

FINAL REPORT

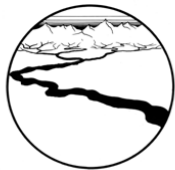
A Summary of the Effects of Bulkheads, Piers, and Other Artificial Structures and Shorezone Development on ESA-listed Salmonids in Lakes

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Introduction

Purpose

This report is a product of a literature review initiated to determine our state-of-knowledge about the impacts of lakeshore development on salmonids, in the context of the Endangered Species Act (ESA). There is much uncertainty surrounding the impacts of various shorezone structures and activities on salmonids within lakes in the Tri-County area (King, Pierce and Snohomish Counties, WA). Identifying the level of current knowledge, areas of uncertainty, and future research needs could provide a more consistent review and evaluation of shoreline development proposals, and decrease potential impacts to threatened salmon populations if the information is utilized in the course of permit review and assimilated into regulations.

This review was initiated by the City of Bellevue, WA, to provide a digest, a library of pertinent literature, and an annotated bibliography in the form of a Microsoft™ Access database, detailing the potential impacts of lakeshore development on ESA-listed species. The review is primarily focused on Lake Washington and Lake Sammamish, but most of the information should be applicable for other lakes within the Tri-County area that have threatened salmonid species. Review products will be utilized by decision-makers at the local through federal levels of government to assist in the creation of guidelines and policies related to lakeshore development. It should be emphasized that, due to time constraints, this “review” is not an exhaustive compendium of all available resources, but rather a measured digest of what was readily accessible in the time allowed. Collection was directed at literature that related directly to shorezone structures and activities in cold, freshwater lakes, but pertinent literature on warmwater, riverine, and marine systems was also included. Despite time constraints, over 350 literature sources were collected and examined, and the salient points from those sources were incorporated into this digest. Primary searches targeted all relevant electronic databases, followed by secondary searches of the references sections of pertinent literature collected during primary searches. Literature collected includes peer-reviewed journal articles, theses/dissertations, books, and technical documents. The literature collected for the review constitutes the majority of available relevant documents, with only the most inaccessible documents omitted. In addition, personal communications with respected local scientists were included where pertinent, current research was not yet published.

Potential Impacts

Shoreline development may seem innocuous to most people. The average property owner on Lakes Washington and Sammamish has no intention of harming salmonids when they propose to build a pier or bulkhead or otherwise modify their shoreline. The property owner’s intention is often to reduce erosion, to develop a tidy shoreline, and/or to improve water access. Thus, it is initially important to define which structures and activities are being analyzed for impacts, and to identify how they may affect ESA-listed species. In Lakes Washington and Sammamish, shoreline development activities and structures that have a federal nexus (i.e., if a federal agency funds, constructs, or

permits the proposed project) and are thus subjected to review under the ESA, include those activities/structures that require a U.S. Army Corps of Engineers (USACE) permit. Specifically, USACE permits are required for the construction, replacement, or repair of piers, docks, boat canopies, boathouses, and shoreline armoring structures (i.e., bulkheads). Adverse impacts on listed species can result from construction or maintenance activities, or from the existence of the structure.

Proposed projects having a federal nexus are analyzed for potential impacts at a variety of levels. Under the ESA, "take," means any potential adverse effects to a listed species that can occur as a direct result of a proposed project, or as an indirect or interrelated result. Direct impacts to a listed species could occur from water quality impacts during construction, or result from increased opportunities for predation from the presence of a new pier. Such a take would be a direct effect. Indirect effects are less obvious. The reduction in water quality that might occur from a fuel spillage during boat fueling at a new residential pier would be an indirect effect. If the pier had not been constructed, the fuel spill would not have occurred.

The following are some specific potential impacts of shorezone structures and activities that have been identified through discussions between The Watershed Company and the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) (collectively called the Services) during 1999 and 2000 as part of the Biological Assessment (BA) process of Section 7 of the ESA. The Services are concerned about potential adverse impacts of shorezone development on juvenile and adult chinook and coho salmon, and bull trout. Despite the Services concern for these ongoing impacts, they can only act on those concerns when an application for a project requiring a federal permit (or with some other federal nexus) is proposed.

- 1) Piers, piles, boatlifts, and moored boats may provide cover, shade, and focal points for exotic predators of juvenile chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*) salmon such as smallmouth bass (*Micropterus dolomieu*) and largemouth bass (*M. salmoides*). Note: native predators such as cutthroat trout (*O. clarki*) and piscivorous birds may also benefit from shorezone structures, but they have yet to be considered in BAs for proposed shorezone development in the Lake Washington system.
- 2) Shading from piers, boat canopies, boathouses, and moored boats may reduce the abundance of prey organisms available to juvenile chinook and coho salmon, and to forage fish of bull trout (*Salvelinus confluentus*) by reducing aquatic vegetation and phytoplankton abundance. Any reduction in aquatic vegetation may also reduce complex refuge habitat. To date, the Services have made no distinction between native and non-native aquatic vegetation.
- 3) The temporary turbidity associated with construction may reduce water quality to the detriment of chinook and coho salmon, bull trout, and forage fish of bull trout.
- 4) Pile driving may disrupt the distribution and behavior of, or injure, chinook and coho salmon, bull trout, and forage fish of bull trout.

- 5) Piers and/or bulkheads may disrupt the migratory and rearing behavior of juvenile chinook and coho salmon.
- 6) The boating activity that accompanies piers could disturb rearing or migrating chinook and coho salmon.
- 7) Chemicals used to preserve or clean wood structures, and hydrocarbons from boats and personal watercraft could be acutely or chronically toxic to chinook and coho salmon, bull trout, or prey items of those species.
- 8) Pier lighting may facilitate nocturnal predation on juvenile chinook and coho salmon by visual predators like smallmouth bass, cutthroat trout, and piscivorous birds.
- 9) The removal of vegetation during bulkhead construction or replacement could eliminate a potential source of cover and food (allochthonous input of terrestrial insects and detritus for foraging aquatic insects) for juvenile chinook and coho salmon, and forage fish of bull trout. Vegetation removed is typically not replaced with native woody species that could provide a future source of woody debris to the lake (see impact # 13, below). Instead, shoreline property owners generally favor lawn or ornamental shrubby species that preserve lake views.
- 10) Bulkheads prevent the recruitment of native sediment to the lake, resulting in a loss of heterogeneous substrate, and resulting in shoreline erosion at the toe or along the shore downwind of the bulkhead. This could affect the availability of spawning and rearing habitat, and the forage base for a variety of fish species.
- 11) Bulkheads eliminate shallow-water habitat, which is critical as refuge and foraging habitat for juvenile salmonids and other small fish.
- 12) Bulkheads reflect wave energy at the shoreline, resulting in the scour of sediment at the bulkhead toe, and creating an inhospitable high-energy environment for juvenile fish.
- 13) The permanent removal of woody debris during bulkhead and/or pier construction reduces the availability of complex refuge habitat for small fish, and attachment surfaces for periphyton.

While the above list of identified potential impacts of shorezone structures and activity on ESA listed species is not exhaustive, it does illustrate the intricacy of the problem. There are potentially many ways that the existing and future development of lakeshores could adversely affect ESA-listed salmonids. A significant shortcoming of the above list is the exclusion of non-ESA-listed salmonids such as sockeye salmon, kokanee (*O. nerka*), and steelhead (*O. mykiss*). While the focus of this report is on salmonid species that are listed or considered for listing under the ESA, shorezone development could have the same potential impacts on many other native fish within Tri-County-area lakes. The listing or proposed listing of chinook and coho salmon, and bull trout under the ESA indicates that these species face serious threats to their perpetuation within the region.

Such threats are likely being faced by other salmonids as well. Thus, in most cases it would be appropriate to consider all salmonids, and perhaps native non-salmonids, as the subjects of the following discussion of impacts.

The potential impacts identified above of docks and bulkheads provided the focus for the review. For discussion purposes, the above list of impacts has been condensed into six categories: chemical contaminants associated with piers, docks, and bulkheads; disruption of natural physical processes; effects on predation and prey refuge habitat; effects on productivity; effects on migration; and recreation and construction activities. The following is a summary of the information contained within the collected literature. The summary will begin with a discussion of the salmonid species that are present in the lakes of the Tri-County area, and are listed, or candidates for listing under the ESA. Second will be a discussion of pertinent research regarding the potential impacts. Finally, the summary will conclude with recommendations for best management practices (BMPs), mitigation options, and further study.

Fish Ecology

Federally Listed Salmonid Species

In order to analyze how an activity or structure could adversely affect any species, it is necessary to understand the ecology and biology of that species, particularly within the subject area. The subject area includes Lakes Washington and Sammamish, but the information should be applicable to other lakes within the Tri-County area that are utilized by ESA-listed or Candidate species. Chinook salmon stocks occurring in the Puget Sound Evolutionary Significant Unit (ESU), which includes tributaries of Lakes Washington and Sammamish, are listed as Threatened by NMFS under the ESA (U.S. Federal Register, 24 March 1999). NMFS has also designated coho salmon stocks in the Puget Sound-Strait of Georgia ESU as Candidates, which are eligible for listing under the ESA (U.S. Federal Register, 25 July 1995), but NMFS has not chosen to list them at this time. In addition, the USFWS has designated bull trout in the Coastal-Puget Sound Distinct Population Segment (DPS) as Threatened (U.S. Federal Register, 1 November 1999) (Table 1). A petition was submitted to the USFWS on 16 March 2000 for an emergency listing of the early run Issaquah Creek kokanee as Endangered. The petition occurred following both the scope development, and the literature collection phases of this review. Thus, kokanee-specific data could not be incorporated into this report.

Table 1. Listed and Candidate Fish Species in the Lake Washington Watershed.

Species	Federal Status	State Status	ESU/DPS ¹
Chinook salmon <i>Oncorhynchus tshawytscha</i>	Threatened March 1999	Candidate	Puget Sound ESU
Coho salmon <i>Oncorhynchus kisutch</i>	Candidate July 1995	None	Puget Sound -Strait of Georgia ESU
Bull trout <i>Salvelinus confluentus</i>	Threatened November 1999	None	Coastal-Puget Sound DPS

¹ ESU = Evolutionarily Significant Unit, the species definition used by the National Marine Fisheries Service

DPS = Distinct Population Segment, the species definition used by the U.S. Fish and Wildlife Service

Chinook Salmon

Chinook salmon are found along the Pacific Coast from the Ventura River in southern California to Point Hope, Alaska (Wydoski and Whitney 1979). In Washington, chinook salmon spawn in streams in the Columbia River Basin, Puget Sound, and coastal drainages (Wydoski and Whitney 1979). In the Lake Washington watershed, fall-run chinook salmon migrate through Lake Washington to reach spawning grounds in the Cedar and Sammamish River systems and in other Lake Washington tributaries. Washington Department of Fish and Wildlife (WDFW) hatchery staff allow returning progeny of the Issaquah Hatchery to migrate beyond the hatchery weir only if egg-take goals have been achieved. Occasional beach spawning within Lake Washington has been observed (Roberson 1967; Fresh, pers. comm., 28 March 2000). Adults begin migrating into fresh water in June, peaking in August, and spawn from mid-August to mid-December (Myers et al. 1998). After spawning, females guard redds for up to three weeks before dying; males attempt to fertilize other redds before dying (U.S. Federal Register, 9 March 1998). Chinook salmon eggs hatch after 90 to 150 days, depending on water temperature (Wydoski and Whitney 1979).

Life History Strategies

The Puget Sound ESU exhibits an “ocean-type” life history (Myers et al. 1998). In general, ocean-type fish move relatively rapidly through fresh water into coastal or estuarine rearing areas, compared to their stream-type counterparts (U.S. Federal Register, 9 March 1998; Wydoski and Whitney 1979). The ocean-type chinook in the Lake Washington basin typically begin their downstream migration as sub-yearlings (Myers et al. 1998). Most chinook emigrate as fry after emerging from the gravel (Myers et al. 1998), reaching Lake Washington in early January to March (Fresh, pers. comm., 2 August 1999). A second wave of juvenile fingerlings enters Lake Washington in May and June (Fresh, pers. comm., 2 August 1999). Chinook fry are an average of 40 mm (1.6 in.) in length when they enter Lake Washington; chinook fingerlings are an average of 100 mm (4 in.) in length when they enter Lake Washington (Fresh, pers. comm., 2 August 1999). In addition to the contribution of natural spawners, WDFW’s Issaquah

Creek hatchery has an annual production goal of releasing 2 million age-0+ chinook each May into Issaquah Creek, many of which enter Lake Washington via the Sammamish River. On average, that goal has been met over the last five years (Mahovich, pers. comm., 7 June 2000). The University of Washington hatchery has an annual production goal, which is consistently met, of releasing 180,000 chinook smolts each May (Tetrick, pers. comm., 10 July 2000). The majority of chinook smolts leave Lake Washington in May and June (Fresh, pers. comm., 9 September 1999).

A study by Reimers (1971) in Sixes River, Oregon, demonstrated that juvenile migration timing within the "ocean-type" designation occurs as a continuum rather than a discreet event. Evidence that most juvenile chinook begin entering the lake in early January and are leaving Lake Washington as smolts by early July, suggests that juvenile chinook in the lake are exhibiting a "type-2" life history (as per Reimers 1971). However, the WDFW Cedar River fry trap that has provided most of the data on migration timing is typically operational only through the end of June or early July. Sampling at the mouth of the Cedar River has found that small numbers of juvenile chinook continue entering Lake Washington as late as 29 July (Fresh, pers. comm., 9 September 1999). Outmigrating chinook smolts have been observed at the Ballard locks in late August (Fresh, pers. comm., 7 June 2000).

Yearling and older chinook (monthly mean fork lengths ranging from 256-323 mm) were captured in littoral gill nets (2-8 m deep) in all regions of Lake Washington from January through October in 1984-1985 (Beauchamp, Univ. of Washington, unpubl. data). Tabor and Chan (1996) captured two juvenile chinook yearlings (234 and 280 mm fork length) in south Lake Washington in March 1995. Although it is not known whether these yearlings reared in the lake or in a tributary, their large size is typical of lake-rearing fish. The appearance of small numbers of age-1+ and age-2+ chinook juveniles in Lake Washington provides additional evidence that extended freshwater rearing occurs in the Lake Washington system (Fresh, pers. comm., 9 September 1999). Haw and Buckley (1962) reported extended freshwater rearing of juvenile chinook in Lakes Washington and Sammamish, with age-1+, and -2+ smolts representing 21 percent and 12 percent respectively of sampled returning adults. The majority of age-0+ chinook juveniles in the Lake Washington watershed leave the lake by mid-summer; 66 percent of the returning adults sampled by Haw and Buckley (1962) had been age-0+ smolts. Reimers (1971) found relatively few juvenile fall chinook migrating as yearlings in Sixes River, Oregon, where yearling migrants represented only 3.1 percent of returning adults. Data from the Lake Washington Ecological Studies indicate that resident chinook up to adult size are in the lake at all times of the year; it is not clear whether these fish go to sea eventually or continue rearing in the lake until spawning (Warner, pers. comm., 7 July 2000).

Diet and Distribution

In Lake Washington, juvenile chinook are distributed along shorelines (Fresh, pers. comm., 18 November 1999). Sampling of both the limnetic and littoral zones of Lake Washington has shown that from early February through late May, young-of-the-year chinook occupy the littoral zone exclusively (Warner and Fresh 1999). They feed

primarily on aquatic insects (chironomid pupae) (Fresh, pers. comm., 18 November 2000) and terrestrial insects (Wydoski and Whitney 1979; Tabor and Chan 1996). Rondorf et al. (1990) found that in a Columbia River reservoir, the diet of juvenile chinook salmon consisted primarily of zooplankton and terrestrial insects; in free-flowing river sections the diet consisted mostly of aquatic insects. Juvenile chinook adapt to local prey abundance by modifying their selection of prey items (Rondorf et al. 1990).

Chinook juveniles, predominantly large individuals, begin appearing in limnetic sampling gear in late May and June in Lake Washington (Fresh, pers. comm., 9 September 1999). Increasing use of the limnetic zone may be an ontogenetic response, a response to increasing temperatures in the littoral zone, or merely represent the capture of outmigrating smolts (Fresh, pers. comm., 9 September 1999). Hamilton et al. (1970) observed an ontogenetic shift with increasing fish body size from littoral foraging on aquatic and terrestrial insects, to limnetic foraging on zooplankton, by coho salmon in a reservoir (see below).

Reimers (1971) found that in the Sixes River, chinook fry (~40 mm) occupied shallow water and were closely associated with shoreline features such as logs and debris, but larger fish (~55 mm) occupied the entire channel. Others have noted a similar transition from an affinity for shoreline structure and cover for newly emerged chinook fry, to a progressively offshore distribution as fish size increases (Lister and Genoe 1970; Weitkamp and Campbell 1980; Roper et al. 1994). In the lower Willamette River, Ward et al. (1994) reported that young-of-the-year chinook caught in vertical gill nets were a mean distance of 31 meters from shore. However, all of these studies were in rivers or riverine portions of estuaries, where current velocity and channel width influenced habitat selection. Even though the fish in Reimers' (1971) study were utilizing the entire estuary, they were still occupying habitat that did not exceed 5 meters in depth.

