7 Direct and Indirect Effects

7.1 Aquatic Vegetation Modifications

7.1.1 Introduction

Aquatic plants are a key element of marine, estuarine, riverine, and lacustrine ecosystems. Aquatic vegetation can be broadly sorted into "marine" and "freshwater" categories. In this paper, the term "marine aquatic vegetation" refers to eelgrass (Zostera spp.); kelps (e.g., bull kelp [Nereocystis leutkeana]); other green, brown, and red macroalgae species; and intertidal wetland vascular plants (i.e., saltmarsh plants) that grow in the marine and estuarine habitats of the state. "Freshwater aquatic vegetation" is submerged and emergent plant material that is rooted below the ordinary high water line (OHWL) of freshwater bodies such as rivers, streams, lakes, ponds, and open-water wetlands.

Aquatic plants provide shelter, habitat and clinging substrate for a variety of aquatic invertebrate species, including insects and zooplankton (Petr 2000). Fish use aquatic plants for cover, and terrestrial wildlife species (in addition to potentially covered species) use emergent aquatic plants for food and habitat (Petr 2000). Aquatic plants provide energy to aquatic ecosystems through photosynthesis, nutrient cycling, and provide food for herbivores and detritivores (Petr 2000). Emergent aquatic vegetation can reduce wave-induced bank erosion (Coops et al. 1996). Aquatic vegetation can modify its physicochemical environment by slowing water velocity, trapping sediment, and altering temperature and water quality (Chambers et al. 1999).

The benefits of aquatic plants listed by Cowx and Welcomme 1998 *(1988?),* in Petr 2000 include:
1. Water purification, both direct (for example, by oxygenation and conversion of toxic ammonia to usable nitrates) and indirect (for example, by plants providing a huge surface area for microbes to do the same tasks).
2. Nutrient recycling, including nutrient removal during the growth season and return during senescence .
3. Physical link between water and air for many invertebrates, e.g., larvae and nymphs of caddis flies, mayflies, and chironomids, which are food for fish and have aquatic larval stages and aerial adults.
4. Refugia for zooplankton, which graze phytoplankton and keep water clear.
5. Cover for a large variety of invertebrates, many of which are food for fish.
6. Cover for fish, which varies as to value and type with the age and species of fish, as well as type of vegetation.
7. Spawning areas and sites of oviposition for many fish species, including Olympic mudminnow, a potentially covered species.
8. Food sources for herbivorous and detritivorous fish, indirect food sources from invertebrate prey living on vegetation surfaces.
9. Effects on flow patterns, i.e., accretion of sediments and deflection of flow, thus providing quiescent waters and faster shallows.

10. Creation of discrete habitat that is as functional as physical structure.

Alterations to aquatic vegetation by HPA-permitted projects may result in altered habitat complexity. This includes changes or reduction in three dimensional habitat structure, refuge and edge habitat, foraging opportunities, and altered autochthonous inputs (alteration of primary productivity, nutrient cycling, and conversion of dissolved organic material into biomass available for grazers, affecting food web productivity). However, HPA-permitted projects are not the only factor that affects the distribution of aquatic vegetation; it is also limited by the ecological conditions of the water body and the requirements of aquatic plant species (Chambers et al. 1999). For example, substrate type can play a role in the survivability of aquatic plants (Koch 2001).

7.1.2 *Aquatic Vegetation and Altered Habitat Complexity*

Changes to aquatic vegetation may cause alterations to habitat complexity through altered autochthonous (primary productivity and other in-water) inputs, changes in the food web, and changes in habitat structure. In some case, alterations have been linked to specific types of HPA-permitted projects. For both fish and invertebrates, altered habitat could reduce foraging opportunity as well as the availability of nutrients, resulting in decreased survival, growth, and fitness.

7.1.2.1 *General Effects: All Environments*

HPA-permitted activities may uproot or permanently displace aquatic vegetation during construction, maintenance, and operation of structures. Short-term construction-related impacts may temporarily modify vegetation. In-water ground disturbance has been used as a measure of habitat take in ESA biological opinions (NMFS 2006e). Vessels used during installation, operation, and/or maintenance of HPA-permitted structures may physically disturb submerged vegetation through increased velocity from propeller wash. Lagler et al. (1950, in Carrasquero 2001) reported that outboard motor use has been shown to clear a swath when used within 1 foot (30 centimeters [cm]) of aquatic vegetation. In addition, propeller use may entrain air bubbles and cause sediment suspension that result in a temporary reduction in light availability (Haas et al. 2002).

HPA-permitted structures can cause losses of aquatic vegetation by several pathways in addition to direct disturbance.

- Hydrologic and geomorphic alterations may induce changes in substrate composition and stability and alter habitat suitability for aquatic vegetation. HPA-permitted activities may increase wave reflection, causing unsuitable energy conditions for vegetation, and may flush out small substrate that supports plant growth. Increased flow velocities and substrate characteristics caused by hydraulic and geomorphic modifications can scour algae and damage macrophytes. Conversely, changes in substrate composition with an increase in fine sediment transport can bury aquatic vegetation. Dewatered channels may experience loss of aquatic vegetation, depending on the duration of dewatering and the vulnerability of the affected plants to desiccation and drought stress.

- Increased turbidity is known to compromise the survivability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. Increased turbidity can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted an approximate 50 percent decline in the number of seagrass species and a precipitous decline in seagrass biomass with a 15 percent increase in the clay content of the sediments.

- Numerous studies have shown that macrophytes and algae in both marine and freshwater environments reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). The processing and retention of sediment, nutrients, and pollutants in aquatic systems is accelerated by the presence of aquatic vegetation (Clarke 2002). Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. Aquatic vegetation does more than reduce nutrient and sediment concentrations; the plants themselves can sequester harmful trace metal pollutants and are frequently planted in wetland treatment systems with that intended function. In a comparative study of heavy metal uptake in terrestrial, emergent, and submerged vegetation, Fritioff and Greger (2003) noted that submerged vegetation was efficient at removing zinc, copper, cadmium, and lead from influent stormwater.

- HPA-permitted activities may affect aquatic vegetation through changes to ambient light. The growth and survival of submerged aquatic plants (benthic and planktonic) are dependent on identified light levels, known as photosynthetically active radiation (PAR). Light levels falling below PAR are known to limit the photosynthesis for a suite of aquatic photosynthesizers, such as diatoms, epiphytes, eelgrass, and other autotrophs important to HCP species. On average in Puget Sound, instantaneous mid-day PAR greater than approximately 150 $\mu M/m^2/sec$ is required to maintain eelgrass growth. Instantaneous PAR of approximately 325 $\mu M/m^2/sec$ is required to support maximum densities (Thom et al. 1998). The light requirements of different species vary, but reduced light in the littoral zone of freshwater environments can potentially limit aquatic vegetation (Chambers et al. 1999). The availability of light is also a crucial parameter for seagrasses and other marine aquatic vegetation (Hall et al. 1999). The loss of vegetation from shade could pose an indirect effect on the HCP species that rely on the species supported by that vegetation. Light limitations can lead to a local reduction in autochthonous primary production (i.e., organic matter produced by aquatic plants within a water body) and a reduction in the other functions of aquatic vegetation, including cover, substrate for invertebrate species, and food for herbivores (Hruby et al. 1999).

Structures can limit ambient light availability. Many types of HPA-permitted structures, including overwater structures, marinas, terminals, bridges, and culverts have been shown to shade the area underneath, within, and adjacent to the structures. The orientation of the structures, density of the structure (solid or open), culvert height, height above water (for bridges, overwater structures, marinas and terminals), length and width of the structure, size and spacing of supports (piers or piles), water depth, and tidal range all affect the extent and degree of shading (Nightingale and Simenstad 2001b).

