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Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead:

An Evaluation of the Effects of Selected Factors on Salmonid Population Viability

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Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead:

An Evaluation of the Effects of Selected Factors on Salmonid Population Viability

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Executive Summary

The primary purpose of this technical memorandum is to evaluate the potential of selected factors associated with the Columbia River estuary to improve viability of listed anadromous salmonids in the Columbia River basin. We examine how the estuary supports viability of anadromous salmonid populations, review what is known about juvenile salmon in the Columbia River estuary, examine how changes in selected factors associated with the estuary have potentially affected salmon, and describe an approach to value the contribution to salmon population viability derived from potentially reducing a factor's impact. This evaluation was conducted in support of efforts by NOAA Fisheries Service to revise deficiencies identified by Judge J. A. Redden in the 2000 Biological Opinion on the effects of the federal Columbia River Power System (FCRPS) on listed anadromous populations in the basin.

For this review, we defined the Columbia River estuary broadly to encompass the entire continuum where tidal forces and river flows interact. The upstream extent of the estuary is Bonneville Dam and the downstream extent includes the plume. Geomorphic features, ecological functions, and physical characteristics vary broadly within this area and give rise to a mix of habitats that juvenile salmon can potentially occupy. The importance or function of any unit of estuarine habitat depends on site specific or patch scale attributes such as vegetation type, substrate type, and salinity regime. In addition, habitat functions depend on the landscape context of that habitat, such as its size, shape, location in the estuary, the composition of surrounding habitat, and connectivity with other habitats. Throughout the entire estuary, the distribution and quality of habitats has been affected (and continues to be affected) by a variety of anthropogenic (e.g., urbanization) and natural (e.g., climate) factors.

Our understanding of the role of the estuary in the life history and ecology of salmonid populations has changed considerably over the past century. Initial perspectives about the estuary were that it was unimportant or irrelevant because the estuary (and ocean) was considered to be limitless in its ability to support salmon. Thus the major factors affecting salmon were considered to be density-dependent factors occurring in freshwater. Eventually, scientists became aware that nonfreshwater factors had an important influence on numbers of returning adult salmon and began to consider the role that the estuary and ocean played in salmon population fluctuations. The estuary was viewed as a bottleneck or limiting factor to the numbers of adults that could be produced. Attempts to remove or bypass the bottleneck (e.g., by releasing juveniles from hatcheries or transporting fish rapidly through the estuary) proved unsatisfying at increasing abundance of adults. In more recent years, the estuary has come to be regarded as part of the continuum of ecosystems that salmon need to utilize in order to complete their life cycle, rather than a place that salmon need to avoid.

Fundamental to the view that estuaries are an important part of the life history of salmon is the concept that anadromous salmonids are comprised of populations, or discrete breeding units, that vary with respect to their spatial and temporal use of habitats. Variability in climate, instream flow conditions, harvest practices, hatchery operations, and accessibility of habitats by adults and juveniles help define how populations use estuarine habitats, including arrival timing in the estuary, duration of estuarine residence, and fish size at the time of estuarine entry.

NOAA Fisheries Service defines the status of anadromous salmonids based on the viability of populations or groups of populations (evolutionarily significant units or ESUs) over long time-frames. For populations and ESUs to recover, the risk that they will go extinct needs to decline. Four viable salmonid population (VSP) performance criteria are used to define viability: abundance, productivity, spatial structure, and diversity. Levels of these attributes in aggregate define extinction risk or persistence of the population; all four VSP criteria are critical to recovery and are interrelated. This approach to evaluating population status differs from traditional salmon management, which assessed the status of anadromous salmonids from a production perspective using primarily numbers of harvested fish, numbers of reproducing adults, or survival. Estuarine habitats clearly contribute to the viability and persistence of salmon populations in a number of ways. The amount of estuarine habitat that is accessible affects the abundance and productivity of a population. The distribution, connectivity, number, sizes, and shapes of estuarine habitats affect both the life history diversity and the spatial structure of a population. Attributes of estuarine habitats (e.g., temperature and salinity regimes, food web relationships) affect diversity and productivity of populations as life-stage specific survivals vary with habitat characteristics.

Ideally, we would like to be able to link factors in the estuary to their potential to affect the viability of each listed population or ESU. However, because we do not have specific, empirical information describing estuarine habitat use at the population level, we used an alternate approach. We evaluated effects of candidate factors on the main life history type expressed by an ESU and the range of life history strategies utilized by each ESU. As each ESU is comprised of a number of populations, we then inferred responses of the populations based on what we predict will occur for the ESU.

Each listed ESU in the Columbia River basin—the lower Columbia River coho salmon (*Oncorhynchus kisutch*) ESU was also included—was classified as expressing either a stream-type or ocean-type behavior based on when juveniles arrive in the estuary and their size at arrival. When viewed over long time-scales, most members of ocean-type populations migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. Most members of stream-type fish migrate to sea after rearing for more extended periods in freshwater, usually at least a year. Thus ocean-type fish tend to spend longer periods in ocean habitats compared to stream-type populations.

Individual members within each population employ a variety of alternative life history strategies or approaches to using available habitat. A life history strategy describes if, how, when, and for how long individual salmon use various habitats throughout the freshwater, estuarine, and ocean landscape. We used two attributes associated with use of estuarine habitats—size of fish at estuarine entry and time of estuarine entry—to define six general life history strategies that can potentially be expressed by all anadromous populations: 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Although each life history type can produce members that use each strategy, the relative proportion of members associated with each strategy varies by life history type. Ocean-type populations are dominated by the fry and fingerling strategies while stream-type populations are dominated by the yearling strategy.

Of the possible estuarine factors that could potentially affect viability, we considered the effects of four on salmon in the Columbia River estuary: flow, predation, habitat, and contaminants/toxics. These four were selected from a larger list of factors affecting salmon in the estuary based on whether: 1) a significant change in the factor from historical conditions was evident, 2) the factor could potentially affect population viability, 3) there was quantitative data available that could be used to analyze effects of the factor, and 4) the factor could be linked to hydropower operations in the Columbia River basin. The selection of factors relied primarily on the first three criteria. We did not use the fourth criteria to exclude evaluation of any factor.

A brief evaluation of changes in each factor and how the factor could affect population viability was conducted based on existing data and analyses. We only considered impacts on juvenile life stages and did not include potential impacts on adults during their return migration. From these overviews, we developed a series of hypotheses or assumptions about each factor that helped guide how we rated their relative importance for each ESU.

It is important to note that our analyses were not inclusive of all factors that may have a significant affect on salmon population viability. For example, we expect that water temperatures have warmed from historical levels, which could exclude some habitats from use by juveniles during part of the year and affect metabolic processes of both salmon and their predators. These changes in water temperature could alter mortality rates. The intent of our evaluation was to lay out a framework for future consideration of other factors that may affect juvenile salmon use and benefits from estuarine habitat as more comprehensive information becomes available.

River flow is a fundamental factor affecting characteristics of juvenile salmon and their habitat in the estuary and plume. The interaction of flow and tides with the land creates and maintains estuarine habitat. Large-scale changes in flow occur as a result of spatially explicit interactions of short- and long-term climate cycles—El Niño-Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO), respectively—with the watershed. Operations of the Federal Hydropower system (e.g., generation of electricity, flood control, and irrigation) have had significant affects on attributes of flow, including reducing the mean annual flow, reducing the size of spring freshets, almost completely eliminating overbank flows, and changing the timing of ecologically important flow events. The hydrological changes, along with floodplain diking, represent a fundamental shift in the physical state of the Columbia River estuarine ecosystem.

Major changes in the estuary resulting from flow alterations that are especially relevant to salmon include a loss of vegetated, shallow water habitats and changes in the size, seasonality, and behavior of the plume. Such changes potentially have significant consequences for both expression of salmonid diversity and productivity. In particular, because the changes in habitat are most pronounced in shallow water areas, we predict that effects on the ESUs and life history strategies (the fry and fingerling strategies) that make the most extensive use of these shallow water areas will be especially significant. Further, we predict that altering and reducing plume size, shape, and intensity will also affect ESUs expressing the yearling life history strategy.

The location and types of habitats present in the Columbia River estuary have been substantially altered from historical conditions. Although the entire estuary has not yet been surveyed, the main changes that have been identified to date have been a major loss of emergent marsh, tidal swamp, and forested wetlands; shifts in organic matter important to estuarine food webs; and changes in features of the plume. Shallow water-dependent life history strategies (fry and fingerlings) have been most affected by the loss of shallow water-vegetated habitat types in the estuary, while larger life history strategies have been most affected by changes in the plume. Alterations in attributes of flow and the construction of dikes and levees have caused these changes. Diking is an especially significant change because it severs the connection of the habitat with the river, eliminating any direct (use) or indirect (export of organic matter for food webs) benefit to the fish.

Exposure to waterborne and sediment-associated chemical contaminants has the potential to affect survival and productivity of both ocean- and stream-type stocks in the estuary. Stream-type ESUs are likely to be most affected by short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals. These chemicals can disrupt olfactory function and interfere with such behaviors as capturing prey, avoiding predators, imprinting, and homing. Ocean-type ESUs may also be exposed to these types of contaminants, but will also be affected by persistent, bioaccumulative toxicants such as poly chlorinated biphenyls (PCBs) and DDTs, which they may absorb during their more extended estuarine residence. Consequently, we expect that the impact of contaminants on ESUs exhibiting the ocean life history type will be more significant.

Predation is a major source of mortality of all salmonid populations. Although many predator-prey interactions in the Columbia River estuary appear to have changed from historical conditions, we have little quantitative data on most predators. One exception is Caspian tern (*Sterna caspia*) predation, which has significantly increased recently due to a change in nesting habits of the birds in the Columbia River estuary. The main impact of tern predation is on ESUs with stream-type life history types, especially steelhead (*Oncorhynchus mykiss*). This is primarily because the dominant migratory periods employed by salmonids with a stream-type life history most overlap with the nesting period of the terns. Improvements to productivity of populations by managing terns would be expected to especially benefit stream-type ESUs, although lesser benefits to other salmonid ESUs in the basin should also occur.

To evaluate effects of factors on population viability, we developed a rating system that ranked each factor as having a high, medium, or low ability to improve the status of anadromous salmon populations if the impact of the factor was substantially reduced. We drew inferences about how a factor affects an ESU based on the life history type of the ESU and how the factor would affect the different life history strategies that characterized that life history type. Because the currently available information regarding use of the estuary does not allow resolution at the level of a population, limiting factors for all stream-type ESUs were ranked similarly while those for ocean-type ESUs were ranked similarly. Ratings were developed by considering each factor relative to other estuarine factors within an ESU; we did not consider the effect of factors relative to other nonestuary factors.

The rating system used to evaluate effects of factors on population viability consisted of two screening levels; each level addressed two major issues. The level 1 screen evaluated whether the factor was likely of concern to an ESU based on its affects on VSP and change in the factor relative to historical conditions. The level 2 screens asked how the factor affected an ESU based on where the affects occurred. Each question was evaluated for each factor for each ESU based on whether the ESU was ocean-type or stream-types. Scoring was done using guidance from the principles/hypotheses developed in the analyses of the limiting factors. Because of limitations in our knowledge base, we aggregated the estuary into two zones—river mouth to Bonneville Dam and the plume. From the river mouth to Bonneville Dam, we only differentiated two habitat types—shallow and deep.