Temperature

The distribution and residence time of juvenile chinook in Lake Washington may be influenced by temperature. Bjornn and Reiser (1991) reported the preferred temperature for chinook as 12 to 14°C, and temperatures from 23 to 25°C could be lethal and were actively avoided. Offshore temperatures at a depth of 1 meter in Lake Washington typically exceed 15°C from mid-May through mid-October, and exceed 20°C from mid-July into September, with maxima generally near 25°C in early August (<http://dnr.metrokc.gov/wlr/waterres/lakes/wash.htm>). Nearshore summer surface-temperatures may exceed those offshore. Despite high temperatures, chinook can be found in the littoral zone during summer. During 1984, chinook were captured throughout the summer in littoral gill nets (3-10 m deep) and in offshore vertical gill nets from mid-July to mid-August at depths of 12 to 18 meters, which corresponded with the thermocline (Beauchamp, Univ. of Washington, unpubl. data). Chinook smolts (and adults) are often found near the surface in water above 18°C (Warner, pers. comm., 7 July 2000).

Behavioral thermoregulation by salmonids is common. Roper et al. (1994) suggested that high temperature (20°C) in the lower reaches of Jackson Creek, Oregon, caused

the emigration of age-0+ chinook. Biro (1998) reported that some young-of-year brook trout (*Salvelinus fontinalis*) in an Ontario lake remained in the littoral zone during periods of high water temperature (23-27°C), but they held and defended positions at cold groundwater seeps. Given the propensity for littoral foraging by juvenile chinook, and that summer temperatures in the littoral zone may be undesirable or potentially lethal, chinook may either leave Lake Washington in mid-summer, limit the timing of their activity, or limit their distribution to cooler areas of the lake. The energetic consequences of such avoidance behavior could reduce the growth rate of juvenile chinook, potentially reducing marine survival. Smolt size has been found to positively correlate with marine survival (Quinn and Peterson 1996). Observations of juvenile chinook in the littoral zone of Lake Washington decline in early to mid-July; this phenomenon could possibly be a behavioral response to high temperatures, but could also be due to smolts leaving the lake (Fresh, pers. comm. 18 November 1999). An avoidance of the littoral zone by chinook during summer would segregate them from shore-based sampling efforts.

Both returning adult and juvenile chinook in Lake Sammamish and Lake Union must also contend with anoxic conditions in the hypolimnion from July through October (<http://dnr.metrokc.gov/wlr/waterres/lakes/thermo.htm>). High temperatures in the epilimnion restrict chinook to depths below 5 to 10 meters, while anoxic conditions below depths of 15 to 20 meters prevent chinook use, thus concentrating them in the relatively narrow (5-10 m) metalimnion (<http://dnr.metrokc.gov/wlr/waterres/lakes/thermo.htm>). These physical restrictions of chinook distribution limit juvenile foraging opportunities, and expose juvenile fish to predators occupying habitat in the metalimnion. In addition, these physical conditions are a stress to holding adults that could cause pre-spawning mortality and reduced egg survival for those adults that survive to spawn.

Coho Salmon

Coho salmon are found along the Pacific Coast from Monterey Bay in central California to Point Hope, Alaska (Wydoski and Whitney 1979). In Washington, coho salmon spawn in streams in the Columbia River Basin, Puget Sound, and coastal drainages (Wydoski and Whitney 1979). In the Lake Washington system, coho salmon stocks have been divided into the Lake Washington/Sammamish Tributary stock and the Cedar River stock (Washington Department of Fisheries [WDF] et al. 1993). Adult coho salmon migrate through Lake Washington and Lake Sammamish to reach spawning grounds in the Cedar and Sammamish River systems, and in small tributaries to the lakes. Adults begin migrating into fresh water in August, and spawn from late October through December in most systems, and through mid-March in the Cedar River (WDF et al. 1993). Coho salmon eggs hatch after 45 to 60 days, depending on water temperature (Wydoski and Whitney 1979).

Life History Strategies

Coho juveniles typically rear in fresh water for one year, but may spend two years in fresh water or migrate to sea as fry during their first spring (Groot and Margolis 1991;

Hartman et al. 1982). While in fresh water, juveniles utilize all accessible reaches of their natal stream systems for rearing, including lakes, seasonally wetted areas, off-channel ponds, sloughs, swamps, and their tributaries (Pollard et al. 1997; Bryant et al. 1996; Swales et al. 1988; Hartman and Brown 1987; Cederholm and Scarlett 1981; Skeesick 1970). Some physical characteristics of habitat typically selected by coho fry and parr include depths greater than 8 centimeters, low current velocity, and availability of cover (Fransen et al. 1993; Fausch 1993; Shirvell 1990; Bugert et al. 1991). Coho juveniles overwintering in two small Vancouver Island lakes were closely associated with the shoreline, with the highest CPUEs from traps set as close as possible to the shore (Swales et al. 1988).

Migration

Juvenile coho generally begin migrating to sea as smolts during their second spring, although Irvine and Ward (1989) found that 10 percent of the smolts migrating from the Keogh River watershed were age-2+ fish. In general, peak outmigration is in May and average smolt size is 90 to 115 mm (Weitkamp et al. 1995).

Studies at the mouth of the Cedar River have shown that most coho enter Lake Washington in May and June, and are 100 mm or greater (Fresh, pers. comm., 2 August 1999). Beyond the contribution of natural spawners, WDFW's Issaquah Creek hatchery has an annual production goal of releasing 1 million age-0+ coho and 450,000 yearlings into Issaquah Creek each spring, many of which enter Lake Washington via the Sammamish River. Additionally, 90,000 coho smolts are also released each May from the University of Washington hatchery (Tetrick, pers. comm., 10 July 2000).

Emigrating coho smolts in a Lewis River reservoir traveled in schools of a few to several hundred fish, and generally remained in the upper 3 meters (Hamilton et al. 1970). Allen (1968) tested whether coho smolts migrating 56 km to saltwater from the Issaquah Creek hatchery through a lake system populated with piscivorous fish would experience higher mortality than coho from the same cohort released from the University of Washington hatchery (8 km from saltwater). Allen (1968) found that the fish migrating from the Issaquah Creek hatchery had an approximately 22 percent higher mortality rate than those from the University of Washington hatchery.

McMahon and Holtby (1992) found that coho smolts in a river and estuary aggregated and sought cover near large woody debris, overhanging banks, and riparian vegetation. Moser et al. (1991) reported that progress of the smolt migration in the Chehalis River was saltatory, with periods of movement with the current interspersed with periods of holding in low-velocity areas that also provided cover.

The distribution of juvenile coho salmon in Lakes Washington and Sammamish is poorly understood. There is evidence that juvenile coho are migrating and feeding along the Lake Washington shoreline (Fresh, pers. comm., 2 August 1999). Gill net sampling in all zones of Lake Washington by Bartoo (1972) indicated that coho juveniles were present during May, June, and July. Beauchamp (Univ. of Washington, unpubl. data) captured juvenile coho in all sampled littoral areas except during July and August. Beak

Consultants Incorporated (1998) reported that peak smolt migration from the Sammamish River was April through mid-May, although their sampling ended in mid-June. Tabor and Chan (1996) found coho smolts in south Lake Washington from April to early June, with peak abundance in early May. Coho juveniles are relatively rare compared to chinook and sockeye juveniles (Walter, pers. comm., 7 July 2000).

Temperature

Water temperature affects the distribution of coho salmon in lakes and reservoirs. Bjornn and Reiser (1991) reported the preferred temperature for coho as 12 to 14°C, and that temperatures from 23 to 25°C could be lethal and were actively avoided by most salmonids. In a stocked Wisconsin lake and in a Washington reservoir, coho inhabited nearshore areas in the spring and fall, and moved below the thermocline into the metalimnion in the summer where temperatures remained below 17°C (Engel and Magnuson 1976; Hamilton et al. 1970). However, coho fry remained in the littoral zone throughout the summer in Margaret Lake in southeast Alaska, especially near the cooler tributary mouths (Bryant et al. 1996). Summer afternoon surface temperatures occasionally exceeded 18°C in Margaret Lake, but most coho were caught while beach seining at dawn and dusk when temperatures were lower, and at the mouths of cool tributaries (Wright, pers. comm., 13 August 1999). This preference for lower water temperatures may be a factor in selection of migration corridors, at least in the latter part of coho migration into and through Lakes Washington and Sammamish. Juvenile coho may avoid the high temperatures in the littoral zone during the summer, segregating themselves from shore-based sampling efforts. The late summer distribution of coho would also be restricted to the narrow metalimnion in Lake Sammamish, in a similar manner as chinook (see above). Like chinook, the result of this restricted distribution would be increased stress on holding adult fish, reduced juvenile foraging opportunities, and potentially higher predation mortality.

Diet

Diet may also be a factor in the distribution of juvenile coho in lakes. Hamilton et al. (1970) found that large coho juveniles were limnetic, with zooplankton as their primary prey. In Chignik Lake, Alaska, yearling coho fed heavily on newly emerged sockeye salmon fry around shoreline spawning and incubation areas (Ruggerone and Rogers 1992). Coho in the littoral zone of Margaret Lake, Alaska, fed entirely on insects (Cartwright and Beauchamp 1995). Aquatic insects comprised 75 percent of the diet of coho smolts in south Lake Washington during spring (February-June) 1995, and juvenile fish another 15 percent (Tabor and Chan 1996). Smolt-sized coho in a Wisconsin lake fed at the surface in the littoral zone during April, and ate aquatic and terrestrial insects (Engel and Magnuson 1971). In late spring and early summer, coho occupied both the littoral and epilimnetic zones, and continued to feed at the surface on terrestrial insects and some aquatic insects (Engel and Magnuson 1971). In late summer (mid-July to October), when water temperature in the epilimnion exceeded 16°C, coho avoided the littoral and epilimnetic zones, and fed on aquatic insects and zooplankton in the metalimnion (Engel and Magnuson 1971). Coho appear to adapt to existence in either the littoral or limnetic zones, exploiting available prey items.

Because of this adaptability, temperature may be the most important determinant of coho distribution in lakes.

Bull Trout

Several thorough reviews of bull trout literature were surveyed in preparation for this species description. Rather than repeat their work here, the following is a summary of the salient points from those reviews cited collectively, with information from other sources cited separately. The collective citation for the bulk of this description follows: Brown (1992), Rieman and McIntyre (1993), Sanborn et al. (1998), and U.S. Federal Register (1 November 1999).

The historical range of bull trout extended from the McCloud River in California to the Yukon River in Alaska, west of the Continental Divide within the contiguous United States except in tributaries of the Saskatchewan River, but east of the Continental Divide in the Saskatchewan and MacKenzie river systems in Canada. In Washington, bull trout occur within the Columbia River system, in rivers of Puget Sound, and in coastal rivers from Grays Harbor north. Two subpopulations of bull trout are considered within the Lake Washington basin: the Chester Morse Reservoir subpopulation and the Issaquah Creek-Sammamish River subpopulation (U.S. Federal Register, 1 November 1999; Washington Department of Fish and Wildlife [WDFW] 1998). In the mid- to late 1990s, less than 10 spawning sites were found, and fry abundance was low in the Chester Morse Reservoir; however, the population in the reservoir was estimated at approximately 3,000 adults based on hydroacoustic surveys, and did not include estimates of juveniles in the tributaries (Seattle Public Utilities, unpubl. data). In the past 10 years, only two “native char” [either bull trout or Dolly Varden (*Salvelinus malma*)] have been reported in Issaquah Creek and none have been reported in the Sammamish River (U.S. Federal Register, 1 November 1999; WDFW 1998). The USFWS is not certain that the latter subpopulation is “viable.” There is no known spawning subpopulation resident in Lake Washington or Lake Sammamish. However, subadult and adult native char are occasionally found in the lakes (USFWS 1999). Two subadult native char (300-400 mm fork length) were captured in horizontal gill nets (3-10 m deep) off the Cedar River delta in March 1985, and an adult (635 mm fork length) in the same location in April 1985; a subadult native char (300 mm fork length) was also captured in August 1984 at a depth of 60 meters (Beauchamp, Univ. of Washington, unpubl. data). Other native char have been caught or observed in Lake Washington, as well as in the Cedar River, Lake Sammamish, Carey Creek (Issaquah Creek tributary), and at the Ballard Locks (KCDNR 2000).

Life History

Several life history forms of bull trout occur, and all may be present within the same population. Fish exhibiting the resident life history strategy are non-migratory, spending their entire lives within their spawning stream. Migratory life history strategies include fluvial, adfluvial, and anadromous. Migratory bull trout reside as adults and subadults in larger rivers (fluvial), lakes or reservoirs (adfluvial), or marine waters (anadromous), and spawn and rear as juveniles in headwater tributaries. Due to differences in productivity

between small headwater streams and larger rivers, lakes, and marine environments, resident fish are typically smaller than migratory fish. Resident fish seldom exceed 300 mm, while migratory forms can exceed 900 mm. Anadromous forms are common in Puget Sound drainages from the Snohomish River north (Kraemer in prep.).

The majority of bull trout spawning occurs between late August and early November. Spawning migrations occur during the summer, but may start as early as April in some systems (Ratliff et al. 1996). In river systems of north Puget Sound, spawners typically arrive in holding areas near spawning grounds from several weeks, to up to four months before spawning (Kraemer in prep.). Characteristics of holding areas are: depth of at least one meter; cover in the form of turbulent water, undercut banks, woody debris, or overhanging vegetation; and cool temperatures, often provided by groundwater input. Spawning typically does not commence until stream temperatures drop to 8°C. In the North Puget Sound region, “the downstream limit of successful spawning is always upstream of the winter snow line (that elevation at which snow is present on the ground for much of the winter)” (WDFW 1999). In Montana, no spawning occurred in 1st order streams, only limited spawning by non-migratory bull trout occurred in 2nd order streams, and the majority of spawning by migratory fish occurred in 4th order streams. Bull trout spawning habitat typically consists of gravel/cobble substrates, although spawning has been observed in sand, and also in cobbles too large to be dislodged by female digging attempts (Kraemer in prep.). Close proximity to cover, low gradient (even within a high-gradient reach), and depths greater than 10 cm appear to be important spawning site-selection criteria. Once sexually mature, resident, fluvial and anadromous bull trout in north Puget Sound spawn annually (Kraemer in prep.). Following spawning, adult bull trout move downstream quickly, remaining in deep pools in larger rivers, or in lakes for the winter. Spawned-out bull trout have been observed in November feeding on loose eggs in salmon spawning grounds (Kraemer in prep.). Beach spawning of native char in Lake Washington and Lake Sammamish is improbable. Confirmed observations of beach spawning bull trout are limited to extreme downwelling conditions in cold, high-elevation lakes (WDFW 1998); water temperatures in Lake Washington and Lake Sammamish are too high for successful incubation.

Temperature

Successful egg incubation requires temperatures less than 5°C (WDFW 1999), with maximum survival between 2 and 4°C. Incubation usually takes from 100 to 145 days, depending on temperature. Egg-to-fry survival declines with increasing percentages of fine particles in the substrate (particles smaller than 6.35 mm). Fine sediments decrease egg-to-fry survival by impeding the flow of water to the eggs or by physically preventing fry emergence (entombment). Maintaining water flow to the developing eggs is necessary to remove metabolic wastes and deliver dissolved oxygen.

Juvenile bull trout are rarely found in streams with summer temperatures that exceed 15°C. Cold groundwater seeps can provide temperature refuge for bull trout in streams with summer temperatures that exceed 15°C. Fry are closely associated with the substrate while foraging, and rely on interstitial spaces for cover. This strong association with the substrate decreases with body size and is substituted by an

association with woody debris and large boulders. Juveniles are benthic foragers, feeding on aquatic invertebrates until they are large enough to become piscivores (> 100 mm). In allopatry, bull trout forage throughout the water column. Bull trout juveniles show a preference for low-velocity habitat; fry are often found in backwater areas, stream margins, and side channels, while larger juveniles occupy pools. There is some evidence for a diel habitat shift between concealment during the day and foraging in deep, fast water at night. Juveniles disperse widely from the spawning area, and should be expected even in tributaries that do not support spawning unless access is obstructed by a passage barrier. Juveniles that adopt a migratory life history strategy usually move downstream to a mainstem river, lake, or ocean following two or three years of rearing in headwater streams. Migration is possibly related to the need for a larger prey base that arises with the onset of piscivory. The timing of this migration varies between and within systems, and is not confined to spring.

Like juveniles, adult and subadult bull trout are typically found only in streams with summer temperatures that do not exceed 15°C. Pools with groundwater seeps may function as thermal refuges where stream temperatures exceed 15°C. Adults and subadults select low-velocity habitats, typically large pools, with abundant cover and large substrate. The diet of adults and subadults in streams consists primarily of fish, including juvenile salmon and trout, and whitefish (*Prosopium* spp.). A diel habitat shift similar to that of juveniles has been observed in adults and subadults in streams. Non-spawning movements are generally associated with thermal requirements, either seeking warmer water in winter (non-coastal populations) or colder water in summer.

