

7.2 Construction, Maintenance and Operations

7.2.1 Introduction

The construction, maintenance and operation activities mechanism of impact includes several submechanisms of impact, capturing a range of activities that are short-lived but intensive and are required to build facilities as well as to provide or maintain access to these facilities. The five submechanisms that have been identified for analysis in these white paper(s) include:

- (1) Elevated underwater sound:
 - Pile driving sound
 - Non-pile driving sounds
- (2) Channel/work area dewatering
- (3) Navigation/maintenance, filling and dredging
- (4) Grounding, Anchoring and or Prop Wash
- (5) Ambient Light Modifications

These five sub-mechanisms are likely to affect HCP species as they occur within the water. However, activities occurring landward of water bodies can also affect HCP species. These activities include staging and equipment access, including the use of heavy equipment around the wetted perimeter in riparian (marine, lacustrine, and riverine environments) and floodplain areas.

7.2.2 Elevated Underwater Sound

Projects permitted under the WDFW HPA program can produce underwater noise through a variety of mechanisms. These mechanisms include construction-related noise impacts from impulsive sources (i.e., short duration, high intensity noise from sources such as pile driving or materials placement), as well as continuous noise sources (e.g., vessel or equipment operation). 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 (2006a).

7.2.2.1 Measurement of Underwater Sound

Units of measurement: dBpeak and dBrms

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: $1\mu Pa$) (Urick 1983). The dB_{PEAK} measure represents the instantaneous maximum sound pressure observed during each pulse. The RMS level represents the square root of the total sound pressure energy divided by the impulse duration, which provides a measure of the total sound pressure level produced by an impulsive source. The majority of literature uses dB_{PEAK} re: $1\mu Pa$ sound pressures to evaluate potential injury to fish.

Up until recently, the USFWS and NOAA Fisheries have used both dB_{PEAK} (for injury) and dB_{RMS} (for behavioral effects) re: $1\mu Pa$ threshold values to evaluate adverse injury and disturbance effects on fish, marine mammals, and diving birds (Stadler 2007; Teachout 2007). dB_{RMS} values are used to define disturbance thresholds in fish species, meaning the sound pressure level at which fish noticeably alter their behavior in response to the stimulus (e.g., through avoidance or a “startle” response). dB_{PEAK} values are used to define injury thresholds in salmonids, meaning the sound pressure level at which injury from barotraumas may occur (i.e., physical damage to body tissues caused by a sharp pressure gradient between a gas or fluid-filled space inside the body and the surrounding gas or liquid).

Based on a new agreement in Principle for Interim Criteria for Injury to Fish from Pile Driving Memorandum, June 12, 2008, Federal Highways Administration (FHWA), lead by research performed by NOAA and USFWS have set new criteria for effects from pile driving. These new pile driving criteria are shown below in Table X1, ***Interim Hydroacoustic Criteria for the Physical and Behavioral Effect From Pile Driving on Fish***. Based on this new criteria, with emphasis on the accumulated sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as $dBre: 1\mu Pa^2/second$), effects from repeated pile strikes, it is likely the current findings for the effects of pile driving in the OWS/Marinas white papers does not provide accurate measurement of impact to fish life.

As noted in the agreement: “The agreed upon criteria identify sound pressure levels of 206 dB peak and 187 dB accumulated sound exposure level (SEL) for all listed fish except those that are less than 2 grams.” Fish less than 2 grams “the criteria for the accumulated SEL will be 183 dB.”

The criteria used for the onset of physical injury and adverse behavioral effects are in the table below. The onset of physical injury uses dual criteria - peak pressure (dB_{peak}) and sound exposure level (SEL). The onset of physical injury is expected if either of these criteria are exceeded. The criteria for accumulated sound exposure level (SEL) is based upon the mass of the fishes under consideration. If fishes smaller than 2 grams are present, the more conservative SEL (183 dB (re: $1\mu Pa^2*sec$)) criteria may be required.

Table 7-1: Interim Hydroacoustic Criteria for the Physical and Behavioral Effect From Pile Driving on Fish. June 11, 2008.

Effect	Metric	Fish mass (grams)	Theshold
Onset of physical injury	Peak pressure	N/A	206 dB (re: 1 μ Pa)
	Accumulated Sound Exposure Level (SEL)	≥ 2 g	187 dB (re: 1 μ Pa ² •sec)
		< 2 g	183 dB (re: 1 μ Pa ² •sec)
Adverse behavioral effects	Root Mean Square Pressure (RMS)	N/A	150 dB (re: 1 μ Pa)

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

Underwater sound 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 sound behavior, but it does not account for a number of other factors that can significantly affect sound propagation. For example, decreasing temperature with depth can create significant shadow zones where actual sound pressure levels can be as much as 30 dB lower than calculated because sound bends toward the colder, denser water (Urlick 1983). Haloclines, current mixing, water depth, acoustic wavelength, sound flanking (i.e., sound transmission through

bottom sediments), and the reflective properties of the surface and the bottom can all influence sound propagation in ways that are difficult to predict.

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

Though installation of piles “in the dry” might or might not require an HPA, it is worthwhile to note potential impacts of pile installations adjacent to water bodies. It is often assumed that installation of piles “in the dry” will result in minimal, or undetectable, sound production levels (SPL) in the water. Monitoring data from impact installation indicates that SPLs in the adjacent waterbody can be significantly elevated (Battelle Marine Sciences Laboratory, 2004; Reyff, 2006.). Hydroacoustic monitoring during impact installation of 48-inch steel piles that were 5 m from a river in California detected SPLs as high as 201 dB_{peak} and 188 dB_{rms} at 10 meters from the pile (Reyff 2006). As sound pressure travels through the substrate, its waveform might be altered, resulting in longer (and therefore less damaging) rise times, but this has not been adequately investigated. Also, during monitoring of vibratory installation of piles adjacent to a river, Reyff (2006) noted that there was clearly noticeable vibration in the river. Project-Related Sound Sources

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

Although there are many sources of sound in the underwater environment, the following are typical sources of underwater sound associated with in-water construction:

- Ambient sound levels
- Project construction and maintenance: equipment operation and materials placement
- Project operation: vessel operation, equipment operation.

7.2.2.2 Ambient Sound Levels

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

Ambient sound 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 dBpeak (Laughlin 2005).
- Ambient levels at the Mukilteo ferry terminal reached approximately 145 dBpeak in the absence of ferry traffic (WSDOT 2006a).
- Ambient underwater sound levels measured in the vicinity of the Friday Harbor ferry terminal project ranged between 131 and 136 dBpeak (WSDOT 2005).
- Carlson et al. (2005) measured the underwater baseline for the Hood Canal and found it to range from 115 to 135 dBRMS.
- Heathershaw et al. (2001) reported open-ocean ambient sound levels to be between 74 and 100 dBpeak off the coast of central California.

These ambient sound levels are typical conditions, and typical conditions can be punctuated by atypical natural events. For example, lightning strikes can produce underwater sound levels as high as 260 dBpeak in the immediate vicinity (Urick 1983).

Limited data are available on ambient sound levels in freshwater environments, but it is reasonable to conclude that they vary considerably based on available information. High-gradient rivers, fast-flowing rivers, and large rivers and lakes with significant human activity are likely to produce more sound than lakes and slow-flowing rivers in more natural environments.

Burgess and Blackwell (2003) measured ambient sounds in the Duwamish River in Seattle, Washington, (averaged over 20 seconds to 5 minutes) and found the sound to

vary between 110 and 130 dB continuous sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dB re: $1\mu\text{Pa}^2/\text{second}$). Amoser and Ladich (2005) measured ambient sound 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 sound 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 sound 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).

7.2.2.3 Materials Placement (Pile Driving) Sound Levels

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

Two major types of pile driving hammers are in common use, impact hammers and vibratory hammers.

- **Impact Hammer:** There are four kinds of impact hammers: diesel, air or steam driven, hydraulic, and drop hammer (typically used for smaller timber piles). Impact hammers produce sharp sound pressure waves with rapid rise times, the equivalent of a punch versus a push in comparison to vibratory hammers. The sharp sound pressure waves associated with impact hammers represent a rapid change in water pressure level, with greater potential to cause injury or mortality in fish and invertebrates.
- **Vibratory Hammer.** Vibratory hammers produce a more rounded sound pressure wave with a slower rise time. 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 sound produced during pile driving. Driven piles are typically composed of one of three basic material types: timber, concrete, or steel (although other special materials such as plastic may be used). Steel piles are often used as casings for pouring concrete piles. Sound levels associated with each of these types of piles are summarized in Table 7-1.

Reference sound levels are denoted in both dB_{PEAK} and dB_{RMS} values, at the specified measurement reference distance.

Table 7-2. Reference sound levels by structure type.

