature and scale of the screen design in question. Small, modular screens with piped bypass systems placed by hand will produce lower intensity impact mechanisms than permanent screen structures scaled to large water diversions. The latter type of structure will commonly require at least some component of in-water work where the bypass system (either channel or pipe) discharges to the aquatic environment. Piped bypass systems are expected to involve minimal in-water work. In the case of bypass channels, more extensive in-water work is likely to be required. However, these effects are considered a component of artificial channel creation, which are addressed in the Channel Modifications white paper (Herrera 2007b).

On this basis, risk of take from off-channel screen construction and maintenance is expected to range from insignificant to low in the case of modular and smaller permanent screen systems. Most systems will require little in-water construction, meaning few if any impacts from equipment operation and materials placement. Similarly, dewatering and fish handling will most likely not be required.

### 9.2.2 Operations

Many off-channel fish screen designs will produce continuous noise and physical disturbance of the water column associated with active debris-clearing mechanisms. These effects represent essentially permanent, intermittent alteration of the environment lasting for as long as the screen is required. However, because off-channel screens are typically implemented in artificial channels and are configured to limit loitering by organisms drawn into the head ditch, exposure to these stressors will be limited and are therefore associated with a low risk of take.

The risk of take from entrainment and impingement described in Section 9.3.2 (*Operations*) applies to off-channel screen designs. In general, this impact mechanism is associated with a high risk of take due to the potential for mortality and injury, with the recognition that a lessened probability of impingement or entrainment is preferable to unmitigated entrainment into unscreened diversions.

### 9.2.3 Hydraulic and Geomorphic Modifications

Hydraulic and geomorphic modifications associated with fish screens are expected to be relatively limited in the majority of cases relative to the flow control structures or channel modifications they are associated with. Larger fish screen systems may certainly have considerable hydraulic and geomorphic effects due to their physical footprint, but because they are typically integrated with flow control structures, they are considered to be the result of those structures. These effects are discussed the Flow Control Structures white paper (Herrera 2007a).

As noted in Section 7.2.5 (*Hydraulic and Geomorphic Modifications*), however, in certain circumstances off-channel structures have at least some potential to impose hydraulic and geomorphic effects. These effects can occur as the result of flow-moderated changes in vegetation encroachment.

Risk of take associated with hydraulic and geomorphic modification impact submechanisms is assessed in the following sections from this standpoint, incorporating the worst-case scenario perspective. In practice, however, this risk must be considered in the context of the broader effects of flow diversion on channel conditions, as well as the beneficial effects of fish screens. In many cases, the design parameters of fish screens provide a means for controlling diversion flows, limiting diversion rates that exceed water rights. This provides a mechanism for preservation of base flows that may negate the influence of bypass system operation on base flow conditions.

#### *9.2.3.1 Altered Flow Conditions, Channel Geometry, and Substrate Composition and Stability*

Flow regime, channel geometry, and substrate composition and stability are dominant factors determining aquatic habitat structure in riverine environments. Because off-channel screens are typically intended for long-term use, these habitat alterations can be essentially permanent and continuous in the bypassed reach. This is particularly the case if flow-mediated vegetation encroachment changes the trajectory of channel evolution. If these effects are extensive, they can alter the productivity of the affected habitat for spawning, foraging, rearing, refuge, and other uses by HCP species.

In cases where hydraulic and geomorphic modifications are extensive, a broad array of research has demonstrated that detrimental effects on survival, growth, and fitness are likely to occur for many of the HCP species that occur in riverine environments. Using the criteria defined for the purpose of this white paper, effects of this nature equate to a high risk of take, with the recognition that the circumstances where this is likely to occur are rare.

### 9.2.4 Ecosystem Fragmentation

Ecosystem fragmentation refers to the disruption of ecological processes by reducing the connectivity between different components of the ecosystem, or the disruption of ecological processes. The following impact submechanisms have been defined to describe the ecosystem fragmentation effects potentially imposed by fish screen projects:

- Barriers to fish passage
- Modified upstream transport of allochthonous nutrients
- Modified downstream transport of woody debris and organic material.

The risk of take resulting from ecological stressors imposed by these submechanisms is described in the following subsections.

#### *9.2.4.1 Barriers to Fish Passage and Modified Upstream Transport of Allochthonous Nutrients*

Off-channel fish screens have the potential to impose a number of barrier conditions that could potentially lead to take of HCP species. Specifically, fish screens may unintentionally delay or otherwise hinder passage of downstream migrants due to design limitations. Conversely, specific off-channel screen configurations may also delay upstream migrants. A fish screen may delay or affect passage of only certain species, and may place unintended selection pressures on affected populations that limit or alter phenotypic diversity. Screens may become less effective at avoiding entrainment effects, or may create passage barriers over time if improperly designed for the conditions present or if maintenance is neglected. It is important to recognize, however, that overall effects on fish passage are relatively minor in comparison to the effects imposed by the flow control structures and channel modifications associated with water diversions and withdraws. While the magnitude of effects imposed by fish screens is expected to be limited overall, the long-term nature of these effects is consistent with a high risk of take for HCP species exposed to stressors resulting from this submechanism.

While the extent of effects on HCP invertebrate species is likely to be limited, indirect effects on upstream dispersal through direct effects on the migration and productivity of host-fish populations is possible.

In the riverine context, limitations on upstream fish passage may in turn result in long-term reductions in the abundance of migratory fish reaching areas upstream of screens. This in turn may result in an incremental decrease in food web productivity through reduced delivery of nutrients derived from allochthonous sources. Again, however, the overall extent of these effects is expected to be limited relative to those imposed by the related flow control structures. On this basis, the risk of take associated with this impact mechanism is expected to be insignificant. Upstream transport of nutrients is not relevant in marine and lacustrine environments.

#### *9.2.4.2 Modified Downstream Transport of Woody Debris and Organic Material*

Modification of downstream transport processes can lead to alteration in habitat complexity, changes in nutrient cycling, and subsequent hydraulic and geomorphic modifications. Each of these perturbations is associated with some risk of take. Given the long-term nature of these effects and the significance of altered ecosystem function, the risk of take would typically be considered high. However, the extent to which fish screens impose this effect is expected to be limited overall. Fish screens with the potential to impose this submechanism include those designs that collect debris in troughs for disposal, or that divert water into bypass channels that may require maintenance clearing. While this potential exists, the actual amount of wood and organic debris trapped on fish screens is not likely to represent a significant proportion of the natural flux. In addition, the incremental effect of the fish screen on this submechanism is likely to be minor in comparison to the flow control structure or channel modification associated with the water diversion. Because the extent of this effect on the environment is not quantified, the risk of take associated with this submechanism is unknown.

# 9.3 Risk of Take Associated with Common Impact Mechanisms

This section identifies the estimated risk of take for HCP species associated with the ecological stressors imposed by impact mechanisms common across the two fish screen subactivity types. The intent of this combined discussion is to maintain parallel organizational structure with Section 7.3 (*Effects of Common Impact Mechanisms and Stressors*) for ease of reference. This section provides a general discussion of the risk of take associated with each impact mechanism by component submechanism, as well as the rationale for the rating selected. The information presented in this section is intended to provide supporting context and detail for the risk of take discussion for each subactivity type provided in Sections 9.1 (*In-Channel Screens*) and 9.2 (*Off-Channel Screens*), and the species-specific risk of take ratings provided in Tables 9-1 and 9-2.

The following are common impact mechanisms associated with all fish screen subactivity types:

- Construction and maintenance
- Operations
- Water quality modifications
- Hydraulic and geomorphic modifications.

The risk of take associated with these common impact mechanisms is discussed in the following sections.

The risk of take ratings presented in the following sections represent the worst-case scenario of construction impacts associated with each subactivity type. When interpreting these ratings for common stressors, the mitigating factors affecting risk of take specific to each subactivity type must be considered, as described in Sections 9.1 and 9.2 (*In-Channel Screens* and *Off-Channel Screens*, respectively).

## 9.3.1 Construction and Maintenance

The construction and maintenance of any type of fish screen will inherently impose ecological stressors of varying severity that pose potential risk of take of HCP species. Risk of take ratings for each construction and maintenance submechanism are presented below.

### *9.3.1.1 Equipment Operation and Materials Placement*

The construction of fish screen structures may involve the use of heavy machinery and the placement of structural materials in and around the stream channel. Use of machinery (e.g., excavators) will generate noise and visual and physical disturbance. Larger, complex screen structures, such as in-channel or bankline structures may require the development of a dewatered exclusion area for construction using cofferdams. Cofferdams are constructed using a variety of methods, including placement of sheet piles with a pile driver, placement of bulk bags or ecology blocks, or erection of temporary earthen berms. Each of these practices creates physical, visual, and noise-related disturbance, with pile driving presenting the greatest potential for adverse impacts from underwater noise. Although no studies have addressed equipment noise associated specifically with construction of fish screens, many studies have addressed noise associated with pile driving, general underwater construction, and underwater tool use (see Section 7.3.1 [*Construction and Maintenance*]).

The risk of take associated with this impact submechanism will vary depending on the type of structure and the intensity of construction-related activities. At a minimum, underwater noise and visual and physical disturbance are likely to displace HCP fish species from occupied habitats, and to otherwise modify their behavior in ways that could affect survival, growth, and fitness. Using the criteria defined to rate risk of take, these short-term stressors would equate to a moderate risk of take. At worst, construction activities that produce intense underwater noise (e.g., installation of sheet piles for temporary construction cofferdams using an impact hammer) could lead to direct injury or mortality. This equates to a high risk of take.

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. Injury and mortality resulting from underwater noise exposure is equated with a high risk of take. 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. This would equate to a moderate risk of take.

The risk of take for HCP invertebrate species associated with underwater noise and visual disturbance is far less certain. Understanding of the sensitivity of invertebrate species to underwater noise and visual disturbance is limited. However, direct physical disturbance certainly imposes some risk of take. Depending on the nature and severity of the disturbance, the risk of take could range from moderate (e.g., from displacement) to high (e.g., from crushing or other forms of mechanical injury).

#### *9.3.1.2 Dewatering and Handling*

Temporary dewatering and flow bypass with fish removal and relocation from work areas are common and necessary practices during fish screen installation and possibly during maintenance. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants as well as to reduce the magnitude of stressor exposure. This construction and maintenance activity may pose a relatively high risk of take. Well-designed protocols and trained personnel are necessary to avoid high levels of mortality. Even with appropriate protocols and experienced field crews, high levels of mortality may result. For example, NOAA Fisheries evaluated take associated with dewatering and handling in a recent biological opinion. They estimated that cumulative salmonid mortality rates may range as high as 13 percent, even when trained personnel are used, from the combined effects of stranding and electroshock mortality. They assumed an electroshocking related 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 cryptic to collect and relocate effectively (e.g., juvenile salmonids hiding in cobble interstices, river lamprey amocoetes buried in fine substrate, larval or juvenile dace, larvae and juveniles of HCP invertebrate species). Mortality is the expected outcome for any individuals stranded within the exclusion area. Even in the absence of mortality, fish (and invertebrate) handling and relocation may result in stress and injury, as well as increased competition for forage and refuge in the relocation habitat. Moreover, the act of capture, handling, or forced behavioral modification of an ESA-listed species constitutes harassment, which is considered a potential form of take. Thus, the permitting of channel and work area dewatering poses a high risk of take of varying levels of severity, depending on habitat and species and life-history stage-specific factors.

In addition to these effects, the act of dewatering the stream and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects constitute a moderate risk of take of HCP species with migratory life-history stages.

#### *9.3.1.3 Dredging and Fill*

Dredging and fill activities associated with construction would ideally be conducted within a dewatered exclusion area to limit risk of take on HCP species. However, should this activity occur in the open channel, it presents the potential for high risk of take from two specific stressors, burial and entrainment. The sensitivity to these stressors generally varies by species and life-history stage. However, each HCP species that occurs in freshwater environments where fish screen subactivity types are likely to occur has at least one life-history stage with a high likelihood of suffering mortality or injury when exposed to either of these two stressors. Therefore, dredging and fill activities are considered to be associated with a high risk of take.

### 9.3.2 Operations

The purpose of a correctly operating fish screen is to avoid and minimize adverse effects on aquatic species caused by water withdrawal or diversion. While the benefits of a correctly operating screen are fairly clear, the fact that these structures are continuously interacting with the aquatic environment indicates the potential for adverse effects on HCP species. The risk of take associated with operational impact submechanisms is described in the following sections.

#### *9.3.2.1 Noise, Visual, and Physical Disturbance*

Both fish screen subactivity types include designs that produce some type of noise, visual, and physical disturbance when in operation. The principal difference between the two subactivity types is the nature of the disturbance. Some in-channel screen designs, typically the larger systems associated with large industrial or agricultural water intake systems, incorporate hydraulic jet or air burst debris-clearing systems that are activated periodically. The related disturbance is intermittent in frequency and short-term in duration. Stressor response is expected to vary depending on the sensitivity of the species exposed, with the most extensive effects involving behavioral alteration and habitat avoidance. Under a worst-case scenario, the long-term operations of these types of systems would be associated with a high risk of take for HCP species that are sensitive to low-level disturbance (e.g., hearing specialists species such as suckers and dace), while species that are relatively insensitive (i.e., HCP invertebrates) would be expected to experience an insignificant risk of take.