Effects on fish associated with decreased light penetration and decreased vegetation include reduced foraging success and altered migration timing due to a reduction in primary productivity and associated reductions in prey species. Decreased light penetration and shading impacts on vegetation could also result in fish expending increased energy or being increasingly exposed to predation because of loss of suitable habitat and cover.

- HPA-permitted activities may introduce noxious aquatic weeds, or may alter ecosystems in ways that sustain their growth.

7.1.2.2   Ecosystem-Specific Effects: Marine and Estuarine

Marine aquatic vegetation is a fundamental structural and ecological component of the nearshore ecosystem and substantially influences the physical and chemical properties of the nearshore environment (Nightingale and Simenstad 2001b).

The Washington State Hydraulic Code Rules (WAC 220-110-250) designate eelgrass, kelp, and intertidal vascular plants as saltwater habitats of special concern and require that hydraulic projects result in no net loss of these habitats. The hydraulic code rules require that overwater structures be designed or located to avoid shading or other impacts that could result in the loss of eelgrass and kelp habitat (WAC 220-110-240 through 330).

Eelgrass is associated with important rearing habitats for a suite of marine fishes, such as Pacific cod, Pacific salmon, rockfish, Pacific herring, walleye pollock, and rockfish (Gustafson et al. 2000; Murphy et al. 2000; Nightingale and Simenstad 2001a; Simenstad et al. 1999).

7.1.2.2.1 Vegetation and primary productivity

The basis for nearly all life in the sea is the photosynthetic activity of aquatic autotrophs such as algae, cyanobacteria, benthic microalgae, benthic macroalgae (kelps and seaweeds), and seed plants (such as seagrasses, mangroves, and salt-marsh plants) (Nybakken and Bertness 2005). These photosynthesizers rely on the availability of light for photosynthesis (Govindjee 1975). Marine aquatic vegetation forms an important component of the base of the aquatic (and terrestrial) food web (Seliskar and Gallagher 1983).

Vegetation growth, survival, and depth of water column penetration are directly related to light availability, with the maximum depth of plant survival increasing with increasing light penetration into the water column (Dennison 1987; Dennison et al. 1993; Kenworthy and Haunert 1991). The level of light penetration is dependent upon water depth, water clarity (dissolved particulates reflect, refract, absorb, and scatter incident radiation), and light absorption by plant material in the water column.

Because of its importance to habitat structure and food webs, eelgrass is one type of aquatic vegetation that has received a lot of attention in the Pacific Northwest. Eelgrass is considered a "saltwater habitats of special concern" (WAC 220-110-250).

Phillips (1984) and Wyllie-Echeverria and Phillips (1994) describe eelgrass ecology in the Pacific Northwest. Two species of eelgrass (*Zostera* spp.) grow in Washington State: the native eelgrass, *Zostera marina*, and the smaller Asian species, *Zostera japonica* (Wyllie-Echeverria and Phillips 1994). Typically, *Z. marina* grows at lower elevations than *Z. japonica* and may either form extensive beds covering many acres or exist in smaller patches (Phillips 1984). Native eelgrass distributions range from approximately +2 feet mean lower low water (MLLW) to -22 feet MLLW (PSAT 2001), although light penetration conditions in many portions of Puget Sound typically limit the lower elevation to the range of -8 feet to -12 feet MLLW. *Z. japonica* is generally found at higher elevations than *Z. marina* and typically grows in patches or a narrow fringe (Phillips 1984). Eelgrass typically grows in sand and mud substrates in sheltered or turbulent waters (Phillips 1984).

Many species of macroalgae also grow in the marine waters of Washington, generally attached to rocky substrates and always within the nearshore photic zone (Kozloff 1983). Macroalgae have a wider elevation range than eelgrass, and macroalgae such as rockweed (*Fucus gardneri*) can grow as high as mean higher high water (MHHW). At the other extreme, brown algae (kelp) may grow at elevations as low as -100 feet MLLW where the water is clear enough and the substrate supports algal attachment (WDNR 2004). However, in Puget Sound, the depth to which sufficient light penetrates to support plant growth (i.e., photic zone) is considered to be -33 feet (-10 m) MLLW (PSNERP 2003).

In the estuarine salt marshes of Washington, bulrushes (*Scirpus* spp.) and sedges (*Carex* spp.) are the most commonly found vegetation types (Kozloff 1993). Other common vegetation types include rushes (*Juncus* spp.), salt grass (*Distichlis spicata*), dune grass (*Leymus mollis*), and pickleweed (*Salicornia virginica*) (Seliskar and Gallagher 1983).

Eelgrass can retard current velocity at the sediment-water interface, allowing fine particulates to settle (Phillips 1984). This action can affect sediment dynamics and local sediment characteristics, favoring continued growth and survival of eelgrass (Phillips 1984). The vertical structure of kelp forests also affords some dissipation of wave energy (Jackson 1984), which can offer some shoreline protection for other sensitive shoreline habitats.

Simenstad et al. (1999) describe the potential effects of propeller wash on eelgrass. Flume studies have shown that current velocities of 1 to 1.5 knots (50 to 80 centimeters per second [cm/sec]) may be sufficient to cause sediment disturbance around eelgrass and that velocities of 3.5 knots (180 cm/sec) can cause severe erosion of eelgrass patch edges. However, eelgrass patches in Puget Sound thrive in currents of up to 3.9 knots (200 cm/sec) (Thom et al. 1996, in Nightingale and Simenstad 2001b). The effect of vessels used during installation of HPA-permitted structures on eelgrass and macroalgae depends on local current and sediment conditions, as well as on maximum current velocity at the sediment surface. In addition to the direct effects of propeller wash on submerged vegetation, propeller wash can entrain bubbles and suspend sediment, causing reduced light availability that can indirectly affect eelgrass and, to a lesser extent, macroalgae (Simenstad et al. 1999).

7.1.2.2.2 Food web interactions

Marine littoral vegetation is important for the colonization of organisms that are important prey resources for potentially covered species such as Newcomb's littorine snail, Pacific sand lance, Pacific herring, Pacific cod, northern abalone, surf smelt, steelhead and coastal cutthroat trout, salmon (pink, chum, coho, and Chinook), Olympia oyster, bull trout, Dolly Varden, rockfish, longfin smelt, eulachon; and walleye pollock (Busby et al. 1996; Chambers et al. 1999; Couch and Hassler 1989; Gardner 1981; Goetz et al. 2004; Johnson et al. 1999; Larsen et al. 1995; Matthews 1987; Myers et al. 1998; Norris 1991; NRS Canada 2004; Orth et al. 1984; Pauley et al. 1988; WDNR 2006a, 2006b; West 1997).

Eelgrass, macroalgae, and saltmarsh plants are very productive and support marine food webs through their plant biomass and detritus. (Phillips 1984; Seliskar and Gallagher 1983; Simenstad 1983). Eelgrass provides a necessary structural surface for a community of epibenthic organisms, making eelgrass communities one of the most productive ecotones in the Pacific Northwest (Ferraro and Cole 2007). These epibenthic prey assemblages of copepods, such as the harpacticoids, are known to feed on bacteria, epiphytes, plant detritus, and diatoms. They in turn are prey for juvenile salmon, shiner perch, and other species (Cordell 1986; Nightingale and Simenstad 2001b; Simenstad and Salo 1980; Simenstad et al. 1980, 1988; Thom et al. 1988 (1989?)).