The distribution of subadults and adults in lakes and reservoirs appears to be temperature mediated, with fish generally avoiding temperatures greater than 15°C, and preferring temperatures less than 10°C. Following stratification of lakes in the spring, bull trout are mostly found below the thermocline, and generally near the lake bottom. The diet of bull trout in lakes consists almost entirely of fish, and the species composition within the diet varies with the relative abundance of prey species in the lake. Cyprinids, catostomids, cottids, and salmonids [kokanee, cutthroat trout, smaller bull trout, whitefish] represent a substantial portion of the diet of lake-dwelling bull trout in various studies. The presence of warm-water prey species, such as yellow perch (*Perca flavescens*), in the diet of bull trout indicates that they either make occasional forays into warmer (17–20 °C) nearshore waters or exploit these prey during winter and spring. Bull trout have also been observed aggregating to take advantage of localized prey abundance such as concentrations of spawning prey fish.

Anadromous Form

The anadromous life history strategy in bull trout is not well understood. Historically, anadromous char were all considered Dolly Varden. The separation of Dolly Varden and bull trout into distinct species, and recent investigations of native char populations in Puget Sound, have suggested that Dolly Varden and bull trout in north Puget Sound are sympatric and equally anadromous. Anadromous bull trout spend two to three years in fresh water before migrating in the spring to the estuary or nearshore marine environment (Kraemer in prep.). While in the marine environment, they feed on smaller

fish such as surf smelt (*Hypomesus pretiosus*), Pacific herring (*Clupea harengus pallasii*), Pacific sand lance (*Ammodytes hexapterus*), and pink (*O. gorbuscha*) and chum (*O. keta*) salmon smolts, closely following the distribution of the prey fish (Kraemer in prep.). Subadults usually spend two summers in the marine environment before they mature (Kraemer in prep.). Anadromous bull trout return to fresh water to overwinter, and immature and non-spawning adult fish migrate upstream with the spawners in late summer (Kraemer in prep.).

Local Lake Washington/Sammamish Sub-Populations

The only likely viable bull trout subpopulation in the Lake Washington watershed is the Chester Morse Reservoir subpopulation. Spawners have not been confirmed within the Sammamish River-Issaquah Creek subpopulation (WDFW 1998), and only two bull trout have been observed in the system within the last 10 years. The thermal requirements for spawning and successful egg incubation may prevent bull trout reproduction in the Sammamish River-Issaquah Creek system. Only a few streams within the Issaquah Creek system approach the winter snow line (see above), and they may be too small to be used by migratory spawners. Vestigial pockets of resident spawners could reside within thermal refugia in the upper reaches of Issaquah Creek (i.e., Holder Creek). Successful spawning by these remnants could produce migratory offspring that may never successfully reproduce. Adfluvial or anadromous offspring of remnant resident spawners could be present within the system. High summer water temperatures and an anoxic hypolimnion (<http://dnr.metrokc.gov/wlr/waterres/lakes/thermo.htm>) would likely deter bull trout from residing in Lake Sammamish.

The Chester Morse Reservoir subpopulation is above an anadromous barrier and is a glacial relic population (WDFW 1998). The population exhibits an adfluvial life history strategy, although residents could exist in the upper watershed (WDFW 1998). Because all life history strategies can arise from the same population, it is possible that some fish emigrate from the Chester Morse Reservoir to exhibit anadromy or to reside in Lake Washington. Water temperatures in the lower Cedar River are probably too high to support a fluvial population (WDFW 1998). Cedar River tributaries below Chester Morse Reservoir likely do not meet the thermal requirements for spawning and successful egg incubation; thus, bull trout that emigrate from Chester Morse Reservoir would not represent a viable spawning population. However, these fish may spawn in a non-natal system. Char are known to exhibit "pioneering" behavior, spawning in areas other than their native stream (WDFW 1999). Anadromous bull trout and Dolly Varden overwinter in freshwater, and may overwinter in systems other than their natal system.

Bull trout that occupy Lake Washington or Lake Sammamish should exhibit similar distribution and behavior to that of bull trout observed in other lakes. Juveniles (length 150-300 mm) would migrate to the lake after rearing one to three years in headwater streams. Spawners would begin upstream migrations from April through July, including anadromous fish migrating through the lake. Adults would likely spawn annually as observed in other North Puget Sound populations (Kraemer in prep.). Immature fish residing in the lake would be likely to migrate upstream with the spawners. Those immature fish and any non-spawning adults remaining in the lake during the summer

would avoid temperatures above 15°C. High temperature avoidance would likely confine bull trout below the thermocline (> 15 m) from mid-June through mid-October, with some annual variation due to climatic differences. The presence of a large prey base in limnetic [e.g., sockeye salmon and longfin smelt (*Spirinchus thaleichthys*)] and deep benthic (e.g., *Neomysis mercedis* and prickly sculpin [*Cottus asper*]) regions would reduce the need for summer forays into nearshore areas. Native char that were captured at the mouth of the Cedar River in March and April were eating spawning longfin smelt and outmigrating sockeye fry (Beauchamp, Univ. of Washington, unpubl. data).

Other Species

Ajwani (1956) reported 20 native and 15 introduced fish species in the Lake Washington watershed. Of those species, 14 are considered common or abundant in Lake Washington, including prickly sculpin, longfin smelt, juvenile sockeye salmon, three-spine stickleback (*Gasterosteus aculeatus*), peamouth (*Mylocheilus caurinus*), yellow perch, rainbow trout (*O. mykiss*), northern pikeminnow (*Ptychocheilus oregonensis*), largescale sucker (*Catostomus macrocheilus*), brown bullhead (*Ictalurus nebulosus*), cutthroat trout, smallmouth bass, largemouth bass, and common carp (*Cyprinus carpio*) (Beauchamp 1990). The relationships of any of the common fish species to ESA-listed salmonids would be related to the timing and duration of distributional overlaps among species. The primary zone of overlap for juvenile chinook and coho with other fish species would be the littoral zone. The expected timing of littoral zone occupation for each species is shown in Table 2.

Bass

Bass have been studied extensively throughout their range, including in Lakes Washington and Sammamish. Direct studies on the relationship between piers and other shorezone structures, and bass predation on salmonids are currently underway (Roger Tabor, USFWS; Kurt Fresh, WDFW; Rod Malcom and Eric Warner, Muckleshoot Tribal Fisheries). Conjecture about that relationship can be supported with information from other studies and personal communications from local scientists. Both largemouth and smallmouth bass demonstrate an affinity for structural elements, and both are piscivorous, preying on salmonids when available.

Stein (1970) found that largemouth bass in Lake Washington preferred heavy log and brush cover to all other available habitat (including docks), and considered the lack of this habitat to be a limiting factor. Largemouth bass were often found under docks in early spring in Lake Washington (Stein 1970). One third of the largemouth bass in Lake Baldwin, Florida showed a significant preference for piers in the absence of aquatic vegetation (Colle et al. 1989). Largemouth bass preferred moderate to dense vegetation and silt or sand substrate in Lake Sammamish (Pflug 1981). Nests were constructed at depths from 0.6 to 1.5 meters, in vegetated areas with soft-sediment to gravel substrates, on moderate to steep slopes (Pflug 1981). Others have noted preferences for nest locations adjacent to a structural feature such as a rock, stump, or a slope (Heidinger 1975; Allan and Romero 1975), and locations that provide cover

(Vogele and Rainwater 1975). In general, largemouth bass select soft substrates; cover in the form of logs, brush, aquatic vegetation, or other structures; and utilize a variety of prey-capture tactics.

Both smallmouth and largemouth bass utilize docks and piles in addition to natural cover. However, smallmouth bass generally select hard substrates without aquatic vegetation, drop-offs or outcroppings, and cover in the form of logs or rocks, whereas largemouth bass generally prefer softer-bottom substrates and aquatic macrophytes (Coble 1975). Smallmouth bass in Lake Sammamish often selected residence areas with overhead cover such as docks, submerged logs, or overhanging vegetation, and preferred areas with cobble/gravel substrate and drop-offs, without aquatic vegetation (Pflug 1981; Pflug and Pauley 1984). Smallmouth bass were the only species that Bryan and Scarnecchia (1992) consistently found in equal or greater abundance in developed sites than in undeveloped sites in Spirit Lake, Iowa. Smallmouth bass in a Texas reservoir selected rock outcroppings more than other habitat types (Kraai et al. 1991). Male smallmouth bass in Lake Sammamish generally located nests within 7 to 20 meters of shore, on gently sloping gravel/cobble substrates, devoid of vegetation, at depths of 1 to 3 meters, and associated with a structural element such as a log, boulder, pile, or other artificial structure (Pflug and Pauley 1984; Malcom, pers. comm., 13 April 2000).

Largemouth and smallmouth bass spawned beside fallen trees in water as shallow as 80 cm in lakes of the Eastern Region national forests (Bassett 1994). Danehy and Ringler (1991) reported that smallmouth bass displayed two different foraging and habitat selection strategies: those occupying cobble/rubble shoals were strongly associated with the substrate and fed primarily on benthos (mostly crayfish), and those found over sandy substrates were piscivorous and exhibited an active hunting behavior, feeding pelagically without an association with the substrate (Danehy and Ringler 1991). Haines and Butler (1969) showed that structures that provided darkness were selected most frequently by yearling smallmouth bass. Fallen trees in less than 1.5 meters of water were generally used by juvenile smallmouth bass, but not by adults in Eastern Region national forest lakes (Bassett 1994). Bassett (1994) reported that artificial structures placed at depths of 3 to 6 meters were most effective at attracting centrarchids during summer. In Lake Joseph, Ontario, young-of-the-year smallmouth bass were the only fish group whose density was not significantly related to coarse woody debris (CWD); instead, their density was highest in areas with high concentrations of shorezone structures (Brown 1998). Helfman (1979) experimented with fish attraction to shade-producing, floating objects in Cazenovia Lake, New York, and found that smallmouth bass were not attracted to the floats (sized from 1.1-3.6 m²); largemouth bass occasionally hovered below the experimental floats, but were more common under larger swimming floats at similar depths. Helfman (1979) speculated that the response of largemouth bass might be indicative of an attraction to "more massive structure" than the experimental floats provided.

Key Predators

Table 2 illustrates that the distributions of all of the common fish species in Lake Washington overlap the distribution of juvenile chinook salmon at some point. Of primary interest is the distribution of potential predators on juvenile chinook. Information provided at a recent workshop convened to present a report on the progress of the ongoing Lake Washington Ecological Studies provided an updated perspective on the predators of juvenile salmonids. The primary native fish species identified as potential predators in the Lake Washington system were: river lamprey, cutthroat trout, rainbow trout, coho salmon, and five species of sculpin; other native species were: mountain whitefish, bull trout, longfin smelt, chinook salmon, and sockeye salmon (Lake Washington Sockeye Studies Interim Workshop 2000). The primary introduced fish species identified as potential predators in the Lake Washington system were: smallmouth bass, largemouth bass, and yellow perch; other species include: brown bullhead, black crappie, white crappie, pumpkinseed, Atlantic salmon, bluegill, and warmouth (Lake Washington Sockeye Studies Interim Workshop 2000). The most important nearshore predator of sockeye fry in Lake Washington was identified as cutthroat trout less than 250 mm; other predators noted were juvenile coho salmon, and rainbow trout (Lake Washington Sockeye Studies Interim Workshop 2000). The most important limnetic predator of juvenile salmonids was identified as cutthroat trout over 250 mm, whose diet consists of approximately 50 percent salmonids (Lake Washington Sockeye Studies Interim Workshop 2000). Prickly sculpin larger than 125 mm were also identified as the most important benthic predator (Lake Washington Sockeye Studies Interim Workshop 2000). Although the focus of the Lake Washington Sockeye Studies was on sockeye salmon, identified predators also prey on other juvenile salmonids, including chinook.

The primary predator of juvenile chinook occupying the littoral zone from January through June, and the limnetic zone for the remainder of the year, would be cutthroat trout (Warner, pers. comm., 7 July 2000). A small proportion of northern pikeminnow, yellow perch, and smallmouth bass reside in nearshore regions during winter, but the majority move inshore in the spring as temperatures in nearshore areas warm (Bartoo 1972; Olney 1975; Coutant 1975). The distributions of these fishes overlap primarily with the peak out-migration of chinook through the littoral zone, whereas the overlap of cutthroat and chinook distributions is continuous. Sculpins are present in the littoral zone year-round and are known to eat chinook (Tabor et al. 1998). In mid-summer, temperatures in the littoral zone become undesirable for juvenile chinook and coho salmon, and the majority leave the lake or seek cooler temperatures away from the littoral zone, thus segregating themselves from littoral predators, but remaining vulnerable to cutthroat trout and potentially prickly sculpin.

The magnitude of avian predation on salmonids in Lake Washington is unknown. Studies from other systems indicate that consumption rates can be substantial. Double-crested cormorants (*Phalacrocorax auritus*) and western grebes (*Aechmophorus occidentalis*) consumed more than 31 percent of the spring plant of trout fry in a Utah reservoir over a two-week period, and nearly 33,000 larger subadult trout over eight months (Modde et al. 1996). Wood (1987a) estimated that common mergansers

(*Mergus merganser*) consumed as much as 39 percent of the potential coho smolt production from the Big Qualicum River, BC. Suter (1995) estimated that cormorants (*P. carbo*) consumed from 5 to 22 percent of the annual standing crop of grayling (*Thymallus thymallus*) in two Swiss rivers. Salmonids comprised the major dietary component of both red-breasted mergansers (*M. serrator*) (Feltham 1990) and common mergansers (Wood 1987b) in their respective studies. Wood (1987b) stated that “mergansers rank among the largest (in terms of appetite) and most efficient predators of juvenile salmon” and they “congregate wherever salmon density is high.” Alexander (1979) was able to attribute 15 percent of annual mortality of age 0 to 1 brook trout in a Michigan River to avian predators and 58 percent to piscivorous brown trout (*Salmo trutta*).

Because of the presence of predatory birds in Lake Washington and Lake Sammamish, avian predation must be considered among potential threats to juvenile salmonids. Common mergansers are abundant in the spring. Double-crested cormorants are common in Lake Washington, typically perching on the log booms at Union Bay and May Creek rather than on docks and bulkheads. Cormorants also commonly perch on individual piles (Warner, pers. comm., 7 July 2000). Western grebes inhabit enclosed bays (and some marinas). Gulls are common in the lake, perching on log booms and on low docks (Warner, pers. comm., 7 July 2000). Gulls are known predators of juvenile salmonids (Ruggerone 1986).

Table 2. The timing of the annual littoral zone occurrence of the common fish species in Lake Washington (similar data were not obtained for Lake Sammamish). Dashed black lines indicate presence, blank areas indicate absence. Footnotes appear on the following page.

Species	Presence in the Littoral Zone											
	Jan.	Feb.	March	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Prickly Sculpin ¹	-----											
Longfin Smelt ²	-----											
Sockeye Salmon ³	-----											
Chinook Salmon ⁴	-----											
Coho Salmon ⁵	-----											
Rainbow Trout ⁶	-----											
Cutthroat Trout ⁷	-----											
Native Char ⁸	-----											
Stickleback ⁹	-----											
Peamouth ¹⁰	-----											
*Yellow Perch ¹¹	-----											
N. Pikeminnow ¹²	-----											
Largescale Sucker ¹³	-----											
*Brown Bullhead ¹⁴	-----											
*Smallmouth Bass ¹⁵	-----											
*Largemouth Bass ¹⁶	-----											
*Common Carp ¹⁷	-----											
Kokanee	Insufficient data											
Mountain Whitefish ¹⁸	-----											
*Pumpkinseed ¹⁹	-----											
*Tench ²⁰	-----											
Month	Jan.	Feb.	March	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.

* Indicates an exotic species.

¹Rickard (1978)

²Beauchamp (1982, unpublished data), Chigbu (1993)

³Martz (1996), Beauchamp (unpublished data)

^{4,5}Fresh (unpublished data), Beauchamp (unpublished data)

⁶Beauchamp (1987, 1990)

⁷Beauchamp et al. (1992)

⁸Beauchamp (unpublished data), USFWS (unpublished data)

⁹Traynor (1973), Beauchamp (unpublished data)

¹⁰Nishimoto (1973), Beauchamp (unpublished data)

¹¹Bartoo (1972), Nishimoto (1973), Nelson (1977)

¹²Bartoo (1972), Olney (1975), Brocksmith (1999)

¹³Beauchamp (unpublished data)

¹⁴Beauchamp (unpublished data)

¹⁵Beauchamp (unpublished data), Fayram (1996)

¹⁶Stein (1970)

¹⁷⁻²⁰Beauchamp (unpublished data)

Shorezone Structures and Salmonid Predators.