Material Type and Size	Impact Hammer Type	Reference Sound 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 (2006a)], (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 2006a)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(Hastings and Popper 2005)
30-inch steel piles	Diesel	208 @ 10 m	192 @ 10 m	Marine	(Hastings and Popper 2005)
66-inch steel piles	Hydraulic	210 @ 10 m	195 @ 10 m	Marine	(Reyff et al. 2003)
96-inch steel piles	Hydraulic	220 @ 10 m	205 @ 10 m	Marine	(Reyff et al. 2003)
126-inch steel piles	Hydraulic	191 @ 11 m	180-206 @ 11 m	Marine	(Reyff et al. 2003)
150-inch steel piles	Hydraulic	200 @ 100 m	185 @ 100 m	Marine	(Reyff et al. 2003)

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

7.2.2.4 Pile Driving Noise Impacts to Fish

Hastings and Popper (2005) recently performed a comprehensive literature review to evaluate the current best available science regarding noise thresholds at which fish would be injured by the percussive sound generated by pile driving. Much of the information presented below has been extracted from that review.

Most fish sense sounds, vibrations, and other displacements of water in their environment through their inner ear and with the lateral line running the length of each side of the fish and on the head. The lateral line is a mechano-sensory system that plays an indirect role in hearing through its sensitivity to pressure changes at close range. The hearing organs and lateral line system are collectively referred to as the acoustico-lateralis system. The hearing thresholds of different fish species vary depending on the structure and sensitivity of this system.

Anatomical variations of the inner ear, swim bladder, esophagus, lateral line, and other structures determine how fish hear and feel sound pressure (Hastings and Popper 2005). All fish fall into two hearing categories: “hearing generalists” such as salmon and trout, and “hearing specialists” such as herring and eulachon (Hastings and Popper 2005).

Hearing specialists have particular adaptations that enhance their hearing bandwidth and sensitivity (Hastings and Popper 2005). Hearing specialists found on the Pacific coast include the sardine and related Clupeiforms such as herring, shad, menhaden, and anchovy (Hastings and Popper 2005).

The majority of fish on the Pacific coast are hearing generalists and do not have specialized hearing capabilities apart from their swim bladder, inner ear, and lateral line (Hastings and Popper 2005). Hearing generalists sense sound directly through the inner ear, and some use the inner ear coupled with the swim bladder to sense additional energy (Hastings and Popper 2005).

Both hearing generalists and hearing specialists are found in many taxonomic groups (Hastings and Popper 2005). Ideally, fish should be compared based on biomechanical properties of their swim bladder and any other internal gas-filled chamber, hearing capabilities, and aspects of their behavior (Hastings and Popper 2005). However, when such data are not available, it is probably more appropriate to extrapolate between species that have somewhat similar auditory structures or pressure-detecting mechanisms (most notably the swim bladder) and species of similar size, mass, and anatomical variety (Hastings and Popper 2005). This would enable at least a first-order approximation of extrapolation to fishes such as salmonids and other teleost fishes that presumably do not have hearing specialization (e.g., rockfish). The results are less easily extrapolated to teleosts without a swim bladder, such as sand lance and lingcod, and to fish with very different ear structures, such as lamprey and sturgeon (Hastings and Popper 2005).

Table 7-3 outlines the known and presumed hearing categories of potentially covered fish species.

Table 7-3: Hearing Categories for Potentially Covered Fish Species

Common Name (Scientific Name)	Hearing Category	Notes and/or References
Trout and salmon (Salvelinus, Onchorynchus spp.)	Generalist	Popper and Carlson 1998
Sturgeon (Acipenser spp.)	Undetermined	Popper (2005) states that sturgeon can detect an extremely wide range of sounds, and several studies have found that some sturgeon produce sounds that may be used to facilitate breeding. However, further studies are necessary to determine how sturgeon vocalize, what levels of sound are produced in the natural environment, and how their vocalizations are used in their behavior.
Eulachon (Thaleichthys pacificus)	Specialist	Blaxter et al. 1981, in Scholik and Yan 2001a
Rockfish (Sebastes spp.)	Generalist	Hastings and Popper 2005
Lake chub (Couesius plumbeus)	Specialist	Hastings and Popper 2005; Popper et al. 2005
Dace (Rhynchichthys spp.)	Unknown/ Presumed	Not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)

Common Name (Scientific Name)	Hearing Category	Notes and/or References
	Generalist	
Lingcod (<i>Ophiodon elongates</i>)	Generalist	Does not have a swim bladder, which is generally an indication of poor hearing (Moyle and Cech 2004; Kapoor and Khanna 2004)
Surf smelt (<i>Hypomesus pretiosus</i>)	Generalist	Included in the taxonomic order Salmoniformes – hearing generalists (Hastings and Popper 2005)
Lamprey (<i>Lampetra</i> spp.)	Generalist	Popper 2005
Margined sculpin (<i>Cottus marginatus</i>)	Generalist	Closely related to the bullhead (<i>Cottus scorpius</i>), which is identified as a generalist (Fay and Popper 1999); also not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)
Mountain sucker (<i>Catostomus platyrhynchus</i>)	Unknown/ Presumed Specialist	<i>Catostomus</i> spp. are known to have weberian ossicles to assist with hearing (Krumholz 1943)
Olympic mudminnow (<i>Novumbra hubbsi</i>)	Unknown/ Presumed Specialist	May have weberian ossicles to assist with hearing (Moyle and Cech 2004). Many closely related fish (minnows, pikeminnow cyprinids) are specialists (Scholik and Yan 2001b; Popper 2005).
Pacific cod (<i>Gadus macrocephalus</i>)	Generalist	<i>Gadus</i> sp. more sensitive than most generalists (Astrup and Mohl 1998, in Scholik and Yan 2002; Hastings and Popper 2005)
Pacific hake (<i>Merluccius productus</i>)	Unknown/ Presumed Generalist	Not a member of a family or grouping identified as hearing specialists (Fay and Popper 1999)
Pacific herring (<i>Clupea harengus pallasii</i>)	Specialist	Hastings and Popper 2005
Pacific sand lance (<i>Ammodytes hexapterus</i>)	Generalist	Does not have a swim bladder, which is generally an indication of poor hearing (Moyle and Cech 2004; Kapoor and Khanna 2004)
Pygmy whitefish (<i>Prosopium coulteri</i>)	Generalist	Of the order Salmoniformes – hearing generalists (Hastings and Popper 2005)
Walleye pollock (<i>Theragra chalcogramma</i>)	Unknown/ Presumed Generalist	Not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)

Physical impacts to fish from intense noises may include temporary hearing loss (referred to as temporary threshold shift), permanent hearing loss (referred to as permanent threshold shift), damage or rupture to gas organs such as the swim bladder and the surrounding tissues, rupture of capillaries in the skin, neurotrauma, and eye hemorrhage (Popper and Fay 1973, 1993, Hastings and Popper 2005). The more serious of these impacts could cause instantaneous death or later death from injuries (e.g., breakdown of tissues in some organs) (NMFS 2003a).

7.2.2.4.1 Lethal Physiological Effects to Fish

- In general, injury and mortality effects from underwater sound 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.

- A study by Abbot (Abbott, R.R., E. Bing-Sawyer, and R. Blizard, 2002) on caged fish demonstrated that energy accumulates over multiple pile driving strikes. This is demonstrated by the fact that fish that received exposure to multiple strikes had extreme internal injuries (in some cases their internal organs were homogenized). Abbott's work also demonstrated that fish with serious internal injuries might not appear harmed to observers (Abbott et al. 2002).
- Impacts on Eggs and Larvae. Although it is possible that some (but not all) fish species would swim away from a sound source, thereby decreasing exposure to sound, larvae and eggs are often at the mercy of currents, move slowly, or are sedentary (Hastings and Popper 2005). Data on the effects of sound on developing eggs and larvae are limited, although in a study by Banner and Hyatt (1973), increased mortality was found in eggs and embryos of sheepshead minnow (*Cyprinodon variegatus*) exposed to broadband noise (100 to 1,000 hertz) that was about 15 dB above the ambient sound level. Hatched fry of sheepshead minnow and fry of longnose killifish (*Fundulus similis*) were not affected in this study. Jensen (2003) noted possible effects of sound impacts on the development of salmonid eggs.
- 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.

Although hearing loss is not a lethal effect of pile driving, the subsequent long-term effects, such as predator/prey detection may result in mortality.

7.2.2.4.2 Sub-Lethal Physiological Effects to Fish

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

that of the lake chub. The hearing sensitivities of lake chub and northern pike returned to their respective normal thresholds after 18 to 24 hours.

- 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 oscar 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.

7.2.2.4.3 Behavioral Effects to Fish

Behavioral responses to sound stimuli are well established in the literature for many fish species. As noted under the *Lethal Physiological Effects* and *Sub-Lethal Effects* section, effected hearing ability of fish may result in long-term mortality. These types of effects on behavior are summarized below:

- 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.
- Moore and Newman (1956) reported that the classic fright response of salmonids to instantaneous sound stimuli was the "startle" or "start" behavior, where a fish rapidly darts away from the noise source.
- Knudsen et al. (1992) found that in response to low-frequency (10 Hz range) sound, salmonids 1.6–2.4 in (40–60 mm) in length exhibited an initial startle response followed by habituation, while higher frequency sound caused no response even at high intensity.
- In a study of the effects of observed pile driving activities on the behavior and distribution of juvenile pink and chum salmon, Feist et al. (1992) found that pile-driving operations were associated with changes in the distribution and behavior of fish schools in the vicinity. Fish schools were two-fold more abundant during normal construction days in comparison to periods when pile driving took place.