For ocean-type ESUs, for example, Columbia River chum salmon (*Oncorhynchus keta*) and Snake River fall Chinook salmon (*O. tshawytscha*), flow and habitat were rated as having a high ability to affect population viability. As noted above, flow changes in the basin are primarily a result of dam operations, whereas habitat changes are a function of both hydropower operations and other, nonhydro issues, notably the construction of dikes and levees in the estuary. The combined affect of flow and habitat changes on estuarine habitat has been to reduce the amount of shallow water habitat (especially vegetated habitat such as swamps and marshes) and disrupt organic matter inputs from these vegetated habitats. The dominant life history strategies of ocean-type Chinook salmon extensively use shallow water habitat, which is where the main flow and habitat changes have occurred. Moreover, the beneficial use of shallow water habitat by each ESU is likely a function of location in the estuary. Whereas all estuarine-dependent life history strategies expressed by each ESU will use the lower portion of the estuary (mouth of the Columbia River to RM 40), upriver ESUs (e.g. Snake River fall Chinook salmon) will be more dependent on tidally influenced shallow freshwater habitats between Bonneville Dam (their point of entry to the Columbia River estuarine system) and approximately RM 40. Tern predation was considered to have a low affect on the ocean life history type because terns do not prey significantly on fry and fingerling-sized fish (the dominant ones associated with this life history type). Contaminants received a medium score. Both waterborne and sediment contaminants can affect fry and fingerling life history strategies in shallow water areas.

For stream type ESUs (e.g., Snake River spring/summer Chinook salmon and mid-Columbia River steelhead), the primary estuarine factors affecting population viability were tern predation and flow; both of these factors were ranked as medium. Tern predation was ranked in the medium category because it is primarily directed at subyearling and yearling size fish, which are the dominant strategies in stream-type ESUs. In addition, these larger fish occur in deeper water channel habitats where they are most vulnerable to the terns and migrate at a time when they are most susceptible to tern predation. Flow changes were ranked medium for stream-type ESUs because both abundance and productivity were affected by changes in plume habitat. In addition, the main life history strategies for these ESUs are most abundant in plume habitats where they would be expected to be most vulnerable to flow-related changes in plume habitat.

From the perspective of the estuary, we conclude that population viability of stream-type ESUs is most affected by tern predation and flow, while flow and habitat most affect ocean-type ESUs. At this time, we do not know how much of a change in each factor is required to affect improvements in population responses of relevant ESUs. Based on available information, we hypothesize that the greatest opportunity to affect ESUs in the Columbia River basin by the manipulation of estuarine factors is with restoration of shallow water habitat. These actions will primarily affect ocean-type ESUs and the shallow water-dependent strategies of stream-type

ESUs. This is because there is a strong linkage between the fry and fingerling life history strategies, which dominate ocean-type ESUs, and shallow water habitat. Thus the main affect on ocean-type ESUs of making changes in habitat and flow will be realized as gains in abundance and productivity. The main affect on stream-type ESUs of reducing tern predation and altering flow will also be realized as gains in abundance and productivity.

There is a large amount of vegetated shallow water habitat that has been lost due to the combined effects of flow changes and diking. These two factors must be considered in concert in considering restoration approaches because of the strong effects of flow on functions of shallow water habitat. Restoration of shallow water habitat can be done without changing hydropower operations (which can have other, unintended consequences, such as an increase in gas bubble disease). However, we expect that studies now underway will provide greater insight into how much change and where such change in shallow water is both possible and needed to affect population viability.

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Introduction

Since 1991, 12 different groupings or evolutionarily significant units (ESUs) of anadromous salmonids that reproduce in the Columbia River basin have been listed as threatened or endangered under the Endangered Species Act (ESA) of the United States. These include steelhead (*Oncorhynchus mykiss*), chum salmon (*O. keta*), Chinook salmon (*O. tshawytscha*), and sockeye salmon (*O. nerka*) populations that spawn from the upper Snake River basin to tributaries of the lower river below Bonneville Dam. Every subbasin of the Columbia that is currently accessible to anadromous salmonids contains at least one threatened or endangered population. The Federal Columbia River Power System (FCRPS) has had a variety of well documented impacts on anadromous salmonids in the basin, including the loss and degradation of spawning and rearing habitat and increased mortality of upstream and downstream migrating fish during passage at hydroelectric facilities (ISG 2000, Williams et al. 2005). As a result, efforts to recover these populations at risk have focused almost exclusively on identifying and modifying risk factors directly associated with the large hydroelectric dams constructed throughout the basin.

In recent years, increasing attention has been directed at the role of other hydropower related issues and nonhydropower-related issues in the decline and recovery of salmonids in the basin. One such issue is changes in the condition and availability of habitats in the estuary. The growing recognition that the estuary has a role in the recovery of Columbia Basin salmonids represents a significant departure from previous management efforts in the system. Several developments appear to be responsible for this change. First, legislation Congress passed in 1996 amended the Power Act and required the Northwest Power Planning Council (NWPPC) to consider the effect of ocean conditions on fish and wildlife populations when recommending hydropower mitigation projects for the Columbia River basin. This legislation focused new attention on the estuary, plume, and coastal ocean habitats.

Second, life stage risk and sensitivity modeling analyses of Columbia River salmon populations by Kareiva et al. (2000) and McClure et al. (2003) suggested that additional actions beyond passage improvements were needed to recover salmonid populations. Two life stages identified as sensitive to perturbations included the first years spent rearing in the river, estuary, and ocean. Kareiva et al. (2000) used a matrix population model to conclude that the maximum potential to contribute to anadromous salmonid recovery was associated with these life stages. However, they could not discriminate between these life stages nor could they determine how much of a change in survival was possible.

Third, scientific perspectives of the life history and ecology of anadromous salmonids have shifted in recent years (Bottom et al. 2005). Previously, habitats and life stages important to salmon were considered in isolation with the goal of identifying single limiting factors restricting salmon production. We now recognize that marine, estuarine, and riverine environments are each components of an extended salmon ecosystem that cannot be treated independently (NRC 1996, Bisbal and McConnaha 1998, ISG 2000). Thus the estuary, which includes the plume, is part of the continuum of landscapes all juvenile and adult anadromous salmonids use that originate from the Columbia River basin. They connect freshwater and marine habitats and are used by all life stages to some degree for feeding, refugia from predators, and physiological transition (McCabe et al. 1983, 1986, Bottom and Jones 1990).

Finally, our understanding of the relationships between habitats and the persistence of salmon populations has evolved in recent years. It is clear that habitats cannot be valued simply on the basis of their role in producing fish biomass or numbers (Bottom 1997). Instead, diverse habitats and the expression of life history strategies based on use of these habitats are directly linked to salmon population viability (i.e., persistence) over long time scales (McElhany et al. 2000). These linkages were explicitly recognized by the Independent Scientific Group (ISG) for the NWPPC, which concluded that estuary and ocean dynamics help to control salmon productivity (Beamish and Bouillon 1993, Beamish et al. 1999) and that salmon biodiversity (including the diversity of estuarine life histories) helps to reduce the effects of fluctuations in ocean, and presumably freshwater, conditions (ISG 2000). Alterations and loss of estuarine habitats thus has direct implications for salmonid population viability.

The challenges of identifying, designing, implementing, and evaluating recovery actions in the estuary are significant, in part because we know little about the estuary and the salmon that use the estuary. While ongoing research efforts will significantly upgrade our knowledge base in upcoming years, much of what we now know is conceptual or based on research from other areas, such as Puget Sound. Estuary restoration at any scale is a challenge and in the Columbia River estuary it is an especially daunting challenge because of the massive size of this system. Further, the estuary is among the most heavily modified portions of the basin (Thomas 1983) due to the long history of coastal development and the cumulative effects of flow regulation, habitat modification in the estuary, and other changes upriver which have altered sediment transport and salinity regimes in the system (Simenstad et al. 1992, Weitkamp 1994). In the last 100 years, these and other changes have decreased the amount of some types of wetland habitats in this region by as much as 70% from historical levels (LCREP 1999).

The primary purpose of this technical memorandum is to evaluate the potential of selected factors associated with the estuary to improve viability of listed anadromous salmonids in the Columbia River basin. Accordingly, we:

1) examine how the estuary supports viability of anadromous salmonid populations,
2) review what is known about juvenile salmon in the Columbia River estuary (which includes the plume),
3) examine how changes in selected factors associated with the estuary have potentially affected salmon populations, and
4) describe an approach to value the contribution to salmon population viability resulting from reducing the impact of a factor.

These analyses were conducted in support of efforts by NOAA Fisheries Service to address deficiencies identified by Judge J. A. Redden in the 2000 Biological Opinion on the effects of the Federal Columbia River Power System (FCRPS) on listed anadromous populations in the basin.

Columbia River Estuary

An estuary is generally defined as a semienclosed coastal body of water with a free connection to the open ocean in which salt water is diluted with runoff from the land (Pritchard 1967). For this review, however, we define the Columbia River estuary more broadly to encompass the entire continuum where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. Thus the upstream boundary of the estuary is Bonneville Dam, which is the extent of tidal influence, while the downstream boundary includes the plume (Figure 1).

The estuary can be divided into different zones based on a variety of attributes, such as geomorphic features, ecological functions, tidal conditions, salinity regimes, and physical characteristics. A number of approaches have been employed to describe and classify the different zones of the estuary (e.g., Johnson et al. 2003). In this report, we consider only two zones of the estuary, primarily because our information on use of the estuary by juvenile salmon does not yet allow us to discriminate use at a finer scale. The first zone extends from the mouth upstream to Bonneville Dam and includes conditions ranging from the tidally influenced freshwater in the upper estuary to higher salinities and higher wave energies near the river's mouth. The second zone is the river plume, which is generally defined by a reduced-salinity contour near the ocean surface of 31 parts per thousand. During high flows, the plume front is readily visible as a sharp interface between sediment-laden river water and the clear ocean. Its geographic position varies greatly with seasonal changes in river discharge, prevailing nearshore winds, and ocean currents. During summer months, the plume extends far to the south and offshore along the Oregon coast; during the winter it shifts northward and inshore along the Washington coast. Strong density gradients between ocean and plume waters create relatively stable habitat features where organic matter and organisms can be concentrated.

Throughout the estuary is a mix of habitats that the juvenile salmon can occupy. Habitat is the physical, biological, and chemical characteristics of a specific unit of the environment occupied by a specific plant or animal. Thus habitat is unique to specific organisms and encompasses all the physiochemical and biological requirements of that organism within a spatial unit. The function of any estuarine habitat for juvenile salmon depends on site specific or patch scale attributes such as vegetation type, substrate type, and salinity regime (Simenstad and Cordell 2000). In addition, habitat functions depend on the landscape context of that habitat (Simenstad 2000). Landscape context refers to the spatial arrangement of habitat, including its size and shape; location of the habitat within the estuary; the composition of surrounding habitat; and connectivity with other habitats (Turner 1989). A variety of systems have been employed (and others continue to be developed) to classify and describe the diverse array of habitats present in the estuary that vary in their basic philosophical approaches, information that is used, scope, complexity, and application. Table 1 presents a general classification of major habitat types within the estuary below RM 46, after Thomas (1983) and Johnson et al. (2003); habitat classification systems have not yet been developed for other parts of the estuary.