Studies of eelgrass communities in Padilla Bay show that a specific group of copepods (*Harpacticus uniremis* and other copepods of the genera *Zaus* and *Tisbe*) is unique to the eelgrass epiphyte assemblage and is the principal prey of juvenile chum salmon, Pacific herring, Pacific sand lance, and surf smelt (Nightingale and Simenstad 2001b), with *Harpacticus* spp. less likely to be found in low light conditions and *Tisbe* spp. found in areas high in detritus, irrespective of light levels.

Pacific herring spawn on the blades of eelgrass and macroalgae (WDNR 2006a). Pacific herring are, in turn, a direct food source of larger predators, including adult Chinook salmon, bull trout (Nightingale and Simenstad 2001b), Pacific hake (Bailey 1982; NMFS 1990; Quirollo 1992; McFarlane and Beamish 1986, in NRC 2001), Pacific lamprey,

rockfish (WDNR 2006a), and many other species (WDNR 2006a). Thus, a reduction in Pacific herring productivity could produce indirect adverse impacts on a number of additional potentially covered fish species.

In studies on outmigrating juvenile chum in Hood Canal, Simenstad and Salo (1980) found juvenile chum fry (1.2–1.8 in [30–45 mm]) feeding extensively upon small, densely distributed harpacticoid copepods, selecting the largest copepods available. Similarly, Miller et al. (1976) reported that juvenile chum fed predominantly on epibenthic harpacticoid copepods. As the fish grew in size, their diet content was composed of larger epibenthos and pelagic crustaceans. Consistent with other studies, the highest densities of harpacticoid copepods occurred in magnitudes 4–5 times higher in eelgrass stands than in sand habitat without eelgrass. Also, recent dietary investigations in central Puget Sound found that juvenile Chinook salmon fed extensively on a polychaete worm (*Platynereis bincanaliculata*) that builds tubes on eelgrass and macroalgae (Brennan et al. 2004).

In a study of the Drayton Harbor marina, Thom et al. (1988) reported that during the study period from September 1987 to October 1988, juvenile salmon density was by far the highest on April 29 at the eelgrass habitat site that was also found to support, by far, the highest salmon prey density and the highest epibenthos density on that date. Total fish density increased dramatically immediately following a peak in maximum epibenthos and the most rapid increase in Zostera biomass (Thom et al. 1988 *(1989?)*). The limitation of habitat for key prey resources likely affects migration patterns and the survival of many juvenile fish species. For smaller fish less than 1.97 in (50 mm) in length, residence times along particular shorelines are thought to be a function of prey abundance (Simenstad and Salo 1980).

The complex structure of eelgrass communities and their associated epifauna and epiflora are thought to limit the success of predators that typically associate and feed in unvegetated communities (Heck and Orth 1980, in Nightingale and Simenstad 2001b; Heck and Thoman 1984). Given the strong association of important fish prey resources with eelgrass, reductions in eelgrass extent or vigor may reduce prey resources for fish.

Northern abalone could be impacted by alterations to marine aquatic vegetation because of its typical association with kelp beds (Gardner 1981). Northern abalone typically cling to rocks in thick kelp beds (Pacific Biodiversity Institute 2006). Larger northern abalone feed on detached, drifting algae and their growth rate can be influenced by the amount of algae available (Jamieson 1999).

Juvenile Dungeness crab, an important salmonid prey species, show a preference for eelgrass compared to other benthic habitats; this is thought to be due in part to the abundance of food items in eelgrass habitat (Pauley et al. 1989).

7.1.2.2.3 Predator avoidance and refugia

Salt marshes provide important feeding opportunities and predator refuge for fishes. For example, juvenile chum and fall Chinook salmon have been observed to selectively forage for chironomid larvae and adults in a restored marsh, suggesting that restored salt marsh provides indirect benefits to fisheries by production of preferred prey items through detritus-based food chains (Shreffler et al. 1992). Seliskar and Gallagher (1983) identified seven of the potentially covered species as associated with marsh habitats: Chinook, chum, coho, sockeye, and pink salmon, longfin smelt, and surf smelt. In the Fraser River estuary, most of the fish populations are dependent upon a small array of benthic invertebrates, many of which are tidal marsh inhabitants (Northcote et al. 1979). Salt marshes are highly sensitive to human disturbance (Seliskar and Gallagher 1983). The destruction of salt marshes can significantly impact the refuge habitat available for fish and the important functions that salt marshes provide at the base of the food web. Levy and Northcote (1981) concluded that juvenile salmon use the entire length of tidal channels and, therefore, that bank protection structures along any part of a marsh area can significantly reduce the estuary's capacity as a rearing area.

Aquatic vegetation, in particular eelgrass, is important cover for juvenile fish and invertebrates (Phillips 1984). Bostrom and Mattila (1999) note that eelgrass plays a role in protecting invertebrates from both fish and avian predators. It is uncertain what role eelgrass plays in the protection of potentially covered invertebrate species, but the generality of the existing work in the field would suggest that a loss of eelgrass would result in increased predation of those species that co-occur with eelgrass.

7.1.2.2.4 Spawning substrate

Both eelgrass and macroalgae provide substrate for herring spawning (Bargmann 1998). Herring is a key species in the nutrient and energy dynamics of the Puget Sound environment, providing an important link between zooplankton and larger predators, including Chinook salmon, bull trout, and other salmonid species (Bargmann 1998).

7.1.2.2.5 Habitat structure

Eelgrass and macroalgae provide vertical structure in nearshore marine habitats and facilitate several important ecological functions. Nightingale and Simenstad (2001b) note that eelgrass provides shelter and influences the physical and chemical properties of the nearshore environment. The vertical structure off the seafloor (substrate) that marine aquatic vegetation provides is important habitat for fish and invertebrates, including salmon, forage fish, and juvenile rockfish (Phillips 1984). The vertical structure of marine aquatic vegetation can also trap and stabilize sediments, and vegetation that grows through the entire water column, such as bull kelp, can dissipate wave energy before it hits the shoreline (Jackson 1984.)

Blackmon et al. (2006) provides a synopsis of research on the use of seagrass and kelp habitats by fish, including many of the marine potentially covered species. This synopsis noted that forage fish and juvenile Pacific salmon species preferentially use eelgrass over other habitats. Juvenile salmon are found in kelp habitat as well (Blackmon et al. 2006). Rockfish (*Sebastes* sp.) produce planktonic larvae that settle in eelgrass, shallow kelp

beds, and floating kelp mats (Blackmon et al. 2006). Juvenile rockfish occupy shallow vegetated habitats, especially areas with eelgrass and kelp, during the summer growing period (Byerly et al. [no date]; Murphy et al. 2000), likely due to the enhanced forage opportunities and refuge from predators that the vertical structure can provide. Juvenile Dungeness crab (young of the year) (a major prey species for some rearing salmonids) are more frequently found in eelgrass and *Ulva* beds than in other benthic habitats, and eelgrass beds are considered valuable nursery habitat for Dungeness crab. This is thought to be due in part to the abundance of food items in eelgrass habitat (Pauley et al. 1989).

Newcomb's littorine snail is found primarily in association with a narrow band of nearshore intertidal habitat that contains certain marsh plant species (Larsen et al. 1995). Because detailed reproductive and habitat needs of Newcomb's littorine snail are not known, it might be conservatively assumed that Newcomb's littorine snail is subject to habitat loss if marine aquatic vegetation, particularly marsh plant species, is displaced by bank protection structures.