The off-channel screen subactivity type will generally create disturbance that is more continuous in nature. For example, motorized rotating barrel screens or designs with mechanical debris-clearing systems produce continuous underwater noise, splashing, and visual disturbance during operation. The level of disturbance produced is generally expected to be limited to levels associated with behavioral avoidance, or potential habituation. Risk of take resulting from these stressors varies by species and life-history stage. Species such as HCP invertebrates that are insensitive to disturbance would be expected to face an insignificant risk of take from this submechanism. In contrast, fish species that become habituated to continuous disturbance may experience auditory masking effects that result in increased vulnerability to predation or reduced foraging success. These effects are associated with a high risk of take for hearing specialist species such as cyprinids (which include HCP dace, chub, and suckers). However, this risk is mitigated by the fact that off-channel screens produce this stressor primarily in an artificially constructed environment (the diversion channel). This means that exposure would occur only for those species that are entrained into and occupy the diversion channel for extended periods.

Hearing generalist species, such as salmonids, would be expected to be less sensitive to these effects.

#### *9.3.2.2 Entrainment and Impingement*

Both in-channel and off-channel fish screens pose some unavoidable risk of entrainment or impingement of aquatic organisms when in operation. For certain species, specifically those with planktonic life-history stages, entrainment of free-floating eggs or larvae may simply be unavoidable if they are in the water column when an intake or diversion is in operation. This effect is in fact a function of the withdrawal that cannot be overcome due to the limitations of fish screen design. Therefore, a high risk of take is assigned for sensitive species exposed to this submechanism to reflect these shortcomings in screen design. Operational entrainment risk may also occur due to site-specific design limitations, or poor performance due to improper maintenance. In such cases, a high risk of take would extend to smaller species and life-history stages that are vulnerable to entrainment.

While a high risk of take rating is appropriate based on entrainment risk, the actual potential for population-level effects from this submechanisms varies considerably by species. For example, considerable numbers of Olympia oyster larvae may be entrained by a screened intake structure, but the resulting risk of take may be insignificant relative to natural larval mortality rates. In such a case, even though larval mortality may occur, the actual effect on population productivity would likely be insignificant. In contrast, the same intake may entrain larval lingcod at rates that greatly exceed natural mortality, suggesting the potential for significant population-level effects. These factors should be considered when assessing the significance of this impact submechanism.

Risk of impingement is a function of screen design, operation, and maintenance, as well as the swimming ability of the HCP species in question. As discussed in Section 7.3.2 (*Operations*), research has demonstrated that many fish species, including HCP species such as bull trout, can withstand short periods of screen impingement with no apparent ill effects. Applying a worst-case scenario perspective, however, the severity of impingement-related stressors may increase if screens are inadequately maintained, to the point that injury or mortality may occur. Moreover, the needs of weak-swimming organisms may not be fully accommodated by current screen design criteria, creating the potential for more serious effects. Given the potential for direct injury or mortality under these circumstances, impingement on fish screens must be equated with a high risk of take. Again, it is necessary to qualify this risk against the level of take that would likely occur from unmitigated entrainment of organisms into unscreened intakes or diversions.

HCP species with a high risk of impingement exposure include those that are likely to be in proximity to screens during their juvenile life-history stage, and/or are small in body size as adults. Fish species that come into proximity with fish screens only as large adults are less likely to experience impingement due to their stronger swimming ability. Similarly, low-motility HCP invertebrate species (e.g., Olympia oyster, freshwater mussels) are unlikely to come into contact with fish screens as adults. Risk of take for these species/life-history stages from entrainment is rated as insignificant.

### 9.3.3 Water Quality Modifications

The installation and operation of fish screen structures can result in water quality modifications during construction, maintenance, and operations. These effects are expected to be predominantly short term in nature. The risk of take associated with water quality impact submechanisms is described in the following sections.

#### *9.3.3.1 Altered Water Temperatures*

Fish screen operation has a limited capacity to affect water temperatures in specific circumstances. There are two mechanisms through which this can occur: (1) alteration of riparian vegetation associated with bypass system development, reducing shading, altering ambient air temperatures, and altering groundwater/surface water interactions; and (2) circumstances where bypass channel operation results in dewatering and stranding.

As discussed in Section 7.3.4 (*Riparian Vegetation Modifications*), the extent of riparian vegetation modification associated with the fish screen structures is expected to be limited. Riparian modification associated with bypass channel creation should be considered a component of intake or diversion development and/or artificial channel creation, which are addressed in separate white papers (Herrera 2007a, 2007b). Piped bypass systems are more arguably attributable to the fish screen system, but the magnitude of riparian vegetation modification associated with these structures is expected to be limited. On this basis, the temperature effects resulting from this impact mechanism are expected to be similarly limited and the related risk of take insignificant relative to the effects of flow diversion.

In contrast, bypass system operation can create circumstances where higher risk temperature effects could occur. Organisms inhabiting or transiting bypass channels can become stranded when the intake and screen is shut off and the channel is dewatered. In the absence of flowing water, stranded organisms may be exposed to rapidly increasing or decreasing temperatures, creating the risk of injury or mortality from thermal stress (in addition to other effects, including increased predation exposure and lack of forage). This potential equates to a high risk of take, with the recognition that this risk can be limited through screen design and operation. Rapid dewatering of bypass channels that are recognized to provide habitat functions for aquatic species of interest is not permitted. Bypass flows are often maintained in these channels to support beneficial habitat functions.

#### *9.3.3.2 Elevated Suspended Sediments*

Construction of fish screens is likely to result in bank and channel disturbance through the use of heavy equipment, materials placement, dredging and fill, and rewatering of exclusion areas. This disturbance is in turn likely to produce a short-term increase in suspended sediment loading. The effects of elevated suspended sediments vary depending on the magnitude of the stressor and the sensitivity of the species or life-history stage exposed to the stressor.

Nonmotile species or life-history stages exposed to pulses of high concentrations of suspended sediments may suffer direct mortality, injury, or extreme physiological stress, while motile species may be able to avoid these stressors. Stressors of this magnitude would typically be expected during the construction phase and would occur, most likely as short-term construction-related impacts. Given the potential for short-term injury or mortality resulting from elevated suspended sediment levels associated with construction, a high risk of take must be assumed for this submechanism for HCP species that occur in any habitat type where this activity type is likely to be used.

#### *9.3.3.3 Altered Dissolved Oxygen*

Generally, the direct effects of the fish screen subactivity types on dissolved oxygen conditions are expected to be limited, and the risk of take associated with these effects insignificant. However, indirect effects on dissolved oxygen conditions are possible in certain circumstances, specifically when operation of bypass systems (i.e., rapid dewatering) exposes organisms in bypass channels to stranding. This is a recognized potential issue in the context of screen operations, and the rapid dewatering is not permitted in channels that are known to be used as rearing habitat by aquatic organisms. However, under a worst-case scenario perspective, the potential for decreased DO conditions to emerge in conjunction with stranding must be recognized. Moreover, when this stressor occurs, it will commonly be associated with increased water temperatures as well, increasing the likelihood of injury or lethality. On this basis, this stressor is equated with a high risk of take.

#### *9.3.3.4 Altered pH*

The construction of fish screen structures may in some cases lead to the temporary alteration of pH levels. Many fish screens are constructed using concrete, a material that produces caustic leachate while curing. Concrete leachate released to surface waters from runoff or curing surfaces “in the wet” can increase pH levels well beyond levels capable of causing injury or mortality of all HCP species. In-water curing of concrete can take as long as 3 to 6 months, during which time pH levels in the immediate vicinity of the curing surface can reach levels as high as 13. This intermediate-term effect moderates over time as the concrete cures and can be minimized using appropriate BMPs. However, due to the significant level of potential adverse effects, this stressor is equated with a high risk of take.

#### *9.3.3.5 Introduction of Toxic Substances*

Fish screens could introduce toxic substances into the aquatic environment through two primary pathways: accidental spills from heavy equipment during construction and maintenance, and failure of mechanical equipment (i.e., debris-clearing systems) during operations. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects on exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of nonmotile larvae to fuel spills). More commonly, intermittent 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.

### 9.3.4 Riparian Vegetation Modifications

Installation of bankline in-channel screens and all off-channel screen types may result in some level of riparian vegetation modification to install the bypass system. The scale of the bypass system may range from a simple pipe with erosion protection at the outfall, to excavation of an artificial channel, the development of which is likely to have extensive effects on riparian vegetation. The extent of effects on riparian vegetation, and the resulting risk of take, is expected to vary depending on the scale of the screen and bypass system in question. For example, piped diversion systems associated with modular off-channel screens on small diversions would not be expected to have extensive effects on riparian vegetation. The resulting risk of take associated with these designs would be expected to range from insignificant to low. In contrast, excavation of artificial bypass channels to support large off-channel or bankline screens would be expected to have potentially significant effects on riparian vegetation, resulting in a high risk of take. The latter represents the worst-case employed for the purpose of this analysis, the effects of which and the resulting risk of take are considered to be similar to those described for channel creation in the Channel Modifications white paper (Herrera 2007b).

**Table 9-1. Species- and habitat-specific risk of take for mechanisms of impact associated with in-channel fish screens.**

| Species | Construction & Maintenance Activities | | | Operations | | | Water Quality Modifications | | | Riparian Vegetation Modifications | | | Hydraulic & Geomorphic Modifications | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine |
| Chinook salmon | H | H | H | H | H | H | H | H | H | L | L | L | I | I | I |
| Coho salmon | H | H | H | H | L | H | H | L | H | L | U | L | I | I | I |
| Chum salmon | H | H | I | H | H | I | H | H | I | L | L | I | I | I | I |
| Pink salmon | H | H | I | H | H | I | H | H | I | L | L | I | I | I | I |
| Sockeye salmon | H | H | H | H | L | H | H | L | H | L | U | L | I | I | I |
| Steelhead | H | H | H | H | L | H | H | L | H | L | U | L | I | I | I |
| Coastal cutthroat trout | H | H | H | H | L | H | H | L | H | L | U | L | I | I | I |
| Redband trout | H | N | H | H | N | H | H | N | H | I | N | L | I | N | I |
| Westslope cutthroat trout | H | N | H | H | N | H | H | N | H | I | N | L | I | N | I |
| Bull trout | H | H | H | H | L | H | H | L | H | L | L | L | I | I | I |
| Dolly Varden | H | H | H | H | L | H | H | L | H | L | L | L | I | I | I |
| Pygmy whitefish | H | N | H | H | N | H | H | N | H | I | N | I | I | N | I |
| Olympic mudminnow | H | N | H | H | N | H | H | N | H | I | N | I | I | N | I |
| Lake chub | H | N | H | H | N | H | H | N | H | I | N | I | I | N | I |
| Leopard dace | H | N | H | H | N | H | H | N | H | L | N | L | I | N | I |
| Margined sculpin | H | N | H | H | N | H | H | N | H | I | N | I | I | N | I |
| Mountain sucker | H | N | H | H | N | H | H | N | H | L | N | L | I | N | I |
| Umatilla dace | H | N | H | H | N | H | H | N | H | L | N | L | I | N | I |
| Pacific lamprey | H | H | H | H | H | H | H | L | H | L | I | L | I | I | I |
| River lamprey | H | H | H | H | H | H | H | L | H | L | U | L | I | I | I |
| Western brook lamprey | H | N | H | H | N | H | H | N | H | I | N | I | I | N | I |
| Green sturgeon | N | H | N | N | L | N | N | L | N | N | I | N | N | I | N |
| White sturgeon | H | H | H | H | L | H | H | L | H | L | I | L | I | I | I |
| Longfin smelt | H | H | H | H | H | H | H | H | H | I | I | L | I | I | I |
| Eulachon | H | H | N | H | H | N | H | H | N | I | I | N | I | I | N |
| Pacific sand lance | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Surf smelt | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Pacific herring | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Lingcod | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Pacific cod | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |

**Table 9-1 (continued). Species- and habitat-specific risk of take for mechanisms of impact associated with in-channel fish screens.**

| Species | Construction & Maintenance Activities | | | Operations | | | Water Quality Modifications | | | Riparian Vegetation Modifications | | | Hydraulic & Geomorphic Modifications | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine |
| Pacific hake | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Walleye pollock | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Black rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Bocaccio rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Brown rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Canary rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| China rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Copper rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Greenstriped rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Quillback rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Redstripe rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Tiger rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Widow rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Yelloweye rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Yellowtail rockfish | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Olympia oyster | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Northern abalone | N | H | N | N | H | N | N | H | N | N | L | N | N | I | N |
| Newcomb's littorine snail | N | H | N | N | N | N | N | L | N | N | L | N | N | I | N |
| Giant Columbia River limpet | H | N | H | I | N | I | M | N | N | L | N | N | I | N | N |
| Great Columbia River spire snail | H | N | H | I | N | I | M | N | N | L | N | N | I | N | N |
| California floater (mussel) | H | N | H | H | N | H | M | N | I | L | N | N | I | N | N |
| Western ridged mussel | H | N | H | H | N | H | M | N | M | L | N | N | I | N | I |