Figure 1. The Columbia River estuary extends from the upper extent of tidal influence at Bonneville Dam (RKm 235), through the oligohaline zone of the river mouth, into the coastal zone including the plume in the Pacific Ocean. (Reprinted from Bottom et al. 2005.)

Table 1. Major types of estuarine habitats and some of their important attributes in the Columbia River below RM 46. (Adapted from Thomas 1983 and Johnson et al. 2003.)

Major habitat types	Important attributes
Tidal swamps	Vegetation is mostly shrub and woody species. Higher elevations. Low water velocities.
Tidal marshes	Dominant vegetation varies. Includes emergent marshes. Tidal channels often present. Depths generally range from mean lower low water (MLLW) to above mean higher high water (MHHW). Low water velocities.
Tidal flats	Depths range between MLLW and 6 ft below MLLW. Usually not vegetated.
Medium deep	Depths range from 3–18 ft below MLLW. Mostly associated with medium sized and larger channels. Higher water velocities.
Deep	Depths >18 ft. Mostly associated with the main channel. Higher water velocities.

Use of Estuarine Habitats by Columbia River Salmonids

There have been few studies of habitat use of the Columbia River estuary by wild juvenile salmon and steelhead, and most of this work has been conducted downstream of Puget Island. Research recently initiated by NOAA Fisheries Service in the middle and lower estuary and plume will significantly upgrade our knowledge about how juvenile salmon, specific populations, and ESUs use the estuary. Most of our knowledge about how juvenile salmon use estuaries has come relatively recently from studies in Puget Sound, British Columbia, and Alaska (e.g., Parker 1971, Stober and Salo 1973, Kaczynski et al. 1973, Reimers 1973, Mason 1974, Bailey et al. 1975, Fresh et al. 1979, Salo et al. 1980, Healey 1979, 1980, 1982, Simenstad et al. 1982). It is remarkable that given the overall importance of Columbia River for salmon in the Pacific Northwest and the large size and diversity of this estuarine system, empirical knowledge about how salmon use this estuary is distinctly lacking compared to other estuarine systems in the Pacific Northwest.

An important factor that needs to be considered in any analysis of estuarine habitat use by wild salmon and steelhead is the occurrence of hatchery fish. Because our ability to separate wild and hatchery fish captured in the estuary has been limited and remains so even at present, many of the spatial and temporal patterns observed in previous data sets may apply to hatchery fish rather than wild fish (e.g., Dawley et al. 1985, 1986).

River Mouth to Bonneville Dam

In the portion of the estuary between Bonneville Dam and the river's mouth, one study that provided enough information to distinguish use of different estuarine habitats by juvenile salmonids was by Columbia River Estuary Data Development Program (CREDDP); however, sampling in this program was limited to the lower estuary (to the upstream end of Puget Island). In general, McCabe et al. (1986) found that subyearling Chinook in shallow intertidal habitats of the Columbia River were smaller than subyearlings captured in deeper pelagic areas. Larger yearling migrants spent little time in shallow estuarine habitats and more time in deeper channel areas (Bottom et al. 1984, McCabe et al. 1986).

Most of what is known about juvenile salmon use of the estuary concerns timing of fish passage through the estuary and was derived from seining studies conducted to recapture coded wire tagged (CWT) fish below Bonneville Dam. In the late 1970s and early 1980s, subyearling Chinook salmon (these are all nonyearling fish combined) were present year round in the estuary (Bottom et al. 1984, Dawley et al. 1986, McCabe et al. 1986). While in many of the years studied overall peak abundance occurred from May to September, there were years when a bimodal distribution was observed (Dawley et al. 1985, 1986). There was also evidence of specific patterns in seasonal timing that particular populations exhibited (e.g., Lewis River) that were different than the patterns for all populations considered in aggregate (Dawley et al. 1985). Peak catches at Jones Beach, where much of the estuary timing work has been conducted, often were highly correlated with the timing of hatchery releases.

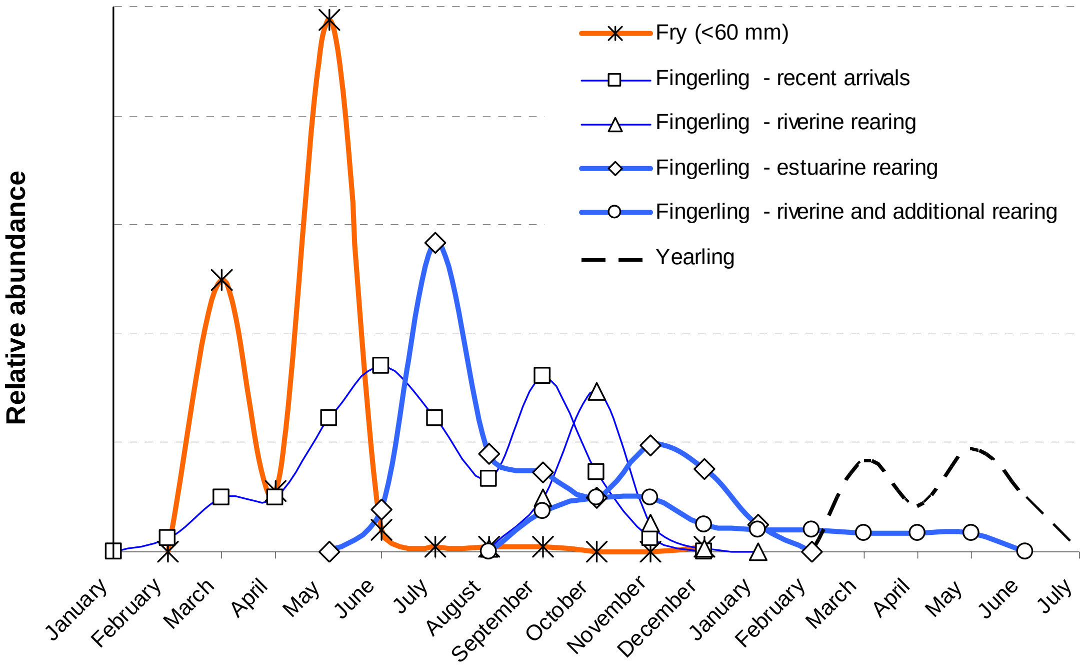
Since 2001 NOAA Fisheries Service has investigated habitat relationships of juvenile salmon in the area upstream of the river mouth. Similar to the work by Dawley et al. (1985), juvenile salmon were present in the estuary year round. However, the number of size classes of juvenile salmon passing through the estuary is considerably reduced from historical levels (Figure 2).

Although knowledge of habitat use in the Columbia River estuary is limited, the information that does exist, in combination with studies in other estuaries of the Pacific Northwest, provides insight into how juvenile salmonids use habitats in this large estuary. Estuarine research has demonstrated that juvenile salmon are generally distributed based on water depth (e.g., Healey 1980, Levy and Northcote 1982, Simenstad et al. 1982, Bottom et al. 1984, Levings et al. 1986, McCabe et al. 1986, Miller and Sadro 2003). The smallest size classes tend to be the most closely associated with shallow water. As fish size increases, fish use a broader array of depths; fish size can change as a result of growth that occurs in the estuary, growth in freshwater, or some combination of rearing in the two environments. Habitat shifts by the juvenile salmon do not necessarily occur continuously as fish size increases, but may occur at specific size thresholds. For example, Simenstad et al. (1980) suggested that juvenile chum salmon shifted from use of littoral to offshore habitats at a size threshold of about 50 mm. Based on this size based model, the smallest juvenile salmon in the Columbia River estuary (fry and fingerlings) will be primarily associated with the shallowest, most peripheral, wetland type of habitat, while the larger subyearlings and yearlings will be found in deeper pelagic areas. Coincident with the fish size/depth relationship, smaller salmon tend to spend longer in the estuary and larger yearling migrants spend less time.

Columbia River Plume

Studies of use of the Columbia River plume were initiated in 1998 by NOAA Fisheries Service. In general, the plume is primarily used by yearlings and less so by subyearlings which appear to stay closer to shore (Emmett et al. 2004). Thus far, because sampling has been conducted in late spring and early summer, the distribution of fish in summer is unknown. Although they were not specifically studying use of the plume, Fisher and Pearcy (1995) found relatively few yearlings near the mouth of the Columbia River in summer. The evidence obtained to date suggests the plume serves juvenile salmon in multiple ways. Pearcy (1992) hypothesized that one function of the plume was to distribute juvenile salmon offshore, away from predation pressure closer to the shore. Findings by NOAA Fisheries Service are consistent with the hypothesis proposed by Pearcy (1992). In May and June when flows are higher, juveniles are found further offshore, in the low saline waters, than when flows are lower (Figure 3). During years when river flow out of the Columbia River during the freshet period is reduced, salmon are more localized around the mouth of the Columbia River.

In addition, the plume appears to provide a place for juvenile salmon to feed and grow (Schabetsberger et al. 2003). It may support juvenile salmon foraging by facilitating primary production during the spring freshet period. During low-flow years, such as observed in 2001,



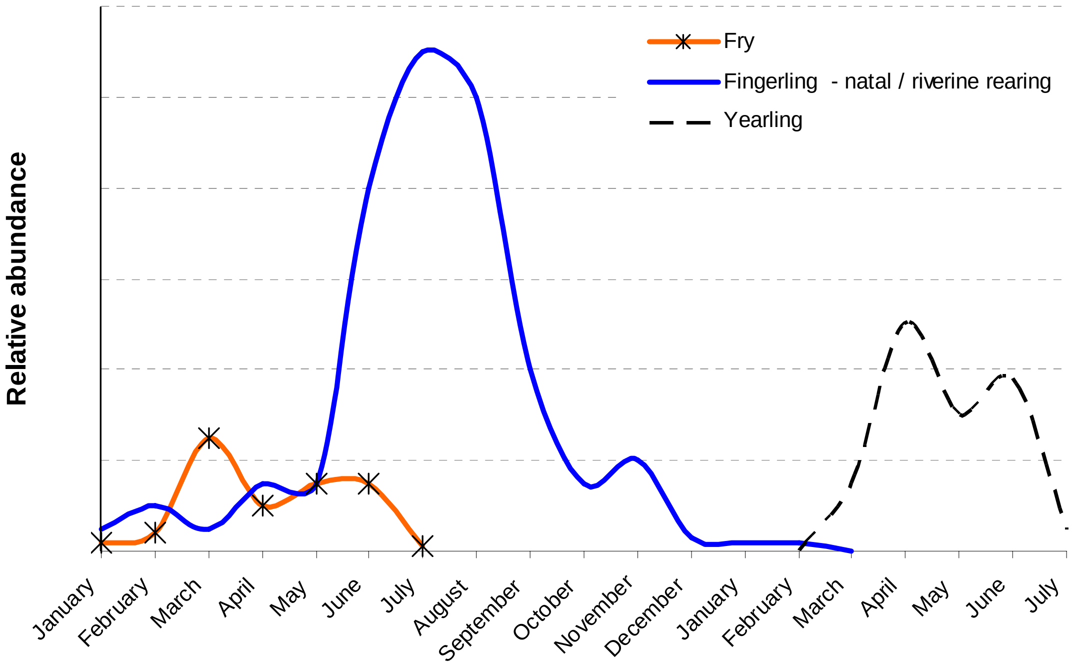


Figure 2. Historical and contemporary early life history types for one broodyear of Chinook salmon in the Columbia River estuary. Historical timing and relative abundance (top) inferred from historical sampling throughout the lower estuary (Rich 1920). Contemporary timing and relative abundance (bottom) derived from Dawley et al. (1985) sampling at Jones Beach. (Adapted from Burke 2005.)