7.1.2.2.6 Large Woody Debris and Aquatic Vegetation

There have been few studies of the influence of LWD in marine environments. Depositional coastal areas throughout Washington are replete with large and small wood. Much of this wood has originated from forest harvesting practices from the turn-of-the-century. Therefore, it is difficult to determine a "natural" concentration of LWD for coastal areas. It is generally thought that small quantities of LWD can increase nearshore habitat quality; however, no significant research has been conducted to verify this hypothesis. Maclennan (2005) studied wood dynamics in two sites in Washington: Elger Bay on Camano Island, and Sullivan-Minor marsh in Padilla Bay. She noted that drift logs, or LWD, tend to accumulate in the upper portions of the intertidal zone and that the wracked pieces in the seaward sections of the estuaries became mobilized during storms and high tides. The mobile logs raked the surface of the marsh and prevented the establishment of emergent vegetation. Meanwhile, stable logs in the higher elevation sections of the estuaries acted as substrate for upland vegetation which could not have otherwise become established (Maclennan 2005). Other studies have shown that these stable logs can act as habitat for macroinvertebrates including grass shrimp (Everett and Ruiz 1993), chironomids, other dipterans, talitrids, homopterans, coleopterans, and collembolans and that beaches with abundant wrack have higher taxa richness than equivalent artificially hardened beaches (Sobocinski 2003). Outside of these few studies, little research has measured the ecological importance of LWD for the formation and persistence of marine aquatic vegetation communities.

7.1.2.2.7 Water Quality

In tidal areas, seagrasses have been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve.

Increased turbidity is known to compromise the survivability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) (Bash et al. 2001; Newcombe and Jensen 1996) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. It can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). Eelgrass is associated with important rearing habitats for a suite of marine fishes including Pacific cod, Pacific salmon, rockfish, Pacific herring, and walleye pollock (Nightingale and Simenstad 2001a; Simenstad et al. 1999). Increased turbidity can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted an approximate 50 percent decline in the number of seagrass species and a precipitous decline in seagrass biomass with a 15 percent increase in the clay content of the sediments.

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

7.1.2.3   Ecosystem-Specific Effects: Riverine

7.1.2.3.1 Vegetation and primary productivity

Riverine aquatic vegetation includes benthic algae (microscopic unicellular algae, forming thin layers or assemblages called periphyton). Macrophytes include angiosperms rooted in the stream bottom, along with mosses, and other bryophytes. These include many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998).

Although aquatic primary production is sometimes underrated due to the small amount of algae and plants present in many streams, it is a basic energy source for freshwater ecosystems. A small algal biomass in a stream can support a much larger biomass of consumers due to the rapid turnover in biomass (Hershey and Lamberti 1992; McIntire 1973; Murphy 1998).

Light is a controlling factor of primary production, with increased light and nutrients stimulating primary production and increasing the production of invertebrates and fish, or decreased light reducing overall productivity (Murphy 1998). Although aquatic plants and algae are adapted to low light intensity, there is a critical light level at which respiration equals photosynthesis, known as the compensation point. Below the compensation point, such plants would eventually starve to death as they would respire food faster than they could produce it (Murphy 1998).

In a study of macrophyte impact on sediment and nutrient retention in Danish streams, Sand-Jensen (1998) reported that dense-stemmed macrophytes created conditions conducive to sediment deposition and that the sediments retained within the macrophyte stands were fine-grained and nutrient-rich. He noted that enrichment of sediment within macrophyte beds relative to the surrounding substratum was 0.1597 lb organic matter per ft^2 (780 g /m^2), 0.006 lb nitrogen per ft^2 (30 g/m^2) and 0.005 lb phosphorus per ft^2 (25 g per m^2). Therefore, any large-scale modification of aquatic vegetation will likely result in increased suspended sediments, increased nutrient loading, and changes in hyporheic exchange, all adversely affecting HCP species.

Freshwater macrophytes are also known to modify their physicochemical environment by slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic content and capturing smaller grain size than substrate in uncolonized areas (Chambers et al. 1999). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange (White 1990), which in turn will promote nutrient cycling.

7.1.2.3.2 Food web interactions

Potentially covered species that depend on freshwater aquatic vegetation for one or more of their life-history stages include green sturgeon, white sturgeon, California floater and western ridged mussels, mountain sucker, lake chub, great Columbia River limpet, pygmy whitefish, leopard and Umatilla dace, Olympic mudminnow, bull trout, Dolly Varden, and Pacific salmon (Frest and Johannes 1995; Hallock and Mongillo 1998, 1999; Hughes and Peden 1989; Mongillo and Hallock 1998; Watters 1999). More specifically, adhesive eggs of the Olympic mudminnow rely on attachment to aquatic vegetation for egg and larval development (Coutant 2004).

Freshwater aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of stream ecosystems. The uptake of carbon, nitrogen, and phosphorous by aquatic vegetation and conversion into biologically available biomass provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous (instream) organic matter and part of the source of allochthonous (terrestrial) matter in each stream reach.

A reduction in aquatic vegetation may alter nutrient loading within stream and river ecosystems. Modification or removal of aquatic vegetation will result in reduced autochthonous production, which provides important energy sources in aquatic food webs. Primary productivity in fresh water is reduced commensurate with the degree that shade reduces the light level of the aquatic environment.

Invertebrate grazing of vegetation by snails, caddisflies, isopods, minnows, and other organisms is an important pathway of energy flow. For stream herbivores, for example, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of

invertebrate scrapers (Gregory 1983; Hawkins and Sedell 1981), an important food source for fish.

Although terrestrial and adult aquatic insects are important (Bjornn and Reiser 1991), juvenile salmon in streams have been found to be primarily supported by autochthonous organic matter (Bilby and Bisson 1992). Invertebrate scrapers and collector–gatherers are known to be most frequently eaten by salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991).

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

7.1.2.3.3 Predator avoidance and refugia

Aquatic vegetation has been found to reduce predation rates by providing cover refuge for prey fish (Gregory and Levings 1996).

The effects of aquatic vegetation removal on invertebrates are not well known. However, the California floater in the Eel River (California) is commonly associated with aquatic vegetation, which is used for protection from high flows (Howard and Cuffey 2003).

7.1.2.3.4 Habitat structure

Activity that mechanically removes or by other means affects aquatic vegetation may reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from the removal of aquatic vegetation may cause increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity.

In particular, the Olympic mudminnow may be most vulnerable to changes in aquatic vegetation because it requires areas with dense aquatic vegetation (Harris 1974) and has been shown to no longer occupy areas where vegetation was removed (Mongillo and Hallock 1999).

7.1.2.4   Ecosystem-Specific Effects: Lacustrine

7.1.2.4.1 Vegetation and primary productivity

Lacustrine aquatic vegetation includes algae and plants. Lakes host both floating and benthic algae (microscopic unicellular algae, forming thin layers or assemblages called periphyton). Lacustrine macrophytes include both angiosperms and bryophytes, in many

forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998). Large woody debris acts as a substrate for periphyton growth and secondary production in lakes (Smokorowski et al. 2006.)

Impacts from shading aquatic vegetation in lakes are similar to those found in marine, estuarine, and riverine systems (Garrison et al. 2005; Jennings et al. 2003; White 1975).

The introduction of noxious weeds can be a concern in aquatic environments (Chambers et al. 1999; WNWCB 2006). In Washington, concern about the impacts of aquatic noxious weeds seems to be higher for lacustrine ecosystems than for either riverine or marine systems. Under the right conditions, noxious weeds can out-compete native vegetation and can reduce habitat quality for native fish species (Chambers et al. 1999).

For example, the Lake Washington shorelines have developed extensive beds of Eurasian milfoil since it was first observed in the lake in 1974 (WNWCB 2005). Eurasian milfoil can cause several adverse habitat conditions, including reduced dissolved oxygen and reduced access to habitat (Chambers et al. 1999). Interlake transfer from boats is thought to be the chief means by which Eurasian milfoil is spread (WNWCB 2005).