Many of the predators that juvenile chinook are exposed to are active, cruising hunters (i.e., other salmonids, many piscivorous birds, northern pikeminnow). Smallmouth and largemouth bass generally utilize ambush or habituation foraging strategies (Hobson 1979). Fayram and Sibley (2000) determined that smallmouth bass in Lake Washington occupied littoral home ranges that radiated 100 to 200 meters from the focal point and generally did not extend below 8-meter depths. Because of this propensity for ambush foraging and shoreline orientation, bass are expected to benefit from artificial structures placed in the littoral zone. Yellow perch utilize “non-structural” areas (Paxton and Stevenson 1979). Recent evidence of the role of cutthroat trout as the major predator of juvenile salmonids in Lake Washington (Lake Washington Sockeye Studies Interim Workshop 2000), raises the question of whether cutthroat trout might also benefit from shorezone development, regardless of their foraging method. Shoreline development could potentially increase the rate of predation on juvenile chinook by several principal means: 1) reducing prey refuge habitat by modifying the structure of the shoreline (critical in all predator-prey interactions, but especially critical for prey of mobile predators such as cutthroat trout); 2) providing concealment structures for ambush predators such as bass and sculpin; 3) creating enough structure to reduce bass home range sizes; 4) providing artificial lighting that allows for around-the-clock foraging by predators; 5) potentially increasing migration routes for smolts and rearing fry, thus increasing exposure to predators; and 6) potentially increasing the bass population by increasing the amount of potential spawning habitat.

Bass are generalist piscivores, eating salmonids when their distributions overlap. In a study of the impacts of introduced fish in lakes of the northeastern United States, Whittier and Kincaid (1999) observed that native brook trout populations had been nearly extirpated, or had to be maintained by stocking, in lakes where smallmouth bass had been introduced. Salmonids were a greater proportion of the diet of largemouth bass than of smallmouth bass in Lake Sammamish (Pflug 1981). Tabor et al. (2000) observed the opposite in the Lake Washington Ship Canal. Pflug (1981) proposed that bass exploitation of the seasonal abundance of outmigrating salmonids was responsible for the unusually high growth rate of bass in Lake Sammamish. An analogous situation exists in coastal Massachusetts lakes where exploitation of anadromous herring contributes to the production of “trophy” bass (Yako et al. 2000).

Much of the bass predation on salmonids in the Lake Washington system corresponds with the out-migration of smolts in the spring and summer (Stein 1970; Pflug 1981; Pflug and Pauley 1984; Fayram and Sibley 2000; Tabor et al. 2000). This phenomenon has also been observed in the Columbia River (Gray and Rondorf 1986; Vigg et al. 1991; Poe et al. 1991; Zimmerman 1999). In the mid-Columbia River, ocean-type chinook fry were the only identified salmonids found in smallmouth bass stomachs by Tabor et al. (1993). The Columbia River studies indicated that salmonids were only seasonally abundant in bass diets, and that other fish species, crayfish, and other invertebrates provided the bulk of bass prey items. The Lake Washington Ship Canal may be an exception due to the tight bottleneck that it imposes on outmigrating salmon smolts. Preliminary evidence from a study by Tabor et al. (2000) indicates large populations of

both largemouth and smallmouth bass in the Ship Canal coinciding with the outmigration of salmon smolts. Analysis of stomach contents indicated that age-0+ chinook were the predominant salmonid prey item, constituting approximately 50 percent of the diet of smallmouth bass; preliminary consumption rates for April – July 1999 were 0.3 smolts/stomach for smallmouth bass and 0.1 smolts/stomach for largemouth bass (Tabor et al. 2000).

As discussed above, patterns of bass predation in the Lake Washington Ship Canal are similar to those in the Columbia River. Preliminary analysis of Ship Canal smallmouth bass stomach-contents from May through July 1999 found that salmonids represented approximately 60 percent of the diet of bass from 200 to 249 mm, and 50 percent of the diet in bass 250 mm and larger (Tabor et al. 2000). The large numbers of bass in the Ship Canal, and their high rates of consumption of salmon smolts (primarily chinook but also coho and sockeye), pose a substantial threat to chinook salmon migrating from the Lake Washington system; however, actual losses due to predation and the proportion of the smolt population these losses represent have not been computed yet (Tabor et al. 2000). Consumption estimates from limited preliminary sampling in 1997 by the Muckleshoot Indian Tribe indicate that as many as 100,000 chinook smolts could have been consumed in a 90-day migration period (Warner, pers. comm., 7 July 2000).

The distributional overlap of chinook with bass in the Lake Washington system is more prolonged than in the mid-Columbia River due to a temperature regime that favors bass. Water temperatures in Lake Washington typically exceed 10°C by mid-April, and 15°C by June (<http://dnr.metrokc.gov/wlr/waterres/lakes/wash.htm>). Thus, the peak of chinook outmigration from Lake Washington (June) corresponds with increasing bass activity and metabolic demands. Furthermore, the outmigration of chinook juveniles continues into late August, prolonging the distributional overlap of chinook and bass. Predation rates in the Ship Canal have likely increased in response to climate change and its effects on predator metabolism. Increasing water temperatures in Lake Union from 1973 to 1996 have produced an estimated increase in predation rates of 18 percent, 16 percent, 13 percent and 9 percent for smallmouth bass, rainbow trout, northern pikeminnow, and largemouth bass, respectively (Stock et al. 2000). Thus, with projected increases in global temperatures over time, predation rates on juvenile salmonids will likely continue to increase even if predator populations and other habitat variables remain constant.

Potential Impacts of Shoreline Development

Historical Changes

The conditions currently experienced by chinook and coho salmon, and bull trout in the Lake Washington watershed result from considerable human alterations of the environment. The lowering of Lake Washington that resulted from the construction of the Ship Canal and Hiram Chittenden Locks (completed in 1916), and concurrent elimination of the Black River and diversion of the Cedar River into Lake Washington were the most monumental modifications. Lake Union was connected to Lake

Washington via the Montlake Cut, and the former outlet to Lake Union was enlarged to form the Fremont Cut. Locating the locks near the western terminus of Salmon Bay converted the formerly saltwater inlet into a freshwater channel, eliminating over 7 km (4 mi.) of estuarine habitat (Chrzastowski 1983). Lowering the lake and diverting the Cedar River affected both the fish populations and the condition of the habitat. Cedar River fish stocks were locally adapted to a riverine migration and an extensive estuary, instead of the current lengthy lacustrine migration and an abrupt entry from warm, fresh water into significantly colder, more saline conditions below the locks. Lake Washington and Lake Sammamish fish stocks, while accustomed to the lengthy lacustrine migration, were also adapted to an extensive estuary. The approximately 9-foot reduction in lake level eliminated much of the available shallow-water and freshwater marsh habitat, and decreased the length of the shoreline. Chrzastowski (1983) reports a loss of 15.3 km (9.5 miles) of shoreline, and an estimated loss of 410 hectares (1,013 acres) of wetland resulting from the lowering of the lake.

The channelization and straightening of the Sammamish River, at a loss of approximately 12 miles of river, eliminated the majority of riverine and off-channel rearing-habitat for juvenile salmon, disconnected the river from its floodplain, and eliminated riparian vegetation. The outlet weir on Lake Sammamish controls summer lake level and restricts flow to the Sammamish River during adult salmon migration. The Sammamish River now represents a substantial thermal migration-barrier to adult spawners. An additional consequence of the Sammamish River channelization, subsequent river trail construction, and removal of the riparian vegetation, is that migrating salmon lack significant refuge habitat and holding areas, and are subjected to disturbance by passing trail users (Malcom, pers. comm., 22 November 1999).

The regulation of the water level in Lake Washington by the USACE has eliminated the annual flood-driven seasonal inundation of the shoreline that shaped the structure of the macrophyte community. Historically, the lake would fluctuate as much as 7 feet during flood events (Chrzastowski 1983). The previously hardstem bulrush- and willow-dominated shoreline community has been replaced by developed shorelines with landscaped yards. The loss of natural shoreline has reduced complex shoreline features such as overhanging and emergent vegetation, woody debris (especially fallen trees with branches and/or rootwads intact), and gravel/cobble beaches. Evermann and Meek (1897) noted in 1896 that “the shore of Lake Washington is not well adapted to collecting with a seine” due to the abundant submerged woody debris, and dense underbrush, small trees, and tule (hardstem bulrush) that fringed the shoreline. Development of the shoreline of Lake Sammamish has reduced the once abundant overhanging vegetation and woody debris. Evermann and Meek (1897) noted in 1896 that the trees along the shore of Lake Sammamish were so dense that when water levels were high “...it is difficult to walk any distance along the shore without swinging from one bough of a tree to another.” The loss of native shoreline vegetation and wetlands in the two lakes has likely reduced allochthonous input of detritus and terrestrial insects. USACE water-level regulation has not similarly affected the shoreline of Lake Union, as the historical water-level fluctuation was similar to the present. Losses of wetland and shoreline vegetation in Lake Union are attributable to filling and shoreline development.

The woody debris, once abundant along the shoreline of Lakes Washington and Sammamish in their historical conditions, has been replaced with structurally simple piers. A survey of 1991 aerial photos estimated that 4 percent of the shallow-water habitat in Lake Washington and 2.5 percent in Lake Sammamish within 30.5 meters of the shore was covered by residential piers (ignoring coverage by commercial structures and vessels) (Malcom, pers. comm., 22 November 1999). Approximately 81 percent of the shoreline of Lake Washington is bulkheaded (Warner and Fresh 1999). The loss of complex habitat features (i.e., woody debris, overhanging vegetation, emergent vegetation), and shallow-water habitat in Lakes Washington and Sammamish has reduced the availability of refuge habitat and forage for juvenile salmonids. Restoring and preventing future losses of these habitat features should be priorities for agencies responsible for managing lakeshore development.

Chemical Contaminants Associated with Piers, Docks and Bulkheads

1. Hydrocarbons within Lake Washington

Direct causal links between bulkheads, piers and other artificial shorezone structures and hydrocarbon inputs are few. Those links that have been identified include the use of creosote-treated lumber (hydrocarbon flux) and localized increases in watercraft powered by internal combustion engines (hydrocarbon spillage and exhaust). Within urbanized regions, such as the Lake Washington watershed, the quantity of aliphatic and cyclic hydrocarbons entering the lake by boating activity and creosote-treated wood is most likely insignificant relative to the quantity of anthropogenic and biogenic hydrocarbons of all species entering the lake through urban runoff, atmospheric particulate fallout, and fluvial inputs (Latimer and Quinn 1998; Wakeham 1977; Jones et al. 1980; Green and Trett 1989). However, hydrocarbon input from boats and treated wood should not be considered trivial.

A. Creosote and PAHs - Creosote is primarily composed of polycyclic aromatic hydrocarbons (PAHs) (Hyötyläinen and Oikari 1998), which are known carcinogens, mutagens and teratogens (Chen et al. 1997; Hussain et al. 1998; Green and Trett 1989). PAHs are a common pyrolytic byproduct of all internal combustion engines and are now commonly found in most aquatic systems near industrialized and urbanized centers (Green and Trett 1989). PAHs are known to bio-accumulate proportionally to the number of carbon rings composing the parent compound, and have bio-accumulation factors ranging between 10,000 to 130,000 (Green and Trett 1989). Although PAHs are more commonly found as a contaminant within urban runoff (Latimer and Quinn 1998), boat exhaust (Smith et al. 1987), fuel spills, and other varied sources, under rare circumstances large quantities of creosote-treated lumber within confined bodies of water (such as San Diego Bay, California) can emit more PAHs (metric tons per year) than all other sources combined (Katz et al. 1995). In British Columbia, Goyette and Brooks (1998) found that significant PAH sediment contamination occurred within 7.5 meters of newly installed creosote piles in a marine inlet, but that significant biological effects on the infaunal community occurred within 0.65 meter. In the same study, Goyette and Brooks (1998) predicted that maximum sediment PAH concentrations should occur approximately 1,000 days post installation.

Overall, relatively little is known about the impacts of PAHs to aquatic organisms. Evidence for immunosuppression resulting from exposure to PAHs was reported by Arkoosh et al. (1998), who determined that chinook smolts from urban estuaries exhibited a higher cumulative mortality after exposure to the marine pathogen *Vibrio anguillarum* than smolts from a non-urban estuary. Tissue examinations of the chinook smolts indicated that those from the urban estuary had been exposed to higher levels of PAHs and PCBs than smolts from the non-urban estuary (Arkoosh et al. 1998). Studies of impacts to freshwater aquatic organisms by PAHs report inhibition of phytoplankton electron transport (Marwood et al. 1999); depression of growth and reproduction of *Daphnia* (Geiger and Buikema 1982); liver stress and formation of liver tumors and cataracts in rainbow trout (Hyötyläinen and Oikari 1998; Black et al. 1988; Laycock et al. 1999); and acute toxicity to the duckweed *Lemna gibba* (Ren et al. 1993). One study relating PAHs in freshwater sediments to human health quantified a greater than negligible lifetime increase in risk of skin cancer to persons swimming in areas with PAH-contaminated sediments (Hussain et al. 1998).

Although a few of these studies standardize and quantify lethal and sub-lethal effects (96 hr EC-50/LC-50 values), many studies employ unique exposure regimes and relative toxicity tests of unknown PAH mixtures, (I think a word may be missing), making comparisons between studies difficult (Malins 1982). Un-saturated PAHs appear to be most toxic, with effect concentration values for various freshwater and marine organisms ranging from 2 to 1,000 ppm, although many fishes have 96 hour LC-50 values between 10 and 100 ppm (Green and Trett 1989). Several studies have identified increased toxicity of PAHs exposed to ultra-violet light (Marwood et al. 1999; Laycock et al. 1999; Ren et al. 1993), an environmental condition more likely to occur within shallow littoral habitats. Additionally, photo-degraded hydrocarbons also tend to be more soluble than parent compounds, thus increasing their bioavailability (Green and Trett 1989).

B. Watercraft Exhaust Emissions - Little study has been given to the role of watercraft exhaust, as a source of environmental aliphatic and cyclic hydrocarbon contamination. In the late 1960s and early '70s several papers were published that discussed two-cycle outboard motor exhaust, although these documents provided only qualitative information regarding the extent of hydrocarbon input (Jones et al. 1980) (Citations A, B, C, none of which were acquired within the constraints of this effort). There seems to be agreement within this group of papers that two-cycle engine oil is a major source of hydrocarbon pollution within freshwater lakes (Jones et al. 1980), although other studies have estimated that relative to the overall hydrocarbon input into urbanized lakes such as Lake Washington, outboard engine operation likely contributes a very small fraction of total input, less than 1 percent (Wakeham 1977).

Two recent studies investigating the effects of watercraft exhaust in relatively pristine water bodies (one marine and one freshwater) have been conducted. The first study detected low levels of PAHs in sediments near power boat moorings at Green Island in the Great Barrier Reef (marine), but PAHs were not detected in measurable concentrations in water and clam tissue at the same locations or within sediments away from power boat moorings (Smith et al. 1987). The second study implicated watercraft

exhaust as the cause of seasonal increases in the water concentration of methyl tert-butyl ether (MTBE) at Donner Lake in California (Reuter et al. 1998), which is added to fuel mixtures to decrease emission of unburned hydrocarbons. MTBE is a known carcinogen, although very few studies have been conducted to determine this compound's effects on aquatic organisms. Note: a significant reference on this topic (Correll 1999) was located at press time, but was not incorporated into this report due to lack of time.

2. Heavy Metal Contamination by CCA-Treated Wood

As a preventative to decay, lumber used to construct piers, docks, bulkheads and other structures experiencing regular marine and freshwater inundation are pressure treated with chromated copper arsenate (CCA). CCA is a mixture of metal oxides (chromium, copper and arsenic), each of which are highly toxic to marine and freshwater organisms in dissolved ionic form (Weis et al. 1998). Although the processes of pressure fixation of CCA to wood fibers is intended to prevent dissolution of toxic metals into the surrounding environment, contamination of water, sediment and biological organisms in proximity to CCA-treated wooden structures, especially within the first three weeks after installation, is common (Brooks 1994; Weis et al. 1998).

The toxicity of CCA leachates to freshwater and marine organisms is high, especially for copper ions which are toxic to most aquatic organisms even in comparatively low concentration, 10 to 100 ppb (Brooks 1996). Although the majority of toxicological research assessing the effects of copper, chromium and arsenic on aquatic biota has been conducted within marine waters, all three metals are known to be toxic to freshwater organisms (copper and arsenic slightly less toxic, chromium slightly more toxic) (Brooks 1996).

While substantial toxicity data for individual aquatic organisms/individual metal contaminants EC-50/LC-50 values exists, studies quantifying the impacts of leached metals in proximity to CCA-treated piers and bulkheads via *in situ* observations are few and limited to marine systems. Those studies that do exist have clearly demonstrated the capacity for these metals to leach into the environment in proximity to structures utilizing treated lumber. In certain circumstances, accumulations of metals leached from treated lumber have been detected in tissues of resident biota, although measurable impacts to individual organisms have been limited (Brooks 1996; Weis and Weis 1993). While new installations of CCA-treated wood structures can cause state and federal water quality standards for copper, chromium and arsenic to be exceeded in the short term (first 3 weeks), long term concentrations of these metals leaching from treated lumber is expected to remain below the effect levels for most aquatic organisms, especially in aquatic environments that are well flushed (Brooks 1996).