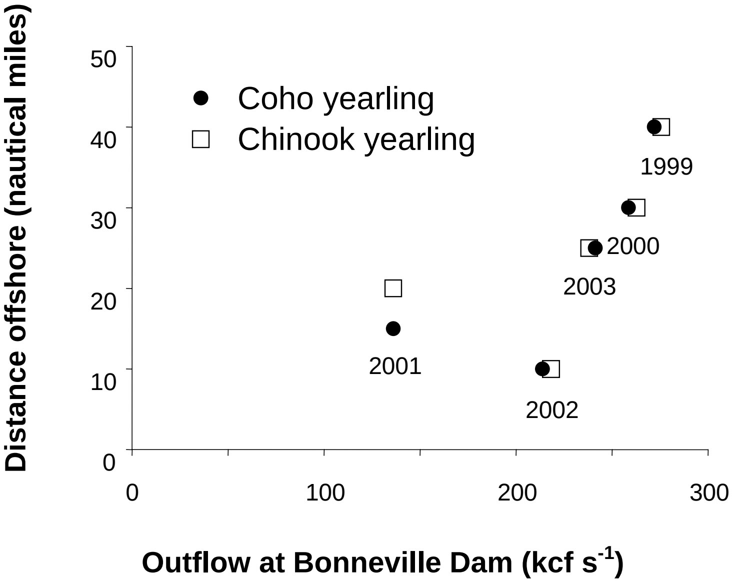


Figure 3. Relationship between average river flow (thousand cubic feet per second) registered at Bonneville Dam for 10 days prior to sampling in the Columbia River plume in May, and the maximum distance offshore juvenile Chinook and coho salmon were captured in surface trawls along a transect extending east along latitude 46.15, just south of the mouth of the Columbia River. (From B. Peterson, NWFSC.)

the amount of chlorophyll evident off of the Oregon and Washington coast affiliated with the plume, as characterized with satellite observations by SeaWifs, was much lower than that observed when more normal flows occurred, such as observed in 1999 (Thomas et al. 2003). Zooplankton biomass is also highly associated with frontal features at the plume margins (Figure 4) and less so either within the plume or oceanic zones.

Some of the ongoing plume research being conducted by NOAA Fisheries Service has focused on identifying attributes of the plume that can be used to define habitat important to juvenile salmon. Features such as the surface area of the plume, the volume of the plume waters, the extent and intensity of frontal features, and the extent and distance offshore of plume waters are considered physical attributes defining habitat important to salmon. One hypothesis that has been studied is that juvenile salmon would preferentially utilize frontal features; this, however, has not been validated. Juvenile salmon abundance was not higher exclusively around frontal features. Studies did suggest that smaller juvenile salmon showed a significant preference for the plume and front habitats as compared to the more marine, oceanic habitats (Figure 5). NOAA Fisheries Service has also found that salmon continue their preference for the low saline environment of the plume as they retain their orientation to the surface region (Emmett et al. in press). The higher turbidity associated with the low salinity plume waters may provide a refuge from predators.

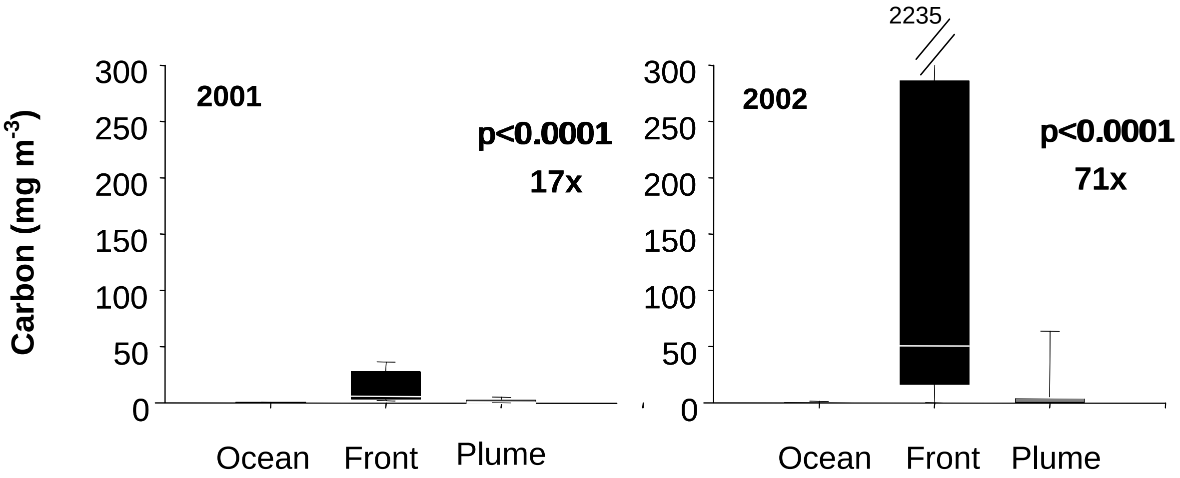


Figure 4. Biomass (milligrams carbon per cubic meter) of megalopae (*Cancer magister*) captured in May 2001 and 2002 in the ocean, front, and plume habitats using a neuston net. Box plots demarcate the 10th, 25th, 50th, 75th, and 90th percentiles. An analysis of variance (ANOVA) (blocked) was used to identify significant differences. Biomass of this species was 17 and 71 times higher in the front habitat compared to the average of the ocean and plume habitats (Morgan et al. in press).

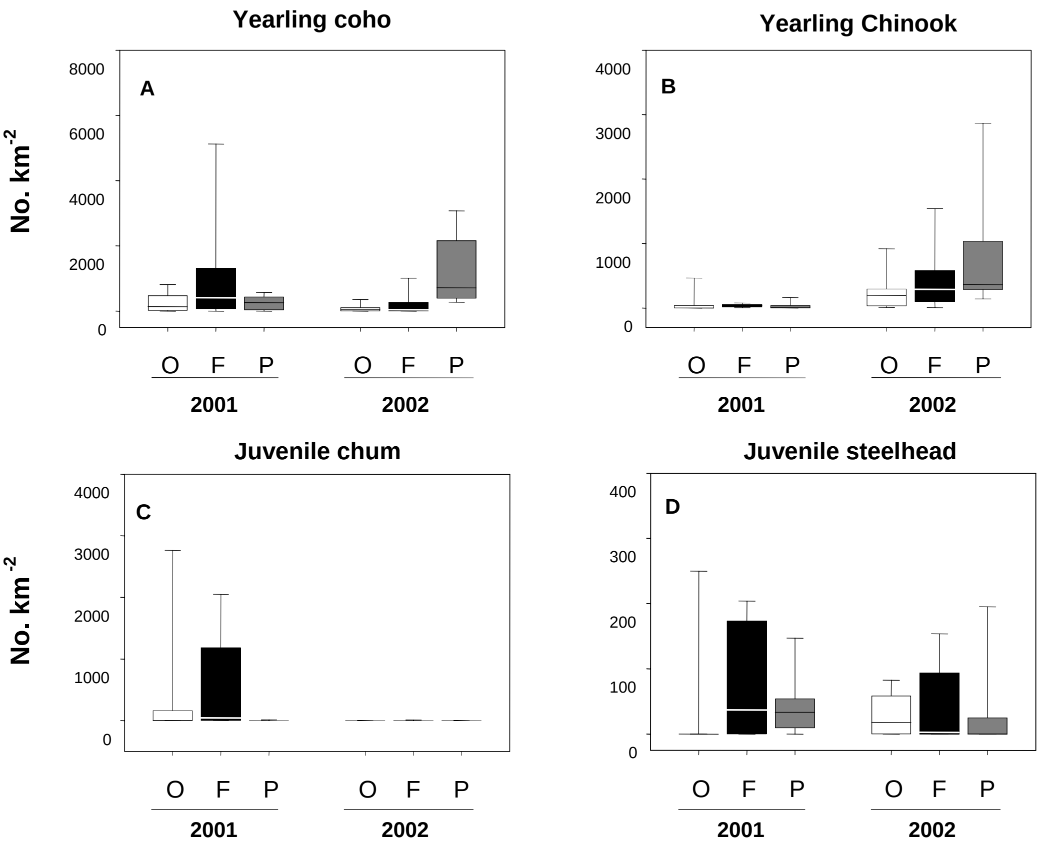


Figure 5. Abundance of A) yearling coho salmon, B) yearling Chinook, C) juvenile chum salmon, and D) juvenile steelhead captured in ocean (O), front (F), and plume (P) habitats using a Nordic rope trawl. Box plots demarcate the 10th, 25th, 50th, 75th, and 90th percentiles of densities. (From A. D. Robertis, NWFSC.)

Role of the Estuary in the Recovery and Management of Salmon Populations

The following section describes our conceptual framework of the role of the estuary in the life history of anadromous salmonids and how this perspective has evolved. It is intended to help structure our evaluation of the potential of selected factors to affect salmonids in the Columbia River basin and to help facilitate recovery and management of anadromous salmonids. The framework describes and organizes our current understanding of how the estuary supports anadromous salmonid species and integrates information on the life history and ecology of salmon, conservation biology, and how estuarine ecosystems function. We incorporated elements of the model that Bottom et al. (2005) developed to guide research on juvenile salmonids in the Columbia River estuary. Because of the lack of knowledge specifically about juvenile salmon in the Columbia River estuary, we used relevant information from throughout the Pacific Northwest to help develop the framework. We recognize that the estuary is much larger than most systems on the Pacific coast where juvenile salmon use of estuaries has been studied, so information from other systems needs to be applied carefully. But, because the salmon are using the estuary for the same fundamental purposes and estuaries have many of the same attributes regardless of size, we believe that principles of estuarine use can be developed from these other systems and applied in the Columbia River estuary.

Historical Views of the Role of the Estuary

Our understanding of the role of the estuary in the life history and ecology of salmon has evolved as we have learned more about how juvenile salmon use estuarine habitats. Initially, the estuarine and ocean environments were considered limitless in their ability to support salmon and so were believed to be unimportant or irrelevant. At this time, freshwater, density dependent factors were assumed to regulate salmon species (Bottom 1997, Bottom et al. 2005), with more adults expected simply as a result of an increase in the number of eggs and fry. A major goal of early salmon research and management was to understand freshwater sources of mortality so that they could be more easily manipulated and more adults could be produced (Bottom 1997). Salmon were viewed primarily from a production perspective as simply another agricultural product or crop to be managed for the benefit of people (Bottom 1997). The output of this crop was defined as short term changes in the numbers of harvestable or reproducing adults.