- Blaxter et al. (1981) found Atlantic herring to exhibit an avoidance response to both continuous pulsed sound stimuli with habituation to more continuous stimuli occurring over time, and Schwarz and Greer (1984) found similar responses on the part of Pacific herring.
- Sound has also been shown to affect growth rates, fat stores, and reproduction (Banner and Hyatt 1973; Meier and Horseman 1977).
- Prolonged underwater noise can also reduce the sensitivity of fish to underwater noise stimuli, with potentially important effects on survival, growth, and fitness. The fish auditory system is likely one of the most important mechanisms fish use to detect and respond to prey, predators, and social interaction (Amoser and Ladich 2005; Fay 1988; Hawkins 1986; Kalmijn 1988; Myrberg 1972; Myrberg and Riggio 1985; Nelson 1965; Nelson et al. 1969; Richard 1968; Scholik and Yan 2001; Scholik and Yan 2002; Wisby et al. 1964). Scholik and Yan (2001) studied the auditory responses of the cyprinid fathead minnow to underwater noise levels typical of human-related activities (e.g., a 50 horsepower outboard motor). They found that prolonged exposure decreased noise sensitivity, increasing the threshold level required to elicit a disturbance response for as long as 14 days after the exposure. Amoser and Ladich (2005) reported similar findings in common carp in the Danube River, noting that auditory ability in this hearing specialist species was measurably masked in environments with higher background noise. They reported similar but far less pronounced responses in hearing generalist species such as perch. These data suggest that elevated ambient noise levels have the potential to impair hearing ability in a variety of fish species, which may in turn adversely affect the ability to detect prey and avoid predators, but that this effect is variable depending on the specific sensitivity of the species in question.
- Feist et al. (1992) similarly theorized that it was possible that auditory masking and habituation to loud continuous noise from machinery may decrease the ability of salmonids to detect approaching predators.

7.2.2.5 Pile Driving Noise Impacts to Invertebrates

Although studies of noise impacts on invertebrates have consistently shown that very high sound pressure levels (in excess of 217 dB) can cause serious injury, the information is sparse, is poorly reported, and was obtained without due experimental rigor (Turnpenny et al. 1994). The studies reported in Turnpenny et al. (1994) exposed mussels, periwinkles, amphipods, squid, scallops, and sea urchins to high airgun and slow-rise-time sounds at between 217 dB and 260 dB. Mussels, periwinkles, and amphipods showed no detectable effect at 229 dB (Kosheleva 1992, in Turnpenny et al. 1994), although one Iceland scallop suffered a split shell after being exposed to 217 dB from a single airgun strike (Matishov 1992, in Turnpenny et al. 1994), suggesting the potential for serious injury when percussive underwater noise exceeds these levels.

7.2.2.6 Vessel/Equipment Operation and Materials Placement (Non-Pile Driving) Sound Levels

In comparison to pile driving, data on sound levels produced by placement of other construction-related materials are limited. For example, measured sound levels associated with work on the Friday Harbor ferry terminal ranged between 133 dB_{peak} and 140 dB_{peak}, excluding pile driving. These sound 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 sound associated with in-water tool use, placement of large rock and similar material, vessel operation, and in-water operation of heavy machinery, generally produce substantially lower sound levels than those associated with pile driving. However, other construction-related noises may generate continuous noise for longer periods, with the effect of elevating ambient noise levels or masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators.

This effect may be of particular concern for projects that result in changes in vessel operation or equipment use that change ambient noise levels for longer periods (e.g., days to years). For example, vessel operation can significantly influence ambient noise levels. Large vessel engines can produce underwater sound up to 198 dB, and depth sounders can produce noise in excess of 180 dB (Buck 1995; Heathershaw et al. 2001).

Hazelwood and Connelly (2005) monitored fishing vessel noise over a broad octave range from 10 Hz–40 kHz and documented noise levels ranging from 140–185 dB_{peak}, with the loudest noise occurring at the lower end of the octave range.

Commercial sonar devices operating in a frequency range of 15–200 kHz can produce underwater noise ranging from 150–215 dB at maximum levels (Stocker 2002).

Equipment and vessels necessary to dig trenches, place riprap, support equipment over water, and perform other activities associated with the construction of overwater structures also produce underwater noise. Construction equipment tends to produce the same type of slow-rise-time noise as do motor boats and ship engines.

- Jones and Stokes (2006) estimated that noise produced by a rather large ocean-cable-installation vessel is about 154 dB_{RMS}.
- JASCO (2005) estimated that noise produced by a rock-dumping vessel is approximately 177 dB (neither peak nor RMS identified) at 3.28 feet (1 m)
- Richardson et al. (1995, in Jones and Stokes 2006) estimated that an equipment support vessel produces noise levels of 152 dB_{peak} at 3.28 feet (1 m).

Sounds of this amplitude may affect the behavior or physiology of fishes, depending on their hearing sensitivity and proximity to the sound.

7.2.2.6.1 Effects on Fish

Operational activities are expected to produce intermittent, continuous noise from facilities and vessels for the life of the facility. In general, noise levels produced by small to moderate sized vessel operations are relatively low in comparison to those levels shown to cause injury in construction projects. Responses to these effects may range from minor changes in behavior, to increased predation risk or lowered foraging efficiency, to potential injury.

7.2.2.6.2 Effects on Invertebrates

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

7.2.3 Channel/Work Area Dewatering

7.2.3.1 Impacts to Fish.

7.2.3.1.1 Fish Removal and Exclusion

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. Channel dewatering occurs primarily in freshwater streams and is typically associated with the need to work “in the dry” during installation, construction, or replacement of culverts and bridges, or other in-water type activities. To reduce stranding, fish removal and exclusion from the construction zone is usually part of channel dewatering activities. This is typically accomplished through passive methods, such as the volitional movement of fish from the construction area during its slow dewatering, or through active methods, such as the use of hand nets, beach seines, or electrofishing equipment to capture and move fish from the construction area that will be dewatered (NMFS 2006). These activities have the potential to cause 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.

7.2.3.1.2 Bypass System

Use of a flow bypass system is a common means of creating exclusion areas via dewatering and flow reduction. This type of activity has the potential to entrain fish within the flow bypass system. If pumps are used to bypass water around a work site or to dewater residual pools within a portion of the dewatered channel, the hose or pipe pulling water from the channel is typically fitted with a protective screen to prevent entrainment of aquatic life into the intake hose/pipe of the pump. Such measures are required for all pumped diversions (WAC 220-110-190), and specific criteria for screens, including approach velocity, mesh size, and screen location, have been developed by NMFS (2008) and WDFW (1998).

Installation of a flow bypass system typically requires in-water work, which can disturb substrates and bank material and cause an increase in turbidity levels. Once the system is installed, operation of a flow bypass system generally will not result in disturbance to the streambed or cause an elevation in turbidity levels, unless the discharge at the outlet results in scouring of substrate material or erosion of streambanks. Energy dissipaters are generally required to preclude such scouring from occurring. Diversion outfalls require temporary erosion-protection measures to prevent scour at the point of return flow from the diversion channel or pipe (WDFW 2004). Removal of the flow bypass also requires in-water work and results in some disturbance to the streambed and banks as the cofferdam is removed and flow is returned to the channel. Generally, the downstream cofferdam is removed first to allow backwatering of a portion of the channel that was dewatered. Then the upstream cofferdam is removed, and flow is slowly returned to the channel to minimize resuspension of fine sediments and increases in turbidity.

7.2.3.1.3 Passive Capture

Passive capture of fish typically involves installing an upstream block net and a cofferdam and slowly dewatering the construction area. This type of passive fish removal eliminates the need to capture and handle some fish. Less commonly, active methods of fish removal may be used, such as the use of a beach seine to “herd” fish downstream to a point beyond the construction area and/or the use of electrofishing equipment to remove fish.

7.2.3.1.4 Partial Dewatering

Partial dewatering is a technique used to reduce the volume of water in the work area to make capture methods more efficient. In riverine habitats, this method is used to move fish out of affected habitats to reduce the number of individuals exposed to capture and handling stress and potential injury and mortality. NOAA Fisheries has estimated that 50–75 percent of fish in an affected reach will volitionally move out of an affected reach when flows are reduced by 80 percent (NMFS 2006). However, volitional movement

will lead to concentration of fish in unaffected habitats, increasing competition for available space and resources.

The following sections describe the physical impacts of channel dewatering on potentially covered fish species.