Three potentially useful literature reviews of the environmental risks associated with three common wood-preservatives used in aquatic environments were identified following the substantial completion of the present review. Brooks (1995a) reviewed creosote-treated wood products, Brooks (1995b) reviewed CCA-treated wood products, and Brooks (1995c) reviewed ammoniacal copper zinc arsenate (ACZA)-treated wood

products. These review documents are available in PDF or MS™ Word formats from the Western Wood Preservers Institute at <http://www.wwpinstitute.org/researchdocs/>.

Another wood preservative, copper-8-quinolinolate (solubilized) (Cu-8), is commonly used for above-water components of shorezone structures. Studies of the effects of Cu-8 on aquatic organisms were not located in the course of this review. Cu-8 is considered non-toxic, and is approved for use by the U.S. Food and Drug Administration in wood-preserving applications where treated wood may contact foodstuffs (American Wood-Preservers' Association 1962). It has also been found to not leach from treated textiles in running water (American Wood-Preservers' Association 1962).

3. Remaining Issues

Two issues pertaining to piers, docks and bulkheads in the near-shore environment have not yet been addressed. Hydrocarbon-contaminated sediments may be disturbed during new pier, dock or bulkhead construction. There is a lack of published research regarding the location and degree of hydrocarbon sediment contamination levels within Lakes Washington and Sammamish, and a lack of research related to the effects of disturbance of hydrocarbon-contaminated sediments on aquatic organisms.

Household or industrial cleaning and preserving agents that may be applied to piers and docks could have adverse effects on aquatic organisms. No published research on this topic was identified. Additionally, the quantity of chemicals used for this purpose, and the types of chemicals used are unknown. The potential for adverse impacts from household cleaning products is exemplified by the recent fish kill in Thornton Creek, which was suspected to have been caused by a concrete cleaner, treated swimming pool water, or a combination (Birkland 2000).

The additional issue of the potential impacts of lawn-care products on aquatic systems was also not addressed due to time constraints. Many lawn-care products are labeled with warnings of the hazards to aquatic organisms that could result from the inappropriate use of those products. The propensity of the typical waterfront landowner to have a manicured lawn indicates the probability that a variety of pesticides, herbicides, and fertilizers may be routinely applied to those lawns.

Citations we did not collect, but would like to include.

- A. Howard, H.H. and R. Stewart. 1968. Water pollution by outboard motors. *Conservationist*, June-July, 6-8 and 31.
- B. Hunnefeld, G.R. 1966. Oil pollution in surface waters caused by the operation of outboard motors. *Dt. gewasserk. Mitt.*, 10, 57-59.
- C. Shuster, W. W. and L. Clesceri. 1974. Effects of exhaust from two-cycle outboard engines. U.S. Environmental Protection Agency, Environmental Protection Technology Series, EPA-670/2-74-063.

Disruption of Physical Processes

Attenuation of wave energy, and sediment recruitment and transport are the primary physical processes that are altered by shoreline structures. When shallow littoral waters become partially or completely isolated from circulation with the main body of the lake, localized thermal, chemical, and physical regimes become established. These alterations can be beneficial to some species, but detrimental to others. When separated from the buffering effects of the main water body, littoral waters exhibit generally warmer and much greater diel temperature fluctuations. Larval forms of some fishes (e.g., catostomids and cyprinids) may benefit from the lack of wave energy and elevated temperatures, as could Eurasian milfoil (*Myriophyllum spicatum* L.). Erosion of an undeveloped shoreline by wave action results in a continuous input of sediment that is episodically supplemented by large inputs from slope failures. Sediment added to the system by erosion or slope failure is transported along shore by wave energy in the direction of prevailing winds (Lawrence and Davidson-Arnott 1997). Irregular shoreline orientation creates distinct areas of deposition and erosion (Nordstrom 1989). Shoreline areas lacking in sediment supply are prone to increased erosion of existing beach substrate, and the reduction of sediment sources in one area results in erosion in other areas (Lawrence and Davidson-Arnott 1997). Shoreline structures could potentially interrupt the process of sediment transport by preventing the input of sediment from the shore, disrupting wave energy, or blocking the movement of sediment along the shoreline.

Marine

Macdonald et al. (1994) concluded that shoreline armoring could deprive beaches of sediment recruitment from the uplands, resulting in scour of the existing substrate and ultimately reducing the beach to substrate particles too large to be mobilized by wave action. Erosion in front of bulkheads is exacerbated by reflected wave energy (Macdonald et al. 1994). However, Kraus and McDougal (1996), in an updated literature review of the effects of seawalls on beaches, concluded that reflected wave energy is “probably not a significant contributor to beach profile change or to scour in front of seawalls,” but the scour is more likely a result of along-shore processes, or a combination of long-shore and cross-shore processes. A gradient in sediment flux is necessary for scour or a reduction in beach profile to occur, and gradients typically occur along shore as a result of insufficient sediment supply (Kraus and McDougal 1996). Thus, the deprivation of sediment supply caused by bulkheads could ultimately result in lowering beach profiles in front of bulkheads and increased erosion at neighboring properties, including properties without bulkheads. Reflected wave energy at seawalls could contribute to the suspension of particles, thus facilitating their transport (Kraus and McDougal 1996).

These processes would be expected to occur in Lakes Washington and Sammamish at a smaller scale due to the relatively reduced energy of the lacustrine systems. The implications for ESA-listed salmonids are uncertain. Changes in sediment composition could affect prey availability, habitat availability, and spawning. Juvenile fall chinook in the Columbia and Snake Rivers showed a preference for shallow water over sand

substrates (Key et al. 1994a and 1994b; Garland and Tiffan 1999), and preliminary data indicates a similar preference by chinook fry in Lake Washington (Tabor, pers. comm., 9 June 2000). A reduction in fine sediments (from an interruption of supply) and shallow water due to bulkhead construction could reduce the availability of shallow sandy habitat.

Lakes/Rivers

Lorang et al. (1993) investigated shoreline changes that resulted from lake-level regulation in Flathead Lake, Montana, and found that docks and seawalls “intercept the natural supply and migration of plunge zone gravels, resulting in accelerated, localized backshore erosion on the downdrift side and heavy aggregation of migrating gravels on the updrift side.” An effect of the regulation of the level of Flathead Lake was that the extended high lake level (similar to Lake Washington - early spring filling, extended into late fall) prevented the summer establishment of riparian vegetation (Lorang et al. 1993). Historically, peak water levels in Flathead Lake increased steeply in May with spring runoff, peaked briefly in June, and declined steeply in July and August; beaches formed at high lake level were colonized by riparian vegetation over the summer (Lorang et al. 1993). These findings could have implications for the establishment of riparian and emergent vegetation in Lake Washington. Additionally, if gravels are captured around shorezone structures in Lake Washington, these accumulations could further enhance the attraction of smallmouth bass, and potentially increase the availability of desirable spawning substrates associated with these structures (see observations by Malcom, pers. comms., 1999).

Bonham (1983) proposed emergent vegetation as effective wave-energy attenuation and scour prevention, and as an alternative to armored shorelines. Bonham (1983) tested the ability of four species of emergent vegetation to attenuate wave energy in large British canals and rivers, and found that a 2-meter-wide bed of any of the four species on a 1v to 4h slope, was capable of dissipating approximately two-thirds of boat wake energy and inhibiting wave break. Rolletschek and Huehl (1997) described the impacts of reed-protecting structures on shorelines; however, only the abstract was available. The structures, apparently designed to protect shoreline marshes from wave action, enhanced the accumulation of organic sediments within the marshes, modifying sediment chemical properties (Rolletschek and Huehl 1997).

Effects on Predation and Prey-Refuge Habitat

Shorezone structures are expected to affect predation on ESA-listed salmonids by simplifying the shoreline (bulkheads eliminate shallow water, complex woody debris, overhanging vegetation, and complex substrate, and create energetically unfavorable shoreline conditions); providing foraging habitat (shade and overhead cover) for structurally-oriented ambush predators, specifically largemouth and smallmouth bass (piers); and by extending the duration of predation by allowing visual predators to forage at night (piers with artificial lighting). Due to their life history strategies, largemouth and smallmouth bass are the predators most likely to benefit from shorezone structures; additional discussion of the ecology of largemouth and smallmouth bass is warranted

(see “Bass” section above), as well as a brief discussion of general predator-prey interactions. However, cutthroat trout are the primary predator of salmonids in the nearshore from February through June, and in the offshore for the rest of the year (Warner, pers. comm., 7 July 2000). The effects of shorezone structures on the efficiency of cutthroat predation on salmonids are unknown. It is likely that the loss of complex refuge habitat resulting from shorezone development would represent a disadvantage to juvenile salmonids in the presence of mobile predators such as cutthroat trout.

Predator-prey Interactions

For juvenile salmonids, the net loss in complex cover resulting from the replacement of natural shorelines with docks and bulkheads may be critical. Historically, the littoral zone of Lake Washington contained abundant aquatic and shoreline vegetation, and woody debris (Evermann and Meek 1897; Stein 1970). The lowering of the water level and substantial shoreline development have eliminated much of the shallow-water habitat available to juvenile salmon. Docks and piles may provide shallow-water cover for juvenile salmon, but they also provide cover for bass. Cooper and Crowder (1979) stated that “reducing structural complexity may remove prey refuges and subject the remaining prey to high risk until they are decimated.” Docks, piles, and bulkheads are relatively simple structural elements compared with rootwads and trees with branches, and other forms of natural cover found along undisturbed shorelines.

Sustainable predator-prey interactions in general require the existence of prey refuge to prevent the extermination of the prey organism. Numerous studies have reported increased use of complex cover (e.g., aquatic vegetation, woody debris, substrate interstices, and undercut banks) by prey fishes in the presence of predators, and reduced foraging efficiency of predators due to habitat complexity (e.g., Bugert and Bjornn 1991; Persson and Eklov 1995; Werner and Hall 1988; Tabor and Wurtsbaugh 1991; Wood and Hand 1985). Savino and Stein (1989) demonstrated that refuge is critical for prey fish survival; their study found that largemouth bass captured all prey fish that strayed from areas with aquatic vegetation into open water. Bass also eliminated all prey fish from pools that provided no refuge in a study by Schlosser (1987), while predator and prey were able to coexist in pools with complex cover. Hixon and Beets (1993) provided evidence of the value of complexity in a study of marine reef fish; prey fish were most abundant on reefs where refuge size closely matched the body size of the prey species, and where the number of refuge holes was not limiting. Lynch and Johnson (1989) showed similar results for juvenile bluegill (*Lepomis macrochirus*) in fresh water. Gotceitas and Colgan (1989) found that prey fish in fresh water preferentially selected refuge habitat with greater complexity than was necessary to significantly reduce foraging success of predators. Helfman (1979) suggested that the utilization of small floating objects on bright days by prey fish was related to the visual advantage the prey fish gained by being shaded over a predator approaching from the brightly lit surrounding area.

Shallow water functions as a refuge from predation for small fish, especially in the absence of complex habitat features such as woody debris or submerged vegetation. In

Schlosser's study (1987), bass eliminated prey fish from structurally simple pools either by direct consumption, or by forcing the prey fish into shallow-water habitats, thus subjecting prey fish to potentially decreased feeding opportunities. Bass predation also excluded grazing minnows from all but the shallow sections of pools in Oklahoma streams studied by Power et al. (1985). Ruiz et al. (1993) reported that mummichogs (*Fundulus heteroclitus*) (< 51 mm) in a subestuary of Chesapeake Bay preferentially occupied shallow water (< 35 cm) in the absence of submerged aquatic vegetation. Collins et al. (1995a) found that feeding rates by small fish (< 100 mm) in two Ontario shield lakes were 10 times higher in shallow water (< 20 cm) than in the rest of the littoral zone. Littoral slope has been negatively correlated with fish numbers and positively correlated with fish size (Randall et al. 1996). Brown (1998) observed no piscivores in "littoral fringe" (within 2.5 m of shore) transects in Lake Joseph, Ontario.

While most of the above studies on predator-prey interactions were from warmwater systems, studies of juvenile salmonid response to predators are analogous. Juvenile salmonids modify their behavior in the presence of predators by seeking or orienting to complex refuge (Gregory and Levings 1996; Reinhardt and Healey 1997), emigrating from areas with predators (Bugert and Bjornn 1991), aggregating (Tabor and Wurtsbaugh 1991), and adopting diel vertical migrations (Eggers 1978). The response of juvenile salmonids to predators increases with experience (Healey and Reinhardt 1995) and body size (Reinhardt and Healey 1997). Behavioral responses can be influenced by environmental factors such as visibility. Turbidity reduces predator-avoidance behavior in salmonids (Gregory and Levings 1996; Gregory 1993), and reduces prey mortality rates by reducing the prey-encounter rates of predators (Ginetz and Larkin 1976; Gregory and Levings 1998; Beauchamp et al. 1999). Salmonid predators also modify their behaviors in response to habitat complexity. Piscivorous brook trout in Quebec lakes switched foraging tactics from active cruising to ambushing when prey refuges were present (East and Magnan 1991).

Simplification of shoreline habitat, reducing the availability of prey refuge-habitat, should be avoided. Predator-prey interactions modify the behavior of both predator and prey species. Prey refuges are essential for the continued existence of vulnerable prey species. Complex habitat features that exclude predators, physically or through risk-aversion, can function as prey refuge. Examples of effective prey refuge may include shallow water, complex substrate, aquatic and emergent vegetation, overhanging terrestrial vegetation, undercut banks, and woody debris. Efforts to restore habitat function along lakeshores should be encouraged.

Shorezone Structures

Separating the effects of shorezone structures on juvenile salmon into discussions of the effects of individual structures in isolation may not yield the most appropriate conclusions since development seldom occurs as an isolated structure. The effects of shoreline development in its entirety should also be included in the discussion. Jennings et al. (1999) stated that "fish do not respond to shoreline structures: rather, they respond to a suite of habitat characteristics that are the result of the structure, changes to the riparian zone associated with its placement (vegetation and woody

structure removal), and often, intensive riparian zone management that occurs on developed properties.” Brazner (1997) found that sites adjacent to human development in Green Bay, Lake Michigan had fewer fish and species, and had more disturbance-tolerant fish assemblages. Fish species richness and abundance were highest in undeveloped wetland habitats (Brazner 1997). Species richness and total fish abundance were less at developed sites than at undeveloped sites in the littoral zone of Spirit Lake, Iowa (Bryan and Scarnecchia 1992). Poe et al. (1986) found that an undeveloped bay was characterized by a percid-cyprinid-cyprinodontid assemblage, while a developed bay (bulkheaded shoreline, frequent dredging, low macrophyte species richness, reduced water quality) was dominated by a centrarchid (bass, sunfish) assemblage. Both Poe et al. (1999) and (Bryan and Scarnecchia 1992) found that fish species richness was positively correlated with macrophyte species richness. Lange (1999) provided evidence that residential shoreline development is “a likely agent in causing system-wide disruption to fish...” Sites with combinations of development structures (i.e., dock and bank stabilization) had low fish abundance and richness (Lange 1999). Lange (1999) generally concluded that the results of cluster analysis indicated that “sites associated with high occurrence of all forms of development and low occurrence of vegetation, tended to have the lowest total abundance and species richness, regardless of observational scale.” Both Jennings et al. (1999) and Lange (1999) found that the scale of one’s observations affects conclusions, and the cumulative impacts of multiple development features may be substantial. With at least 81 percent of Lake Washington shoreline bulkheaded and at least 2.5 percent and 4 percent of the shallow-water habitat covered with residential piers in Lakes Sammamish and Washington, respectively, the potential for cumulative adverse impacts is significant.

It is within this context - that shoreline development in general degrades aquatic communities - that we examine the effects of individual structure types on those communities. Individual structure types often occur together, confounding inference about their respective impacts. Additionally, the extrapolation of results among systems can be uncertain due to the physical and biological differences between systems. However, a negative response to human disturbance and habitat alteration is consistent among diverse aquatic/marine communities.

Piers

As discussed above, bass utilize structural features (natural or artificial) for both foraging and spawning. Christensen et al. (1996) found a significantly negative correlation of lakeshore development with CWD in 16 lakes in northern Wisconsin and upper Michigan. Qualitative observations in Lake Washington indicate that little woody debris can be found along developed sections of the shoreline. Piers provide alternative sources of shade, overhead cover, and in-water structure (piles and boatlifts) that bass could utilize for foraging and spawning, in the absence of natural features. Observations by Stein (1970) and Pflug (1981) in Lakes Washington and Sammamish respectively, indicate that bass do occasionally occupy piers.