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 impact associated with off-channel fish screens.**

| Species | Construction & Maintenance Activities | | | Operations | | | Water Quality Modifications | | | Riparian Vegetation Modification | | | Hydraulic & Geomorphic Modifications | | | Ecosystem Fragmentation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine |
| Chinook salmon | L | L | L | H | H | H | H | H | H | L | L | L | H | I | I | I | I | I |
| Coho salmon | L | L | L | H | L | H | H | L | H | L | U | L | H | I | I | I | I | I |
| Chum salmon | L | L | I | H | H | I | H | H | I | L | L | I | H | I | I | I | I | I |
| Pink salmon | L | L | I | H | H | I | H | H | I | L | L | I | H | I | I | I | I | I |
| Sockeye salmon | L | L | L | H | L | H | H | L | H | L | U | L | H | I | I | I | I | I |
| Steelhead | L | L | L | H | L | H | H | L | H | L | U | L | H | I | I | I | I | I |
| Coastal cutthroat trout | L | L | L | H | L | H | H | L | H | L | U | L | H | I | I | I | I | I |
| Redband trout | L | N | L | H | N | H | H | N | H | I | N | L | H | N | I | I | N | I |
| Westslope cutthroat trout | L | N | L | H | N | H | H | N | H | I | N | L | H | N | I | I | N | I |
| Bull trout | L | L | L | H | L | H | H | L | H | L | L | L | H | I | I | I | I | I |
| Dolly Varden | L | L | L | H | L | H | H | L | H | L | L | L | H | I | I | I | I | I |
| Pygmy whitefish | L | N | L | H | N | H | H | N | H | I | N | I | H | N | I | I | N | I |
| Olympic mudminnow | L | N | L | H | N | H | H | N | H | I | N | I | H | N | I | I | N | N |
| Lake chub | L | N | L | H | N | H | H | N | H | I | N | I | H | N | I | I | N | I |
| Leopard dace | L | N | L | H | N | H | H | N | H | L | N | L | H | N | I | I | N | I |
| Margined sculpin | L | N | L | H | N | H | H | N | H | I | N | I | H | N | I | I | N | I |
| Mountain sucker | L | N | L | H | N | H | H | N | H | L | N | L | H | N | I | I | N | I |
| Umatilla dace | L | N | L | H | N | H | H | N | H | L | N | L | H | N | I | I | N | I |
| Pacific lamprey | L | L | L | H | I | H | H | L | H | L | I | L | H | I | I | I | I | I |
| River lamprey | L | L | L | H | H | H | H | L | H | L | U | L | H | I | I | I | I | I |
| Western brook lamprey | L | N | L | H | N | H | H | N | H | I | N | I | H | N | I | I | N | I |
| Green sturgeon | N | L | N | N | I | N | N | L | N | N | I | N | N | I | N | N | N | N |

**Table 9-2 (continued).** **Species- and habitat-specific risk of take for mechanisms of impact associated with off-channel fish screens.**

| Species | Construction & Maintenance Activities | | | Operations | | | Water Quality Modifications | | | Riparian Vegetation Modification | | | Hydraulic & Geomorphic Modifications | | | Ecosystem Fragmentation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine |
| White sturgeon | L | L | L | H | I | H | H | L | H | L | I | L | I | I | I | N | N | N |
| Longfin smelt | L | L | L | H | H | H | H | H | H | I | I | L | H | I | I | I | I | I |
| Eulachon | L | L | N | H | H | N | H | H | N | I | I | N | H | I | N | I | I | N |
| Pacific sand lance | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Surf smelt | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Pacific herring | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Lingcod | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Pacific cod | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Pacific hake | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Walleye pollock | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Black rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Bocaccio rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Brown rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Canary rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| China rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Copper rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Greenstriped rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Quillback rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Redstripe rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Tiger rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Widow rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |

**Table 9-2 (continued).** **Species- and habitat-specific risk of take for mechanisms of impact associated with off-channel fish screens.**

| Species | Construction & Maintenance Activities | | | Operations | | | Water Quality Modifications | | | Riparian Vegetation Modification | | | Hydraulic & Geomorphic Modifications | | | Ecosystem Fragmentation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine | Riverine | Marine | Lacustrine |
| Yelloweye rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Yellowtail rockfish | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Olympia oyster | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Northern abalone | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Newcomb's littorine snail | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N | N |
| Giant Columbia River limpet | L | N | L | I | N | I | H | N | N | L | N | N | H | N | I | ? | N | I |
| Great Columbia River spire snail | L | N | L | I | N | I | H | N | N | L | N | N | H | N | I | ? | N | I |
| California floater (mussel) | L | N | L | H | N | H | H | N | H | L | N | N | H | N | I | I | N | I |
| Western ridged mussel | L | N | L | H | N | H | H | N | H | L | N | N | H | N | I | I | N | I |

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.

# 10.0 Data Gaps

This section summarizes the key data gaps identified during the preparation of this white paper that are relevant to fish screens. When viewed across both subactivity types, it is clear that key data gaps remain in the following general areas:

1. Knowledge of the movement patterns of HCP species at different life-history stages relevant to the development of design and operational guidance for fish screens.

2. Knowledge of the behavioral and physiological limits on the swimming ability of HCP species, sufficient to guide definition of screen design criteria, particularly for nonsalmonid species.

3. Lack of useful design criteria across the range of environment types and conditions where screens are employed.

4. Clear demonstration that fish screens are an effective tool for protecting the productivity and diversity of HCP species (relative to other conservation measures).

These issues apply generally across both fish screen subactivity types and are therefore discussed further here. Data gaps relevant to a specific subactivity type are discussed in the following sections, followed by a discussion of data gaps specific to common impact mechanisms and stressors.

General data gap 1 is relevant to fish screen operation, but it encompasses an issue driven more so by water removal from the system. Essentially, fish screens can only impose operational effects when an intake or diversion is active, meaning that both fish screen design and managing the timing of water withdrawals are two available tools to limit adverse effects on HCP species. A better understanding of the range of species and life-history stages likely to occur, as well as the timing of their occurrence, is necessary to select the most appropriate screen design. In cases where the protection provided by fish screens is fundamentally limited, knowledge of when sensitive species and/or life-history stages are present can be used to manage the timing of water withdrawals. For example, intake systems that will unavoidably entrain fish larvae at high mortality rates could be shut down when larvae are most likely to be present. These management practices would require an expansion of WDFW's authority to regulate water withdrawals, which is currently limited.

With regard to general data gap 2, available data show that screen effectiveness may vary by species, depending on factors such as swimming physiology, behavior, sensitivity to bypass entrainment, the unintended effects of stimuli that might be used to guide them toward or away from intakes, and other factors. For example, fish screens designed to protect salmon may also be effective at protecting fish species with similar swimming physiology, but may not be protective for weaker-swimming species such as juvenile lamprey (Close et al. 1998). Most screen research in Washington State has focused on protection of salmonids, resulting in criteria that may not provide adequate protection for other native species. However, at least some research is available on the response of nonsalmonid species to fish screens. Better understanding of the tradeoffs between screen function and species protection will allow for more effective design and operational guidance. This issue is discussed in more detail in Section 10.3.2.2 (*Entrainment and Impingement*).

With regard to general data gap 3, while uniform design guidance would be desirable, the bulk of available research indicates that it is impractical to develop guidance applicable for all environments and uses. The factors that determine the most appropriate screen design for a given situation are highly dependent on both the type of withdrawal (intake or diversion) and site-specific conditions. For example, an effective screen design for an agricultural diversion must consider a number of competing factors, such as the diversion flow rate, flow conditions and variability of the source body, the expected volume of naturally transported debris that must be cleared or passed, and the swimming physiology and sensitivity of the full range of HCP species that occur in the affected environment. This presents a complex set of demands that are not easily addressed by uniform design guidance. This suggests a need for a broader set of assessment steps that can be used to develop site-appropriate designs.

With regard to common data gap 4, it is not clear that fish screens provide a conservation benefit for all species and all circumstances. While the issue of fish entrainment in industrial and power plant water intake systems is effectively mitigated by fish screens (Goodyear 1977; Hadderingh 1979; Taft and Mussalli 1978; Travnichek et al. 1993), the effectiveness of off-channel screen designs has been less well studied in agricultural applications. Moyle and White (2002) and Moyle and Israel (2005) conducted a broad review of published literature and found that despite policy directives dictating the widespread implementation of fish screens on agricultural diversions, relatively few studies have attempted to evaluate their effectiveness at maintaining or increasing population abundance and productivity. The literature suggests that this lack of evaluation is typical throughout the western United States, despite millions of dollars spent annually on fish screen installation and maintenance (Moyle and Israel 2005). While it can be argued that these studies are unnecessary because the conservation benefits are clear, it may be useful to consider more directed study to identify and prioritize the diversions with the greatest impact on fish populations, and to determine which types of screens provide the best protection for HCP species likely to be exposed.

More generally, the effects of certain forms of disturbance on HCP invertebrate species is generally poorly understood, which limits the understanding of potential impacts from construction and operational activities. Specifically, the sensitivity of HCP invertebrate species to impulsive and continuous underwater noise, visual, and physical disturbance is unclear. While these species are not expected to be sensitive to auditory masking or similar effects that would increase predation vulnerability or interfere with foraging, other potentially important effects may occur that are not readily apparent.

## 10.1 In-Channel Screens

In-channel or end-of-pipe screen systems are relatively simple in design in comparison to off-channel structures, and their effects are more broadly understood. Data gaps related to this subactivity type primarily concern uncertainty about the presence of HCP species with sensitive life-history stages (e.g., small size, planktonic or weak swimming) that cannot be effectively protected by current fish screen designs. For example, flow and velocity requirements necessary to draw various life-history stages of salmon into bypass systems are not well known. Although current designs seem to be effective, they are not likely optimized for either fish passage or flow management because of a lack of empirical data.

This uncertainty can only be addressed by amassing available site-specific data or conducting the necessary research to understand the timing and distribution of sensitive life-history stages in relation to the desired operating parameters of the water intake system. This understanding can be used to set operational limits as necessary to overcome limitations in screen performance. Guidance criteria for the siting, design, and operation of infiltration gallery screens are currently lacking. Additional research should be conducted to determine if this technology has practical utility and, if so, to identify appropriate uses and develop design criteria.

## 10.2 Off-Channel Screens

The off-channel screen subactivity type encompasses a number of screen designs that range from relatively simple to complex. The design requirements for these structures are highly site specific. Although generalized guidance can provide some basis for selecting an appropriate design, site-specific assessments and research will be necessary to develop these designs fully.

Recognizing this, a number of data gaps have been identified that–when addressed–could improve both the general guidance for species protection and an understanding of the limitations of certain screen designs. These include:

- ***Passage-related effects of fish screen designs:*** The potential for certain types of off-channel fish screens, specifically those with integrated bypass channels, to create attraction flows that unintentionally delay adult migration has been identified as an issue of concern from a design perspective by WDFW (2001a). However, empirical data necessary to provide clear design guidance on this subject are currently lacking. Similarly, screens with bypass channels must produce adequate sweeping flows to avoid delaying downstream migrant salmonids. While sweeping flow requirements are fairly well understood for salmonids and some other fish species, the needs of some HCP species (e.g., lamprey) appear to be less clear.

- ***Upstream movement requirements of HCP invertebrate species:*** The freshwater HCP invertebrate species vary in terms of the mechanisms they use to influence dispersal in flowing water environments. Unionid mussels, as is well known, rely on host-fish species to disperse their parasitic larvae to upstream environments. However, these species have also been shown to disperse upstream for short distances by crawling along the bottom using their muscular foot and byssal thread attachments (Vaughan 2002). Other HCP invertebrate species, such as the giant Columbia River limpet and great Columbia River spire snail, crawl along hard substrates and are theoretically capable of navigating upstream for short distances. The degree to which fish screens may help or hinder these dispersal mechanisms and the ramifications for population health are an area requiring additional study.

- ***The ecosystem fragmentation effects of screens:*** Delayed migration and other fish passage-related ecosystem fragmentation effects have been widely documented in association with flow control structures and shoreline modifications in freshwater and marine environments. Certain off-channel fish screen designs may also affect upstream and downstream fish passage by delaying migration, or imposing unintended selection pressures on affected populations. Fish screens and associated water withdrawals may also affect the transport of organic material and woody debris. However, the extent and severity of these effects, particularly the cumulative effects of multiple screens distributed across the landscape, are not clear. This is an area that could benefit from additional research. Given the site-specific nature of these effects, however, it may be difficult to produce results that lead to broadly applicable guidance.