The role of emergent vegetation in the lacustrine nearshore on the productivity of habitat has not been explored in detail in Washington waters or with regard to HCP species.

7.1.2.4.2 Food web interactions

Potentially covered species that depend on freshwater aquatic vegetation for one or more of their life-history stages include white sturgeon, lake chub, pygmy whitefish, bull trout, and Chinook and sockeye salmon (Frest and Johannes 1995; Hughes and Peden 1989; Mongillo and Hallock 1998; Mongillo and Hallock 1999; Watters 1999).

Aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of freshwater ecosystems. The uptake of carbon, nitrogen, and phosphorous by these plants provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous organic matter and part of the source of allochthonous matter within all lakes. Grazing of these primary producers by snails, caddisflies, isopods, minnows, and other grazers is an important pathway of energy flow. For herbivores, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of invertebrate scrapers (Hawkins and Sedell 1981), an important food source for fish. Although no studies were found that have linked the productivity of invertebrate communities with an increase in fish numbers, these observations may explain the differences in fish density along disturbed lakeshores (Jennings et al. 1999).

Juvenile salmonids, as drift-feeders, focus on food from autochthonous pathways. Invertebrate scrapers and collector-gatherers are known to be most frequently eaten by

salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991). Although no studies were found that have linked the productivity of invertebrate communities with an increase in fish numbers, these observations may explain the differences in fish density along disturbed lakeshores (Jennings et al. 1999).

Freshwater macrophytes are known to modify their physiochemical environment and contribute to habitat complexity by changing surface water patterns, slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic content and capturing smaller grain size than substrate in uncolonized areas (Fonseca and Bell 1998, Carrasquero 2001, Chambers et al. 1999). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange, which in turn will promote nutrient cycling. For example, White (1990) found that dense vegetation hummocks promote upwelling of porewater into the rootmass, which provides nutrients that encourage and sustain vegetation growth. In these ways, aquatic vegetation can contribute to habitat complexity and food web productivity.

7.1.2.4.3 Predator avoidance and refugia

As with marine aquatic vegetation, lacustrine aquatic vegetation has been shown to dramatically enhance the density of benthic invertebrates in temperate lakes through the relative protection from predation it affords (Beckett et al. 1992). In fact, even invasive vegetation can provide the same functions without compromising invertebrate productivity (Gardner et al. 2001).

Reductions in lacustrine aquatic vegetation may affect potentially covered species by loss of protection from predators due to a loss of cover.

7.1.2.4.4 Habitat structure

As in riverine systems, activity that mechanically removes or by other means affects aquatic vegetation may reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from the removal of aquatic vegetation may cause increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity.

One potentially covered freshwater invertebrate species would be impacted by loss of habitat if freshwater aquatic vegetation were disturbed or removed; the larvae of the California floater mussel in Curlew Lake depend primarily on the Tui chub (*Gila bicolor*) as a host (Pacific Biodiversity Institute 2006), and juvenile Tui chub typically stay close to vegetation until they are longer than 0.5 inch (Wydoski and Whitney 2003).

7.1.3   Activity-Specific Effects

7.1.3.1   Bank Protection

Bank protection structures in marine waters have the potential to affect marine aquatic vegetation through direct or indirect disturbance and displacement. Saltmarsh vegetation growing along the upper intertidal shoreline fringe, the backshore, or in larger saltmarsh complexes is highly susceptible to disturbance through the potential hydraulic disconnection, burial, or conversion of habitat associated with bank protection structures. Marine aquatic vegetation may be uprooted or displaced as the vegetation itself may be removed for projects constructed below the ordinary high water line (OHWL). During construction, vegetation may be trampled or subject to spills from construction equipment in the project area or work corridor. Vegetation may also be disturbed as a result of vessel grounding or propeller wash (Lagler et al. 1950, in Carrasquero 2001; Haas et al. 2002), which can entrain air bubbles and introduce sediment suspension (Haas et al 2002). Because light availability is a fundamental requirement for eelgrass and macroalgae growth, turbid conditions limit their ability to thrive.

Bank protection projects can indirectly impact marine aquatic vegetation distributions through the sediment coarsening that occurs as a result of the altered wave regime or disruption of littoral drift (e.g., Johannessen et al. 2005). The sediment coarsening may result in substrates too large to support marine aquatic vegetation.

Bank protection activities in freshwater settings have the potential to affect freshwater aquatic vegetation through direct disturbance or indirect disturbance and displacement. Bank protection structures positioned waterward of the OHWL will encroach upon areas that support or potentially would support freshwater aquatic vegetation.

7.1.3.2   Conduits

A study of eelgrass recovery at four submarine cable landings in Skagit and San Juan counties demonstrated the disturbance that occurs with conduit (electrical transmission cable) installation and some factors affecting recovery (Jones & Stokes 2005; Wones and Cziesla 2004). This project disturbed eelgrass habitat along a cable route that ranged from 3 to 15 feet in width. Although the upper beaches of these cable landings returned to the pre-excavation appearance within a few weeks through wave action and littoral drift, the disturbance through eelgrass beds required one to four years to fully recover. The sites where eelgrass recovered most rapidly included a site with only a narrow band of eelgrass and a site with an extensive, robust eelgrass bed. Eelgrass was slowest to recover at the deepest extent of eelgrass growth, where light availability may limit the rate of eelgrass recruitment and growth (Jones & Stokes 2002, 2005).

7.1.3.3   Culverts

Impacts to habitats and species may occur through the loss of vegetation resulting from construction of new water crossings.

Maintenance or retrofitting of culverts is not expected to have any appreciable effect on aquatic vegetation. This is due to the fact that the footprint of the existing structure has

already imposed its effects on the vegetation community. Therefore, most work within the footprint of an existing structure will not displace or affect vegetation. Some extended effects are possible, however, if removal or replacement results in hydraulic and geomorphic modifications that change habitat suitability.

7.1.3.4   Fishways

The potential for the fishway subactivity type projects to result in aquatic vegetation modification is generally more limited in comparison to other types of HPA-permitted activity types. This is due to the fact that the in-water footprint of fishway structures is typically small and, in the case of passage around man-made barriers, the fishway is integrated into the barrier structure, and the incremental effect on aquatic vegetation is negligible. Some extended effects on aquatic vegetation are possible, however, if the fishway design results in hydraulic and geomorphic modifications that change habitat suitability for vegetation. For example, exit flows from the fishway may lead to localized alteration of substrate conditions. In general, however, any effects associated with aquatic vegetation modifications are expected to be limited in extent and insignificant in terms of stressors imposed on the aquatic community.

7.1.3.5   Roughened Channels

It is useful to note that roughened channels are often implemented to aid passage of fishes in high-velocity channels. This type of environment is less than ideal for aquatic vegetation and typically does not support extensive aquatic vegetation communities. Therefore, the direct effects of construction are expected to be limited overall. In general, effects associated with aquatic vegetation modifications are expected to be limited in extent and insignificant in terms of stressors imposed on the aquatic community.

The in-water footprint of roughened channels is typically small or nonexistent (in the case of channels constructed through uplands). Some extended effects on aquatic vegetation are possible, however, if the roughened channel design results in hydraulic and geomorphic modifications that change habitat suitability for vegetation immediately upstream or downstream. For example, exit flows from the roughened channel may lead to localized alteration of substrate conditions. Construction-related water quality impacts in the form of elevated suspended sediments may also occur.