Freshwater: As expected, the literature review did not produce any studies of the relationship between piers and bass predation on juvenile salmonids. Studies from freshwater systems also lacked evidence for predator aggregations associated with piers. A study in Lake Washington found no significant differences in catch-per-unit-effort of any fish species between under-pier and control sites, but few bass were captured (White 1975). However, evidence for the use of piers by bass in Lakes Washington and Sammamish was located. Stein (1970) reported that largemouth bass were commonly found under piers in Lake Washington during the spring, but considered natural cover to be their preferred habitat. Unpublished results of a study by the Muckleshoot Indian Tribe in Lake Sammamish indicated that smallmouth bass were preferentially locating nests proximate to residential piers (Malcom, pers. comm., 13 April 2000). Although residential piers only covered approximately 13 percent of the nearshore zone (0-20 m from shore), 32 percent of the smallmouth bass nests were within 2 meters of piers, and 54 percent were within 2 meters of a pier or other artificial structure (i.e., isolated piles, water pipes, boat launch rails, tires, rebar) (Malcom, pers. comm., 13 April 2000). Shade was apparently not a critical attraction feature of piers for spawning smallmouth bass; instead, the attraction was to physical structure provided by piers, further evidenced by the location of nests adjacent to non-shading structures such as isolated piles (Malcom, pers. comm., 13 April 2000). This finding does not indicate that shade was unimportant to *foraging* smallmouth bass, only that bass were not preferentially locating nest sites in shady locations. The findings of Malcom (pers. comm., 13 April 2000) corroborate the findings of Vogele and Rainwater (1975), who also found that smallmouth bass nests were not closely associated with sheltered habitat in Bull Shoals Reservoir. The majority of smallmouth bass nests were beside submerged stumps in gravel and rubble substrates, while largemouth bass nests were either under artificial brush shelters or adjacent to a submerged log, rock, or tree base (Vogele and Rainwater 1975).

Additional evidence for a connection between bass and piers comes from unpublished data. WDFW personnel electrofishing for bass in 50 to 70 local (western Washington) lakes observed that bass were more often associated with natural structures such as brush piles, beaver lodges, and overhanging willows and, to a lesser degree, were found under docks or adjacent to piles, but empirical evidence to support these observations was not collected (Bonar, pers. comm., 13 June 2000). Qualitative observations by Bonar (pers. comm., 13 June 2000) suggest that structures concentrate bass in lakes where structure is limiting. One-third of the largemouth bass in Lake Baldwin, Florida showed a significant preference for piers in the absence of aquatic vegetation (Colle et al. 1989).

Two studies (that did not include bass) of freshwater fish use of piers did not find evidence of predator aggregation. Ward et al. (1994) did not find a relationship between shoreline development (including piers) and northern pikeminnow predation on outmigrating chinook and steelhead in the lower Willamette River, Oregon. Northern pikeminnow were more abundant along undeveloped than developed reaches of the lower Willamette River (Ward et al. 1994). In a study in Lake Tahoe by Beauchamp et al. (1994), day and night patterns in fish density and species composition were similar

between docks and open shoreline. Neither of these studies specifically investigated the relationship between piers and bass or cutthroat predation on salmonids.

Marine or Estuarine: Several studies from East Coast estuarine systems provided contradictory results that may reflect differences in systems and study designs. Low fish abundance and species richness under piers compared with pile field and open-water sites (Able et al. 1998), and low fish growth rates under piers compared with pier edges and open-water (Duffy-Anderson and Able 1999) have been reported in the Hudson River estuary. Conversely, in the Rhode River estuary, Toft et al. (1995) reported significantly greater abundance of several fish species under piers than 10 meters away. Local studies of estuarine systems that included salmonids were more consistent, indicating that juvenile salmonids forage under piers, and that predator aggregations were not observed. Ratté (1985) reported that juvenile chinook and coho salmon foraged under a large commercial pier in the Commencement Bay estuary, Washington, and that no aggregations of predators or selective predation on salmonids was observed. Juvenile chum and pink salmon were attracted to a large pier complex (submarine berth) during daylight at the U.S. Navy Bangor submarine base on Hood Canal, but aggregations of predators were not observed (Prinslow et al. 1979 and 1980). Findings from marine and estuarine systems should be considered to be the least applicable to the Lake Washington system.

Rock Crib Structures: Several studies have examined the effects of shoreline development in general on various indices of fish community structure (e.g., Poe et al. 1986; Brown 1998; Lange 1999; Jennings et al. 1999). Of these studies, Brown (1998) and Lange (1999), while examining the effects of shoreline development in general, included analysis of fish response to moorage structures in Lake Joseph and Lake Simcoe, Ontario, respectively. The moorage structures in these Ontario lakes differ somewhat from the typical structures found in local lakes. The majority of the moorage structures on Lake Washington are piers supported by piles (typically wood piles, 20-30 cm in diameter). The majority (> 85%) of the structures in Brown's (1998) study on Lake Joseph were crib structures, that is docks or boathouses supported by log cribs filled with boulders; only 8 percent were piers supported by piles. While not a typical design for residential piers in Lake Washington, these types of structures are common in the Ship Canal. Lange (1999) examined fish response to a variety of structure types at three different scales in Lake Simcoe, differentiating between structures supported by piles ("temporary docks") or cribs ("permanent docks"). Significant negative effects of temporary docks (most similar to our local piers) on fish richness or abundance in Lake Simcoe were not observed at any scale (Lange 1999). Crib-supported docks did have a significant positive effect on fish abundance at the two largest scales (244 m, 488 m), but not at the smallest scale (122 m) (Lange 1999). Brown (1998) also found that crib structures increased densities of forage fish (< 100 mm) in the littoral fringe on exposed shorelines or in areas where CWD had been removed. Brown (1998) speculated that interstitial spaces within crib structures provided refuge from waves and predation for small fish along exposed shorelines. In Lake Tahoe, up to ten-fold higher densities and a greater diversity of small fishes were associated with rock-crib structures, whereas fish assemblages around pile piers did not differ from paired adjacent areas without shorezone structures (Beauchamp et al. 1994); however, bass and other centrarchids

were absent from the main basin of Lake Tahoe at the time of this study. Brown (1998) observed (qualitatively) large numbers of piscivores beyond the littoral fringe around crib structures in Lake Joseph, and suggested that their presence was a response to the abundance of forage fish.

Conclusions: These findings, when considered with existing knowledge of bass ecology, suggest that bass prefer natural cover for foraging, and preferentially site nests adjacent to structures, but bass utilize piers, piles, and other artificial structures for foraging and nesting in lieu of natural cover or structure. Piers and piles differ from natural cover/structure elements such as brush piles, primarily in their lack of structural complexity. This difference is critical for prey fish, which rely on structural complexity for survival in the presence of predators, particularly mobile predators such as cutthroat trout. In developed lakes, piers become the dominant structural features at the expense of natural complex structures such as woody debris and emergent vegetation. That bass and other predators gain an advantage over prey fish in structurally simple environments is substantiated by findings that bass (especially smallmouth bass) persist or thrive along developed shorelines, while other species decline (Brown 1998; Bryan and Scarnecchia 1992; Poe et al. 1986; Lange 1999). Recognition of this advantage to bass and other predators necessitates a cautious scrutiny of proposed new and modified piers while awaiting results from the direct studies on the relationship between piers and other shorezone structures, and bass predation on salmonids that are currently underway (i.e., the studies of Roger Tabor, USFWS; Kurt Fresh, WDFW; Rod Malcom and Eric Warner, Muckleshoot Tribal Fisheries). Regardless of the development proposal, any project that would potentially reduce the structural complexity of the shorezone should be considered likely to adversely affect ESA-listed salmonids. New piers should also be considered as new, structurally simple habitat elements that provide cover and structure to spawning and foraging bass, and perhaps other predators. Replacement piers and pier modifications should be viewed by regulatory agencies as opportunities to regain some habitat function and minimize overwater coverage (see "Productivity" section below).

It is useful to ask what features of piers make them attractive to bass in the lacustrine environment. Male bass preferentially locate nests adjacent to structural features such as rocks or logs, apparently to reduce the perimeter that must be guarded or to provide visual isolation from nearby conspecifics (Heidinger 1975). Thus, for spawning bass, pier elements that protrude from the substrate (i.e., piles, boatlifts, etc.) may be attractive. The initial data suggests that this is the case for smallmouth bass (Malcom, pers. comm., 13 April 2000). The structure provided by piers and boatlifts may potentially increase spawning habitat and/or reproductive success of bass.

It is less clear what pier features primarily attract foraging bass: shade, overhead structure independent of shade, vertical structural elements, or a combination of features. The finding that largemouth bass were more likely to hover under large swimming floats than small study floats, and that fish were generally not observed under "sham" floats consisting of wood frames only, suggests that shading may be key, and that the dimensions of the area shaded may also be important (Helfman 1979). Anecdotal evidence from anglers supports the hypothesis of the importance of the

dimensions of the overwater area, as fishing efforts are directed at the portions of piers with the most surface area (broad ells). Despite circumstantial evidence and an intuitive connection, direct evidence for a correlation between pier shade-production and bass occupation was not located in the course of this review. The circumstantial evidence does indicate a need for both further study and critical appraisal of pier design in the interim. Studies investigating the effects of light-transmitting devices (prisms, grating) on bass use of piers would also be useful. Prisms are currently being mandated as mitigation for overwater coverage by several municipalities on Lake Washington and Lake Sammamish. However, their ability to reduce bass attraction to piers has not been proven, despite their ability to transmit ambient light.

One additional note on piers in Lake Washington is the prevalence of illegal lake-water withdrawals. Many waterfront property owners have illegal pump systems for withdrawing water from Lake Washington. This may be a critical source of water loss during the dry season, and could ultimately affect fish passage at the Ballard locks.

Bulkheads

Studies of the relationship between shoreline armoring and predation on juvenile chinook or coho salmon in Lake Washington and Lake Sammamish were not found. Cautious conclusions about the effects of shoreline armoring on predation can be drawn from studies of predator-prey interactions and the habitat use by small non-salmonids in other north-temperate lakes, and studies of salmonid habitat use in large rivers and reservoirs. While no direct links were identified between predation and bulkheads, an intuitive connection exists between the loss of complex, shallow-water foraging habitat for juvenile salmonids and an increased exposure to potential predation. Bulkheads could directly affect predation on juvenile salmonids by eliminating shallow-water refuge habitat or, indirectly, by the elimination of shoreline vegetation and in-water woody debris that generally accompanies bulkhead construction. The importance of shallow-water refuge habitat and complex habitat features to small fish has been discussed above. Juvenile fall chinook salmon in the Columbia and Snake Rivers and (preliminary results suggest) Lake Washington have demonstrated a preference for shallow, low-angle shorelines, although the motivation for this observed preference has not been fully investigated (Key et al. 1994a and 1994b; Garland and Tiffan 1999; Tabor, pers. comm., 9 June 2000). Placing bulkheads waterward of ordinary high water (OHW) eliminates the shallow water identified by Collins et al. (1995b) as critical for foraging, refuge, and migration of small fish (< 100 mm). The simplification of the shoreline (i.e., removal of CWD and shoreline vegetation) that typically accompanies bulkhead construction (Christensen et al. 1996) further reduces refuge habitat. Lange (1999) found that bank stabilization (i.e., various forms of erosion control structures referred to as "bulkheads") was negatively correlated to fish abundance and species richness at all spatial scales investigated in Lake Simcoe, Ontario.

The finding that both fish species richness and abundance were negatively correlated with bulkheads at every scale (Lange 1999) indicates that fish in Lake Simcoe generally avoid bulkheads. Juvenile fall chinook in the Columbia and Snake Rivers were found to avoid riprap shorelines (Key et al. 1994a and 1994b; Garland and Tiffan 1999). Young-

of-year bass in Lake Joseph, Ontario, did not exhibit a preference for spatially complex habitat in Brown's (1998) study, and may represent an exception to the avoidance theory. Jennings et al. (1999) found that species richness was greater along riprap bulkheads than smooth vertical bulkheads in 17 Wisconsin lakes. Riprapped shorelines in the study by Jennings et al. (1999) also had greater species richness than unarmored shorelines (does not imply natural, only the lack of armoring structure), but they cautioned that the findings were an artifact of the scale of the investigation, the heterogeneity of the unarmored sites, and the increased effort required to obtain estimates of species richness at unarmored sites. Converting lakes entirely to riprapped shoreline would ultimately reduce species richness at the lake scale, but in situations where hard shoreline armoring is necessary, riprap would be preferred over vertical walls for fish habitat (Jennings et al. 1999) (the specifications of the various shorelines compared by Jennings et al. 1999 are available in a report that was not obtained for this review - Jennings et al. 1996). However, Jennings et al. (1999) were not considering a situation where an endangered species could be potentially jeopardized by the shoreline protection method. Riprap may provide greater habitat heterogeneity and ultimately greater species richness than smooth vertical bulkheads, but the effects of habitat heterogeneity on predation were not investigated.

The use of riprap shoreline protection in Lake Washington could provide concealment habitat to the most abundant native piscivores - cottids. Tabor et al. (1998) reported predation on salmonid juveniles by sculpins greater than or equal to 50 mm in length. Few such sculpin were found over sand/mud substrates relative to gravel/cobble substrates in Lake Washington, due to the lack of refuge habitat in sand/mud substrate (Tabor et al. 1998). In Lake Washington and the Cedar River, cottid size was generally positively correlated with substrate size, and riprap shorelines had large cottids relative to sites with smaller substrate particles (Tabor et al. 1998). Bulkheads in Lakes Washington and Sammamish are typically nearly vertical, and constructed of large boulders with large interstitial spaces. The large interstitial spaces found within riprap shorelines provide concealment to abundant, large native cottids.

In summary, bulkheads eliminate shallow-water habitat and complex habitat features that may function as critical prey-refuge for juvenile chinook and coho salmon. Bulkheads have been shown to reduce the diversity and abundance of all fish species except smallmouth bass in other north-temperate lakes. Riprap bulkheads, which provide interstitial spaces that can be utilized by a variety of invertebrate and fish species, may provide refuge habitat for piscivorous sculpin, while also eliminating the shallow water refuge for juvenile salmonids. No evidence was found for positive effects of shoreline armoring on aquatic species.

Lighting

Studies of the effect of pier lighting, on predation of juvenile salmonids in lakes were not found in the course of this review. Western grebes have been observed foraging at night around artificial lights in Lake Washington (Tabor, pers. comm., 9 June 2000). Grebes, blue herons, and other birds have been observed feeding at night on the Cedar delta in the portion that is lit up by The Boeing Company lights (Warner, pers. comm., 7

July 2000). Prey behavior can influence light-mediated predation rates by both increasing exposure to predators by slowing migration rates through rivers, and reducing capture efficiency by increasing avoidance behavior. In freshwater laboratory experiments, Tabor et al. (1998) found that prickly and torrent sculpin were capable of preying on sockeye fry in complete darkness, but predation rate declined with increasing light intensity. An increase in predator avoidance ability by sockeye fry with increasing light intensity may explain this inverse relationship (Tabor et al. 1998). Petersen and Gadomski (1994) observed a similar relationship (decreasing predation rate with increasing light intensity) between northern squawfish (northern pikeminnow) and juvenile chinook salmon, and offered the same explanation, as did Howick and O'Brien (1983) for bass-bluegill interactions and Mazur (Univ. of Washington, unpubl. data) for juvenile trout responding to lake trout. Alteration (slowing) of migratory behavior and subsequent increased sculpin predation rates on sockeye fry with increasing light intensity were observed in simulated stream experiments (Tabor et al. 1998).

The nocturnal behavior of juvenile chinook and coho and their predators in Lake Washington and Lake Sammamish, and their response to lighting is poorly understood. Chinook fry have been observed primarily resting on the bottom during night snorkel surveys in Lake Washington (Tabor, pers. comm., 11 July 2000). Reimers (1971) observed that juvenile chinook delayed downstream migration until the darkest part of the night in the Sixes River, Oregon, a result similar to the delay in sockeye migration with increasing light observed by Tabor et al. (1998) in the Cedar River. Key et al. (1994b) found that few juvenile fall chinook were caught during night sampling (relative to diurnal catches) in McNary Reservoir, and proposed that the fish were inactive during night seining. Studies of whether or not chinook or coho juveniles exhibit nocturnal inactivity in Lake Washington were not located. It is possible that the artificial ambient lighting regime in the urbanized basin of Lake Washington may produce uncharacteristic behavior in both juvenile salmonids and their predators. Until more information is available, one should not assume that lighting of overwater or shoreline structures does not affect predator-prey interactions.

Studies from a marine system indicated that wharf lighting could attract juvenile chum and pink salmon (Salo et al. 1977; Prinslow et al. 1979; Prinslow et al. 1980). Significant predation on juvenile chum salmon was not observed in the area of a lighted wharf in Hood Canal, Washington (Prinslow et al. 1980). Whether the behavior of juvenile chinook and coho salmon and their predators in freshwater would behave similarly to chum and pink salmon and their predators in the marine environment remains uncertain and should not be assumed.

Effects on Productivity

Light Intensity and Primary Production

Studies from Lake Washington: Overwater structures reduce the amount of light available to phytoplankton and aquatic macrophytes, which can ultimately reduce primary production. White (1975) compared light intensity and primary

production/biomass ratios of phytoplankton at sampling stations under overwater structures, and at control stations outside of overwater structures in Lake Washington. As expected, light intensity was higher at all control stations compared with intensities measured under overwater structures. Surface light intensities at open water stations on sunny days inhibited phytoplankton production in the upper 60 cm of the water column. Production/biomass ratios measured at stations outside of piers reached maximum levels at depths from 1 to 2 meters. Surface phytoplankton production/biomass ratios under narrow residential piers or at the edge of large overwater structures exceeded those measured at open water stations due to the reduction in surface light intensity. However, phytoplankton production/biomass ratios from outside stations exceeded those from under-pier stations from a depth of about 60 cm to the maximum depth measured. White (1975) suggested that, while narrow residential piers do not significantly reduce surface phytoplankton production, the reduction is inversely proportional to shading, as illustrated by the low production/biomass ratios observed under a residential pier with skirting, a boathouse, and an overwater apartment complex.