One outgrowth of the production approach to salmon management has been the use of hatchery fish throughout the Pacific Northwest. Because of habitat destruction and alteration, the high demand for salmon, overharvest, and the expanding human population, there has never been enough harvestable salmon to go around (Lichatowitch and McIntyre 1987). Hatcheries evolved as a way to increase salmon abundance and were based on the freshwater-centric, density-dependent philosophy that more adults would result in direct proportion to the additional number of eggs that survived (Lichatowitch 1999). Hatcheries focused on bypassing as much of the freshwater life of salmon as possible, where the most significant sources of mortality were believed to occur.

The continued failure of hatchery production to increase or even maintain salmon numbers raised new questions about whether passage through estuaries and the ocean might be critical to determining numbers of returning adults. Salmon researchers in the 1950s and 1960s began to recognize that nonfreshwater factors had a role in determining numbers of returning adults (e.g., Manzer and Shepard 1962, Gilhousen 1962). The development and analysis of long-term data sets on salmon production suggested freshwater conditions could not by themselves adequately explain variability in numbers of returning adults (e.g., Salo and Bayliff 1958, Hunter 1959, Gilhousen 1962, Parker 1968, Peterman 1978). Beginning in the late 1960s, research on the estuarine and early marine life of juvenile salmon rapidly expanded throughout the Pacific Northwest.

Initially, scientists adopted the production-oriented view that estuaries were a bottleneck to salmon production. This bottleneck concept was based on research suggesting that juvenile salmon mortality during this period could be especially high (e.g., Parker 1968, Bax 1983). The estuarine and early marine period came to be regarded by many researchers as the critical period of salmon life that significantly affected overall survival rates and adult returns (Kaczynski et al. 1973, Peterman 1978, Healey 1980, Nickelson 1986, Pearcy 1992).

In the 1970s, proposed expansions in hatchery production generated concerns that there could be a limit or carrying capacity to estuarine and coastal environments. Simenstad et al. (1980) expressed concern that the carrying capacity of the Hood Canal, Washington estuarine environment for pink salmon (*O. gorbuscha*) and chum salmon would be exceeded if enough hatchery fish were released. Bailey et al. (1975) suggested that releases of hatchery pink and chum salmon in Alaska coastal areas could exceed carrying capacities of estuaries for hatchery fish and result in fewer than expected adult returns. Studies were initiated to estimate the quantities or carrying capacity of hatchery fish that could be supported by estuaries (e.g., Reimers et al. 1979) and to find the optimum conditions in the estuary that would maximize production. Because some managers questioned whether salmon were really estuarine dependent, further studies were conducted to ascertain whether the estuary could be bypassed altogether and so render moot the issue of carrying capacity in this environment (MacDonald et al. 1988, Solazzi et al. 1991).

Current Views of the Role of the Estuary: Population Perspective

In more recent years, we have shifted from the more production-oriented views of the estuary to the developing perspective that the estuary is part of the continuum of ecosystems that salmon utilize to complete their life cycle, rather than a place that should be avoided. Estuaries, as well as other places used by salmon throughout their lives, are therefore necessary and important parts of the salmon life cycle. Fundamental to this view of estuaries is the concept that salmon species are comprised of populations. Populations are geographically discrete, self perpetuating, and semiisolated (in terms of genetic exchange) groups of reproducing salmon. They are the unit that much of modern research and management of salmon is organized around. The population perspective of salmon is in part an outgrowth of studies by Willis Rich on Columbia River salmon (e.g., Rich 1920, 1939). Rich evaluated scale patterns of juveniles passing through the Columbia River estuary and found a variety of patterns of estuarine use. Fish exhibited a wide diversity in the time at which they arrived in the estuary, the amount of time they were spending there, and the size at which they arrived (Rich 1920, 1939); some salmon were present nearly year round in the estuary.

To explain these observations, Rich (1939) suggested that the Columbia River estuary was a mixing ground of fish from many different sources or populations, each of which exhibited a range of behaviors that were defined by the particular set of conditions found in the full range of spawning and nursery areas available to the population. He concluded that there was not a unique or singular way for a salmon species or population to use the estuary or any other habitat. Instead populations employed a diversity of approaches that were determined by variability in climate, instream flow conditions, origin of the fish, harvest practices, hatchery operations, and accessibility of habitats by adults and juveniles. Groups of salmon became locally adapted to the conditions they experienced.

The concept that habitat use by salmon can depend on the population of origin is supported by a wide body of literature (e.g., Wilmot and Burger 1985, Burger et al. 1985, Beachum and Murray 1987, Burgner 1991, Healey 1991, Wood 1995, Woody et al. 2000, Hodgson and Quinn 2002, Miller and Sadro 2003, Ramstad et al. 2003). Our understanding of the specifics concerning between- and within-population variability in estuarine habitat use is still increasing. However, what we know is consistent with the population perspective of habitat use. For example, Beamer[1] has found that early migrating Chinook salmon fry in the Skagit River, Washington, use a different suite of Puget Sound habitats than later migrating fish. Reimers (1973) was able to define different patterns of estuarine habitat use and seaward movement in Chinook salmon in the Sixes River, Oregon, and link those to their ultimate success (i.e., survival). Carl and Healey (1984) concluded that variations in migration behavior and estuarine use within the Nanaimo River, British Columbia basin were linked to different subpopulations associated with geographically distinct spawning areas.

The Estuary and Viability of Anadromous Salmonid Populations and ESUs

NOAA Fisheries Service's Northwest Fisheries Science Center (NWFSC) developed a conceptual approach to evaluating the status of anadromous populations and ESUs that defines status based on a population's viability over long time scales (McElhany et al. 2000). The authors defined a viable population or ESU as one that has a negligible risk of extinction over a 100-year time period. For populations or ESUs to recover, the ability of a population or ESU to persist must increase over time, or conversely, the risk that they will go extinct needs to decline (McElhany et al. 2000). Four viable salmonid population (VSP) performance criteria are used to define viability (McElhany et al. 2000): abundance, productivity, spatial structure, and diversity. All four VSP criteria are critical to the viability of salmon populations, all are interrelated, and levels of all four attributes in aggregate define extinction risk or the likely persistence of the population or ESU. This approach to evaluating population status differs from traditional salmon management, which typically assessed the status of anadromous salmonids from a production

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[1] E. Beamer, Skagit River System Cooperative, La Connor, WA. Pers. commun., February 2004.

perspective using numbers of harvested fish or reproducing adults. Using the NOAA Fisheries Service VSP criteria incorporates a much broader view of how to define population status. This perspective recognizes that factors affecting salmon populations vary widely, so using a variety of metrics can provide insight into what types of actions are needed to help populations and ESUs recover.

Abundance is a measure of the number of members in the population (e.g., numbers of spawners or returning adults), while productivity is the rate of growth of the population over a given time interval. Productivity can also be expressed as life-stage specific survivals, since the cumulative effects of those survivals results in a population's growth rate over time. Populations that have a lot of members and a positive population growth rate are more likely to persist than populations that do not have these characteristics. Changes in abundance and productivity of a population can be measured over multiple time scales. Evidence clearly suggests that estuarine habitats contribute to the abundance and productivity of salmon populations (e.g., MacDonald et al. 1988). For example, Reimers (1973) demonstrated that for the one broodyear of Chinook salmon that he studied in the Sixes River, Oregon, most adult returns originated from fish that made extensive use of the estuary. Magnusson and Hilborn (2003) similarly concluded that survival to adult return of hatchery Chinook salmon populations in coastal environments was directly and positively correlated with the condition of the estuary. The Kareiva et al. (2001) matrix population model suggested that improving survival during the estuarine and early ocean stages could significantly increase salmon population growth rates. Changes in conditions in the nearshore zone of Puget Sound, which serves as an extension of the estuary, account for significant variability in adult returns of Skagit Bay Chinook salmon.[2]

Although conservation of diversity and spatial structure are emerging paradigms in recovery and management of Pacific Salmon (e.g., McElhany et al. 2000, Waples et al. 2001, Hilborn et al. 2003, Issak et al. 2003) and other fish species (e.g., Gresswell et al. 1994), their application to salmon recovery remains a considerable challenge. For example, within Puget Sound, quantitative abundance and productivity goals have been developed for many threatened Chinook salmon populations, but population specific goals for spatial structure and diversity do not yet exist. Moreover, guidance on improving spatial structure and diversity in salmon recovery has thus far focused primarily on the effects of spawning characteristics (i.e., amount, quantity, location, and diversity of spawning habitats available to a population) rather than on characteristics occurring during other life stages. Because understanding the role of the estuary in affecting spatial structure and diversity is as important as understanding its effect on abundance and productivity, we consider these concepts and their application to salmon recovery efforts in greater detail below.

Spatial structure refers to the geographic distribution of individuals in the population and the processes that generate that distribution. The conceptual basis of spatial structure originates from principles of metapopulation dynamics (Hanski and Gilpin 1996), which describe how groups of populations interact with their habitat and with one another. Metapopulation principles suggest that persistence of a species in a variable environment will depend in part on the spatial distribution of suitable habitat, including the numbers, quality, and quantities of habitat patches occupied; patterns in the use of patches; when patches are occupied; when patches are available;

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[2] C. Greene, NWFSC, Seattle, WA. Pers. commun., February 2004.

and the ability of members to colonize and use habitat patches. At any one time, there may be a wide variety of habitats that can be occupied by members of a population, although not all suitable habitats that are available will necessarily be occupied. The application of metapopulation principles to the conservation of salmonids has attracted considerable interest in recent years (e.g., McElhany et al. 2000, Rieman and Dunham 2000, Issak et al. 2003, Ruckelshaus et al. 2004).

Salmon populations clearly exhibit complex geographic structure that can be defined at multiple spatial scales. For example, at any moment, a population can be distributed across many thousands of square miles, ranging from the headwater spawning areas to Pacific Ocean feeding grounds (e.g., Healey 1991). Within one area, such as the estuary, multiple habitat types such as deep channels, mudflats, and emergent marshes can be simultaneously occupied by members of one population. Distributing members of a population through an array of habitats helps reduce the vulnerability of the population to shifts in environmental conditions (McElhany et al. 2000, Hilborn et al. 2003). There are, however, few studies directly linking geographic structure to observed patterns of estuarine use. Carl and Healey (1984) concluded that variations in migration behavior and estuarine use by Chinook salmon within the Nanaimo River basin were linked to different subpopulations associated with geographically distinct spawning areas.

Diversity consists of the variability in life history and discrete genetic traits exhibited by salmon. Diversity in salmon life histories exists along a continuum and includes individuals, subpopulations, populations, ESUs, and species. Along with spatial structure, phenotypic diversity helps buffer populations from environmental variability (e.g., Taylor 1990, Healey and Prince 1995, Hilborn et al. 2003). Life history diversity can be measured as variability in a wide variety of traits, including body size, fecundity and egg size, timing of life history events such as spawning, where spawning and rearing occurs, residence time in various habitats, habitat use, size at age, age at maturity, ocean distribution patterns, and physiological characteristics (Koski 1971, Healey and Heard 1984, Burger et al. 1985, Beachum and Murray 1987, Tallman and Healey 1991, Taylor 1990, 1991, Quinn and Unwin 1993, Roni and Quinn 1995, NRC 1996, Quinn et al. 2000, Waples et al. 2001, Hodgson and Quinn 2002, Beckman et al. 2003, Miller and Sadro 2003, Ramstad et al. 2003, Brannon et al. 2004). Variability in life histories can arise as a result of genetic variation, variation in environments the fish experience, or both; differentiating these is a considerable challenge. Thus some of the variability we see in life history patterns may be adaptive (i.e. genetically based) and reflect local adaptations that different salmon populations have evolved to cope with the specific conditions that they experience, including variability in estuarine habitats (Taylor 1991, Hansen and Jonsson 1991, Gharrett and Smoker 1993, Quinn et al. 2000). Other genetically based variations may not be adaptive, but instead may be due to random genetic drift (Stearns 1992).