7.1.3.6   Overwater Structures

7.1.3.6.1 Sediment disturbance associated with overwater structures

Vessels associated with overwater structures may cause propeller wash, which may disturb, damage, or uproot aquatic vegetation. The effect of vessels associated with overwater structures on eelgrass and macroalgae depends on local current and sediment conditions, as well as on maximum current velocity at the sediment surface. Flume studies have shown that current velocities of 20 to 31 inches per second (50 to 80 centimeters per second [cm/sec]) may be sufficient to cause sediment disturbance around eelgrass and that velocities of 71 inches per second (180 cm/sec) can cause severe erosion

of eelgrass patch edges. However, eelgrass patches in Puget Sound thrive in currents of up to 79 inches per second (200 cm/sec) (Thom et al. 1996, in Nightingale and Simenstad 2001b).

Thom et al. (1996), in studying the impacts of passenger-only ferries at the Vashon Island terminal, found that at 187 feet (57 meters) from the boat, it is likely that the propeller wash has little effect on existing eelgrass. Thom et al. (1996) also concluded that currents with a velocity above 2.46 feet/second (0.75 meters/second) damaged eelgrass by eroding away overlying sediment and that currents above 3.61 feet/second (1.1 meters/second) caused extensive damage to eelgrass rhizomes.

The vertical and horizontal distance at which current velocity may affect eelgrass depends on the size and shape of the propeller. The U.S. Army Corps of Engineers' Regional General Permit No. 6 prohibits the construction or installation of floats or float support pilings within a 4-foot depth elevation between the top of the float stopper and the elevation of the landward-most edge of a macroalgae bed or eelgrass (USACE 2005). This restriction applies to a zone 25 feet wide on both sides of the float projecting waterward horizontally from the float (USACE 2005).

In a study of 44 shallow, sheltered soft-bottom inlets on the Baltic Sea, Eriksson et al. (2004) found that recreational boating activities and ferry boat traffic significantly affected aquatic vegetation both in percent cover and species richness. These effects were largely attributed to water movement and prop wash generated by boating activity, and turbidity blocking light transmission to vegetation. These findings are consistent with numerous studies on the effects of disturbance on eelgrass beds (Sargent et al. 1995; Loflin 1995; Haas et al. 2002).

Studies in Florida related to the impacts of boating activity on seagrass indicate that the largest concentration of scarring occurs in waters less than 6.5 feet (2 meters) deep (Sargent et al. 1995, in Dawes et al. 2004). In Florida, many shallow flats and mud banks are severely eroded due to constant scarring, ship groundings, chronic wave action from boats, and water-current scouring (Kruer 1994, in Dawes et al. 2004). Removal of seagrass roots and rhizomes due to prop scarring also destabilizes sediments and resuspension occurs, thereby lowering water transparency and retarding seagrass regrowth into the scar (Durako et al. 1992, in Dawes et al. 2004).

Studies in Florida have also found that fragmentation of seagrass beds caused by propeller scarring did not appear to have any consistent effects on some animal populations over a one-year period, as long as the seagrass patch sizes were greater than 3 square feet (1 square meter) (Bell et al. 2002, in Dawes et al. 2004). The numbers of pinfish (*L. rhomboides*), pipefish (*Syngnathus scovelli*), and eight species of epibenthic shrimp were similar in moderately scarred (6 percent to 31 percent loss of the beds) and non-scarred seagrass beds in Tampa Bay (Dawes et al. 2004). The results of these studies suggest that propeller scars that fragment seagrass beds may enhance certain faunal development caused by edge effects along the cuts, as long as they are not too severe (Dawes et al. 2004). Nevertheless, a recent study of scarring in a *T. testudinum* bed in

Puerto Rico revealed a negative effect of scarring on crabs and molluscs up to 16 feet (5 m) from the scar. Also, shrimp species within the scar differed from those in the non-scarred seagrasses. Fish populations did not show an effect from the scarring (Dawes et al. 2004). Further studies are clearly needed to define the effects of moderate scarring compared to those of severe scarring on seagrass productivity (Dawes et al. 2004).

In addition to direct disturbance, propeller wash can entrain bubbles and suspend sediment, causing reduced light availability that can indirectly affect eelgrass and, to a lesser extent, macroalgae (Simenstad et al. 1999). Propeller wash has also been found to resuspend nutrients and contaminants, such as nitrogen and phosphorous, that can stimulate algal blooms as well as increase turbidity (Haas et al. 2002; Michelsen et al. 1999; Parametrix 1996; Thom et al. 1997; Thom and Shreffler 1996). The resulting increased algal growth may lead to eutrophication and reduction of dissolved oxygen levels due to respiration during desiccation of the algal material. The result of increased turbidity and lower dissolved oxygen has effects throughout the food web.

7.1.3.6.2 Effects on light availability associated with overwater structures, bridges and culverts

Many types of HPA-permitted structures, including overwater structures, marinas, terminals, bridges, and culverts cast shade and thus may potentially affect aquatic vegetation.

Where shading reduces photosynthetically active radiation (PAR) levels, eelgrass and macroalgae growth will be affected and may be impaired or prevented (Nightingale and Simenstad 2001b; Penttila and Doty 1990). Penttila and Doty (1990) found that fixed and floating docks and structures largely eliminate existing eelgrass and macroalgae, even when the structures are only partially shading. Such shading impacts to eelgrass can be seen to occur in as little as 18 days (Backman and Barilotti 1976, in Nightingale and Simenstad 2001b), although light reduction capacity varies depending on combinations of both structure design and environmental factors. For example, Penttila and Doty (1990) found no apparent eelgrass loss due to shading under a floating dock secured by anchors and chains. In that case, it was thought that, given the winds and current of the site, the degree of movement allowed by the anchor-chain system resulted in no area beneath the dock being continuously shaded, thereby reducing the effect of shade on the eelgrass bed.

Thom et al. (1998) analyzed the photosynthetically active radiation (PAR) levels at seven Washington State ferry terminal sites. and found that no eelgrass was found where instantaneous mid-day PAR levels were less than about 100 micro-moles of photons within the PAR range of wavelengths striking a square meter in one second (μM/m2/sec). They also found that the lowest eelgrass shoot densities were found where instantaneous mid-day PAR was less than 150 μM/m2/sec. Thom et al. (1998) found that maximum shoot densities required instantaneous PAR of 325 μM/m2/sec. PAR intensities less than about 300 μM/m2/sec can be limiting to eelgrass, whereas intertidal macroalgae may be limited by PAR less than 400 to 600 μM/m2/sec (Thom and Schreffler 1996, cited in Simenstad et al. 1999). Subtidal macroalgae can survive lower light levels and may only

be limited by PAR less than 100 μM/m2/sec (Luning 1981, cited in Simenstad et al. 1999). Light availability observed at multiple terminals in Puget Sound ranges from 0 to almost 9 PAR units under the Kingston terminal, with Port Townsend, Clinton, and Vashon terminals rangiingfrom 0.5 to1 PAR, and light availability varying significantly at different points under the terminal (Simenstad et al.. 1999)

Many other studies have focused on light limitation effects under ferry terminals in Washington State (Backman and Barilotti 1976; Blanton et al. 2001; Bulthuis and Woelkerling 1983; Burdick and Short 1999; Fresh et al. 1995; Glasby 1999; Haas et al. 2002; Loflin 1995; Olson et al. 1997; Parametrix 1996; Penttila and Doty 1990; Reish 1961; Shafer 1999, 2002; Shreffler and Moursund 1999; Simenstad et al. 1988; Thom et al. 1996, 1997; Thom and Shreffler 1996; Visconty 1997). Some of these studies focused on prey resource availability for juvenile salmon at ferry and shipping terminals in Puget Sound and found that these structures negatively affect prey availability (Blanton et al. 2001; Haas et al. 2002). In a study comparing light levels under the Clinton, Bainbridge, and Southworth terminals, Blanton et al. (2001) found terminal orientation to the arc of the sun, terminal height and width, construction materials, and piling type to influence the shadow cast on the nearshore environment and its effect on the littoral vegetation. Similarly, in a study comparing the Clinton, Edmonds, and Port Townsend terminals, Thom and Shreffler (1996) found similar light limitation effects on littoral vegetation, and Haas et al. (2002) found effects of underwater light limitation under the Bainbridge, Clinton, and Southworth ferry terminals.