- Fish that remain in a dewatered reach during construction may encounter lethal conditions. Fish left in the exclusion area would potentially be directly exposed to stranding, dessication and asphyxiation during dewatering or, if left inundated, to mechanical injury and/or high-intensity noise, turbidity, and other pollutants. Trampling, higher-than-normal temperatures, and increased vulnerability to predators could also kill or injure fish. 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). Therefore, they face a higher likelihood of exposure to stranding or entrainment in dewatering pumps, which would be expected to lead to mortality. 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.
- NOAA Fisheries has estimated incidental take resulting from dewatering and fish handling associated with stream crossing projects. In calculating incidental take from these activities, the agency applied an estimated stranding rate of 8 percent for ESA-listed salmonids (which equates to 8 percent mortality) (NMFS 2006), based on an expected 45 percent capture efficiency using three pass electrofishing (Peterson et al. 2004), and assuming a 25 percent injury rate. If bank protection projects require dewatering similar streams, then a similar level of take could be expected.
- Fish removal efforts such as beach seining and electrofishing could inadvertently result in fish mortality. The amount of unintentional mortality (and non-lethal injury) attributed to seining would vary widely depending on the seine used, the ambient conditions, and the expertise of the field crew (NMFS 2006). Professional experience has shown that beach seining in areas of dense aquatic vegetation or in muddy areas could also result in significant mortality of seined fish that become trapped in a mass of vegetation or mud.
- Electrofishing could also kill both juvenile and adult fish if improperly conducted. Mortality could result from direct trauma or from indirect factors (e.g., as a result of disease or subsequent fungal attack due to scale loss).
- There generally would be fewer adverse impacts associated with seining compared to electrofishing, and first using a seine to remove fish would minimize the adverse effects of electrofishing (NMFS 2006).

- The act of capture and handling demonstrably increases physiological stress in fishes (Frisch and Anderson 2000). Primary contributing factors to handling-induced stress and death include exposure to large changes in water temperatures and dissolved oxygen conditions (caused by large differences among the capture, holding, and release environments); duration of time held out of the water; and physical trauma (e.g., due to net abrasion, squeezing, accidental dropping). Even in the absence of injury, stress induced by capture and handling can have a lingering effect on survival and productivity. One study found that stress from handling impaired the salmonids' ability to evade predators for up to 24 hours following release and caused other forms of mortality (Olla et al. 1995). Capture and handling could also reduce fish access to prey.
- Beach seining could affect fish in several ways, including stress, scale loss, physical damage, suffocation, and desiccation. Anesthetics such as tricaine methane sulfonate (also known as MS-222) and clove oil are often used to sedate fish to facilitate easier fish handling and reduce fish stress.
- Electrofishing could also result in sublethal effects, such as spinal injury (NMFS 2006; Snyder 2003). The following excerpt from NMFS (2006) concisely describes the state of the knowledge pertaining to electrofishing impacts:

Most of the studies on the effects of electrofishing 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. Smaller fish intercept a smaller head-to-tail potential than larger fish (Sharber and Carothers 1988) and may therefore be subject to lower injury rates (e.g., Dalbey et al. 1996, Thompson et al. 1997). McMichael et al. (1998) found a 5.1 percent injury rate for juvenile middle Columbia River steelhead captured by electrofishing in the Yakima River subbasin while Ainslie et al. (1998) reported injury rates of 15% for direct current applications on juvenile rainbow trout. The incidence and severity of electrofishing damage is partly related to the type of equipment used and the waveform produced (Dalbey et al. 1996, Dwyer and White 1997, Sharber and Carothers 1988). Continuous direct current or low-frequency (equal or less than 30 Hz) pulsed direct current have been recommended for electrofishing (Dalbey et al. 1996, Fredenberg 1992) because lower spinal injury rates, particularly in salmonids, occur with these waveforms (Ainslie et al. 1998, Dalbey et al. 1996, Fredenberg 1992). Only a few recent studies have examined the long-term effects of electrofishing on salmonid survival and growth (Ainslie et al. 1998, Dalbey et al. 1996). These studies indicate that although some of the fish suffer spinal injury, few die as a result. However, severely injured fish grow at slower rates and sometimes they show no growth at all (Dalbey et al. 1996).

In the absence of additional supporting information, it is reasonable to conclude that these same effects would affect many of the HCP fish species, but this conservative assumption may not be universally accurate. Studies of the effects of electrofishing on other fish species are more limited, but available data indicate that at least some HCP species may be less sensitive to injury-related effects. Holliman et al. (2003) subjected a threatened cyprinid (minnow) species to electrofishing techniques in the laboratory and found that the typical current and voltage parameters used to minimize adverse effects on salmonid species produced no evidence of injury. This suggests that other cyprinids (such as leopard and spotted dace, lake chub, and suckers) may also be less sensitive.

Electrofishing-related injury rates are variable, reflecting a range of factors from fish size and sensitivity, individual site conditions (e.g., water conductivity, visibility, etc.), 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).

- Channel dewatering decreases benthic prey availability for young salmonid life stages and other species that feed upon benthic prey in the area near the dewatered zone. Bell (1991) reported that the permanent wetted area of a channel is the governing factor in food production for salmonids because aquatic food supplies do not shift in streams as water levels rise or fall. The loss of prey is generally temporary, and as flow is returned to the dewatered portion of the channel, benthic macroinvertebrates from outside the dewatered area and those that sought refuge in the hyporheic zone recolonize the previously dewatered channel. The amount of time necessary for the benthic macroinvertebrate community to recolonize a dewatered reach will depend upon the size and duration of dewatering, the size and life cycles of the benthic macroinvertebrate community in nearby areas, and the season of disturbance (NMFS 2001b, 2005).
- Another potential impact is related to displacement of fish, either naturally to avoid high turbidity or as a result of fish removal. This will increase the density of fish in the area being utilized, and increase competition for food and space. In addition, dewatering, or water diversions may also effect migration patterns of fish.
- Dewatering and diversion may also disrupt migration patterns to various fish species and life history strategies.

7.2.3.2 Impacts to Invertebrates

Typically, potentially covered benthic invertebrate species are not removed during channel dewatering and so would be subject to injury or mortality. Loss of

macroinvertebrates can result from excavation, installation of bank protection structures, and placement of associated fill material.

HCP invertebrate species demonstrate different sensitivity to the effects of dewatering and relocation than fish, with many species being relatively insensitive to the effects of handling, at least during adult life-history stages. For example, Krueger et al. (2007) studied the effects of suction dredge entrainment on adult western ridged and western pearlshell mussels in the Similkameen River (Washington) and found no evidence of mortality or significant injury. Suction dredge entrainment is expected to be a more traumatic stressor than removal and relocation by hand. These findings suggest that careful handling would be unlikely to cause injury. However, the authors cautioned that these findings were limited to adult mussels, and the potential for injury and mortality in juveniles remains unknown.

Mussels provide a good example of potentially covered invertebrate species that may be affected by desiccation, as they exhibit sensitivities related to periodicity of inundation as well as temperature. Although no studies were located that specifically examined the impacts of construction-related dewatering, several studies have examined the influence of dam operations on freshwater mussel habitats, providing insight on the potential impacts from construction dewatering (summarized in Watters 1999). Depending on the use of the dam, water levels may fluctuate at regular intervals (for hydroelectric purposes) or random intervals (for flood control). In some areas, water levels may become shallow enough that thermal buffering is lost, allowing extreme temperatures to occur (Watters 1999). Blinn et al. (1995, in Watters 1999) reported that substrate subjected to 2- to 12-hour exposures to air required more than four months for mussels to regain a biomass similar to that in unexposed habitat. Federally endangered mussel species were reported by Neck and Howells (1994, in Watters 1999) as casualties of scheduled dewatering processes, and Riggs and Webb (1956) reported that several thousand mussels died in the tailwaters of Lake Texoma, an impoundment of the Red River formed by Denison Dam, when water levels dropped, in turn allowing water temperatures to become excessively warm (greater than 79 degrees Fahrenheit [F], 26 degrees Celsius [C]).

Combined with desiccation, exposure to cold air may be equally lethal to mussels. Nagel (1987, in Watters 1999) suggested that mussels would be more sensitive to cold water during frosts than to warm water during temporary droughts. Blinn et al. (1995) showed that a single overnight exposure to subzero temperatures resulted in at least a 90 percent loss of invertebrate biomass, and Valovirta (1990) reported that mussels were killed when water froze to the river bottom.

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

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

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

7.2.4 Navigation or Maintenance Filling and Dredging

Navigation or maintenance dredging is by far the most frequent form of dredging in Washington State. This type of dredging can convert intertidal habitat to subtidal habitat and shallower subtidal habitats to deeper subtidal habitats through periodic deepening to remove accumulated sediments that impede navigation to and from marinas/terminals. There are several different means by which dredging affects fish and invertebrates, the most significant being alteration of bathymetry, removal of aquatic vegetation, entrainment of benthic organisms, and turbidity and resuspension of contaminated sediments. These stressors are discussed below.

7.2.4.1 Altered Bathymetry and Substrate Composition

Large channel deepening projects can markedly alter ecological relationships through the change of freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing. Miller et al. (1990) reported that only through comprehensive areal surveys over a minimum of four seasons before dredging, with follow-up surveys after dredging, could impacts of channel deepening on aquatic resources be determined. In a comparison between dredged and undredged areas in the Port of Everett's public marina, Pentec (1991) found catches of fish to be higher in the dredged area before dredging than after dredging. Catches decreased from about 90 fish per tow to about 3 fish per tow and from eight species to five species.