White (1975) did not comment on the reduced production/biomass ratios of shaded sites at depth, compared to open-water control sites, nor did he measure periphyton or macrophyte abundance and production under and outside of overwater structures. The reduced light intensity observed under all overwater structures when compared with open-water control sites, resulted in reduced total water-column phytoplankton production/biomass ratios, and likely reduced periphyton and macrophyte production as well. White (1975) did not attempt to measure the cumulative loss in primary productivity that would ultimately result from the collective overwater coverage of numerous residential and commercial structures. While the loss in productivity from a single, narrow, residential structure may be insignificant, the cumulative impact of thousands of narrow piers is likely to be a significant reduction in primary productivity.

A comparison of the benthic invertebrates under, and outside of, overwater structures in Lake Washington revealed complex patterns of abundance and/or distribution in the benthic community, with some organisms more abundant under, than outside of, piers in spring and less abundant in fall (White 1975). Possible explanations for the observed patterns include variations in phototaxis with life-history stage and differences in forage availability during fall and spring (White 1975). Macrophytes were absent or sparse under piers; grazing invertebrates would be found outside piers where macrophytes were abundant in the fall, but under piers, where they could graze on periphyton, in the spring when macrophytes were sparse (White 1975).

Studies from other systems: Loflin (1995) reported that docks in two Florida marine locations produced distinct areas in their shadow that were nearly devoid of seagrass, and that were significantly correlated with total dock surface area. Shading from docks also produced changes in seagrass species composition and reduced epiphytic loading on grass blades (Loflin 1995). The percent cover of epifauna on primary kelp blades was less under piers than on perimeter piles at a marine site in Portsea, Australia (Fletcher and Day 1983). Shaded piles in Sydney, Australia had different epibiotic assemblages than unshaded piles or adjacent rocky reefs (Glasby 1999a; 1999b).

Epibiotic assemblages on unshaded piles were composed of filamentous and foliose algae (primary producers), while communities on shaded piles were composed of filter feeders (Glasby 1999b).

At a Long Island Sound location, Iannuzzi et al. (1996) predicted that construction of an 800-slip marina would reduce macroalgal production by 17%, but that reduction would be compensated for by microalgal production on the hard attachment surfaces of the marina. The Long Island Sound site was a high-energy marine system with radical changes in energy, sediment composition, and turbidity expected to result from construction of the marina (Iannuzzi et al. 1996). The references from this and the preceding paragraph were studies of marine systems. Extrapolating their results to freshwater systems may not be entirely appropriate. However, the responses of macro- and microalgae to reductions in light intensity resulting from overwater structures would be expected to be similar among systems.

Another expected effect of shoreline structures includes the loss of allochthonous nutrient input resulting from the removal of shoreline vegetation. France and Peters (1995) estimated the annual litter input to a northern Ontario lake from 1 m of forested shoreline to be 32-g dry weight. Allochthonous litter input per unit offshore distance was related to the size of riparian trees, their proximity to the shoreline, and the elevation of their canopy (France and Peters 1995). Riparian deforestation resulted in annual reductions of up to 17.8 g of dissolved organic carbon and 2.9 g of total phosphorous per meter of shoreline in oligotrophic Canadian shield lakes, reducing primary production by up to 9 percent (France et al. 1996). The implications are less serious for mesotrophic urban lakes where increased phosphorous loading is problematic. However, Eggers et al. (1978) concluded that the littoral benthic community in Lake Washington was resource limited. Increasing allochthonous litter input from shoreline vegetation on Lakes Washington and Sammamish could increase forage for juvenile salmonids and the forage fish of bull trout in the littoral zones. The permanent removal of shoreline vegetation for bulkhead construction, and for unobstructed views may affect the forage base of ESA-listed salmonids by reducing allochthonous input to the littoral zone. An incidental effect of shoreline vegetation removal is likely to be an increase in diel temperature fluctuation in the littoral zone due to loss of shade (Steedman et al. 1998), especially in littoral areas that have been isolated from the main water mass by artificial structures.

In summary: Evidence from Lake Washington indicates that single narrow residential piers do not significantly reduce surface planktonic primary productivity, but the productivity losses below 60 cm, and from reduced macrophyte productivity were not investigated. Cumulative reductions in primary productivity resulting from numerous overwater structures were not measured, but could be substantial. Comparisons of benthic primary production (epiphytes) in Lake Washington were not made. The effects of overwater structures on invertebrate production in Lake Washington have not been conclusively established. Evidence from marine systems indicated that epibiotic assemblages are affected by shade, with primary producers being replaced by consumers. Removal of shoreline vegetation could reduce allochthonous input of nutrients.

While the deck prisms mentioned above that are being mandated by some municipalities as mitigation for overwater coverage have not been tested for efficacy at reducing bass attraction to piers, they do transmit ambient light. Thus, they may be useful for retrofitting existing piers to restore the primary productivity loss from those structures.

Aquatic Vegetation Control

Control of “nuisance” aquatic macrophytes by lakeshore property owners is another potential source of lost productivity. Aside from the importance of macrophytes in primary production, numerous studies have indicated the importance of littoral vegetation for increased fish production (e.g., Randall et al. 1996). The most significant effect on fish of development of the shoreline of an Iowa lake was the removal of aquatic macrophytes by lakeshore residents (Bryan and Scarnecchia 1992). Methods for vegetation control include active harvesting, chemical controls, and covering of the substrate with materials that block vegetation growth. Engel (1984) compared removable and non-removable materials for aquatic vegetation control, and concluded that even removable screens with large pore sizes nearly eliminated benthic invertebrates. Despite the generally undesirable effects of macrophyte removal, there are situations where the reduction of aquatic macrophytes may benefit fish. Dissolved oxygen (DO) levels under dense patches of Eurasian milfoil and fragrant white water lily (*Nymphaea odorata*) were lethal to caged steelhead trout in Lake Washington in a study by Frodge et al. (1995). Native species of aquatic macrophytes found in Lake Washington typically do not form large monotypic stands with dense surface mats such as those found to reduce DO concentrations (Frodge, pers. comm., 10 July 2000).

Physical Effects

Lake Washington: The physical effects of bulkheads on benthic organisms are expected to depend upon both the designs of bulkheads and the material from which bulkheads are constructed. White (1975) compared benthic invertebrate abundance at various depths in front of bulkheaded shorelines, developed shorelines without bulkheads, and along natural shorelines in Lake Washington. White’s (1975) results were inconclusive, indicating no clear trends in invertebrate abundance. However, White (1975) did not report the position relative to OHW of the bulkheads in his study, nor did he measure invertebrate abundance immediately waterward (at the toe) of bulkheads.

Results from White’s (1979) study provide inconclusive evidence for an adverse affect of bulkheads on the benthic community within local lakes. Smooth vertical structures would be expected to reflect wave energy in a non-random manner. Complex non-vertical bulkheads, such as those constructed of boulders, would be expected to reflect wave energy in a random manner. Reflected wave energy would produce a chaotic, high-energy environment for epibenthic and infaunal invertebrates within a zone adjacent to the bulkhead. The benthic community within this zone would be expected to have lower invertebrate abundance, richness, and diversity than lower energy zones.

Estuarine Systems: Only one study was obtained that specifically examined the effects of shoreline armoring on infaunal organisms, and the study system was an estuary with a sand beach. Spalding (1998) concluded that sediments were finer and better sorted immediately adjacent to bulkheads than at similar elevations at control sites, and that meiofauna densities increased with distance from bulkheads in a New Jersey estuary. Bulkheads had the greatest influence on sediment characteristics and meiofaunal densities when located waterward of wavebreak, or when subjected to high wave energy (Spalding 1998). Spalding's (1998) study was of sandy estuarine beaches with tidal influences - drastically different from the typical beaches in local lakes; she was also sampling organisms within the swash zone of the beach. Thus, Spalding's (1998) results may have limited applicability to our local lakeshores. However, the findings of Spalding's (1998) study should not be ignored. Chironomid larvae are the primary prey items for juvenile chinook and coho salmon, and could be adversely affected by bulkhead-induced changes in sediment composition. A lack of affect of bulkheads on infaunal organisms should not be concluded without further investigation. Byrne (1995) described a year-long study designed to test whether there was a difference in the species composition and abundance of macroinvertebrates and fish inhabiting bulkheaded and non-bulkheaded shore zones in a manmade estuarine lagoon. Results from Byrne's (1995) study could not be obtained, only a summary of the study design.

A comparison of the macroinvertebrate community structure on rip-rap bulkheads and smooth retaining walls was conducted using simulated rip-rap and patio blocks of similar surface area placed along three different shorelines (rip-rap, vertical retaining wall, and natural shoreline) in three Wisconsin lakes (Schmude et al. 1998). As expected, the complex artificial substrate had significantly greater abundance and taxa richness of macroinvertebrates than the two-dimensional patio blocks, but no significant differences in abundance or richness were observed among shoreline types.

In summary, only one study investigating the effects of shoreline armoring on benthic organisms in freshwater was located, and it did not provide conclusive evidence for an adverse affect. The findings of Spalding (1998) indicating a change in sediment composition and meiofauna density in front of bulkheads along sandy estuarine shorelines imply that bulkheads could adversely affect benthic organisms in freshwater lakes, and that further investigation is necessary.

Effects on Fish Migration

There is concern that bulkheads and piers may interrupt the migration of juvenile salmonids along the shorelines of Tri-County lakes. No studies investigating the migration of chinook and coho salmon along armored shorelines or under piers in lakes were located during this review. Chinook fry in Lake Washington appear to avoid riprap and cobble shorelines (Tabor, pers. comm., 9 June 2000). Studies of juvenile fall chinook in the Columbia and Snake Rivers found that they avoid riprap shorelines (Key et al. 1994a and 1994b; Garland and Tiffan 1999). Studies of non-salmonids in other north-temperate lakes were located. Collins et al. (1995b) observed that daytime small fish (<100 mm) traffic in the shallow "fringe zone" along natural shorelines in three Ontario lakes was 2.5 times higher than along shorelines with manicured lawns. The

shorelines with manicured lawns in Collins' (1995b) study were not bulkheaded, but did have characteristics typical of bulkheaded shorelines - no overhanging vegetation or complex woody debris. Juvenile salmonids and small fish in general may be reluctant to traverse shoreline habitat that does not provide prey refuge in some form (i.e., shallow water, overhead cover, and complex structure). Jennings et al. (1999) found that vertical bulkheads had the greatest average water depth compared with riprap and unarmored shorelines in Wisconsin lakes. Vertical, smooth bulkheads in deep water without overhanging vegetation would represent the least favorable passage and forage conditions.

Studies from marine/estuarine systems have concentrated on the effects of marinas. Concerns about juvenile salmonids and baitfish entrapment, and subsequent increased predation were unconfirmed; schooling of juvenile salmonids in marinas appeared to be volitional, and the primary concern was water quality related to poor marina design (Cardwell and Koons 1981). Taylor and Willey (1997) found no evidence for disrupted migration of juvenile chinook or coho salmon, or increased predation in an Elliott Bay marina, but theirs was an observational study only, not a controlled study. Heiser and Finn (1970), in a qualitative study, observed that juvenile pink and chum salmon were reluctant to migrate past a vertical or near-vertical bulkhead, but that riprap bulkheads with an approximately 45° slope did not hinder passage. The critical differences identified by Heiser and Finn (1970) between the vertical bulkheads and riprap bulkheads were that the vertical structures lacked complex refuge and shallow water, features preferred by migrating juvenile pink and chum salmon. These observations may not apply to freshwater lakes. Bulkheads in Lakes Washington and Sammamish are typically nearly vertical and constructed of large boulders, unlike the riprap bulkheads in Heiser and Finns' (1970) report.

Piers did not affect migration of sub-yearling or yearling chinook migrating through the lower Willamette River in Oregon (Ward et al. 1994). The only indication of an influence on behavior was that sub-yearling chinook were found uncharacteristically close to shore in shallow water around a wharf with closely-spaced piles that created a large backwater with a soft bottom (Ward et al. 1994). The chinook in the Ward et al. (1994) study may have been responding more to the velocity refuge provided by the piles than other factors; velocity refuge would not be a factor in the lacustrine environment. Ratté (1985) observed no reluctance in juvenile chinook or coho to swim under large commercial piers in Commencement Bay, Washington. Outmigrating pink and chum salmon in Hood Canal were attracted to the explosives-handling wharf at the U.S. Navy's Bangor submarine base, but migration was not significantly delayed (Prinslow et al. 1979; Prinslow et al. 1980). Overall, none of these studies provided evidence for a delay in migration caused by piers in riverine or estuarine systems. No studies of the effects of piers on salmonid migrations in lakes were identified.

Summary: No studies were located that specifically investigated the effects of piers and armored shorelines on the migration of juvenile chinook and coho salmon along lakeshores. Evidence from the Columbia and Snake Rivers indicates that juvenile chinook avoid riprap shorelines, and studies from north-temperate lakes in Ontario indicate that small non-salmonids are less likely to traverse structurally simple

shorelines than naturally complex shorelines. Collectively, these studies indicate that concern over a migrational disruption caused by bulkheads in lakes is warranted. Information about pier-related migrational disruptions in lakes was not obtained. The question remains whether juvenile salmonids in lakes migrate under, or otherwise utilize, piers, or if they avoid them and/or traverse their perimeter.

Effects of Recreational and Construction Activity

Construction Activity

The activities necessary for construction of shorezone structures can have direct impacts on ESA-listed salmonids. As enumerated above, the primary impacts of construction activities considered by NMFS, aside from the actual crushing of individual fish, are turbidity produced during pile driving and bulkhead construction/removal, and the effects of shock waves produced by pile driving. Numerous studies have documented the detrimental affects of chronic exposure of salmonids to turbid water in riverine environments (e.g., Sigler et al. 1984). Physiological effects are only apparent after prolonged exposure (3-5 days in the case of Sigler et al. 1984), and chinook and coho juveniles will emigrate from turbid water (Scrivener 1994; Murray and Rosenau 1989; Skeesick 1970; Sigler et al. 1984). However, even low turbidity levels can produce a variety of sublethal effects (i.e., reduced survival, reduced growth, reduced food conversion, reduced feeding, altered diet, stress, disease, avoidance, displacement, altered behavior [including predator avoidance behavior]) that could ultimately reduce the fitness of the individual (reviewed in Lloyd 1987). Further, even minor increases in turbidity in a clear lake can result in significant reductions in primary productivity both through reduction in light penetration and physical coverage of the benthos (Lloyd 1987). Contractors conforming to BMPs specified by WDFW utilize floating “sedimentation control curtains” to contain turbid water, allowing turbidity to settle before removing the curtain. Turbidity from an individual construction activity would not represent a permanent sediment source and would not produce conditions of chronic exposure, but it could be acute. The possibility also exists that fish could be trapped within the sedimentation control curtain, and thus exposed to potentially lethal turbidity levels. Nevertheless, to minimize potential impacts, the Services restrict construction activity to periods when ESA-listed salmonids are least likely to inhabit the area of construction. In the case of NMFS, “allowable construction windows” for the protection of chinook have not been established.

Pile Driving: The expected effects of pile driving on juvenile salmonids can be generally summarized as disruptions of normal behavior. The shock waves generated by pile driving could potentially disrupt the foraging behavior of juvenile salmonids, cause them to move away from the shoreline or exhibit a startle response, or delay migratory progress. Only one published study (Feist et al. 1996) of the effects of pile driving on juvenile salmonids was located. Feist et al. (1996) studied the effects of vibratory and drop hammer pile driving on the behavior of juvenile chum and pink salmon in Puget Sound at the Everett Homeport. They determined that salmonids were capable of detecting the sound of drop-hammer pile driving at least 600 meters away, and that the sound was at least 20 dB above ambient levels at 593 meters. Data collected from the

vibratory hammer location was insufficient to allow appropriate analysis; thus, only data analysis from the drop-hammer location was reported. Juvenile pink and chum salmon did not change their distance from shore or cease foraging in response to pile driving, but there were significant differences in the distributions and sizes of fish schools, and behavior within schools on pile driving days versus non-pile-driving days. On pile driving days, there were nearly half the number of fish schools on the construction side of the site than on non-pile-driving days. One concern with pile driving is that the sound will “mask” the sound of an approaching predator, or that salmon would become habituated to the sound and fail to hear the approach of a predator. Qualitative observations indicated that fish had habituated to the sound of pile driving (Feist et al. 1996).

Caveats about the study by Feist et al. (1996) are that this was a study of pink and chum juveniles in a marine environment, not chinook juveniles in freshwater. Second, the study did not investigate the impacts of pile driving on adult salmon behavior. Third, the study investigates one site during one season, and relies predominantly on human observation in the assessment of fish behavior. The extrapolation of these results to other locations, particularly freshwater systems with different species and age classes, may not be valid. Despite these caveats, the implications are that salmonids do respond to pile driving, and until pile-driving effects on freshwater systems are investigated, regulators should assume the potential for adverse affects on ESA-listed salmonids in lakes.