In a study on the influence of piers and bulkheads on the aquatic organisms in Lake Washington, White (1975) reported that light levels under piers were consistently lower, and this light reduction resulted in reduced phytoplankton production. In general, the larger the overwater structure, the larger the area of light limitation and reduction of phytoplankton production. Also, macrophytes were generally absent or sparse under piers. In the fall, grazing invertebrates were found outside of piers where macrophytes were abundant; in the spring, grazing invertebrates were found under piers where they could graze on periphyton during the spring (White 1975).

Studies conducted at piers on lakes in Wisconsin also report the loss of submerged lake vegetation due to dock shading (Garrison et al. 2005; Jennings et al. 2003). A dock study in Montauk, New York (Ludwig et al. 1997) reported the exclusion of eelgrass near a floating pier due to insufficient light in the float's impact zone. Burdick and Short (1999) found that floating docks severely impact eelgrass. Three of the four floating docks they studied had no rooted eelgrass under them. Increased dock height above the bottom was identified to be the most important dock characteristic correlating to eelgrass bed quality, and a similar effect would be expected for bridges. Burdick and Short (1999) also found light to be the most important variable affecting canopy structure (i.e., shoot density and height) and eelgrass bed quality.

7.1.3.7   Pilings

Pilings that support overwater structures such as bridges may also reduce eelgrass recruitment and survival through biotic interactions with the piling reef community

(Nightingale and Simenstad 2001b). Pilings in marine waters become encrusted with mussels and other sessile organisms. Shell material from these organisms is then deposited around the pilings over time, altering the local substrate (Nightingale and Simenstad 2001b). The piling reef habitat provides food for sea stars and the shell bottom is prime settling habitat for juvenile Dungeness crabs (Nightingale and Simenstad 2001b). The burrowing activities of large numbers of crabs can affect the establishment of eelgrass (Nightingale and Simenstad 2001b).

7.1.3.8   Dredging

Dredging operations directly entrain and eliminate aquatic vegetation from the dredged site. In both marine and freshwater environments, dredging generally removes or disturbs benthic vegetation. Altering the vegetation also potentially alters the distribution and abundance of prey resources, the availability of refugia, and water quality benefits provided by vegetation to various species. The time period between dredging activities will determine the capacity for recolonization of the submerged aquatic vegetation.

A special case of freshwater dredging is the use of a hydraulic suction dredge for the specific goal of vegetation management in freshwater environments. The suction dredge is typically used to remove invasive aquatic vegetation; however, the technique is not plant-specific and can entrain aquatic vegetation that is beneficial to HCP species.

A potential threat from dredging is the burial of aquatic vegetation in adjacent areas due to increased sedimentation (Wilber and Clarke 2001). Dredging operations broadcast fine sediments over a broad area, the extent of which depends on tidal currents and basin geometry (Hossain et al. 2004). Burial of the submerged aquatic vegetation community in marine environments can lead to decreased primary and secondary productivity, which in turn may affect overall food-web productivity.

Another threat to aquatic vegetation is sedimentation from open-sea or nearshore beach nourishment disposal sites. Although open-sea disposal has been demonstrated to be protective of aquatic vegetation when disposal rates are regulated to produce low rates (less than 1 inch) of sedimentation over the course of the project (Simonini et al. 2005), the extreme sensitivity of eelgrass to burial makes it vulnerable even in areas that are distant from the disposal site (Mills and Fonseca 2003).

Gravel mining and sediment capping may also result in the burial of aquatic vegetation.

7.1.3.9   Channel Creation and Realignment

Aquatic vegetation will initially be destroyed by channel creation and alignment activities. Studies of gravel augmentation show that this vegetation recovers quickly following minor disturbances (Merz et al. 2004). Elevated nutrient levels during the growing season will accelerate primary production and post-project vegetation recovery. Consequently, initial impacts on aquatic vegetation may only be ephemeral and thus the associated impact on HCP species will be minimal. The loss of aquatic vegetation as part

of channel creation and alignment includes altered autochthonous production and altered habitat complexity.

7.1.3.10 Dams

Dams can cause losses of aquatic vegetation by several pathways. Increased velocities can scour algae downstream and damage macrophytes, reducing cover for fish. Second, changes in substrate composition with an increase in fine sediment transport can bury aquatic vegetation. Finally, modification may occur directly from construction and maintenance activities.

Dam removals have shown contrasting results with respect to impacts on aquatic vegetation. In one study, the removal of a dam resulted in increases in aquatic macrophytes leading to increased cover and habitat for fish (Hill et al. 1993). However, dam removal may kill off some vegetation from sediment released during the removal process and subsequently cause abrasion of roots and stems (Wood and Armitage 1997). Dam removals may increase the scour of algae and insects, thereby altering food web interactions and food quality, particularly if algae or leaf accumulations are buried (Doeg and Koehn 1994; Newcombe and MacDonald 1991; Wood and Armitage 1997).

7.1.3.11 Outfalls

Effluent flows from outfalls may cause scour and loss of aquatic vegetation. Increased nutrient loading from outfall effluent may result in an indirect effect on aquatic vegetation. Increased nutrient loading can stimulate primary productivity and lead to decreased dissolved oxygen. Eutrophication can lower dissolved oxygen concentrations and negatively affect HCP species.

7.1.3.12 Beaver Dam Removal

Macrophytes which populate many floodplain ponds are most abundant in the littoral zones (Ahearn et al. 2006). Consequently, when a beaver pond is dewatered much of the aquatic vegetation is left isolated in dry upland areas. The result is a substantial decrease in aquatic vegetation and the standing vegetation stock (Naiman et al. 1988). This, coupled with the loss of shallow water refugia, may represent the single greatest impact on fish and invertebrates associated with beaver extirpation and dam removal.

Beaver dam removal will decrease the amount of shallow water habitat available for macrophyte and algal production. The decrease in autogenic production will cause a shift in the form of the food resources available to primary consumers. Shredders and scrappers may replace grazers, and the changing macroinvertebrate population may have an overall impact on higher trophic level species (Winkelmann et al. 2007). Altered food web complexity may, as a result, affect foraging opportunities for HCP species that feed on aquatic macroinvertebrates in beaver ponds and similar environments during some phase of their life history.

7.1.3.13 Large Woody Debris Placement/Movement/Removal

Large woody debris acts as a substrate for periphyton growth and secondary production in channels (Atilla et al. 2003; Bowen et al. 1998; Hoffmann 2000; Warmke and Hering 2000) and lakes (Smokorowski et al. 2006), but few studies have shown that the addition of this substrate increases primary productivity in the system as a whole [see (Atilla et al. 2003)] for an exception in marine systems). However, LWD can function to induce floodplain and backwater connection with the main channel. By creating more connectivity with productive backwaters, the presence of wood within the channel can increase aquatic productivity and access to aquatic vegetation. Consequently, LWD removal may be associated with a decrease in access to these productive areas.