Depending on site characteristics, maintenance dredging may occur annually or at intervals of 10 years or longer. These different dredging timelines represent different disturbance regimes both in terms of the ability of the benthos to recolonize prior to redisturbance and the magnitude of benthic productivity affected by dredging. In a literature review report on dredge and disposal effects, Morton (1977) reported the range of effects on invertebrate communities to be from negligible to severe, with impacts ranging from short to long term. In general, this literature review found that short-term, small-scale dredging and dredge disposal projects affected benthic communities less than long-term, large-scale projects. This is likely due to the fact that benthic communities are more likely (and quicker) to recover from short-term, less intense, small-scale disturbances than from large-scale and intense disturbances over long time periods (Guerra-García et al. 2003; Dernie et al 2002). For example, in experiments conducted in

sheltered sand flats, the benthic community recovered from lower intensity disturbance (i.e., sediment removal to a depth of 3.9 inches [10 cm]) within 64 days, whereas recovery from higher intensity disturbance (i.e., sediment removal to a 7.9-inch [20-cm] depth) required 208 days postdisturbance (Dernie et al 2002).

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

Dredging is often required during marina and terminal projects as a component of facility development, as well as during routine maintenance to maintain navigability. In marine environments dredging converts intertidal into subtidal habitats, affecting the plant and animal assemblages that are uniquely adapted to the particular light, current, and substrate regimes of intertidal areas. By altering bathymetry and bottom substrates, such conversions are described as producing a habitat “trade-off” of intertidal and shallow-subtidal communities for deeper, subtidal communities. In lacustrine environments, dredging converts shallow-water littoral habitats into deeper water environments and may create a steeper bathymetric transition. This change in habitat characteristics may change the size and species distribution of fish in the localized environment, altering predator/prey dynamics. The effects of dredging on riverine environments are more complex still, because localized alteration of channel morphology can lead to dynamic shifts in channel form as the system adjusts to the changed conditions. These effects can extend a considerable distance beyond the bounds of the original dredging project.

Dredging activities result in short-term direct effects, including entrainment and potential mortality; periodic removal of potentially suitable habitats for fish and invertebrates; alteration of water circulation and subsequent nutrient, prey, and habitat availability; and increased turbidity and potential resuspension of contaminants. In addition, long-term and food web indirect effects can occur, such as reconfiguration of the benthos and the availability of nutrient and prey resources. Resulting impacts, include mortality, injury, decreased foraging opportunity, decreased growth and fitness, and physiological and behavioral responses. Deposition of dredge spoils can bury existing habitats and benthic organisms, resulting in a similar suite of impacts. For invertebrates at dredge disposal sites, research has shown potential increases in densities.

7.2.4.2 Effects of Entrainment and Burial on Fish

Entrainment occurs when an organism is trapped in the uptake of sediments and water being removed by dredging machinery (Reine and Clark 1998). Demersal fish, such as sand lance, sculpins, and pricklebacks, likely have the highest rates of entrainment as they reside on or in the bottom substrates, with life-history strategies of burrowing or hiding in the bottom substrate. This is also true in freshwater environments. For example, lamprey ammocoetes likely have a high risk of vulnerability to dredging due to the lengthy residence time in freshwater sediments in their early life-history stages. In

general, larval fish that have little or no swimming capacity to avoid direct dredge impacts are also at significant risk of entrainment in dredge sites. Of particular concern for the purpose of this analysis are the HCP groundfish (lingcod, rockfish, Pacific cod, pollock, hake) and the forage fishes (herring, sand lance, and surf smelt), all of which have larval or juvenile life-history stages with low motility. The juvenile life-history stage of the groundfish species typically rear in shallow nearshore habitats, where dredging is likely to occur. Due to their demersal nature and limited motility, they face a higher risk of dredging entrainment.

Larger fish may also be susceptible to entrainment. Armstrong et al. (1982) found that larger fish were not necessarily able to avoid the hopper dredge, with the largest specimen being a 9.2-in (234-mm) tomcod. Tests of excluders mounted on the draghead of a hopper dredge showed that 66 percent fewer fishes (mostly flatfish and gunnels in the study) could be saved from entrainment through use of the device (Shaw 1996).

Buell (1992) found entrainment of juvenile white sturgeon (11.8–19.6 in [300–500 mm]) at a rate of 0.015 fish/cy. In another study, juvenile salmonids and eulachons were the dominant entrained taxa due to the dredge location in a constricted waterway, making it more difficult for salmonids to avoid the dredge operation (McGraw and Armstrong 1990; Larson and Moehl 1990).

Entrained bivalve larvae, such as larval oysters, are assumed to suffer 100 percent mortality by sediment smothering, anoxia, starvation, or desiccation even without direct mechanical impacts from pumping. However, the population-level effects of these stressors may be relatively limited. For example, concern for oyster larvae entrainment in Chesapeake Bay resulted in the development of a population model using conservative temporal and spatial distributions (Lunz 1985). The model predicted that entrainment would have minimal negative effect on the population, with the calculated mortality rate ranging between 0.005 and 0.3 percent of larval abundance. Lunz (1985) concluded that this represented no significant impact as the dredge entrained only a small fraction of the total water volume flowing past the dredge. 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). Therefore, the potential mortality from entrainment is relatively insignificant in comparison (Lunz 1985).

7.2.4.3 Effects of Entrainment and Burial on Invertebrates

Benthic infauna are particularly vulnerable to being entrained by dredging uptake, but mobile epibenthic and demersal organisms such as burrowing shrimp, crabs, and fish also can be susceptible to entrainment. Entrainment rates are usually described by the number of organisms entrained per cubic yard (cy) of sediment dredged (Armstrong et al. 1982).

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.

- 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 (50 mg/m^2), with mortality and inhibition of feeding at higher levels of fine sediment (200 mg/m^2) (Airoldi and Hawkins 2007). The mechanism of effect is postulated to be the clogging of 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 highly sensitive to the effects of entrainment and burial and are assumed to suffer high mortality from mechanical injury, smothering, anoxia, starvation, or desiccation. However, in the case of freshwater mussels, stressor exposure would have to be extensive to result in significant population-level effects. As an example, the issue of larval oyster mortality caused by dredge entrainment was studied in detail 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 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.

7.2.5 Ambient Light Modifications

Along marine, riverine, and lake shorelines, marinas (as a collection of individual piers) and shipping or ferry terminals are known to affect light availability and the aquatic habitats upon which HCP species depend. A considerable body of literature provides evidence that shading from these structures can reduce ambient daytime aquatic light availability to levels below the light threshold levels required for aquatic plant photosynthesis and fish feeding and movement. Effects of reduced light availability on plants is discussed under Aquatic Vegetation Modifications. These facilities can also alter ambient nighttime light through the use of artificial light. In the case of terminals that berth large vessels, documented shade casting includes the reflective effects of sediment resuspension and bubbles generated by high propulsion prop wash in shallow environments (Blanton et al. 2001; Haas et al. 2002; Thom et al. 1996).

7.2.5.1 Fish Vision

Light perception by fish is dependent upon the light transmission qualities of the water environment coupled with the spectral qualities of the fish retinal visual pigments (Ali 1959, 1975; Brett and Groot 1963; Fields 1966; Hoar 1951; Hoar et al. 1957; McDonald 1960; McFarland and Munz 1975; Mork and Gulbrandsen 1994; Nemeth 1989).

Habitat and genetics determine the light absorption capacities of fish visual pigments. Capacities differ across the solar spectral compositions specific to the habitats upon which these species depend for growth and survival (Browman et al. 1993; Coughlin and Hawryshyn 1993; Hawryshyn and Harosi 1993, Novales-Flamarique and Hawryshyn 1996; Wald et al. 1957).

Light is received by the fish retina. This light reception triggers physiologic responses. The visual cell layers consist of two types of photoreceptors, rods, and cones. These retinal pigments have different light thresholds and respond to light and dark with changes in their relative positions. When the light intensity is above the retinal pigment and cone thresholds, the eye assumes the light-adapted state. When the light intensity falls below threshold values, the cones expand away, and the eye assumes a dark-adapted state (Ali 1959). In freshwater laboratory studies, Ali (1959) found that when the light drops below particular thresholds, the school disbands and feeding by visual means ceases, with the extent of expansion and elongation dependent upon ambient conditions (Ali 1975).