## 10.3 Data Gaps for Common Impact Submechanisms and Stressors

Impact mechanisms common across both fish screen subactivity types impose a range of potential stressors that could affect HCP species. Several data gaps exist with regard to species-specific sensitivity to these stressors, as well as the range of possible effects of stressor exposure. These data gaps are discussed by submechanism in the following sections.

### 10.3.1 Construction and Maintenance

#### *10.3.1.1 Equipment Operation and Materials Placement*

The following data gaps were identified in relation to the effects of noise-related disturbance on HCP species in marine, lacustrine, and riverine environment.

- The sound sensitivity of primitive fishes (such as lamprey) is currently unknown.
- The sound sensitivity of the Olympia oyster is currently a data gap, and the effects of related sound stressors are unknown.
- Effect of underwater noise on mollusks in general is a data gap.

#### *10.3.1.2 Dewatering and Handling*

Few studies have compared the susceptibility of various fish and macroinvertebrate species to different types of handling techniques. More information comparing the susceptibility to injuries associated with these types of techniques is needed to identify potential take for these species. Training and minimum qualifications for personnel performing fish capture and handling (particularly electrofishing) are also needed to define standard protocols that would minimize risk of take. Most of the studies on the effects of fish handling have been performed on electrofishing. Electrofishing effects have been conducted on adult fish greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids indicate that spinal injury rates are substantially lower than they are for large fish. Only a few recent studies have examined the long-term effects of electrofishing on salmonid survival and growth (e.g., Ainslie et al. 1998, Dalbey et al. 1996). Little research has been conducted on the effects of dewatering and fish capture and handling on nonsalmonid HCP species. More directed research is necessary to understand the risk of take resulting from this submechanism for these species.

#### *10.3.1.3 Dredging and Fill*

The effects of minor construction-related dredging and fill are generally well understood. No specific data gaps were identified.

### 10.3.2 Operations

#### *10.3.2.1 Noise, Visual, and Physical Disturbance*

Data gaps associated with this submechanism are similar to those described in Section 10.3.1.1 (*Equipment Operation and Materials Placement*).

#### *10.3.2.2 Entrainment and Impingement*

Entrainment and impingement risk is a subject of continuous and ongoing research as fish screen design advances. Despite a large body of existing research, much of the information necessary to protect the broad range of HCP species potentially exposed to screens from these stressors remains unknown. For example, the bulk of available research has focused on fish with subcarangiform swimming physiology (side-to-side undulation of the posterior one-third to onehalf of body length), a characteristic of most, but not all, HCP fish species. This research provides the primary base of information on swimming performance used to guide design. However, design criteria based on these data are not likely to provide adequate protection for weaker-swimming fish, specifically lampreys, with anguilliform swimming physiology (eel-like full body undulation). Even for well-understood species such as salmonids, several factors such as species, age class (i.e. size), condition, and water temperature can influence swimming performance in ways that are relevant to design. Sensitivity to injury or other adverse effects also varies between species. For example, Zydlewski and Johnson (2002) evaluated fish screens designed for anadromous salmon protection and found that while juvenile bull trout were frequently impinged on the screen, they were able to escape and were effectively passed downstream without apparent injury or adverse effects. In contrast, Swanson et al. (2005) and White et al. (2007) found that even limited screen contact caused stress and injury sufficient to lead to delayed mortality in delta smelt.

Many design criteria in common use today are based on untested theories (Bates 2008). Design criteria that should be subjected to further research and scrutiny include the following:

- The relationship between screen mesh size and approach and sweeping velocity for balancing debris-clearing effectiveness against impingement and entrainment risks
- The efficacy of widely used sweeping velocity parameters for guiding various HCP fish species and life-history stages across screens and into bypass systems
- Effects of nonuniform approach velocity on impingement risk
- Use of turbulence, light, sound, and other mechanisms to deter or guide fish
- Efficacy of various cleaning mechanisms relative to different types of debris (e.g., hydraulic eddy cleaners), and related risks to HCP species
- Investigation and development of new cleaning technologies, such as vortex separators, to continuously clear sediment from screen bays
- Optimization of bypass configuration for fish collection and flow management
- Appropriate bypass depths and velocity for fish protection and water management
- Screening designs for planktonic larval life stages including effects of impingement, handling, and release.

### 10.3.3 Water Quality Modifications

In general, additional information is needed regarding how cumulative impacts related to water quality degradation may affect HCP species. As indicated in Jones & Stokes (2006), information is needed to identify the impacts of suspended sediments on HCP species. Bash et al. (2001) filled many data gaps for freshwater habitats, but additional information is needed to evaluate effects of turbidity and suspended sediment on freshwater HCP species, and more data are required to evaluate impacts on marine habitats and species.

In addition, 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.

### 10.3.4 Hydraulic and Geomorphic Modifications

In other regions of the United States, studies have documented the cumulative impacts on the nearshore environment (e.g., the Great Lakes [Meadows et al. 2005]). Additional data and analysis describing the ecosystem processes affected by certain types of fish screens are desirable. Screen designs of potential concern include large, permanent in-channel structures capable of altering local hydraulic and geomorphic conditions in riverine, marine, and lacustrine environments.

In certain circumstances, off-channel structures may also cause undesirable effects. Specifically, screens that require a significant component of remaining instream flow to operate a bypass system may cause hydraulic and geomorphic effects by encouraging vegetation encroachment. Additional research to identify the types of stream channels sensitive to these effects may be desirable.

With regard to the marine environment, Finlayson (2006) identified five areas where additional research pertaining to physical nearshore processes is needed:

1. Characterizing the role of historical morphology
2. Identifying tide-level controls on littoral phenomena
3. Further development of existing littoral transport models
4. Improved characterization of the role of extreme events in shaping low-energy, mixed-sediment beaches
5. Further testing and adaptation of numerical wave models for fetch-limited environments.

No research has been conducted to study submarine and intertidal groundwater in Puget Sound. It is clear from work elsewhere that such flows are crucial in sustaining nearshore ecosystems

(Gallardo and Marui 2006); however, their role on the nearshore environment throughout Puget Sound is virtually unknown (Finlayson 2006). Again, fish screens are expected to have relatively modest hydraulic and geomorphic effects in comparison to the flow control structure (or other activity types) associated with the related water intake or diversion system.

# 11.0 Habitat Protection, Conservation, Mitigation, and Management Strategies

Risk of take associated with fish screens results from two primary impact mechanisms: the effects of construction, and the effects of operation. Construction-related effects are straightforward, and the risk of take can be minimized by using standard BMPs for in-water construction activities. Operational effects are more complex; however, this area presents several opportunities for improvement in how fish screens are designed, constructed, and operated.

This section provides several strategies for improving how fish screens are used in Washington State. These strategies fall into the following categories, which are addressed in the subsequent subsections:

- Management strategies for improving how fish screens are designed and operated
- Strategies for improving fish screen subactivity type
- General strategies by impact mechanism.

## 11.1 Management Strategies

### 11.1.1 Improved Training and Research

A primary issue that limits the operational effectiveness of fish screens is the fact that a significant proportion of fish screen designers have no training or experience in this unique bio-engineering field. Designing an effective fish screen requires an integrated understanding of the engineering demands of the structure, site-specific performance requirements, and understanding of the biological needs of the species the screen is intended to protect. This combined knowledge is necessary to develop both an effective screen design, and to provide operational parameters for the water withdrawal or diversion when sensitive species are present that cannot be effectively protected.

WDFW currently provides training, design, and installation assistance for screening projects through the Technical Applications (TAPPS) Division. WDFW-sponsored research conducted at the Yakima Screen Shop facility has produced many of the screen concepts and design criteria in current use in the region. There is some level of ongoing coordination among state and federal agencies in the Pacific Northwest on research and practical application of screening technologies. However, funding cuts in recent years have limited research and collaboration, leading to the abandonment of efforts targeted at developing and building effective screening technologies. The TAPPS Division and the screen research program should be strengthened and coordinated with efforts at the federal level and in other states in the region.

Web-based case studies that evaluate the effectiveness of integrated design and operational parameters would be particularly useful.

### 11.1.2 Improved Guidance

The most current WDFW guidance on fish screen design is in incomplete draft form and has not been revised since 2001 (WDFW 2001a). This guidance document should be updated and improved based on the latest technical information and made available to managers and the public. A notable weakness in this and other fish screen guidance documents is the widespread use of inconsistent terminology, resulting in standards that are confusing and at times contradictory. The revised guidance document should be coordinated for consistency with NOAA guidelines, using a parallel format and consistent terminology to allow for easy cross-referencing among documents. Where state standards necessarily depart from federal guidance, the differences should be clearly highlighted and the rationale for the departure explained. The design guidance should also incorporate a set of typical design drawings for common screen designs and a range of flows, as well as provide contact information for manufacturers and vendors. The guidance should be supported by up-to-date web-based technical assistance, including current case studies that are regularly updated.

Currently, fish screens are typically designed conservatively around the most extreme scenarios to provide fish protection (i.e., the smallest and weakest swimming of fish, the most extreme temperature conditions [which affect swimming performance], and the highest flow rates), conditions that are rarely observed in practice. Using the swimming performance and requirements of the smallest and weakest-swimming species and/or life-history stages likely to be exposed to the screen is presumed to provide broad protection for all other species and life-history stages. This leads to screen facilities that might impose a greater burden on the operator due to their operational limits and maintenance requirements, meaning that operators have an incentive to contribute to research.

However, even when properly engineered for site conditions, a fish screen may not be able to protect all HCP species/life stages. For example, planktonic larvae may be unavoidably entrained through even the most protective screen system. WDFW may want to investigate expansion of authority under the Hydraulic Code to allow for withdrawal restrictions to be included as part of the approval process under the HPA program for fish screens.

These restrictions would be enforced in circumstances where screens cannot provide adequate protection when sensitive life-history stages of various species are present. Water users in high-priority habitats (i.e., habitats where HCP species may be acutely vulnerable) should be required to develop an operational plan that is certified by state and federal agencies. Moreover, research should be dedicated to developing effective screen technologies for settings where flow restrictions are not practicable.

### 11.1.3 Improved Performance and Compliance Monitoring

More consistent monitoring and enforcement will greatly benefit the advancement of fish screen science, and help to ensure that existing screens are as protective of HCP species as possible. Performance monitoring is a necessary tool to determine whether existing screens are functioning as intended and how effective they are at avoiding or limiting entrainment and impingement of sensitive species and/or life stages. For nearly two decades, the Bonneville Power Administration (BPA) has funded ongoing monitoring of fish screen systems on several of the larger irrigation diversions on Columbia River tributaries, including the Yakima, Walla Walla, Umatilla, and other river systems (Carter et al. 2003; Knapp 1992; McMichael and Chamness 2001; Vucelick and McMichael 2003; Vucelick et al. 2004). WDFW has received funding from BPA and NMFS through intergovernmental memoranda of agreement to conduct screen inspection and maintenance on screen systems throughout the Columbia River basin, and to a lesser extent in western Washington. WDFW currently operates a statewide screen maintenance and inspection program, partially subsidized by federal funds, that provides maintenance guidance and monitors maintenance compliance and screen performance.

Improved compliance monitoring is an additional and necessary strategy to enhance protection of HCP species. Simply put, even when the best possible screen design and operational criteria are developed, some fish screens will not be operated or maintained as necessary to provide the level of protection desired. Noncompliance with permitting requirements is certain to be an issue of concern regardless of any advances in screen design and operational implementation. Full funding and expansion of the WDFW program would provide a useful and necessary means for training fish screen specialists, and provide case studies for demonstrating successful design and operational procedures. This type of program should consider the following objectives:

- Pre- and postconstruction review of fish screen designs and as-builts for all high-priority screen projects to confirm that the structure was built as intended

- Incorporation of operational certification into the approval process under the HPA program, with a set recertification schedule based on inspection performance

- Routine monitoring of fish screens (e.g., every other year, every 3 years) to evaluate compliance with maintenance and operational requirements for recertification purposes

- Coordination with performance monitoring to provide a mechanism for addressing underperforming structures.

A comprehensive compliance program should include a mandatory but practical pathway for owners of noncompliant screens to address structural and operational issues as quickly as possible. Compliance incentives should first provide funding and technical assistance (building on existing state-level programs) to help owners meet recertification requirements, followed by enforcement and legal action as necessary.

## 11.2 Subactivity Type Specific Strategies

### 11.2.1 In-Channel Screens

In-channel screens vary widely in design configuration, ranging from simple screens on small, private water supply systems to large and complex structures on industrial water intake systems. Given this variety, the strategies identified lean toward recommendations specific to designs for certain applications or of a certain scale. Strategies identified include the following:

- ***Infiltration galleries:*** Guidance criteria for the siting, design, and operation of infiltration gallery screens are currently lacking. This design guidance should be developed and incorporated into the recommended guidance document discussed in the previous section.