7.1.3.14 Spawning Substrate Augmentation

Aquatic vegetation will initially be destroyed in areas where gravel is placed. But, research has shown that this vegetation recovers quickly (Merz et al. 2004). Many augmentations occur below dams. Dams with hypolimnetic release points can elevate growing-season nutrient concentrations in downstream reaches (Ahearn et al. 2005). Elevated nutrient levels during the growing season will accelerate primary production and post-project vegetation recovery. Consequently, initial impacts on aquatic vegetation may only be ephemeral and thus the associated impact on HCP species will be minimal.

7.1.3.15 In-Channel/Off-Channel Habitat Creation/Modifications

The creation of shallow water edge habitat (a goal of some in-channel and off-channel habitat modifications) will generally lead to increased autochthonous production. In a study of rehabilitated incised streams in Denmark, Pederson et al. (2006) found that a re-profiling of the stream bank to create shallow water habitat resulted in increased macrophyte densities. The study concluded that shallow and wide banks allowed for increased autogeny and for a larger migration of macrophytic species from the stream banks into the streams, thereby enhancing species diversity within the stream channel. Rehabilitation of off-channel habitat will promote the exchange of aquatic vegetation between the channel and the floodplain. Schemel et al. (2004) noted a 2-fold increase in chlorophyll-a concentrations (a measure of algal biomass) in a floodplain in California versus the adjacent Sacramento River. Meanwhile, Ahearn et al. (2006) noted chlorophyll-a concentrations five times greater in a Cosumnes River, California floodplain than in an adjacent channel. These studies and others (Hein et al. 2004; Tockner et al. 1999) indicate that autochthonous production is elevated in floodplains relative to channels. This biomass is transported between backwater areas and the main channel during floods, and the resultant increase in food resources within the channel can bolster aquatic food webs and benefit the HCP species.

Freshwater aquatic vegetation provides shelter and clinging substrate for a variety of prey of the HCP species, including mollusks and many fishes (nonsalmonids) with a strong association with this vegetation (Petr 2000). Some of the HCP species which have been shown to commonly utilize vegetated habitat in off-channel areas include cutthroat trout, bull trout, sculpins, Dolly Varden, sockeye and coho salmon, and dace species (Pollock et al. 2003). In fluvial systems, slack water productive patches provide habitat diversity in

systems that are otherwise dominated by high velocity, less-productive habitat (Johnston and Naiman 1990). This is also likely the case in estuarine/slough areas. The creation of off-channel habitat will increase access to aquatic vegetation and benefit many of the HCP species.

7.1.3.16 Wetland Creation/Restoration/Enhancement

Riparian and coastal wetlands provide extensive shallow water habitat where macrophyte and algal species may thrive. Due to shallow water depths, warm temperatures, and protection from the erosive power of flooding and wave action, wetlands are ideal habitat for aquatic vegetation. In turn, aquatic vegetation provides food resources and structural habitat for both fish and invertebrate species. Wetland creation, restoration, and enhancement will increase aquatic vegetation habitat and the ecosystem functions associated with it.

The Schemel (2004), Ahearn (2006), Hein et al. (2004), Tockner et al. (1999), Petr (2000), Pollock et al. (2003), and Johnson and Naiman (1990) studies discussed in the section on "In-Channel/Off-Channel Habitat Creation/Modifications" are also pertinent to aquatic vegetation in wetlands. As with off-channel habitat creation, autochthonous production in wetlands can bolster aquatic food webs and benefit many of the HCP species, especially in areas where aquatic productivity within the channel is low. Nutrient-poor, or oligotrophic systems are common in Washington State, especially since a major pathway of nutrient import, namely marine-derived nutrients from salmon spawning, has drastically decreased over the past 100 years (Naiman et al. 2002). Consequently, any habitat modification measure which increases productivity in waters connected to these oligotrophic systems will benefit aquatic species.

Riparian and estuarine wetlands are characterized by abundant macrophyte growth which has been shown to provide habitat for coho (Swales and Levings 1989), marine invertebrates (Seitz et al. 2005), and numerous other species. The creation, restoration, or enhancement of estuarine and riparian wetlands may increase the amount of, and improve access to, this habitat. The potential increase in habitat may alleviate any density dependent mortality which may be occurring within the system (Greene and Beechie 2004).

Organisms which can access floodplain wetland habitat may benefit from the increased productivity which characterizes those systems by taking advantage of improved foraging and hunting opportunities. However, organisms that remain within the channel may also benefit from riparian wetland productivity because the systems tend to be hydraulically linked and carbon export to the channel will occur during periods of high flow.

Wetlands have been called the kidney of the landscape (Mitsch and Gosselink 2000) because of their ability to sequester and transform nutrients and pollutants. If they are designed with a sufficiently high hydraulic residence time, wetlands can retain pollutants and reduce upland pollutant loadings, including sediment, nutrients, and toxic substances, to downstream aquatic resources. This in turn will benefit any HCP species which may

reside within the downstream freshwater or marine systemHickey and Doran (2004) review buffer widths and include information that is also applicable to wetlands.

7.1.3.17 Beach Nourishment

The primary potential impact on aquatic vegetation from beach nourishment is burial. Eelgrass, the dominant seagrass in western Washington waters, is sensitive to large sedimentation rates (Mills and Fonseca 2003). Therefore, if there is eelgrass present near the activity site, it is possible that the activity will cause an aquatic vegetation loss. As eelgrass is a crucial component to the life history of several HCP species (Phillips 1984), these activities would likely limit or reverse gains in fish populations from the addition of loose, mobile foreshore materials.

7.1.3.18 Reef Creation/Restoration/Enhancement

Depending on the depth of placement, artificial reefs could bury or block light to aquatic vegetation. However, this would only occur in the footprint of the proposed reef. Therefore reefs should not be placed in seagrass meadows.

7.1.3.19 Eelgrass and Other Aquatic Vegetation Creation/Restoration/Enhancement

Aquatic vegetation planting is the least often used technique among all habitat modification subactivities. This rehabilitation technique has been applied with mixed results in Puget Sound (Thom et al. 2005), but it is an attractive alternative considering the relative lack of impacts as compared to other nearshore restoration techniques (e.g., beach nourishment).

Eelgrass is the dominant species of macrophyte in Puget Sound, and is also an important member of the nearshore ecosystem elsewhere in Washington marine waters (Phillips 1984). The effects of introducing eelgrass on other aquatic plant communities are unknown. However, it is clear that eelgrass provides important surface for epiphytes which serve as an important food source for juvenile salmonids and forage fish (Phillips 1984). There are numerous synergistic and competitive interactions between eelgrass and other aquatic flora and fauna (Nelson and Lee 2001). These interactions control the relative abundance of each particular species. As a result, when eelgrass in not present, the likelihood of phytoplankton blooms markedly increases (Hiratsuka et al. 2007).

7.1.3.20 Vessel Traffic

Grounding, anchoring, and/or prop wash can cause benthic disturbance and turbidity, eelgrass and macroalgae disturbance, and freshwater aquatic vegetation disturbance. These effects have been well documented (e.g., Thom et al. 1997 and Thom and Shreffler 1996).

Prop wash or waves produced by boats and personal watercraft can cause shoreline erosion (Gatto and Doe 1987; Mason et al. 1993; Hurst and Brebner 1969) and increase suspended sediments and turbidity (Hilton and Phillips 1982; Kennish 2002; Yousef et al. 1980; Yousef 1974). The effect of this increased turbidity may decrease light levels,

which could potentially affect the growth rates of submerged vegetation, upon which most HCP species depend. Turbidity is also known to be associated with fish respiratory injury (Berg and Northcote 1985) and increased sediment deposition on downstream spawning habitat (Hartman et al. 1996).