The time period for such physiologic changes in response to light variations varies across species and lifestages. At the juvenile stage, the time required for light-adapted chum and pink salmon fry to fully adapt to dark conditions was found to range from 30 to 40 minutes. However, the time required for dark-adapted fry to adapt to increased light conditions was found to range from 20 to 25 minutes (Ali 1959; Brett and Ali 1958; Protasov 1970). During these transition periods, the juvenile chum's visual acuity ranges from periods of blindness to a slightly diminished capacity, depending upon the magnitude of light intensity contrasts. As the animals become older, the time required for light adaptation generally shortens. The time necessary to adapt to the dark, on the other hand, tends to increase with age. The progression of retinal changes from one state to another is influenced by the intensity of the introduced light and the intensity of light to which the fish have been previously exposed (Ali, 1962, 1975; Fields 1966; Protasov 1970; Puckett and Anderson 1987). It is the contrasts in light levels that determine the changes the eye undergoes and the speed of transition from one state to another. Fish previously exposed to higher light intensities become dark-adapted more slowly than those previously exposed to lower light intensities (Ali 1962). A review of the literature covering juvenile salmon behavioral responses to ambient and artificial light also revealed species-specific behavioral differences. Species that occupy and defend stream territories, such as coho, tend to be quiescent at night, while species that disperse to estuaries, such as Chinook, pink, and chum, typically school, show nocturnal activity, and demonstrate an aversion to light (Godin 1982; Hoar 1951).

The teleost fishes, a classification that includes all HCP fish species with the exception of the lampreys and the sturgeons, depend on sight for feeding, prey capture, and schooling. For these fishes, sight is the primary sensory organ used for spatial orientation, prey capture, schooling, predator avoidance, and migration. As juveniles, they utilize nearshore or shallow water habitats and share a sensitivity to ultraviolet wavelengths reflected in shallow-water habitats (Britt 2001, Tribble 2000, both in Nightingale and Simenstad 2001b). By interfering with sight, modification of the underwater light environment may affect these fundamental activities. Shade can affect fish and invertebrates by disrupting normal migration patterns, reducing the ability to avoid predators, capture prey and reducing available refuge (Ali 1962, 1975; Britt 2001; Fields 1966; Hoar et al. 1957; Johnson et al. 1998; McDonald 1960; Mork and Gulbrandsen 1994; Nightingale and Simenstad 2001a; Tribble 2000).

Juvenile and larval fish are primarily visual feeders, with starvation being the major cause of larval mortality in marine fish populations. Survival has been found to be linked to the ability to locate and capture prey and avoid predation (Britt 2001). This ability depends on sufficient light. Tribble (2000) found the swimming and feeding behavior of juvenile and larval sand lance to be reduced with low-light levels. Similar to other juvenile fishes with cone-based vision, the retinal cells of larval sand lance exhibit limited visual acuity in low-light environments. Their visual acuity increases with growth, with an eventual development of rod vision that provides them with vision in light-limited environments. Rods appear to develop at 0.94-in (24-mm) fork length, and full adult visual acuity develops at 1.38-in (35mm) fork length. This visual development prepares them for transition to deeper waters.

Tribble (2000) reports that the visual development of Pacific sand lance reflects the respective habitats they occupy given their size. At 1.97 in (50 mm) in length, they begin to move into deeper pelagic waters where the light environment changes, and their light requirements for prey capture change in response to the light wavelengths characteristic of that habitat. Many juvenile fishes using nearshore habitats, such as the Pacific sand lance (Tribble 2000), salmonids (Ali 1959), and lingcod (Britt 2001), share this sensitivity to ultraviolet (UV) wavelengths reflected in shallow nearshore marine habitats. Similar to salmonids, yellow perch and sand lance have been found to lose UV sensitivity with growth. Browman et al. (1993) reports this loss of UV sensitivities to be size-related rather than age-dependent and to likely correlate with the time that such fishes move from shallow to deeper water habitats and move from feeding on small crustaceans and other zooplankton to larger food items. As zooplankton reflect short wavelength light, such as UV, this provides an advantage for juvenile fishes with UV sensitivity feeding upon zooplankton in shallow nearshore waters. The ability of zooplankton to reflect UV is likely due to high concentrations of amino acids that protect them from the damaging effects of UV radiation.

Figure 7-1 depicts light conditions related to juvenile salmon behavior such as schooling, predator avoidance, feeding, and migratory behavior.

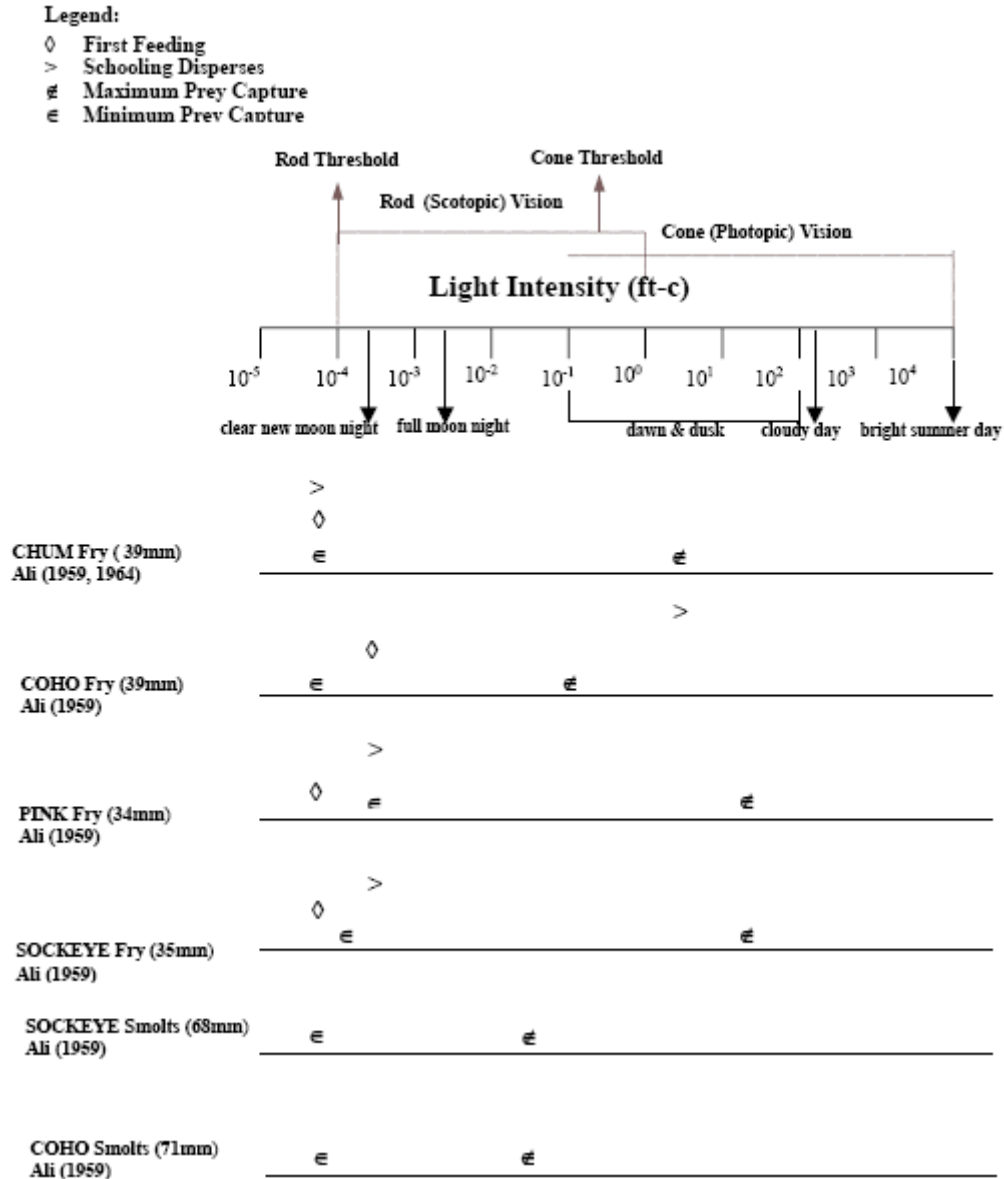


Figure 7-1: Juvenile Salmon Behavior Patterns Related to Light Intensity
 Source: Nightingale and Simenstad 2001b

Tribble (2000, in Nightingale and Simenstad 2001b) found the swimming and feeding behavior of juvenile and larval sand lance (*Ammodytes hexapterus*) to be reduced with low light levels. Similar to other juvenile fishes with cone-based vision, the retinal cells of larval sand lance fall in the violet to green range, with limited visual acuity in low-light environments. Their visual acuity increases with growth as their cone pigments shift from violet to blue sensitivity. Tribble (2000, in Nightingale and Simenstad 2001b) reports that sand lance visual development reflects the habitats they occupy at given total lengths. Rods appear to develop when the fish reach approximately 1 inch (24 millimeters [mm]), and full adult visual acuity develops at 1.4 inches (35 mm). At approximately 2 inches (50 mm) in size, the fish will begin to move into deeper pelagic waters, where the light

environment changes, and their light requirements for prey capture change in response to the light wavelengths characteristic of that habitat. At this point they will largely depart from the range of water depths where they may be affected by overwater structures. A similar change in visual sensitivity has been observed in yellow perch. Brownan and Hawryshyn (1994, in Nightingale and Simenstad 2001b) report this loss of ultraviolet sensitivity to be size-dependent rather than age-dependent and to likely correlates with the time when fishes move from shallow to deeper water. These results suggest that shading effects attributable to overwater structures predominantly affect smaller fish, and that “shading” as an impact includes the loss of both visual and ultraviolet wavelengths of light.