One way to conceptualize life history diversity is as a set of alternate life history strategies or trajectories (we use these terms to refer to the same thing) that individual members of a salmon population can follow (Wissmar and Simenstad 1998). Each trajectory represents an approach to using the spawning, rearing, and migration habitats that are available to the fish in space and time. A variety of phenotypic differences (age at return, size at return, fecundity, and time spent in particular habitats) can exist between trajectories based on how habitats are used. From the perspective of the estuary, studies have demonstrated a broad range in the use of estuarine habitats between and within populations in such attributes as residence time, timing of arrival in the estuary, habitat usage, and size of arrival in the estuary (e.g., Reimers 1973, Carl and Healey 1984, Levings et al. 1986, Quinn and Unwin 1993, Miller and Sadro 2003, Bottom et al. 2005, Bottom[3]).

A major factor affecting the number and quality of life history strategies (quality is defined as how successful the trajectory is at producing recruits) present within a population will be the distribution and quality of habitats that can potentially be used (NRC 1996). If the habitats do not exist because of either natural or anthropogenic factors, then population members cannot use them and distinct life history strategies can potentially be eliminated from the population. This can reduce viability of the population by diminishing productivity, spatial structure, and diversity. For a population to use diverse habitats, the habitats must be available and the right fish must be available to use these habitats (e.g., the appropriate genotype).

Although each member of a population is unique and has its own trajectory, these individual trajectories can be bundled or aggregated into a more limited number of general trajectories based on spatial and temporal patterns in use of habitats (e.g., Reimers 1973, Carl and Healey 1984, Beamer[4]). A variety of metrics associated with these general trajectories can then be used as measures of life history diversity including number of trajectories, condition of the trajectories (e.g., quality of habitats being used), distribution of members across trajectories, and success or survival of members using different strategies. In general, the abundance of members using some strategies will be greater than the abundance of members using other strategies within a population because those strategies are more successful under the prevailing environmental conditions. As conditions change, the distribution or proportion of members associated with each life history strategy can then shift. Over short time scales (e.g., annually), the distribution of strategies can vary in response to annual variability in such factors as flow, water temperature, and the occurrence of El Niño events. A sustained shift in conditions (e.g., climate shift, anthropogenic influences) potentially can produce more significant shifts in the distribution of life history traits (Hilborn et al. 2003).

The complex geographic distribution of members within populations and the alternate approaches to completing life cycles are not unique to salmon or anadromous species (Roughgarden et al. 1988, Sinclair 1988, Secor 1999, Able et al. 2003, King and MacFarlane 2003). As with salmon it is not clear in most cases, whether diversity is more facultative or genetically determined. The relative roles of genetics and the environment in inducing diversity will have a significant effect on any population's persistence in the face of changing environmental conditions. Within the Alagnak River, Alaska, Meka et al. (2003) found that rainbow trout *(Oncorhynchus mykiss)* exhibited three life history patterns based on their migratory movements within this river system. A similar diversity in movements within Yellowstone cutthroat trout *(Oncorhynchus clarki bouvieri)* was described by Gresswell et al. (1994). Although little genetic differentiation existed, they concluded that the range in variation in life history strategies within the cutthroat trout was an adequate basis for providing protection to each life history type. Curry et al. (2002) found a range of tactics related to the use of freshwater and estuarine habitats within one riverine brook trout population. Recent research has also revealed that striped bass *(Morone saxatilis)* exhibit a variety of life history approaches that

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[3] D. Bottom, NWFSC, Newport Research Station, Newport , OR. Pers. commun., February 2004.

[4] E. Beamer, Skagit River System Cooperative, La Conner, WA. Pers. commun., January 2004.

can vary within and between populations in use of freshwater and estuarine landscapes (Secor and Piccoli 1996). Limburg (2001) found that there was differential mortality of American shad (*Alosa sapidissima*) based on fish size and age the fish emigrated from freshwater. Many populations of marine fish also exhibit complex approaches to how they distribute themselves in space and time that are similar to the tactics exhibited by anadromous species (e.g., Roughgarden et al. 1988, Able et al. 2003, King and McFarlane 2003).

In summary, estuaries are part of the continuum of habitats salmon use. Rather than serving as a bottleneck to production, estuarine habitats clearly contribute to the viability and persistence of salmon populations. They accomplish this in a number of ways. The amount of estuarine habitat that is accessible affects the abundance and productivity of a population. The distribution, connectivity, number, sizes, and shapes of estuarine habitat affects both the diversity and the spatial structure of a population. Attributes of estuarine habitats (e.g., temperature and salinity regimes, food web relationships) affect diversity and productivity of populations as life-stage specific survivals vary with habitat characteristics.

Analyzing Effects of Factors on Population Viability

In this section, we describe our approach to analyzing effects of estuarine factors on salmonid population viability. We first describe how we linked estuarine factors to their potential to affect the viability of each listed population. Because we had limited empirical information describing estuarine and plume habitat use by specific anadromous populations or ESUs in the Columbia River estuary and plume, we used an alternate approach. Specifically, effects of candidate factors were linked to viability of an ESU based on the life history type of each ESU and how a factor affected the distribution and quality of life history strategies associated with each life history type. As each ESU consists of an aggregate of populations, we can then infer responses of populations based on what we predict will occur for the ESU. Second, we describe which factors were included in our analyses and how they were selected. Finally, we describe the system we used to rate effects of each factor on viability of ESUs.

Defining Life History Type and Life History Strategy

We first defined the life history type of each ESU as either stream-type or ocean-type. The terms stream-type and ocean-type originated from Gilbert (1912) to discriminate Chinook salmon based on their length of stream residence. Based on banding patterns he observed on scales, Gilbert referred to stream-type fish as those fish having scales with a banding pattern consistent with a period of poor growth in winter in cold freshwater habitats. Ocean-type scales (he used the term sea-type) did not show this poor winter growth, indicating the fish moved into warmer, more productive marine waters before winter. Subsequently, Healey (1991) extended use of the terms to include both juvenile residence patterns and ocean distribution patterns. Healey (1991) proposed that ocean-types and stream-types were separate races that were independent and geographically isolated from one another except in the southern part of their range where they separated temporally in areas of sympatry. Recently, Healey's racial model explaining variability in Chinook salmon life history patterns at broad spatial scales has been challenged (Brannon et al. 2004, Waples et al. 2004).

We use the terms stream-type and ocean-type to separate ESUs into two groups based strictly on certain characteristics exhibited by juveniles during their first year of life, including how long they rear in freshwater, when they outmigrate, and how long they spend in estuarine habitats (Table 2). Populations are referred to as ocean-type if most of the members of the population migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. A shorter period of freshwater rearing is usually correlated with more extensive use of estuarine and oceanic habitats. In contrast, most members of a stream-type population migrate to sea after rearing for at least a year in freshwater (Table 2). Thus ocean-type fish have a greater reliance on estuarine and ocean habitats during their first year of life, while stream-type salmon tend to depend on freshwater habitats during this same period. Differences in other life history characteristics can also occur as a function of juvenile life

Table 2. A summary of the juvenile characteristics of stream and ocean life history types as used in this report.

Stream-type fish	Ocean-type fish
Species	
Coho salmon	Coho salmon
Some Chinook populations	Some Chinook populations
Steelhead	Chum
Sockeye	Pink
Attributes	
Long period of freshwater rearing (>1 yr)	Short period of freshwater rearing
Shorter ocean residence	Longer ocean residence
Short period of estuarine residence	Longer period of estuarine residence
Larger size at time of estuarine entry	Smaller size at time of estuarine entry
Mostly use deeper, main channel estuarine habitats	Mostly use shallow water estuarine habitats, especially vegetated ones

history characteristics including differences in age and size at return and timing of return migrations (Healey 1991). However, there is not a simple relationship between these other life history characteristics and life history type. For example, while stream-type fish are often spring spawners and ocean-type fish are generally fall spawners, there are numerous exceptions (Brannon et al. 2004, Waples et al. 2004).

An ESU was classified as stream type if the majority of populations within that ESU fit the stream-type life history model; an ESU was classified as ocean type if the majority of populations fit that life history type. Information documented in the following species status reviews was used to classify the populations and ESUs: Myers et al. (1998) for Chinook salmon, Johnson et al. (1997) for chum salmon, and Busby et al. (1996) for sockeye and steelhead.

Individual members within a population exhibit a variety of alternative spatial and temporal trajectories, or life history strategies, for using available habitat. We defined alternative life history strategies based solely on the size at estuarine entry and arrival time in the estuary. Size at entrance into the estuary can be used to classify life history strategy because there is a linkage between fish size, habitat use, and residence time (Healey 1980, 1982, Levy and Northcote 1981, 1982, Simenstad et al. 1982, Levings et al. 1986, Tschplanski 1987, Miller and Sadro 2003). In general, residence time in the estuary decreases as the size of the fish entering the estuary increases (with the exception of pink salmon). In addition, juvenile salmon are generally distributed based on water depth, with the depth of the water occupied by the fish increasing as the size of the fish increases (McCabe et al. 1986). Larger fish can result from growth either in estuarine or freshwater habitats. There is not necessarily a continuous relationship between habitat occupied and fish size but rather there may be transitional or threshold sizes where most fish shift from use of one habitat type to another. For example, Simenstad et al. (1980) suggested that juvenile chum shifted from shallow littoral habitats to more offshore habitats in Puget Sound at a size of about 50 mm.

The time the fish arrive in the estuary also varies within a general size class of individuals (Carl and Healey 1984, Bottom et al. 2005). Because available resources and habitats can be different depending on when a fish arrives in the estuary, arrival timing represents a reasonable way to define how the fish use habitats. The wide range in variability in size at estuarine entry and time of entry that can occur is illustrated by Figure 2 from Bottom et al. (2005). In this analysis, Bottom et al. (2005) classified historical early life history strategies for juvenile salmon recovered in the Columbia River estuary based on size at entry, time of entry, freshwater life history (where and how long the fish reared in freshwater), and estuarine growth data generated from scale pattern analyses performed by Rich (1920). For example, under historical conditions, fry (fish < 60 mm at estuarine entry) were found in the estuary nearly year round, while yearlings were present from February to June. The source populations were not identified in these analyses, but it is reasonable to assume that many ESUs contributed to the patterns historically observed in the Columbia River estuary.