- ***Improve fishway screening requirements:*** Current design guidance for fishways and fish ladders does not require screening of auxiliary intake systems. This may unnecessarily expose smaller sensitive fish to entrainment-related injury. Requiring screening of these features is recommended.

- ***Siting of large intake systems:*** In-channel fish screens are currently limited in their ability to avoid entrainment of HCP species with planktonic life-history stages. While the technology is advancing, adverse impacts can be limited or avoided by siting intake systems at locations and depths where sensitive life-history stages are less likely to occur. This highlights the value of incorporating biological expertise into the design of fish screens and related flow control structures.

### 11.2.2 Off-Channel Screens

Off-channels screens also encompass a range of designs appropriate for different conditions; therefore, design strategies are relatively specific to given design types.

- ***General flow control for off-channel screens:*** Overtopping by high flows or due to debris accumulation is the most common cause of screen failure and elevated entrainment risk. Other changes in flow conditions can change diverted flow rates, screen submergence, bypass flows, and other parameters in ways that adversely affect screen performance. Screens should be generally designed to accommodate the hydrologic context of the system in question. Automated headgate systems programmed to respond to changes in flow conditions should be used to maintain design flows.

- ***Flow control for inclined plate screens:*** The screen design should provide for a minimum depth of water over the entire screen face. This depth should be based on expectations of the size and type of debris, size, and condition of fish (or other HCP species) requiring passage, and the potential variation in flow that could reduce the depth to below the desired minimum. To achieve these conditions, a substantial amount of bypass flow is typically required and flow conditions must be carefully monitored. Downward sloping screens require at least several feet of head loss to operate effectively. These constraints typically limit this type of screen to riverine applications. Because of the restrictive control of flow necessary for downward-sloping fixed plate screens to provide fish and debris clearance, this design is not recommended except where constant and precise flow control can be provided.

- ***Avoiding ecosystem fragmentation effects:*** The following factors should be incorporated into off-channel screen design to avoid ecosystem fragmentation effects:

  - Screens employing bypass channels must provide sufficient sweeping velocities to draw downstream migrant and dispersing fish into the bypass system, avoiding delay.

  - Bypass systems should be sited to minimize the length of the bypassed reach, and outlets should be sited to minimize predation on organisms exiting the system.

  - Bypass outlets should not be located in side channels or other channel features where the discharge could create attraction flows that delay upstream movement of migratory species.

  - The potential cumulative effects of migration delays imposed by multiple screen systems should be considered when permitting the screen as well as the related flow control structure or channel modification.

## 11.3 General Strategies by Common Impact Mechanism

This section identifies general strategies and best management practices (BMPs) that can be used to avoid and minimize impacts associated with the common impact mechanisms imposed by fish screens. For ease of reference, this discussion is organized by impact mechanism in parallel format with Sections 7.3 (*Effects of Common Impact Mechanisms and Stressors*) and 9.3 (*Risk of Take Associated with Common Impact Mechanisms*).

### 11.3.1 Construction and Maintenance

#### *11.3.1.1 Dewatering and Fish Handling*

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 channel 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. Following passive fish removal, fish removal by seining is recommended before resorting to electrofishing, which carries a greater risk of mortality (NMFS 2006).

In summary, minimize channel dewatering impacts on HCP species 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.
- Following passive fish removal, fish removal by seining is recommended before resorting to electrofishing, which carries a greater risk of mortality.

Where electrofishing is required, employ the following BMPs:

- 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.

#### *11.3.1.2 Dredging and Fill*

Dredging and fill will be a necessary component of project construction and maintenance for many fish screens. The permitted in-water work window for these structures should consider the full range of HCP species likely to occur in the vicinity and should be timed to avoid the presence of sensitive species and/or life-history stages where practicable. In cases where adverse impacts on HCP species cannot be avoided effectively (e.g., a nursery site for buried lamprey amocoetes), alternative designs that avoid dredging and fill impacts should be considered.

Where practicable, dredging and fill activities should be conducted within an exclusion area (dewatered or watered as appropriate) following fish removal. This will help to limit elevated turbidity and sediment impacts. Creation of exclusion areas and fish removal and relocation should be conducted using standardized protocols for these procedures.

A number of techniques have been developed that may be used to avoid or mitigate the effects of dredging (Smits 1998) and placement of fill materials on sensitive ecosystems such as wetlands (Sheldon et al. 2005). Dredging associated with fish screens is typically coupled with the installation and/or maintenance of a water diversion system. Placement of fill material is typically associated with the installation of water diversion system or may be incidental during construction. Habitat protection, conservation, mitigation, and management strategies applicable to dredging activities as well as placement of fill are discussed in the Channel Modifications white paper (Herrera 2007b).

## 11.3.2 Operations

### *11.3.2.1 Noise, Visual, and Physical Disturbance*

Underwater noise, visual, and physical disturbance are, to a certain extent, unavoidable with screen systems that employ mechanical debris-clearing systems. Mechanical systems should be sound insulated and located above water to the extent practicable to limit continuous underwater noise that could contribute to auditory masking effects or avoidance behavior (except in circumstances where noise is being used as a behavioral deterrent). Air jet or hydraulic debris-clearing systems for in-channel screens should be calibrated to limit impulsive sound below established disturbance thresholds where practicable (e.g., 150 $dB_{RMS}$ for salmonids). Proper siting of in-channel screens should limit behavioral avoidance of suitable habitats or other undesirable effects.

### *11.3.2.2 Entrainment and Impingement*

Screen mesh size, mesh material, and approach velocity are critical factors in determining entrainment and impingement risk. Screen design parameters are typically selected based on the smallest and weakest swimming species or life-history stage expected to occur, and the most extreme temperature and flow conditions. This combination of factors produces conservative design criteria that provide broad protection for the full range of species and life-history stages likely to be exposed to the screen. This is a useful uniform recommendation that should be employed in all screen designs. However, this may impose engineering demands that are infeasible in certain cases. To provide additional protection where performance limitations cannot be overcome through design, it may be useful for WDFW to seek the authority to impose operational limits on water withdrawals.

As noted in Section 10.3.2.2 (*Entrainment and Impingement*), current scientific understanding of the swimming performance and risk of entrainment or impingement-related effects is less than uniform across the range of HCP species. However, these design criteria may not necessarily consider the full range of HCP species likely to occur and therefore may not be as protective as possible.

## 11.3.3 Water Quality Modifications

### *11.3.3.1 Elevated Suspended Sediments*

Based on the findings of Bash et al. (2001) on turbidity effects on salmonids, the following measures are recommended to avoid direct and indirect effects on HCP species:

- Determine background suspended sediment concentrations, including particle size and shape, to understand the ambient turbidity to which animals have adapted.

- Review existing watershed assessments to consider pollution loads that may be from sources outside the project to evaluate the project's cumulative effects on turbidity levels.

- Upon determination of existing turbidity and sources, establish acceptable project increases to background turbidity that are similar to those set in the Implementing Agreement between WSDOT and Ecology (WSDOT and Ecology 1998). These standards allow a mixing zone for turbidity generated by in-water construction, as allowed by WAC 173-201A-1090(4) and (6) if the use of this mixing zone does not result in habitat loss, damage to the ecosystem, or adversely affect public health. A mixing zone not meeting turbidity standards is only authorized following the issuance of all other local and state permits and approvals, as well as the implementation of BMPs to avoid or minimize exceedance of turbidity criteria.

- Require that stormwater runoff be 100 percent contained. Route stormwater from the structure and adjacent impervious surfaces to a treatment system.

- If possible, determine a spatial limit, beyond which no water quality effects will extend. Within this limit, monitoring will be required to ensure that established water quality standards are met. If at any point during construction/dredging/demolition these standards are exceeded, construction/dredging/demolition activities will cease until water quality standards are met.



#### *11.3.3.2 Altered pH*

Existing Washington State Department of Ecology regulatory requirements for Clean Water Act Section 401 certification and the Hydraulic Code limit the in-water curing of concrete as necessary to avoid pH effects and the use of appropriate BMPs to avoid leakage of concrete leachate to surface waters. Proper enforcement of these requirements should be sufficient to avoid pH-related impacts from fish screen installation.

#### *11.3.3.3 Introduction of Toxic Substances*

Fish screens have the potential to introduce toxic substances to the aquatic environment through two primary pathways: (1) accidental spills of fuel, oil, lubricants, or other pollutants during construction and maintenance; and (2) screen equipment failure resulting in the release of toxic lubricants.

Construction and maintenance-related impacts can be avoided by requiring the project proponent or contractor to have an established spill prevention and spill containment plan in place and to use nontoxic, food grade hydraulic fluids and lubricants. These are standard BMPs employed in all construction projects. Operational releases of toxic substances can be minimized through proper equipment maintenance, and the use of non-toxic lubricants. The compliance inspection and monitoring program described in Section 11.1 (*Management Strategies*) will provide an effective means for limiting this type of pollution event.

### 11.3.4 Hydraulic and Geomorphic Modifications

For the purpose of this white paper, hydraulic and geomorphic modifications resulting from fish screens are limited to the effects of the screen itself and not the accompanying water withdrawal, or the accompanying flow control structure or channel modification used to withdraw the water from the system. On this basis, the extent of hydraulic and geomorphic modification is expected to be quite limited, and there are no specific recommendations for fish screens.

The above finding is applicable for both lacustrine and marine habitat types. However, it is necessary to acknowledge that the design limits of fish screens and the requirements of sensitive HCP species may lead to changes in the siting of flow control structures. These changes may in turn alter the nature and extent of hydraulic and geomorphic modifications that a flow structure imposes. For example, a water intake system may have to be extended into deeper water to avoid entrainment of planktonic fish larvae that cannot effectively be screened, leading to more extensive effects. Methods for minimizing and mitigating the hydraulic and geomorphic effects of flow control structures are discussed in the Flow Control Structures white paper (Herrera 2007a).

Certain types of fish screens do impose distinct hydraulic and geomorphic effects that are appropriate for discussion here (specifically, off-channel screen designs that incorporate bypass channels). Because the bypass channel flow is removed from the main channel and is unavailable until it is discharged at some point downstream, the stream segment between the intake and the discharge point is vulnerable to hydraulic and geomorphic effects. These effects can be minimized by limiting the length of the bypass, discharging the return flow as short a distance downstream as practicable. This design criterion must be balanced against the need to provide sufficient head loss to maintain bypass flow velocities necessary to clear debris, and to discharge entrained fish at a safe location (e.g., areas unsuitable for loitering by lie-in-wait predators).

These competing design requirements may lead to relatively long bypass channels. If the length of the affected reach is significant (e.g., greater than five times the average reach width) and the flow required to operate the bypass channel is a significant portion of the streamflow in the channel downstream of the diversion, then undesirable changes in channel morphology may occur due to factors such as vegetation encroachment.

Any adverse effects on habitat conditions should be addressed with appropriate mitigation. Mitigation parameters should be based on the requirements of HCP species that use the affected habitat, as well as on addressing specific habitat-limiting factors imposed by the activity or that are more generally prevalent in the affected system. The species occurrence in potentially affected areas can be determined via surveys, an inventory database, WDNR Aquatic Lands HCP, Forest Practices HCP (WDNR 2005), Streamnet database, and/or the Priority Habitats and Species database. Estimating adverse effects of a proposed project should be guided using a limiting factors analysis. For example, the primary limiting factor (such as loss of spawning habitat) should be included in the determination of adverse effects. Baseline data for limiting factors are available for most Water Resource Inventory Areas (WRIAs) from the Washington State Conservation Commission at http://salmon.scc.wa.gov. The limiting factors quantitative analysis for salmonids is available for most streams using the Ecosystem Diagnosis and Treatment model (see http://www/mobrand.com/edt/).

# 12.0 References


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.

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.

Airoldi, L., and S.J. Hawkins. 2007. Negative Effects of Sediment Deposition on Grazing Activity and Survival of the Limpet *Patella Vulgata*. *Marine Ecology-Progress Series* 332: 235-240.

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.

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.

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).

Babbitt, K.J., and G.W. Tanner. 1998. Effects of Cover and Predator Size on Survival and Development of *Rana Utricularia* Tadpoles. *Oecologia* 114(2): 258-262.

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.

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.

Bash, J., C.H. Berman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. WA-RD 526.1. Olympia, Washington: Washington State Department of Transportation.

Bates, K. 2008. Personal communication with E. Doyle, Herrera Environmental Consultants, Seattle, Washington. January 8, 2008.

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.

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.

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.

Bestgen, K.R., J.M. Bundy, K.A. Zelasko, and T.L. Wahl. 2004. Effectiveness of High-Velocity Inclined Profile-Bar Fish Screens Measured by Exclusion and Survival of Early Life Stages of Fathead Minnow. *North American Journal of Fisheries Management* 24(4): 1228-1239.