7.2.5.2 Daytime Shading and Fish Behavior

In response to daytime shading, fish potentially modify migration direction or behavior, resulting in increased energy expense. Shading can also reduce foraging success and increase potential exposure to predation. In addition, shading can modify species assemblages to a degree that available habitat is rendered unsuitable for native fish or invertebrate species. For invertebrates, shading can alter the suitability of habitat and reduce foraging opportunity as well as the availability of nutrients, resulting in decreased survival, growth, and fitness.

7.2.5.2.1 Shade in Freshwater Systems

Shade cast by overwater structures in the freshwater environment can be used by some fish as cover and can increase predation on juvenile salmonids (Tabor et al. 1998). Indeed, in freshwater environments of western Washington, largemouth and smallmouth bass are common predators of juvenile salmonids, and several authors have documented the use of overwater structures by bass (Carrasquero 2001; Kahler et al. 2000, Tabor et al, 1998, Stein 1970, Helfman 1979 both in Carrasquero 2001). Carrasquero’s (2001) review found that the attraction of fish to floating or overhanging objects is linked to the shade produced by the objects, and Kahler et al. (2000) suggests that piers, piles, boatlifts, and moored boats provide cover, shade, and focal points that benefit exotic predators of juvenile salmon, such as smallmouth and largemouth bass. An alternative explanation of fish attraction to on-water and overwater structures in fresh water was presented by Fresh (pers. comm., in Carrasquero 2001), who explains that both the structures and the shade they cast may provide fishes with physical reference points for orientation.

Interactions of smallmouth bass and juvenile salmonids depend on timing of salmonid outmigration, salmonid species, and residence of the juvenile salmonids and found studies that suggest the attraction of predatory fish (including largemouth bass) to floating or overhanging objects is linked to the shade produced by the objects rather than to the tactile stimulus and that the larger the floating object, the greater the shaded area, and thus the greater the number of fish attracted to such objects (Carrasquero 2001). This assumption suggests that shading from overwater structures alters fish distribution and aggregation in fresh water. In addition, Kahler et al. (2000) states that shading from

overwater structures may reduce the abundance of prey organisms available to juvenile salmonids and forage fish by reducing aquatic vegetation and phytoplankton abundance.

In freshwater environments of Western Washington, largemouth bass and smallmouth bass are common predators of juvenile salmonids, and several authors have documented the use of overwater structures by bass in Western Washington waters. Stein (1970, in Carrasquero 2001) examined the types of cover used by largemouth bass in Lake Washington and found that they prefer areas of heavy log and brush cover over other habitat types (including docks). However, largemouth bass are commonly found under docks in early spring and are thought to be present there until late summer (Stein 1970, in Carrasquero 2001).

Interactions between smallmouth bass and juvenile salmonids depend on factors such as the timing of salmonid outmigration, salmonid species, and residence time of juvenile salmonids in lentic (still-water) or lotic (flowing) environments (Fayram and Sibley 2000, in Carrasquero 2001; Gray et al. 1984; Gray and Rondorf 1986; Pflug and Pauley 1984; Poe et al. 1991; Shively et al. 1991; Tabor et al. 1993; Tabor et al. 2000; Warner 1972;).

Carrasquero (2001) presents the following observations and inferences of predator/prey aggregations in freshwater environments under and around structures:

- Different fish species respond differently to the shade produced by overwater structures.
- Smallmouth bass and largemouth bass have a strong affinity to structures, including piers, docks, and associated pilings.
- Bass have been observed foraging and spawning in the vicinity of docks, piers, and pilings; where vegetation is lacking, largemouth bass seek other forms of structures, such as dock pilings.
- Smallmouth bass are opportunistic predators that consume prey items as they are encountered and are major predators of juvenile salmonids.
- Fish, particularly largemouth bass, seem to be attracted to the shade produced by floats, rather than their physical structure. In contrast, smallmouth bass do not seem to be attracted to the shade produced by such structures.
- In reservoir systems of Eastern Washington, juvenile salmonid predation is specific to the behavior and distribution of each salmonid species and its predator. The behavior and distribution of predator and prey species reportedly depend on temperature, the degree of shore-zone development, the slope and substrate of the shoreline, and the presence of man-made in-water structures.

7.2.5.2.2 Shade In Marine Systems

In marine environments, shading also influences prey abundance and prey capture. Haas et al. (2002) found that densities and assemblages of important epibenthic prey organisms were reduced under large overwater structures. In New York Harbor, Able et al. (1998) found juvenile fish abundance to be reduced under piers when compared to open water or areas with only piles but no overwater structure. This is likely due to both limitations in prey abundance and prey capture under structures. In a New York study of pier impacts on fish growth and prey resource abundance, Duffy-Anderson and Able (1999) compared growth rates of caged juvenile fish under municipal piers to those of fish caged at pier edges and in open water beyond piers. Those fishes caged under the piers showed periods of starvation, which could potentially make these individuals more vulnerable to predation, physiological stress, and disease. Along the pier edge, they found growth rate variability to be extremely high and likely related to light levels. They concluded that light availability is likely an important component of feeding success. They concluded that large piers do not appear to be suitable habitat for some species of juvenile fishes and that increased sunlight enhances growth.

The addition of floating piers is also known to affect nearshore ecology by shifting population structures to non-native species as a result of shading. In southern California, Reish (1961) observed a succession of attached organisms occurring on marina floats with an apparent climax community of the *Mytilus* mussel and *Ulva* algae after the floats were in the water for 6 months.

In abundance, *Ulva* spp., an opportunistic green macroalgae, is known to reduce light and oxygen and create an anoxic environment (Hull 1987; Hernandez et al. 1997). Through shading, the algae *Ulva* is capable of triggering habitat shifts resulting in declines of eelgrass and concomitant increases in *Ulva* (Wilson and Atkinson 1995; Wilson 1993). The Puget Sound Expedition, a survey of nonindigenous species, sampled dock-fouling organisms on floats at 26 marinas throughout the entire Puget Sound region and identified 39 nonindigenous species (Cohen et al. 1998).

In the marine nearshore, daytime light reduction caused by shading under overwater structures could cause migrating juveniles to move into deeper waters, increasing the risk of predation by larger predators that occupy pelagic waters (Heiser and Finn 1981, Pentec 1977, in Nightingale and Simenstad 2001b). Predation mortality may increase through altering predator detection and reducing refugia provided by the schooling behavior of juvenile salmonids (Pentec 1997, in Nightingale and Simenstad 2001b).

Based on a combination of light measurements, visual fish survey, and acoustic tagging and telemetry fish tracking undertaken over a 7-week period between April 20 and June 3, 2005, Southard et al. (2006) found under-terminal light levels at the Anacortes, Bainbridge, Clinton, Edmonds, Fauntleroy, Kingston, Mukilteo, Port Townsend, Southworth, and Vashon terminals to deter or delay juvenile salmon movement along the nearshore. This effect was found to be dependent upon nearshore morphology, tidal level, and terminal design features affecting light availability.

Behaviors important to the growth and survival of fishes, such as migration, schooling, and feeding, are known to be altered by changes in light availability. For example, abrupt transitions from light to dark can cause juvenile Chinook salmon to alter their migration pathway from the nearshore (shallow water) to deeper water or avoid an overwater structure altogether (Tabor et al. 2004.) Some salmonids commence or terminate these behaviors in response to specific light levels or thresholds. In a snorkel and beach seine survey of Seattle marine shorelines, Toft et al. (2004) reported that juvenile salmon avoided swimming beneath overwater structures, while other animals (such as crabs and sculpin) were found in these under-dock habitats. Large groups of juvenile salmonids were found in the vicinity of overwater structure sites; however, most juvenile salmonids were observed at the edge of the overwater structure or farther away, with only one school observed underneath a structure. Similarly, only one Pacific sand lance was observed under an overwater structure, with most being along the periphery or in the general vicinity of the overwater structures. In general, most fish were not observed underneath overwater structures. This study suggests that the under-pier environment, in particular shading effects, could affect the behavior and movement of salmon along the nearshore area (Toft et al. 2004; Simenstad et al. 1999; Able et al. 1998).

Although it is believed that predation risks are elevated when fish move into deeper waters around piers, the actual potential for increased predation due to aggregating predators under structures in marine environments is uncertain (Weitkamp 1981; Taylor and Wiley 1997, in Nightingale and Simenstad 2001b). Taylor and Wiley (1997) found no aggregation of avian predators and Weitkamp (1981) reported no aggregation of aquatic predators during the peak juvenile chum outmigration. Consistent with these findings, Penttila and Aguero (1978, in Nightingale and Simenstad 2001b) found no empirical evidence of predation among the marina floats in Birch Bay, but instead found evidence of competition among fish species for mutually preferred prey resources (i.e., the calanoid and harpacticoid copepods). Fresh and Cardwell (1978, in Nightingale and Simenstad 2001b) list 17 potential predators of juvenile salmon in the southern Puget Sound region and find that only three (maturing Chinook, copper rockfish, and staghorn sculpins) prey extensively on nearshore fishes. Their analysis of food habits found only staghorn sculpins with juvenile salmon in their stomachs, and there was no evidence that staghorn sculpins were in greater abundance under structures than elsewhere in the study area. Additionally, Ratte (1985, in Nightingale and Simenstad 2001b) found sea perch and pile perch, which do not prey on salmonids, to be the most abundant fish species under docks. Nightingale and Simenstad (2001b) and Southard et al. (2006) summarize these and additional studies that pertain to fish behavior, including migration, distribution, and predator/prey relationships potentially associated with overwater structures in marine areas of Puget Sound.