Recreational Activities

The effects of recreational activities on ESA-listed salmonids are unknown. Direct effects of boating or swimming on salmonids are expected to be disruptions of salmonid behavior or physical injury due to contact with the boat or entrainment in the propulsion system. Migrating adult chinook in the ship canal have been tracked at depths as shallow as 1 meter (Warner, pers. comm., 7 July 2000), exposing them to potential boat contact. Beak Consultants Incorporated (1998) cited Weitkamp (1982) as indicating “that juvenile salmonids in marine environments near piers returned to their normal behavior immediately after a boat passed.” The implications of this statement are that juvenile salmonid behavior was temporarily disrupted by boats. However, we were unable to obtain a copy of the Weitkamp (1982) document, and thus cannot verify this implication or describe the conditions under which the potential disturbance occurred. Mosich and Arthington (1998) reviewed the impacts of power boating and water skiing on lakes and reservoirs, and identified direct boat contact and propeller action as sources of injury to aquatic organisms. In the only direct reference to salmonids, Sutherland and Ogle (1975) cited in Mosich and Arthington (1998) describe significant mortality of chinook eggs resulting from pressure fluctuations created by passing jet boats in shallow water.

Indirect effects of boating on salmonids involve adverse impacts to habitat quality. Substantiated impacts that were discussed at a workshop held at Woods Hole Oceanographic Institute in 1994 include: sediment and contaminant resuspension and resultant turbidity, laceration of aquatic vegetation with loss of faunal habitat and

substrate stability, toxic effects of chemical emissions of boat engines, increased turbulence, shearing of plankton, shorebird disturbance, and the biological effects of chemically treated wood used in dock and bulkhead construction (Crawford et al. 1998). The complete proceedings of the workshop (a book) was unavailable, but the abstract was obtained. The abstract indicated that, while the above potential impacts had been identified, impacts remain "inadequately defined and described," and that sufficient evidence exists to infer that recreational boating is not an environmentally benign activity (Crawford et al. 1998). Mosich and Arthington (1998) in their review of literature on lakes presented a list of impacts similar to that of Crawford et al. (1998), but also added chemical impacts from hydrocarbons, erosion of banks and destruction of emergent vegetation, introduction of plant fragments and plankton via jet-propulsion systems and boat propellers and trailers, and the biological impacts of sediment resuspension and erosion (i.e., clogging of respiratory structures of fish and invertebrates, reduced photosynthesis, increased nutrient availability). Crawford et al. (1998) expressed concern that the potential for impacts in temperate climates was exacerbated by the "unfortunate synchrony...between the peak season of boating and the occurrence of planktonic embryonic and larval stages of vertebrates and invertebrates in estuaries and coastal waters." This same concern has application to freshwater environments to some degree.

Bonham (1983) described the effects of boats and their wakes on river and canal shorelines with specific attention to the effects on emergent and submergent vegetation. Boat wakes erode shorelines and wash soil from the roots of emergent vegetation; emergent vegetation is subsequently uprooted by the wakes (Bonham 1983). Loflin (1995) reported scarring of seagrass flats from boat propellers. Asplund and Cook (1999) discussed the advantages and limitations of "no-wake zones" for protecting fragile lakeshore environments. Unfortunately, only the abstract of this document was obtained, so further comment on Asplund and Cook's (1999) discussion is not possible. Mosich and Arthington (1998) indicated that boat traffic close to shore had the greatest erosive affect. Collins et al. (1995b) found that feeding by small fish in Lake Rosseau, Ontario was suppressed by disturbance from boat wakes.

Literature on the impacts of recreational swimming on salmonids was not obtained. As mentioned above, the expected impact is a disruption of salmonid behavior. The rapid movements and splashing of a recreational swimmer would be expected to disturb the foraging behavior of salmonids to some degree. Recreational water use, including boating and swimming (people and pets), could occur independent of the construction and maintenance of docks and bulkheads at private residences. There is recognition that the construction and maintenance of those structures facilitates some recreational activities, and thus some recreational activities are an interrelated or indirect affect of the structures.

Conclusions and Recommendations

Conclusions

As expected, analysis of the literature collected for this review has both answered questions, and revealed new questions. The following will be a summary of the findings of this review arranged in two categories: what the review tells us, and what the review does not tell us.

What the literature review tells us.

1. Most chinook salmon enter Lake Washington as fry from January through March; they are closely oriented to the shoreline during this period, preferring shallow, sandy beaches. A second wave enters the lake as smolts in May and June. Smolt outmigration peaks in May and June, but extends at least through August. The majority of coho salmon enter the lake as smolts in May and June.
2. The primary predator of juvenile salmonids in Lake Washington is cutthroat trout. Their distribution closely overlaps that of chinook fry, subjecting age-0+ chinook to cutthroat predation for the duration of their rearing and migration.
3. The distributions of bass (smallmouth and largemouth) overlap that of chinook juveniles, from April through the end of smolt outmigration.
4. The majority of known bass predation on juvenile salmonids occurs in the Ship Canal. Smallmouth bass are the primary predator, preying most heavily on chinook juveniles due to their small size relative to sockeye and coho yearlings.
5. Bass are structurally oriented for both spawning and foraging, and smallmouth bass may prefer artificial structures such as piles for nest sites. Both species will utilize artificial structures in lieu of natural structure, and piers may concentrate bass in systems that lack natural structure (such as Lakes Washington, Sammamish and Union).
6. Small prey fish (i.e., chinook and coho fry) require complex habitat such as rootwads, undercut banks, overhanging vegetation, and CWD, and also shallow water as refuge habitat from the numerous predators within the Lake Washington system.
7. Piers are structurally simple, lacking the complexity necessary to function as prey-refuge habitat. Thus, they confer an advantage to predatory fish and birds over vulnerable juvenile salmonids.
8. Bulkheads are also structurally simple relative to undisturbed shorelines. Bulkhead construction generally entails the permanent removal of CWD and other complex features such as shoreline vegetation, which eliminates the sources of future CWD. Bulkheads waterward of OHW eliminate shallow-water refuge and foraging habitat of juvenile salmonids. Bulkheads with large interstitial spaces provide concealment

- habitat for sculpin, which prey upon juvenile chinook. Chinook fry appear to avoid bulkheads, which may affect their dispersal.
9. Shorezone development, and specifically the proliferation of bulkheads and bulkheads in combination with piers, consistently results in a reduction of fish and invertebrate diversity and the dominance of the fish assemblage by disturbance-tolerant species (centrarchids - primarily smallmouth bass), and the extirpation of rare species.
 10. Bulkheads interrupt the recruitment and transport of native sediment to lakes, resulting in both local and along-shore erosion. Piers also disrupt sediment transport. The resultant changes in sediment composition and distribution may affect bass and salmonid spawning (as well as other species), and could also affect the distribution and abundance of invertebrate prey items of juvenile salmonids.
 11. Piers reduce primary productivity by both phytoplankton, and macro- and micro-fauna. The amount of reduction is a function of the reduction in light intensity by the structure.
 12. Removal of shoreline vegetation can reduce the allocthonous input (terrestrial insects and detritus) to the lake, ultimately affecting productivity.
 13. Sediment and water contamination can result from leaching of PAHs and heavy metals from treated wood. Hydrocarbon input from two-stroke watercraft engines can be a significant factor in water quality reductions in lakes. However, in urban drainages, and Lake Washington in particular, outboard motors contribute less than 1 percent of PAH input. Chinook smolts exposed to an estuary contaminated with heavy metals and PAHs exhibited suppressed immune responses.
 14. Artificial lighting retards migratory progress of sockeye fry, subjecting them to increased predation. Lights from industrial areas in south Lake Washington facilitate nocturnal foraging by piscivorous birds.
 15. Pile driving can affect the distribution and behavior of juvenile salmonids in the marine environment over long distances, and appears to produce a habituation to the sound that could prevent a fish from sensing an approaching predator.
 16. Minor turbidity increases can cause significant reductions in lake productivity, and produce sublethal effects in salmonids that could reduce their fitness and survival. Acute exposure to high turbidity can be lethal for salmonids.
 17. Boating is not an environmentally benign activity. Potential impacts range from spreading exotic species of plants and plankton to physical injury or death to fish.
 18. The artificial water level regime maintained in Lake Washington for operation of the Ballard locks during boating season may inhibit the establishment of emergent vegetation along the shoreline.

19. Emergent vegetation can be an effective barrier for attenuation of wave energy.

What the literature review does not tell us:

1. How do juvenile salmonids respond to piers, bulkheads, and other artificial structures in local lakes (i.e., do they seek or avoid them, will they swim under or go around)? Is response size-dependent? Do variations in design (configuration, materials) affect prey response to structures?
2. Is there a relationship between piers and predation on juvenile salmonids (by all predators, but especially bass and cutthroat, and avian predators) in local lakes? How are the structures utilized by the various predators?
3. Which characteristics or combination of characteristics (shade, cover, structure, etc.) of piers attract bass (or other predators) in local lakes? How do variations in design (configuration, materials) affect predator attraction?
4. Do prisms and grating change predator or prey response to piers? How effectively do they reduce shading *in situ*?
5. How do bulkheads and piers affect sediment distribution/composition and benthic invertebrate distribution and abundance in local lakes?
6. How does pier lighting affect the behavior of chinook fry and their predators in Lake Washington and Lake Sammamish, and ultimately the predation rate on chinook fry?
7. How do juvenile salmonids and their prey, and adult salmonids respond to drop-hammer and vibratory pile driving in lakes?
8. What are the cumulative impacts of overwater coverage on total lake productivity from the existing structures on Lakes Washington, Sammamish, and Union?
9. How do juvenile salmonids in local lakes respond to temporary construction-related turbidity?
10. What is the current contribution of two-stroke marine engine emissions to PAH contamination in local lakes? How is the productivity of the lakes, and the health of fish, being affected?
11. How pervasive is the use of dock-cleaning chemicals by homeowners around local lakes, and what chemicals are being used? What hazard does this chemical use pose to fish? Same questions for lawn-care products.
12. How do juvenile and adult salmonids respond to local boating and swimming activity?
13. How do changes in sediment distribution/composition affect populations of bass?

With these answers and questions, we have a framework for directing our future research efforts, and for determining our regulatory responsibilities.

Recommendations

These recommendations reflect a conservative approach based on the significant findings and uncertainties identified in the literature review. The most important point of this review is the verification of the intuitive relationship between shorezone development and the loss of properly functioning shorezone habitat (including riparian and littoral zones and their interconnection). The dependence on quality shoreline habitat of sensitive juvenile salmonids and the continual onslaught on the quality of that habitat within an urbanizing watershed was juxtaposed throughout the literature reviewed for this report. The ultimate goal of regulatory personnel charged with protecting ESA-listed salmonid species, or other fish species, should be the protection of that species' critical habitat. With that goal in mind, prevention of the continued simplification of shoreline habitat within the lakes of the Tri-County area is the primary recommendation from this report to policy makers and regulators.

Secondary recommendations follow:

1. New bulkheads should not be permitted unless a critical personal property loss can be demonstrated. In most cases, there are soft alternatives to shoreline armoring. If bulkheads are determined necessary, make every effort to pull the structure shoreward of OHW.
2. Encourage the removal of bulkheads in favor of shoreline restorations [i.e., native emergent and riparian plant species, low-gradient beach (or as appropriate for specific site), native structural elements (CWD, rock) in combination with emergent vegetation for wave energy attenuation]. Such designs have been successfully incorporated into recovery efforts in the Great Lakes (Schollen 1995).
3. Encourage property owners to retain driftwood and fallen trees on their properties.
4. Shorezone construction should not occur during the January through June period when juvenile chinook are most likely to be in the littoral zone. More restrictive allowable construction windows would be appropriate in some areas (i.e., Ship Canal, north end of Lake Washington, south end of Lake Washington, and near mouths of spawning tributaries in Lakes Washington and Sammamish). Consider site proximity to a spawning stream or river, and the likely timing of juvenile and adult migration.
5. Instigate an aggressive bass removal campaign in confined areas, such as the Ship Canal, utilizing unlimited catch restrictions and bounties if necessary.
6. Consider a "no new piers" policy as the best option for protecting fish and fish habitat. Encourage the use of floats or buoys instead. If politically impractical, see #7 below.

7. No net increase in overwater coverage should occur in the Lake Washington system - permits for new construction should be contingent on permits for replacement structures. Only replacement structures that demonstrate a reduction in overwater coverage should be permitted. The amount of overwater coverage eliminated from the replacement pier could be held in a "surface area mitigation bank," which new piers would have to draw from. Gradually lower the total net coverage over local lakes.
8. All piers, both new and replacement structures, should be restricted to a 3.5-foot-wide cantilever bridge that spans the nearshore area to a narrow moorage structure of the minimum size necessary to moor the applicant's boat.
9. Cantilever bridge structures should be grated, and as high off the water as practicable, and moorage structures should be no less than 24 inches above OHW. Floating structures should have maximum light penetration, and be removed annually after boating season.
10. Prisms and grating should be studied to determine their efficacy at providing sufficient ambient light for macrophyte production under piers. The best products should be utilized in all new or replacement overwater structures to minimize losses of primary productivity.
11. Minimize the number of piles used in all structures. Study pile dimension to provide evidence for or against mandated pile dimension. Require the removal of existing treated piles if present.
12. Every shorezone development application should be considered an opportunity for a habitat improvement project. Require shoreline restoration as mitigation for shorezone structures.
13. Do not permit shoreline or pier lighting unless future studies suggest otherwise.
14. Discourage the use of pesticides, herbicides, fertilizers, and cleaners, especially near or over the water.
15. Consider phasing-out sales of two-stroke outboard marine engines.
16. Aggressively enforce a "no lake-water withdrawal" policy with waterfront property owners.

Future research should focus on the 13 questions in the "what the literature review does not tell us" section. Two studies that could result in a relatively rapid regulatory response would be the determination of the effectiveness of prisms and grating for facilitating macrophyte growth under piers, and a study of the effects of the two methods of pile driving on salmonids in local lakes. Results from each of these studies could be used immediately to determine WDFW Hydraulic Project Approval (HPA) or municipal requirements for pier design and construction methods. Another top priority might be a

determination of the response of predators and prey to pier lighting, since having that knowledge could produce a rapid regulatory response.

Studies should be directed at determining the predator population response to shorezone alterations and structures; do these alterations and structures enhance predator abundance or simply concentrate the population in predictable areas? If predator populations are limited by factors other than structure availability, placing additional structures may not increase their abundance.

What is the spatial and temporal correlation between artificial structures, gravel, juvenile salmon and predators? Can cause and effect be demonstrated experimentally (i.e., perform a replicated before-after-control-treatment experiment where piles are introduced in “accretion” and “non-accretion” zones, and monitor for one to two years for changes in substrate and fish use).

Our understanding of how various pier designs affect salmonid/bass (and salmonid/salmonid) interactions remains limited. The basis of bass attraction to piers and piles is poorly understood. Spawning bass are attracted to structures protruding from the substrate, but it is unknown which pier features attract foraging bass. Circumstantial evidence suggests that the amount of shade or the area of overhead coverage provided by a structure is important to foraging bass. Investigations of bass utilization of structures of various designs are necessary. How salmonids and their predators respond to light-transmitting pier design elements (i.e., prisms, grating), overhead cover, and piles of various diameters requires investigation. Until investigations of prisms and grating are complete, they should not be relied upon as a mitigation measure that reduces bass attraction.

Finally, the summer boating season corresponds with the highest water levels in Lake Washington. Erosion caused by power boating can be severe. The high water levels that occur during the periods of heaviest boat use increase the potential of boat wake-induced erosion damage around the lake. Efforts to restore natural shorelines in Lake Washington will be hindered by artificially maintaining a high summer lake-level. The ecological implications of the continuation of the existing water-level management regime in Lake Washington should be critically examined.

Ongoing and Further Study

There are a number of ongoing studies within the Lake Washington system from which results will be available to the public in the near future. Roger Tabor (USFWS) and staff from the Muckleshoot Indian Tribe’s Fisheries Department have been studying the predation by largemouth and smallmouth bass and northern pikeminnow on salmon smolts in the Lake Washington Ship Canal and in the Lake proper. Preliminary results from this study were presented at the North Pacific International Chapter-American Fisheries Society meeting from 10 to 12 April 2000, and were included in this document (Tabor et al. 2000). Kurt Fresh (WDFW) is currently studying the timing of entry and distribution of juvenile chinook salmon in Lake Washington; details on the availability of results are pending. Rod Malcom (MITFD) has three reports in preparation:

1. Preferential association of smallmouth bass nests with residential piers and artificial structures in Lake Sammamish, King County, Washington. 1995. Muckleshoot Indian Tribe.
2. Changes in the number and overwater coverage by residential piers in Lake Sammamish, King County, Washington. 1995. Muckleshoot Indian Tribe.
3. Changes in the number and overwater coverage by residential piers in Lake Washington, King County, Washington. 1995. Muckleshoot Indian Tribe.

Details on the availability of these reports were not available.

Additional studies on docks and bulkheads may be currently in progress in the region but information was unavailable at the time of publication.

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