Bilby, R.E., B.R. Fransen, P.A. Bisson, and J.K. Walter. 1998. Response of Juvenile Coho Salmon (*Oncorhynchus Kisutch*) and Steelhead (*Oncorhynchus Mykiss*) to the Addition of Salmon Carcasses to Two Streams in Southwestern Washington, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 55(8): 1909-1918.

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.

Blackley, T. 2004. Screening Irrigation Offtakes in the Murray-Darling Basin to Reduce Loss of Native Fish. *Proceedings of the Workshop on Downstream Fish Movements. Murray-Darling Basin Commission, Canberra*.

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.

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.

Bohn, C.C., and J.G. King. 2000. Stream Channel Responses to Streamflow Diversion on Small Streams of the Snake River Drainage, Idaho. Research Paper RMRS-RP-20. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, Colorado.

Bolton, S.M., J. Moss, J. Southard, G. Williams, C. Deblois, and N. Evans. 2002. Juvenile Coho Movement Study. Washington State Transportation Center (TRAC).

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.

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.

Bryant, M.D., T. Gomi, and J.J. Piccolo. 2007. Structures Linking Physical and Biological Processes in Headwater Streams of the Maybeso Watershed, Southeast Alaska. *Forest Science* 53(2): 371-383.

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.

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.

Cada, G., T.J. Carlson, J. Ferguson, M. Richmond, and M. Sale. 1999. Exploring the Role of Shear Stress and Severe Turbulence in Downstream Fish Passage. U.S. Department of Energy, Oak Ridge National Laboratory.

Cada, G.F., M.G. Ryon, D.A. Wolf, and B.T. Smith. 2003. Development of a New Technique to Assess Susceptibility to Predation Resulting from Sublethal Stresses (Indirect Mortality). U.S. Department of Energy. ORNL/TM-2003/195. August 2003.

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.

Camargo, J.A., and N.J. Voelz. 1998. Biotic and Abiotic Changes Along the Recovery Gradient of Two Impounded Rivers with Different Impoundment Use. *Environmental Monitoring and Assessment* 50: 143-158.

Cameron, W.A., S.M. Knapp, and R.W. Carmichael. 1997. Evaluation of Juvenile Salmonid Bypass Facilities and Passage at Water Diversions on the Lower Umatilla River; 1991-1995 Final Report. Prepared for U.S. Department of Energy, Bonneville Power Administration by Oregon Department of Fish and Wildlife, Portland, Oregon.

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. Prepared for Washington State Department of Transportation by Battelle Marine Sciences Laboratory, Sequim, Washington.

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, Inc., Seattle, Washington. April 2001.

Carter, J.A., G.A. McMichael, and M.A. Chamness. 2003. Yakima River Basin Phase II Fish Screen Evaluations, 2002. PNNL-14205. United States.

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.

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, L.Y., A.G. Heath, and R.J. Neves. 2001. Comparison of Oxygen Consumption in Freshwater Mussels (Unionidae) from Different Habitats During Declining Dissolved Oxygen Concentration. *Hydrobiologia* 450(1): 209-214.

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.

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.

Clary, W.P., C.I. Thornton, and S.R. Abt. 1996. Riparian Stubble Height and Recovery of Degraded Streambanks. *Rangelands* 18(4): 137-140.

Close, D.A. 2000. Pacific Lamprey Research and Restoration Project: Annual Report 1998. Prepared for U.S. Department of Energy by Tribal Fisheries Program, Department of Natural Resources, Portland, Oregon. May 2000.

Close, D.A., M.S. Fitzpatrick, D.R. Hatch, A.D. Jackson, H. Li, and B.L. Parker. 1998. Pacific Lamprey Research and Restoration: Annual Report 1997. DOE/BP-39067-3. Portland, Oregon: U.S. Department of Energy, Bonneville Power Administration.

Cluett, L. 2005. The Role of Flooding in Morphological Changes in the Regulated Lower Ord River in Tropical Northwestern Australia. *River Research and Applications* 21(2-3): 215-227.

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.

COSEWIC. 2003. Rocky Mountain Ridge Mussel, *Gonidea Angulata*. Status Report. 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.

Coutant, C.C. 2004. A Riparian Habitat Hypothesis for Successful Reproduction of White Sturgeon. *Reviews in Fisheries Science* 12(1): 23-73.

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 the Acoustical Society of America.*

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.

Danley, M.L., S.D. Mayr, P.S. Young, and J.J. Cech Jr. 2002. Swimming Performance and Physiological Stress Responses of Splittail Exposed to a Fish Screen. *North American Journal of Fisheries Management* 22(4): 1241-1249.

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.

Desjardin. 2003. Personal Communication as cited in WSDOT (2006), Biological Assessment Preparation for Transportation Projects, Advanced Training Manual. Version 6. Olympia, Washington: Washington State Department of Transportation.

DFO. 2007. Concrete Wash Water: Characteristics. Canadian Department of Fisheries and Oceans. Available at: http://www-heb.pac.dfo-mpo.gc.ca/water_quality/fish_and_pollution/conc_char_e.htm (accessed June 3, 2007).

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.

Dorn, P., L. Johnson, and C. Darby. 1979. The Swimming Performance of Nine Species of Common California Inshore Fishes. *Transactions of the American Fisheries Society* 108(4): 366-372.

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.

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. Pub. #99-06. Olympia, Washington: 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. Washington State Department of Ecology. Olympia, Washington.

Edinger, J.E., and V.S. Kolluru. 2000. Power Plant Intake Entrainment Analysis. *Journal of Energy Engineering* 126(1): 1-14.

Ellis, M.M. 1942. Fresh-Water Impoundments. *Transactions of the American Fisheries Society, 71st Annual Meeting*: 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.

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.

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 (*Onchorhynchus Gorbuscha*) and Chum (*O. Keta*) Salmon Behavior and Distribution. Seattle, Washington: University of Washington.

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. Ph.D. Thesis, University of Washington, Seattle, 216 pp.

Finstad, A.G., S. Einum, T. Forseth, and O. Ugedal. 2007. Shelter Availability Affects Behaviour, Size-Dependent and Mean Growth of Juvenile Atlantic Salmon. *Freshwater Biology* 52: 1710-1718.

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.

Forrester, G.E., and M.A. Steele. 2004. Predators, Prey Refuges, and the Spatial Scaling of Density-Dependent Prey Mortality. *Ecology* 85(5): 1332-1342.

Francis, R.A. 2006. Allogenic and Autogenic Influences Upon Riparian Vegetation Dynamics. Area 38(4): 453-464.

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. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

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.

Gadomski, D.M., and M.J. Parsley. 2005. Effects of Turbidity, Light Level, and Cover on Predation of White Sturgeon Larvae by Prickly Sculpins. *Transactions of the American Fisheries Society* 134(2): 369-374.

Gale, S.B., and A.V. Zale. 2005. Evaluation of Entrainment Losses of Westslope Cutthroat Trout at Private Irrigation Diversions and the Efficiency of Fish Screens on Skalkaho Creek, Montana. Prepared for Wild Fish Habitat Initiative by Montana Cooperative Fishery Research Unit, U.S. Geological Survey, Department of Ecology, Montana State University, Bozeman, Montana.

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.

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.

Geist, D.R. 2000. Hyporheic Discharge of River Water into Fall Chinook Salmon (*Oncorhynchus Tshawytscha*) Spawning Areas in the Hanford Reach, Columbia River. *Canadian Journal of Fisheries and Aquatic Sciences* 57(8): 1647-1656.

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.

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).

Goodyear, C.P. 1977. Assessing the Impact of Power Plant Mortality on the Compensatory Reserve of Fish Populations with Demersal Eggs. In *Assessing the Impact of Power Plant Induced Mortality on Fish Populations*, edited by J.H.S. Blaxter. New York: Pergammon Press.

Gotthardt, T. 2006. Longfin Smelt. Anchorage, Alaska: Alaska Natural Heritage Program.

Gran, K., and C. Paola. 2001. Riparian Vegetation Controls on Braided Stream Dynamics. Water Resources Research 37(12): 3275-3283.

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.

Gregory, R.S., and T.G. Northcote. 1993. Surface, Planktonic, and Benthic Foraging by Juvenile Chinook Salmon (*Oncorhynchus-Tshawytscha*) in Turbid Laboratory Conditions. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 233-240.

Greig, S.M., D.A. Sear, and P.A. Carling. 2007. A Review of Factors Influencing the Availability of Dissolved Oxygen to Incubating Salmonid Embryos. *Hydrological Processes* 21(3): 323-334.

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.

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.

Hadderingh, R.H. 1979. Fish Intake Mortality at Power Stations the Problem and Its Remedy. *Aquatic Ecology* 13(2): 83-93.

Hadderingh, R.H., and Z. Jager. 2002. Comparison of Fish Impingement by a Thermal Power Station with Fish Populations in the Ems Estuary. *Journal of Fish Biology* 61: 105-124.

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, J.L., and R.C. Wissmar. 2004. Habitat Factors Affecting Sockeye Salmon Redd Site Selection in Off-Channel Ponds of a River Floodplain. *Transactions of the American Fisheries Society* 133(6): 1480-1496.

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.

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*. Academic Press. New York.

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.

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. August 2005.

Hatch, A.C., and G.A. Burton. 1999. Phototoxicity of Fluoranthene to Two Freshwater Crustaceans, *Hyalella Azteca* and *Daphnia Magna*: Measures of Feeding Inhibition as a Toxicological Endpoint. *Hydrobiologia* 400: 243-248.

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.

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 *Proc. I.O.A.* 23: 1-13.

Heintz, R.A., B.D. Nelson, J. Hudson, M. Larsen, L. Holland, and M. Wipfli. 2004. Marine Subsidies in Freshwater: Effects of Salmon Carcasses on Lipid Class and Fatty Acid Composition of Juvenile Coho Salmon. Transactions of the American Fisheries Society 133(3): 559-567.

Helfrich, L.A., C.R. Liston, B. Mefford, and R. Bark. 2001. Survival and Injury of Splittail and Chinook Salmon Passed through a Large Hidrostal Pump. *North American Journal of Fisheries Management* 21(3): 616-623.

Henderson, P.A., and R.M.H. Seaby. 2000. Technical Evaluation of U.S. Environmental Protection Agency Proposed Cooling Water Intake Regulations for New Facilities. Lymington, England: Pices Conservation Ltd.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of Sedimentation and Turbidity on Lotic Food Webs: A Concise Review for Natural Resource Managers. Reviews in Fisheries Science 8(2): 125 - 139.

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.

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. 2007a. Flow Control Structures White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, September 2007.

Herrera. 2007b. Channel Modifications White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, October 2007.

Herrera. 2007c. 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.

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.

Holliman, F.M., R. J.B., and K. T.J. 2003. Electroshock-Induced Injury and Mortality in the Spotfin Chub, a Threatened Minnow. *North American Journal of Fish Management* 23(3): 962-966.

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.

Illingworth and Rodkin Inc. 2001. Noise and Vibration Measurements Associated with the Pile Installation Demonstration Project for the San Francisco-Oakland Bay Bridge East Span, Final Data Report.

Ingermann, R.L., M. Holcomb, M.L. Robinson, and J.G. Cloud. 2002. Carbon Dioxide and pH Affect Sperm Motility of White Sturgeon (*Acipenser Transmontanus*). *Journal of Experimental Biology* 205(18): 2885-2890.

Johnson, G.E., B.D. Ebberts, D.D. Dauble, A.E. Giorgi, P.G. Heisey, R.P. Mueller, and D.A. Neitzel. 2003. Effects of Jet Entry at High-Flow Outfalls on Juvenile Pacific Salmon. *North American Journal of Fisheries Management* 23(2): 441-449.

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, P.N., K. Bouchard, and F.A. Goetz. 2005. Effectiveness of Strobe Lights for Reducing Juvenile Salmonid Entrainment into a Navigation Lock. *North American Journal of Fisheries Management* 25(2): 491-501.

Jones & Stokes. 2006. Overwater Structures and Non Structural Piling (White Paper). Prepared for Washington Department of Fish and Wildlife, Olympia, Washington by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C., and R2 Consultants.

Jones, J.B., S.G. Fisher, and N.B. Grimm. 1995. Nitrification in the Hyporheic Zone of a Desert Stream Ecosystem. *Journal of the North American Benthological Society* 14: 249-258.

Jungwirth, M., O. Moog, and S. Muhar. 1993. Effects of River Bed Restructuring on Fish and Benthos of a 5th-Order Stream, Melk, Austria. *Regulated Rivers-Research & Management* 8(1-2): 195-204.

Kahler, T.H., and T.P. Quinn. 1998. Juvenile and Resident Salmonid Movement and Passage through Culverts. Fisheries Research Institute, School of Fisheries, University of Washington.

Kahler, T.H., P. Roni, and T.P. Quinn. 2001. Summer Movement and Growth of Juvenile Anadromous Salmonids in Small Western Washington Streams. *Canadian Journal of Fisheries & Aquatic Sciences* 58(10): 1947-1956.