7.2.5.3 *Nighttime Artificial Lighting*

Artificial night-light-induced changes to ambient nighttime conditions appear to affect fish migration behavior and place some species at risk of increased predation (Fields 1966; Johnson et al. 1998; Prinslow et al. 1979; Ratte and Salo 1985; Weitkamp and Campbell 1980; Weitkamp and Schadt 1982;). Prinslow et al. (1979) reported changes to fish assemblages and predation rates during a study of the effects of high-intensity

security lights on a naval base (Bangor) in Puget Sound's Hood Canal. At that site, the level of intensity of artificial night lighting appeared to influence the behavior of fishes, with significantly greater light intensities (200–400 lux) attracting aggregations of juvenile chum and other small fishes. This aggregation suggested a potential to delay chum outmigration through the canal. Spiny dogfish, a Puget Sound shark, also appeared to be attracted to security lighting, likely due to the illumination of aggregating prey. Although herring and sand lance were not the subject of the study, Prinslow et al. (1979) reported potential exposure of herring and Pacific sand lance to predation due to the effects of the security lighting. Prinslow et al. (1979) suggested that based on study observations, the continuous use of high-intensity security lighting at the Bangor wharves could contribute to increased predation of HCP species.

Impacts to fish from artificial lighting are often the result of changes in nighttime behaviors such as migration, activity, and location (Nightingale and Simenstad 2001b) and potentially in schooling behavior in juvenile salmonids (Ali 1959, 1962, in Simenstad et al. 1999). Therefore, behavioral differences between species at differing life stages, life histories, and behaviors specific to the local environment must be considered when evaluating potential impacts from artificial light. For instance, different species of salmonids have different nighttime behaviors. Species that occupy and defend stream territories, such as coho salmon and steelhead trout, tend to be quiescent at night (Simenstad et al. 1999), while species that disperse to lakes and estuaries as juveniles, such as sockeye, Chinook, pink, and chum salmon, typically school and show nocturnal activity (Godin 1982, Hoar 1951, both in Nightingale and Simenstad 2001b). Behavioral differences in salmonid responses to artificial lighting have been observed by several authors. Ocean-type juvenile salmon, such as chum and summer and fall run Chinook, are attracted to lights at night (Simenstad et al. 1999). Pucket and Anderson (1988, in Simenstad et al. 1999) and Nemeth (1989, in Simenstad et al. 1999) found that different species of salmon react differently to strobe lights; Mork and Gulbrandsen (1994, in Simenstad et al. 1999) found differing activity levels in reaction to lights at surface and bottom depths in different species of salmon, trout, and char. Fields (1966, in Simenstad et al. 1999) found that spring migrant juvenile salmon were more repulsed by bright lights than were later migrants. Behavior patterns of different salmon species related to different light intensities and other details of artificial light impacts to juvenile salmonids are reviewed by Simenstad et al. (1999).

Impacts to fish also depend on the fish's ability to adapt to dark or lighted conditions and the intensity and type of light. Ali (1959, in Simenstad et al. 1999) found that the eyes of sockeye fry and smolts and coho smolts adapt to light more slowly than do the eyes of coho, Chinook, and pink fry. Other studies by Ali (1959, 1962, in Simenstad et al. 1999) reveal the threshold light intensities for different behaviors of juvenile salmon.

Artificial lighting may be used during the construction of overwater structures, and some kinds of structures also require nighttime lighting for security or operations. Nighttime artificial lighting has been shown to change fish species assemblages by:

- Attracting fish to lighted areas (Prinslow et al. 1979, in Nightingale and Simenstad 2001b; Simenstad et al. 1999; Nightingale and Simenstad 2001b)

- Delaying salmonid migrations (McDonald 1960, in Tabor et al. 1998; Prinslow et al. 1979, in Nightingale and Simenstad 2001b; Tabor et al. 1998)
- Increasing the risk of predation (Kahler et al. 2000; Tabor et al. 1998)
- Altering predator avoidance and detection (Tabor et al. 1998)
- Increasing prey capture success for some species of fish (Nightingale and Simenstad 2001b; Prinslow et al. 1979, in Nightingale and Simenstad 2001b).

Similarly, in a study of lighted and nonlighted areas along the Cedar River in the City of Renton, Washington, Tabor et al. (2001) found increased nighttime lighting intensities to have a profound effect on the behavior of salmon fry. Results indicated that increased levels of nighttime artificial light intensity, measured at lighted building and bridge sites, appeared to cause sockeye fry to delay migration and move to the low-velocity and lighted shoreline habitats, where they were found to be more vulnerable to increased predation. Even small increases in light intensity levels appeared to affect fry behavior. Tabor found nightly downstream migration of sockeye fry to be initiated after light intensity was less than 1 lux. However, with the addition of 32 lux, migration almost completely stopped. Given such changes to the habitat, Tabor et al. (1998) reported that a reduction in the intensity of artificial night lighting could benefit these sockeye salmon. In a study comparing urban and rural nighttime light regimes for lake environments, Moore et al. (2006) found the relative intensity of illumination to increase along the suburban-to-urban gradient, under both clear and cloudy conditions, with the nighttime surface light intensity for urban lakes ranging from 7 to 48 times the light intensity for lakes in rural environments. An effect of the higher nighttime light intensities found in urban environments was the suppression of vertical migration of zooplankton in urban lakes (Moore et al. 2006). Nighttime light intensities have also been found to affect fish foraging, schooling, spawning, and vertical movement in the pelagic zone (Blaxter 1975; Gliwicz 1986; Robertson et al. 1988; Luecke and Wurtsbaugh 1993; Appenzeller and Legget 1995; Contor and Griffith 1995).

A number of studies have shown that fish respond quite differently to various lighting types, such as flickering strobe, mercury, or halogen light sources (Fields and Finger 1954; Hoar et al. 1957; Fields 1966; Prinslow et al. 1979; Puckett and Anderson 1987; Nemeth 1989; Johnson et al. 1998). In Washington State, fish responses to increased nighttime underwater light intensities have been found to pose potentially significant population effects including changes in light-mediated predation rates on fish, reduction in prey capture efficiency by increased fish avoidance behavior, and slowing of migratory behavior (Prinslow et al. 1979; Tabor et al. 1998, 2001).

The few studies that have examined predation rates on juvenile salmonids under varying light intensities have generally shown that within the natural range of light intensities (e.g., overcast skies, moonless nights, clear nights, moonlit nights), predation increases with increasing light (Ginetz and Larkin 1976, Mace 1983, Patten 1971, all in Tabor et al. 1998); however, this occurrence cannot be extrapolated to determine impacts of artificial night lighting and for all species and life stages of fish. Ali (1959, in Simenstad et al. 1999) found that the maximum prey capture success for coho fry and sockeye and coho smolts was at light intensities equivalent to levels found at dawn or dusk, whereas

maximum prey capture success for sockeye and pink fry was found to be equivalent to a cloudy day. Tabor et al. (1998) showed that under freshwater laboratory conditions, sculpin capture success of sockeye fry decreased with increased light. The authors also found that sculpin can capture sockeye fry even in complete darkness. Although sculpin success at capturing sockeye decreased with increasing light in a circular tank, the increased light slowed emigration of sockeye fry in a simulated stream, and predation increased under the lighted conditions due to the slower migration rate. The light may have also caused the fry to migrate in areas of lower water velocity and closer to the bottom, leaving them more susceptible to predation by sculpin (Tabor et al. 1998). Grebes, blue herons, and other birds have been observed feeding at night on the Cedar River delta in an area lit by Boeing Company facilities (Warner, pers. comm., in Kahler et al. 2000), and Tabor (pers. comm., in Kahler et al. 2000) observed grebes foraging under lights at night on Lake Washington. Finally, Kahler et al. (2000) suggests that lighting attached to piers in Lake Washington where bass congregate may benefit bass by extending the duration of predation because it allows the visual predators to forage at night.

Studies examining the use of artificial light for guiding salmonids safely through migration barriers, such as hydroelectric dams, have found measurable differences in different species' responses to both the quantity and quality of the light stimulus. For example, Puckett and Anderson (1987) found juvenile salmon to be attracted to incandescent light when encountering a decrease in ambient light intensity. In the case of steelhead, Puckett and Anderson (1987) found the fish to initially avoid the mercury light and then to swim toward the light, likely following adaptation.

Nighttime lighting can result in altered migration behavior and timing (interruption or stalling as a result of attraction to light sources) as well as increased predation (as a result of aggregation). Subsequently, fish survival is reduced.