Based primarily on size at arrival in the estuary and time of estuarine entry data derived from Burke's (2005) reanalysis of Rich's scale information, we defined six general life history strategies that used the estuary historically (Table 3): 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Fry were defined as fish that enter the estuary at a size less than 60 mm, with early fry entering in approximately March and April and late fry from May to June. Fingerlings were identified as fish that enter the estuary at a larger size than fry, which implies there was some period of freshwater rearing; fingerlings have yet to begin the physiological transition associated with smolting. Fingerlings rear in the estuary for some period, with early fingerlings entering between January and July and late fingerlings from August to December. Subyearlings are fish that rear for less than a year in freshwater, rear little in the estuary, and smolt as they outmigrate during their first year of life. They reside in the estuary for less time than fry or fingerling salmon. Yearlings rear for at least one year in freshwater and then emigrate; these fish generally spend less time in the estuary than fry or fingerlings.

Although any one population can potentially produce all strategies, some strategies will be more abundant or dominant than others within a population. In general, yearlings will tend to be the dominant life history strategy in stream-type populations, while nonyearlings will be most abundant in ocean-type populations. The distribution or proportion of members within a population associated with each life history strategy will depend on the environmental conditions the fish are experiencing (e.g., ocean conditions, freshwater spawning habitat, and predator populations). The distribution of members within different strategies can vary in response to climate changes, flow, water temperature, and ocean conditions.

Considering the dramatic changes that have occurred over the last 100 plus years in climate, estuarine habitats, ocean conditions, and the freshwater spawning and rearing habitats that produce the source populations, it seems reasonable to assume that these changes have altered the distribution of life history strategies within populations and therefore within ESUs. This change is suggested by comparing current and historical use of the estuary by different

Table 3. Some general attributes of life history strategies associated with Columbia River anadromous salmon populations based upon historical use of the system. Information from D. Bottom, NWFSC, and J. Burke, University of Washington School of Aquatic and Fishery Sciences, was used to develop this table. All values should be considered general estimates.

Life history strategy	Attributes
Early fry	Time of estuarine entry: March–April Size at estuarine entry: <50 mm Estuarine residence time: 0–40 days Freshwater rearing: 0–60 days
Late fry	Time of estuarine entry: May–June, present through September Size at estuarine entry: <60 mm Estuarine residence time: < 50 days Freshwater rearing: 20–60 days
Early fingerling	Time of estuarine entry: April–May Size at estuarine entry: 60–100 mm Estuarine residence time: <50 days Freshwater rearing: 60–120 days
Late fingerling	Time of estuarine entry: June–October, present through winter Size at estuarine entry: 60–130 mm Estuarine residence time: 0–80 days Freshwater rearing: 50–180 days
Subyearling (smolt)	Time of estuarine entry: April–October Size at estuarine entry: 70–130 mm Estuarine residence time: <20 days Freshwater rearing: 20–180 days
Yearling	Time of estuarine entry: February–May Size at estuarine entry: >100 mm Estuarine residence time: <20 days Freshwater rearing: >1 year

strategies of Chinook salmon (Figure 2). For the estuary as a whole, all life history strategies were evident for longer periods of time throughout the year (Rich 1920); clearly, the current use of the estuary is more limited now than in the past. We examined possible changes in expression of different life history strategies for each ESU between historical and current conditions. Such a change analysis is restricted by the limited information on the distribution of life history strategies under either current or historical conditions for specific ESUs or populations. We defined the mix of life history strategies that we believe is likely being expressed by each ESU, under current conditions based on a variety of information sources including the life history type of that ESU, origin, unpublished data from the ongoing studies of Columbia River estuary (Bottom unpubl. data), and status reviews. We then extrapolated the likely mix of life history strategies historically employed by each ESU, using such information as Figure 2, and predicted how different types of changes affecting each ESU would have impacted each life history strategy (Table 4). The changes in distribution of life history strategies described in Table 4 should be considered hypotheses.

Selection of Estuarine Factors for Analysis

To facilitate recovery of endangered salmon stocks in the Columbia River basin, factors that currently act to suppress an increase in their viability need to be identified. If the factors are appropriately and correctly pinpointed and can be manipulated, it is logical to conclude that reducing their affect should improve the recovery potential of targeted populations. Further, incorporating the influence of a factor's impact on recovery of salmon populations should improve policy and management decisions.

We recognize that identifying single habitat factor solutions to salmon survival problems in isolation has not been historically effective. A growing body of evidence suggests that a broader scale view of ecosystem restoration that focuses on how alteration to landscape processes and ecosystem attributes affects salmon via habitat changes is more likely to succeed (NRC 1996, Beechie and Bolton 1999, Stouder et al. 1999, Roni et al. 2002, Beechie et al. 2003a, 2003b). Identification of limiting factors in the estuary represents a logical first step that needs to be incorporated into a broad, landscape-scale assessment of strategies to improve the recovery potential for endangered salmon populations. Our goal here is to consider the effects of several estuarine factors on recovery of listed anadromous salmonids.

The major estuarine related factors that we identified that can potentially affect salmonid population viability in the Columbia River estuary are climate and climate change (which can control other factors): water flow; access to and quality of habitats; sediment; salinity; temperature; toxics/contaminants; predators such as Caspian terns (*Sterna caspia*), cormorants (*Phalacrocorax auritus*), marine mammals, and northern pikeminnow (*Ptychocheilus oregonensis*); hatchery practices; and harvest practices. Although it would be useful to evaluate the role of each of these factors, we selected water flow, habitat, Caspian tern predation, and contaminants for detailed analysis. These four were selected based on four criteria:

1. Was there a significant change in the factor from historical condition?
2. Could the factor potentially affect population viability?
3. Was quantitative data available that could be used to analyze effects of the factor?
4. Could the factor be linked to hydropower operations in the Columbia River basin?

We relied primarily on the first three criteria and did not use the fourth to exclude any factor.

We wish to emphasize that factors were not selected based on whether or not we believed they had a significant affect on salmonid population viability. Some factors that were not selected for consideration in this report may have a significant affect on viability. For example, we expect that water temperatures have warmed from historical levels, which could exclude some habitat from use by juveniles during part of the year and affect metabolic processes of both salmon and their predators. These changes in water temperature could alter growth and survival. A major purpose in our evaluation was to lay out a framework for future consideration of other

Table 4. Linkage between anadromous salmonid ESU, life history type (ocean-type or stream-type), and dominant life history strategies of juvenile salmon in the Columbia River. We estimated general contribution to the outmigrant population of each life history strategy as Abundant (>50%), Medium (10–50%), Rare (1–9%), or Absent (<1%) listed for each ESU under historical (early 1900s) $^{(H)}$ and current conditions.$^{(C)}$ We made these estimates using a variety of data sources and our judgment about how the various changes occurring in the system would have affected each strategy within each ESU.

		Life history strategy					
ESU	**Life history type**	Early fry	Late fry	Early fingerling	Late fingerling	Subyearling	Yearling
Columbia River chum salmon	Ocean	Abundant $^{(H)}$ Abundant $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$
Snake River sockeye salmon	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Rare $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Lower Columbia River coho salmon	Stream	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Rare $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Upper Columbia River steelhead	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Snake River steelhead	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Lower Columbia River steelhead	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Medium $^{(H)}$ Rare $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Middle Columbia River steelhead	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Medium $^{(H)}$ Rare $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$
Upper Willamette River steelhead	Stream	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Absent $^{(H)}$ Absent $^{(C)}$	Rare $^{(H)}$ Absent $^{(C)}$	Abundant $^{(H)}$ Abundant $^{(C)}$

Table 4 continued. Linkage between anadromous salmonid ESU, life history type (ocean-type or stream-type), and dominant life history strategies of juvenile salmon in the Columbia River. We estimated general contribution to the outmigrant population of each life history strategy as Abundant (>50%), Medium (10–50%), Rare (1–9%), or Absent (<1%) listed for each ESU under historical (early 1900s) (H) and current conditions.(C) We made these estimates using a variety of data sources and our judgment about how the various changes occurring in the system would have affected each strategy within each ESU.

ESU	**Life history type**	**Life history strategy**					
		Early fry	Late fry	Early fingerling	Late fingerling	Subyearling	Yearling
Snake River fall Chinook salmon	Ocean	Absent (H) Absent (C)	Absent (H) Absent (C)	Medium (H) Rare (C)	Medium (H) Rare (C)	Abundant (H) Abundant (C)	Rare (H) Medium (C)
Upper Willamette River Chinook salmon	Ocean	Rare (H) Absent (C)	Rare (H) Absent (C)	Medium (H) Rare (C)	Medium (H) Rare (C)	Rare (H) Medium (C)	Abundant (H) Abundant (C)
Lower Columbia River fall Chinook salmon	Ocean	Medium (H) Rare (C)	Medium (H) Rare (C)	Medium (H) Rare (C)	Medium (H) Rare (C)	Medium (H) Abundant (C)	Rare (H) Rare (C)
Upper Columbia River spring Chinook salmon	Stream	Absent (H) Absent (C)	Absent (H) Absent (C)	Rare (H) Absent (C)	Rare (H) Absent (C)	Rare (H) Rare (C)	Abundant (H) Abundant (C)
Snake River spring/summer Chinook salmon	Stream	Absent (H) Absent (C)	Absent (H) Absent (C)	Rare (H) Absent (C)	Rare (H) Absent (C)	Rare (H) Rare (C)	Abundant (H) Abundant (C)

factors that may affect juvenile salmon use and that could be used as more comprehensive information becomes available.

For each factor, we considered how it could affect population viability based on available data and analyses. We then developed a series of hypotheses or principles about each factor that helped guide how we rated their relative importance for each ESU. In addition, we assessed changes in each factor from historical conditions. Such a comparison between historical and current conditions is a useful way to help identify causes and mechanisms of ecosystem change (e.g., Borde et al. 2003, Beechie et al. 2003b, Collins et al. 2003).

Analyzing and Rating the Relative Importance of Estuarine Factors

To rate the importance of each factor, we developed a simple rating system that ranked each factor as having a high, medium, or low ability to improve the status of anadromous salmon populations. We defined improvement in population status to mean improvement in population viability (McElhany et al. 2000). We drew inferences about how a factor affects an ESU based on the life history type of that ESU, how we believed the factor would affect the life history strategies that characterized that life history type, and the hypotheses and principles developed from the overviews of each factor. The limiting factors for all stream-type ESUs were ranked similarly while those for ocean-type ESUs were ranked similarly. Ratings were developed by considering each factor relative to other estuarine factors within an ESU; ratings were not considered relative to other nonestuarine factors such as tributary habitat.

The rating system consisted of two levels. The level 1 screen evaluated if the factor was likely a concern for an ESU based on its affects on VSP and changes in that factor from historical conditions. The level 2 screens asked how the factor affected an ESU based on where the affects occurred and the life history strategies affected.

Level 1: Is the Factor Likely a Concern for the ESU?

What is the affect on each VSP parameter?

Each factor will have some affect on each VSP parameter. We assumed, however, that if the factor affected large numbers of individuals in the ESU and individuals associated with multiple life history strategies (again, relative to other factors) that there was a significant affect on abundance and productivity. If a factor primarily affected only one or two particular life history strategies or specific habitat types more than others, we assumed the primary affect of the factor was on spatial structure and diversity. Because most populations in threatened or endangered status are at low levels of abundance, we reasoned that the abundance levels of these depressed populations needed to be increased as soon as possible. Therefore, we doubled the score if the factor affected abundance and productivity.

Has the factor changed from historic conditions and what it is the likelihood that it could be improved relative to the other factors?