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.

Karrow, N.A., H.J. Boermans, D.G. Dixon, A. Hontella, K.R. Solomon, J.J. Whyte, and N.C. Bols. 1999. Characterizing the Immunotoxicity of Creosote to Rainbow Trout (*Oncorhynchus Mykiss*): A Microcosm Study. Aquatic Toxicology 45(4): 223-239.

Kemp, P.S., M.H. Gessel, B.P. Sandford, and J.G. Williams. 2006. The Behavior of Pacific Salmonid Smolts During Passage over Two Experimental Weirs under Light and Dark Conditions. *River Research and Applications* 22(4): 429-440.

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.

Kiefer, R.B., and J.N. Lockhart. 1995. Idaho Department of Fish and Game Intensive Evaluation and Monitoring of Chinook Salmon and Steelhead Production, Crooked River and Upper Salmon River Sites, Annual Progress Report January 1, 1993 - December 31, 1993.

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.

Kitano, S., and K. Shimazaki. 1995. Spawning Habitat and Nest Depth of Female Dolly-Varden *Salvelinus-Malma* of Different Body-Size. *Fisheries Science* 61(5): 776-779.

Knapp, S.M. 1992. Evaluation of Juvenile Fish Bypass and Adult Fish Passage Facilities at Water Diversions in the Umatilla River, Progress Report 1990-1991. DOE/BP-01385-2, Oregon Dept. of Fish and Wildlife, Portland, Oregon.

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.

Kock, T.J., J.L. Congleton, and P.J. Anders. 2006. Effects of Sediment Cover on Survival and Development of White Sturgeon Embryos. *North American Journal of Fisheries Management* 26(1): 134-141.

Kondolf, G.M. 1997. Hungry Water: Effects of Dams and Gravel Mining on River Channels. *Environmental Management* 21(4): 533-551.

Kondolf, G.M., and M.G. Wolman. 1993. The Sizes of Salmonid Spawning Gravels. *Water Resources Research* 29(7): 2275-2285.

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.

Lagardère, J.P. 1982. Effects of Noise on Growth and Reproduction of *Crangon crangon* in Rearing Tanks. *Marine Biology* 71(2): 177-185.

Lagardère, J.P., and M.R. Régnault. 1980. Influence Du Niveau Sonore De Bruit Ambiant Sur Le Métabolisme De Crangon Crangon (Decapoda: Natantia) En Élevage. *Marine Biology* 57(3): 157-164.

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.

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: WSDOT, 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.

Lawler, D.M. 2005. Turbidity and Nephelometry. In *Encyclopedia of Analytical Science*, edited by P.J. Worsfold, A. Townsend, and C.F. Poole. *Elsevier*. pp. 343-351.

Leary, R.F., and F.W. Allendorf. 1997. Genetic Confirmation of Sypatric Bull Trout and Dolly Varden in Western Washington. *Transactions of the American Fisheries Society* 126: 715-20.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. *Fluvial Processes in Geomorphology*. New York: Freeman.

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.

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 Habitats and Population Level Consumption of Groundfish. In *Groundfish Food Habitats 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.

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.

MacAvoy, S.E., S.A. Macko, S.P. McIninch, and G.C. Garman. 2000. Marine Nutrient Contributions to Freshwater Apex Predators. *Oecologia* 122(4): 568-573.

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.

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.

Mayfield, R.B., and J.J. Cech. 2004. Temperature Effects on Green Sturgeon Bioenergetics. *Transactions of the American Fisheries Society* 133(4): 961-970.

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.

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.

McGurk, M.D. 1986. Natural Mortality of Marine Pelagic Fish Eggs and Larvae: Role of Spatial Patchiness. *Marine Ecology Progress Series* 34: 227-242.

McKinnell, S., J.J. Pella, and M.L. Dahlberg. 1997. Populations-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., and M.A. Chamness. 2001. Walla Walla River Basin Fish Screen Evaluations, 2001: Burlingame and Little Walla Walla Sites. DOE/BP-00000652-7. Portland, Oregon: Bonneville Power Administration.

McMichael, G.A., J.A. Vucelick, C.S. Abernethy, and D.A. Neitzel. 2004. Comparing Fish Screen Performance to Physical Design Criteria. *Fisheries* 29(7): 10-16.

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.

McNabb, C.D., C.R. Liston, and S.M. Borthwick. 2003. Passage of Juvenile Chinook Salmon and Other Fish Species through Archimedes Lifts and a Hidrostal Pump at Red Bluff, California. *Transactions of the American Fisheries Society* 132(2): 326-334.

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.

Meeuwig, M.H., J.M. Bayer, and J.G. Seelye. 2005. Effects of Temperature on Survival and Development of Early Life Stage Pacific and Western Brook Lampreys. *Transactions of the American Fisheries Society* 134(1): 19-27.

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*. World Mariculture Society, Eighth Annual Meeting.

Mesa, M.G. 1994. Effects of Multiple Acute Stressors on the Predator Avoidance Ability and Physiology of Juvenile Chinook Salmon. *Transactions of the American Fisheries Society* 123(5): 786-793.

Mesa, M.G., and T.M. Olson. 1993. Prolonged Swimming Performance of Northern Squawfish. *Transactions of the American Fisheries Society* 122(6): 1104-1110.

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.

Misitano, D.A., E. Casillas, and C.R. Haley. 1994. Effects of Contaminated Sediments on Viability, Length, DNA and Protein-Content of Larval Surf Smelt, *Hypomesus-Pretiosus*. *Marine Environmental Research* 37(1): 1-21.

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, J.W., D.E. Schindler, J.L. Carter, J. Fox, J. Griffiths, and G.W. Holtgrieve. 2007. Biotic Control of Stream Fluxes: Spawning Salmon Drive Nutrient and Matter Export. *Ecology* 88(5): 1278-1291.

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 D. White. 2002. Effects of Screening Diversions on Fish Populations in the Central Valley: What Do We Know? University of California, Davis. A report for the Science Board, CALFED Ecosystem Restoration Program. January 2002.

Moyle, P.B., and J.A. Israel. 2005. Untested Assumptions: Effectiveness of Screening Diversions for Conservation of Fish Populations. *Fisheries* 30(5): 20.

Moyle, P.B., and J.J. Cech Jr. 1988. *Fishes: An Introduction to Ichthyology*. Second Edition. New Jersey: Prentice Hall Publishing.

Mulholland, P.J., E.R. Marzolf, J.R. Webster, D.R. Hart, and S.P. Hendricks. 1997. Evidence That Hyporheic Zones Increase Heterotrophic Metabolism and Phosphorus Uptake in Forest Streams. limnology and Oceanography 42: 443-451.

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. *Ecology* 66(3): 1012-1023.

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.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

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.

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.

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.

National Conservation Training Center. 2004. The Analytical Approach to Consultation. Lacey, Washington: Advanced Interagency Consultation - Regional Training curriculum.

Nebeker, A.V. 1972. Effect of Low Oxygen Concentration on Survival and Emergence of Aquatic Insects. *Transactions of the American Fisheries Society* 101(4): 675-679.

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 S. '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. Subacoustics LTD.

Nedwell, J., and B. Edwards. 2002. Measurements of Underwater Noise in the Arun River During Piling at County Wharf, Littlehampton. Subacoustech.

Neitzel, D.A., and T.J. Frest. 1989. Survey of Columbia River Basin Streams for Giant (sic) Columbia River Spire Snail, *Fluminicola Columbiana* and Great (sic) 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.

Neitzel, D.A., C.S. Abernethy, and E.W. Lusty. 1990. A Fisheries Evaluation of the Wapato, Sunnyside, and Toppenish Creek Canal Fish Screening Facilities, Spring 1988. DOE/BP/01830-T1, Pacific Northwest Lab., Richland, Washington.

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.

Newcomb, T.W., and T.A. Flagg. 1983. Some Effects of Mt. 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 Sediments on Aquatic Ecosystems. *North American Journal of Fisheries Management* 11(1): 72-82.

Nightingale, B., and C. Simenstad. 2001. 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.

NMFS. 1990. West Coast of North America Coastal and Ocean Zones Strategic Assessment: Data Atlas. Washington, District of Columbia: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. 2004. Draft - Anadromous Salmonid Passage Facility Guidelines and Criteria. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region, Portland, Oregon.

NMFS. 2005. Endangered Species Act Interagency Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the US Highway 12 Naches River Bank Protection and Habitat Enhancement Project, Yakima County, Washington. NMFS Tracking No: 2004/01714. Seattle, Washington: National Marine Fisheries Service.

NMFS. 2006. Endangered Species Act–Section 7 Formal Consultation Biological and Conference Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. NMFS No. 2005/06396, 2005/07365, and 2005/07366. Seattle, Washington: National Marine Fisheries Service.

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.

Northcote, T.G. 1998. *Migratory Behavior of Fish and Its Significance to Movement through Riverine Fish Passage Facilities.* Fish Migration and Fish Bypasses. Edited by M. Jungwirth, S. Schmutz and S. Weiss. Fishing News Books.

NRC. 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Edited by N.R.C. Committee on Protection and Management of Pacific Northwest Anadromous Salmonids. The National Academies Press.

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.

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.

Pacific Biodiversity Institute. 2007. Homepage. Available at: http://www.pacificbio.org/ (accessed July 2007).

Padma, T.V., R.C. Hale, M.H. Roberts, and R.N. Lipsius. 1999. Toxicity of Creosote Water-Soluble Fractions Generated from Contaminated Sediments to the Bay Mysid. *Ecotoxicology and Environmental Safety* 42: 171-176.

Paola, C., E. Foufoula-Georgiou, W.E. Dietrich, M. Hondzo, D. Mohrig, G. Parker, M.E. Power, I. Rodriguez-Iturbe, V. Voller, and P. Wilcock. 2006. Toward a Unified Science of the Earth's Surface: Opportunities for Synthesis among Hydrology, Geomorphology, Geochemistry, and Ecology. Water Resources Research 42(3).

Paragamian, V.L., G. Kruse, and V. Wakkinen. 2001. Spawning Habitat of Kootenai River White Sturgeon, Post-Libby Dam. *North American Journal of Fisheries Management* 21(1): 22-33.

Patrick, P.H., and R.S. McKinley. 1987. Field Evaluation of a Hidrostal Pump for Live Transfer of American Eels at a Hydroelectric Facility. *North American Journal of Fisheries Management* 7(2): 303-305.

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.

Peake, S. 2004. Effect of Approach Velocity on Impingement of Juvenile Northern Pike at Water Intake Screens. *North American Journal of Fisheries Management* 24(2): 390-396.

Pearson, W.H. 2005. Protocols for Evaluation of Upstream Passage of Juvenile Salmonids in an Experimental Culvert Test Bed. Prepared for Washington State Department of Transportation, Pacific Northwest Division and Planning and Capital Program by Battelle Memorial, Institute. Report No. PNWD-3525, Richland, Washington.

Penttila, D.E. 2000. Forage Fishes of the Puget Sound Region. *NWSC/PSAMP Data Conference,* LaConner, Washington.

Penttila, D. 2001. Effects of Overhanging Shading Vegetation on Egg Survival for Summer-Spawning Surf Smelt on Upper Intertidal Beaches in Northern Puget Sound, Washington. Olympia, Washington: WDFW, Marine Resources Division.

Peterson, H.W.U., and L. Amiotte. 2006. Decline of Skokomish Nation Spot Shrimp Catch in Low Dissolved Oxygen Waters of the 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 Fish 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.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.L. Stromberg. 1997. The Natural Flow Regime. *Bioscience* 47(11): 769.

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., and T.J. Carlson. 1998. Application of Sound and Other Stimuli to Control Fish Behavior. *Transactions of the American Fisheries Society* 127(5): 673-707.

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.

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.

Regnault, M., and J.P. Lagardere. 1983. Effects of Ambient Noise on the Metabolic Level of *Crangon crangon* (Decapoda, Natantia). *Marine Ecology* 11: 71-78.

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.

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.

Richard, J.D. 1968. Fish Attraction with Pulsed Low Frequency Sound. *Journal of Fishery Research Board of Canada* 25(7): 1441-1452.

Richards, K., J. Brasington, and F. Hughes. 2002. Geomorphic Dynamics of Floodplains: Ecological Implications and a Potential Modeling Strategy. *Freshwater Biology* 47(4): 559-579.

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.

Rodgers, D.W., and P.H. Patrick. 1985. Evaluation of a Hidrostal Pump Fish Return System. *North American Journal of Fisheries Management* 5(3a): 393-399.

Rodriguez, M.A. 2002. Restricted Movement in Stream Fish: The Paradigm Is Incomplete, Not Lost. *Ecology* 83(1): 1-13.

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.

Rowntree, K.M., and E.S.J. Dollar. 1999. Vegetation Controls on Channel Stability in the Bell River, Eastern Cape, South Africa. Earth Surface Processes and Landforms 24(2): 127-134.

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.

Salo, E.O., N.J. Bax, T.E. Prinslow, C.J. Whitmus, B.P. 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.

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.

Schille, P.C. 2007. Personal communication (contact with E. Doyle, Herrera Environmental Consultants, Seattle, Washington, regarding the use of pile driving in fish screen installation). WDFW, Yakima Screen Shop, Washington.

Schille, P.C. 2008. Personal communication (contact with E. Doyle, Herrera Environmental Consultants, Seattle, Washington, providing text description of modular screen systems developed by the WDFW Yakima Screen Shop during the 1990s). WDFW, Yakima Screen Shop, Washington.

Schlesinger, W.H. 1997. Biogeochemistry: An Analysis of Global Change. San Diego, Calif.: Academic Press.

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.

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.

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.

Sheibley, R.W., J.H. Duff, A.P. Jackman, and F.J. Triska. 2003. Inorganic Nitrogen Transformations in the Bed of the Shingobee River, Minnesota: Integrating Hydrologic and Biological Processes Using Sediment Perfusion Cores. *Limnology and Oceanography* 48: 1129-1140.

Sheldon, D., T. Hruby, P. Johnson, K. Harper, A. McMillan, T. Granger, S. Stanley, and E. Stockdale. 2005. Wetlands in Washington State - Volume 1: A Synthesis of the Science. Olympia, Washington: Washington State Department of Ecology. Publication #05-06-006. March 2005.

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.

Sibley, P.K., M.L. Harris, K.T. Bestari, T.A. Steele, R.D. Robinson, R.W. Gensemer, K.E. Day, and K.R. Solomon. 2001. Response of Zooplankton and Phytoplankton Communities to Liquid Creosote in Freshwater Microcosms. *Environmental Toxicology and Chemistry* 20(2): 394-405.

Sibley, P.K., M.L. Harris, K.T. Bestari, T.A. Steele, R.D. Robinson, R.W. Gensemer, K.E. Day, and K.R. Solomon. 2004. Response of Zooplankton and Phytoplankton Communities to Creosote-Impregnated Douglas Fir Pilings in Freshwater Microcosms. Archives of *Environment Contamination and Toxicology* 47: 56-66.

Sigler, J.W., T. Bjornn, and E.H. Everest. 1984. Effects of Chronic Turbidity on Density and Growth of Steelheads and Coho Salmon. *Transaction of the American Fisheries Society* 113: 142-150.

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, US Environmental Protection Agency.

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.

Smits, J. 1998. Machines, Methods and Mitigation. Environmental Aspects of Dredging. The Hague, Netherlands: International Association of Dredging Companies (IADC).

Snoeyink, V.L., and D. Jenkins. 1980. *Water Chemistry*. Wiley. New York.

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.

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.

Sommer, T.R., W.C. Harrell, and M.L. Nobriga. 2005. Habitat Use and Stranding Risk of Juvenile Chinook Salmon on a Seasonal Floodplain. *North American Journal of Fisheries Management* 25(4): 1493-1504.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An Ecosystem Approach to Salmonid Conservation. Report No TR-4501-96-6057. ManTech Environmental Research Services Corporation, Corvallis, Oregon.

Stadler, J., NOAA Fisheries, Seattle, Washington. 2007. Email 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.

Stamp, M.L., and J.C. Schmidt. 2006. Predicting Channel Responses to Flow Diversions. Stream Notes: To Aid in Securing Favorable Conditions of Water Flows. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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 Wildlife Service.

Stevens, D.E., D.W. Kohlhorst, L.W. Miller, and D.W. Kelley. 1985. The Decline of Striped Bass in the Sacramento-San Joaquin Estuary, California. *Transactions of the American Fisheries Society* 114(1): 12-30.

Stone, J., and S. Barndt. 2005. Spatial Distribution and Habitat Use of Pacific Lamprey (*Lampetra Tridentata*) Ammocoetes in a Western Washington Stream. *Journal of Freshwater Ecology* 20(1): 171-185.

Stratus. 2005. 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.

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.

Sumida, B.Y., and H.G. Moser. 1984. Food and Feeding of Bocaccio and Comparison with Pacific Hake Larvae in the California Current. Calif. Coop. Oceanic Fish. Invest.

Suttle, K.B., M.E. Power, J.M. Levine, and C. McNeely. 2004. How Fine Sediment in Riverbeds Impairs Growth and Survival of Juvenile Salmonids. *Ecological Applications* 14(4): 969-974.

Swales, S., and C.D. Levings. 1989. Role of Off-Channel Ponds in the Life-Cycle of Coho Salmon (*Oncorhynchus-Kisutch*) and Other Juvenile Salmonids in the Coldwater River, British-Columbia. *Canadian Journal of Fisheries and Aquatic Sciences* 46(2): 232-242.

Swanson, C., P.S. Young, and J.J. Cech. 2005. Close Encounters with a Fish Screen: Integrating Physiological and Behavioral Results to Protect Endangered Species in Exploited Ecosystems. *Transactions of the American Fisheries Society* 134(5): 1111-1123.

Swartz, R.C., P.F. Kemp, D.W. Schults, G.R. Ditsworth, and R.J. Ozretich. 1989. Acute Toxicity of Sediment from Eagle Harbor, Washington, to the Infaunal Amphipod *Rhepoxynius-Abronius*. *Environmental Toxicology and Chemistry* 8(3): 215-222.

Taft, E.P., Ill, and Y.G. Mussalli. 1978. Fish Diversion and Transportation System for Power Plant Application. *Fisheries* 3(3): 2-5.

Takami, T., F. Kitano, and S. Nakano. 1997. High Water Temperature Influences on Foraging Responses and Thermal Deaths of Dolly Varden Salvelinus Malma and White-Spotted Charr S-Leucomaenis in a Laboratory. *Fisheries Science* 63(1): 6-8.

Teachout, E., Fish and Wildlife Biologist with U.S. Fish and Wildlife Service, Lacey, Washington. 2007. Email 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.

Tenera Environmental. 2005. Draft - Carlsbad Desalination Facility Intake Effects Assessment. Prepared for Poseidon Resources Corporation by Tenera Environmental, Lafayette, California.

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.

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.

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.

Travnichek, V.H., A.V. Zale, and W.L. Fisher. 1993. Entrainment of Lchthyoplankton by a Warmwater Hydroelectric Facility. *Transactions of the American Fisheries Society* 122(5): 709-716.

Triska, F.J., V.C. Kennedy, R.J. Avanzino, G.W. Zellweger, and K.E. Bencala. 1989. Retention and Transport of Nutrients in a 3rd-Order Stream in Northwestern California - Hyporheic Processes. *Ecology* 70: 1893-1905.

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.

USFS. 2007. Fish Resources. U.S. Forest Service website. Available at: 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). U.S. Fish and Wildlife Service, Coastal Ecology Group. Lafayette, Louisiana.

Vagle, S. 2003. On the Impact of Underwater Pile Driving Noise on Marine Life. Government Report. Canadian 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. *Ecology* 83(10): 2888-2901.

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.

Van Eenennaam, J.P., J. Linares-Casenave, X. Deng, and S.I. Doroshov. 2005. Effect of Incubation Temperature on Green Sturgeon Embryos, *Acipenser Medirostris*. *Environmental Biology of Fishes* 72(2): 145-154.

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.

Vaughan, D.M. 2002. Potential Impacts of Road-Stream Crossings (Culverts) on the Upstream Passage of Aquatic Macroinvertebrates. Portland, Oregon: The Xerces Society.

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.

Vucelick, J.A., and G.A. McMichael. 2003. Walla Walla River Basin Fish Screen Evaluations, 2003 Nursery Bridge Fishway and Garden City/Lowden Il. Pacific Northwest National Laboratory.

Vucelick, J.A., G.A. McMichael, and M.A. Chamness. 2004. Yakima River Basin Phase II Fish Screen Evaluations, 2003. Prepared for Bonneville Power Administration by Pacific Northwest National Laboratory, PNNL-14627.

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.

Wahl, T.L. 1995. Hydraulic Testing of Static Self-Cleaning Inclined Screens. *The First International Conference on Water Resources Engineering*, San Antonio, Texas, pp. 14-18.

Wahl, T.L. 2003. Design Guidance for Coanda-Effect Screens. *Report. Water Resources Research Laboratory. U. S. Bureau of Reclamation. R-Series*(3): 37.

Wahl, T.L., and R.F. Einhellig. 2000. Laboratory Testing and Numerical Modeling of Coanda-Effect Screens. *Building Partnerships- 2000 Joint Conference on Water Resource Engineering and Water Resources Planning & Management*.

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.

Watson, L.R., A. Milani, and R.P. Hedrick. 1998. Effects of Water Temperature on Experimentally-Induced Infections of Juvenile White Sturgeon (*Acipenser Transmontanus*) with the White Sturgeon Iridovirus (WSIV). *Aquaculture* 166(3-4): 213-228.

Watters, G.T. 1996. Small Dams as Barriers to Freshwater Mussels (*Bivalvia*, *Unionoida*) and Their Hosts. *Biological Conservation* 75(1): 79-85.

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. 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. Available at: http://www.wdfw.wa.gov/fish/forage/smelt.htm (accessed Nov. 3, 2006).

WDFW. 2000. Fishway Guidelines for Washington State. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW. 2001a. Fish Protection Screen Guidelines for Washington State - Draft. Draft Report. Olympia: Washington Department of Fish and Wildlife.

WDFW. 2001b. 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.

WDNR. 2005. Forest Practices – Habitat Conservation Plan. Washington Department of Natural Resources. Olympia, Washington. December 2005.

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.

Weber, E.D., S.M. Borthwick, and L.A. Helfrich. 2002. Plasma Cortisol Stress Response of Juvenile Chinook Salmon to Passage through Archimedes Lifts and a Hidrostal Pump. *North American Journal of Fisheries Management* 22(2): 563-570.

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.

Weisberg, S.B., W.H. Burton, F. Jacobs, and E.A. Ross. 1987. Reductions in Ichthyoplankton Entrainment with Fine-Mesh, Wedge-Wire Screens. *North American Journal of Fisheries Management* 7(3): 386-393.

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. Springer. New York. 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.

Welton, J.S., W.R.C. Beaumont, and R.T. Clarke. 2002. The Efficacy of Air, Sound and Acoustic Bubble Screens in Deflecting Atlantic Salmon, *Salmo Salar L.*, Smolts in the River Frome, UK. *Fisheries Management and Ecology* 9(1): 11-18.

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.

White, D.K., C. Swanson, P.S. Young, J.J. Cech Jr., Z.Q. Chen, and M.L. Kavvas. 2007. Close Encounters with a Fish Screen II: Delta Smelt Behavior before and During Screen Contact. *Transactions of the American Fisheries Society* 136(2): 528-538.

White, D.S. 1990. Biological Relationships to Convective Flow Patterns within Stream Beds *Hydrobiologia* 196: 149-158.

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.

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.

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: Washington Department of Fish and Wildlife.

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 National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

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 and Ecology. 1998. Implementing Agreement Between the Washington State Department of Transportation and the Washington State Department of Ecology Regarding the Compliance with the State of Washington Surface Water Quality Standards. February 13, 1998. Washington State Department of Transportation and Washington State Department of Ecology. Obtained from agency website on April 9, 2007: http://www.wsdot.wa.gov/NR/rdonlyres/7C788425-CD3B-4B21-B4CD-7C7F1211FFB0/0/IA_SurfaceWaterQuality1988.pdf.

WSDOT. 2005. Underwater Sound Levels Associated with the Friday Harbor Ferry Terminal. Agency Report. Washington State Department of Transportation, Office of Air, Noise, and Energy.

WSDOT. 2006. Biological Assessment Preparation for Transportation Projects, Advanced Training Manual. Olympia, Washington: Washington State Department of Transportation.

Wydoski, R.G., and R.S. Wydoski. 2002. Age, Growth, and Reproduction of Mountain Suckers in Lost Creek Reservoir, Utah. *Transactions of the American Fisheries Society* 131(2): 320-328.

Wydoski, R.S., and R.R. Whitney. 2003. *Inland Fishes of Washington, Second Edition*. American Fisheries Society and University of Washington Press.

Young, W.T., and D.L. Scarnecchia. 2005. Habitat Use of Juvenile White Sturgeon in the Kootenai River, Idaho and British Columbia. *Hydrobiologia* 537: 265-271.

Zemlak, R.J., and J.D. McPhail. 2006. The Biology of Pygmy Whitefish, *Prosopium Coulterii*, in a Closed Sub-Boreal Lake: Spatial Distribution and Diel Movements. *Environmental Biology of Fishes* 76(2-4): 317-327.

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.

Zydlewski, G.B., and J.R. Johnson. 2002. Response of Bull Trout Fry to Four Types of Water Diversion Screens. *North American Journal of Fisheries Management* 22(4): 1276-1282.