We considered whether each factor had changed significantly from historical conditions. Because we intentionally selected factors that we believed had changed significantly from historical conditions, this screen did not differentiate much between factors. We also considered from a practical perspective how much change in each factor was possible. A factor could be significantly changed from historical levels but relatively difficult to change relative to other estuarine factors.

Level 2: How Does the Factor Affect the ESU?

Does the factor have a significant affect on the abundance of the dominant life history strategy?

For the dominant life history strategy, we asked how the factor affected the abundance of juveniles of that life history type in shallow water habitats (upstream of the river's mouth), deepwater habitats (upstream of the river's mouth), and plume habitats. We only considered these three areas because available information indicates that use of habitats in the plume and confined portion of the estuary (river mouth to Bonneville Dam) is quite different. Thus separation of the estuary into these two large zones was warranted. Within the zone from Bonneville Dam to the river's mouth, we also differentiated use of shallow, low velocity habitats from medium and deep, higher velocity channel habitats because this was also consistent with available information. Studies indicate that smaller juveniles more extensively use shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats) than medium and deep, higher velocity channel habitats; the opposite pattern exists for larger size classes such as yearlings (Healey 1980, 1982, Levy and Northcote 1981, 1982, Simenstad et al. 1982, Levings et al. 1986, Miller and Sadro 2003).

For the dominant life history strategy, does the factor affect habitat quality, quantity, and opportunity?

For the dominant life history strategy, we asked what type of affect the factor had in shallow water (upstream of the river's mouth), deepwater (upstream of the river's mouth), and plume habitats. We considered effects of the factor on habitat quantity, quality, and opportunity. The concepts of opportunity and quality (or capacity) were proposed by Simenstad and Cordell (2000) and adopted by Bottom et al. (2005) for the Columbia River estuary.

Opportunity attributes relate to the accessibility of habitat to juvenile salmon. In general, opportunity metrics are largely physical and chemical in nature such as tidal elevation, temperature, and location of habitat. For example, extreme high temperatures and diminished flows can constrain accessibility of shallow water habitat. Capacity measures primarily relate to the biotic and ecological functions (i.e., acquiring food and avoiding being eaten) of habitat. Capacity metrics must be considered within the context of the species and life stage using the habitat, and the location of that habitat within the landscape. In addition to capacity and opportunity, we also included quantity of habitat as a separate metric. For toxics, we rated effects separately in shallow water and deepwater estuarine habitat for waterborne and sediment-borne contaminants. For example, if there were risks to the main life history type from both types of contaminants in shallow water, then the score would double.

Each of the four questions listed above was evaluated for each factor and each ESU based on whether the ESU was classified as an ocean or stream life history type. Each cell in a matrix was either scored as a yes (+1) or no (0) with two exceptions: 1) abundance and productivity which were given a +2 score, and 2) toxics in deep and shallow water which each could be scored a +2 if there was effects from both waterborne and sediment-associated toxics. This affected the maximum possible score that could be assigned to a factor. For flow, habitat, and predation, the maximum possible score was 20, whereas the maximum possible toxics score was 28. The final rating was computed as the ratio between the assigned score and maximum possible score; a ratio of >0.66 was assigned a high ranking, 0.34–0.66 a medium ranking, and <0.34 a low ranking.

Analyses of the Effects of Estuarine Factors on Columbia River Basin Salmonids

Water Flow

Water, interacting with the land, creates and maintains the mosaic of estuarine habitats that juvenile salmon occupy. The estuarine habitat features to which salmon have adapted are largely the result of riverine and tidal processes and the physical characteristics of the watershed. The major geologic feature affecting flows through the basin is the Cascade mountain range, which divides the Columbia River drainage basin into interior and western subbasins. The moist and relatively warm western subbasin contains only about 8% of the total surface area of the 660,480-km^2 basin, but contributes almost one quarter of the total river flow (Orem 1968). Most of the western subbasin is at too low an elevation to accumulate a large seasonal snow pack. Thus the highest flows are observed during and after winter storms, between December and March. In contrast, most of the flow in the interior subbasin occurs as the result of the seasonal snow pack melt between April and July. Much of the interior subbasin is relatively arid, but its Canadian component experiences heavy winter snowfall and plays a major role in spring freshet flows.

Effects of Climate on Flow

Natural variations in Columbia River flows associated with both short and long term fluctuations in climate have a significant affect on amount and timing of water delivered to the estuary. These variations directly affect habitat conditions in the estuary, and help determine what areas are wetted and potentially accessible to juvenile salmon and how estuarine salinity gradients vary. Climate-induced variations in Columbia River flow occur on time scales from months to centuries (Chatters and Hoover 1986, 1992). One example of a longer term climate effect is the Pacific Decadel Oscillation, commonly known as the PDO (Francis and Hare 1994, Mantua et al. 1997), which alternates between cold and warm phases at approximately 30-year time scales. During the cold phase, more rainfall is typical in the Pacific Northwest, whereas in the warm phase, less rainfall occurs. The cold phase of the PDO (e.g., the 1945–1976 period) was regarded as benefiting salmonid production in the Pacific Northwest, while being less favorable for salmon originating in northern British Columbia and Alaska (Hare et al. 1999). The opposite circumstance prevails during the warm phase, characteristic of the recent period between 1977 to about 1998, when listing of salmon in the basin occurred. Another cold, wet phase seems to have commenced about 1998 (Peterson and Schwing 2003). PDO-related fluctuations in salmonid survival have been linked to the degree of density stratification of the coastal ocean (Gargett 1997), but they are also likely influenced by conditions within the river and estuary (e.g., salinity, turbidity, and river flow).

Another climate related feature is the El Niño-Southern Oscillation (ENSO; typically 3–7 years in duration) cycles (Redmond and Koch 1991, Kahya and Dracup 1993, Dracup and Kahya

1994, Gershunov et al. 1999, Jay 2001). ENSO cycles are shorter in duration compared to the PDO cycle, typically amplifying conditions associated with the cold or warm phase of the PDO. El Niño winters in the Pacific Northwest often bring high sea level, warm air temperature, low precipitation, low snowpack, and weak subsequent spring freshet flows (Kahya and Dracup 1993, Dracup and Kathya 1994). La Niña winters (the contrast to El Niño) typically exhibit an opposing climate and hydrological response. As a consequence, the annual average flows of the Columbia and the Willamette rivers during years with a strong El Niño winter are 91% and 92% of the long-term annual average, while in case of strong La Niña winters, they are 110% and 111%, respectively. El Niño effects are intensified during a warm-PDO phase, while those of La Niña are enhanced during a cold-PDO phase (Gershunov et al. 1999). Conversely, El Niño effects are suppressed during the cold-PDO phase, as are those of La Niña during the warm-PDO phase (Jay 2001).

The Columbia River basin's climate response is conditioned by its position between $41^{o}30'$ and $54^{o}40'$N lat., within a latitudinal band of strong response to the ENSO cycle and to the PDO (Mantua et al. 1997). However, the affects of climate on flow vary considerably depending on location within the basin. While the river flow per unit area is much larger in the western than in the interior subbasin, there are only modest variations across the basin in response to ENSO or PDO forcing. Still, the relatively large north-south extent of the basin brings about important differences in flow seasonality—the incidence of winter floods and timing of spring snowmelt—even within the interior subbasin.

What Changes Have Occurred in Flow Attributes

Changes in flow attributes, such as when and how much water arrives in the estuary, are an integral measure of alterations in a river system. In a recent analysis and review, Jay (2005) provided a detailed estimate of changes in flows in the Columbia River for the past 100 years, thereby providing an accounting of flow conditions during the historical and current period. One measure of hydrological change is the overall quantity of water delivered to the estuary. Jay concluded that there has been approximately a 16.4% reduction in river flow during the last 100 years. He evaluated the contribution of climate and human perturbation on the observed flow and concluded that approximately half of this change was due to climate (less rainfall) and half to human activities (e.g., water withdrawal for irrigation). In addition, a small percentage of the decrease was assigned to a combination of uncertainty (error) and evaporation from the impoundment of water in reservoirs due to increased surface area in the basin. Jay's conclusions were based on the record of observed flows for the past 100 years and an estimated adjusted flow provided by the U.S. Geological Survey (USGS) to account for reservoir manipulations; this was used to estimate the virgin river flow (flow unadulterated due to hydropower operations and irrigation removal) from approximately 100 years ago (Figure 6).

Reductions in the total amount of water (i.e., annual average flow) are only a small part of the total hydrological changes that have occurred in the Columbia River basin. Seasonal changes, particularly those involving the timing and magnitude of the spring freshet, have been much larger than changes in annual average flow. Spring freshets are extremely important for juvenile salmonids in that high flows (especially overbank flows) provide habitat, transport fish downstream, limit predation by increasing turbidity, and maintain favorable water temperatures during the spring and early summer. Organic matter supplied by the river during the freshet

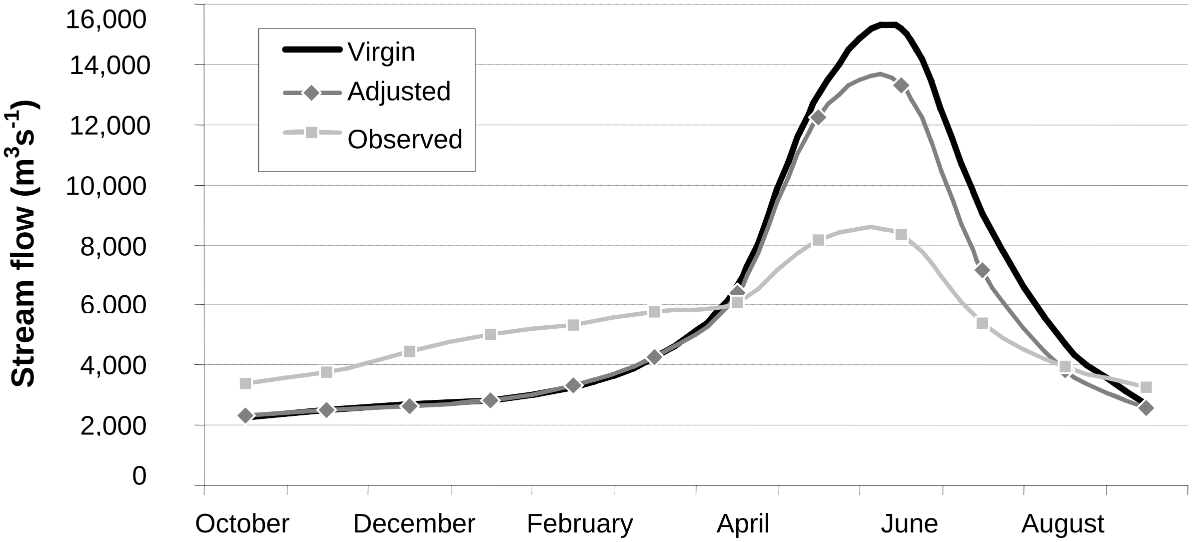

Figure 6. Comparison of the monthly averaged Columbia River eastern subbasin virgin, adjusted, and observed river-flow (cubic meters per second) estimates from 1970 to 1999. Flow regulation and irrigation depletion decreased spring and summer flows (May to August), while fall and winter flows (September to March) increased. (Reprinted from Bottom et al. 2005.)

season is also a major factor maintaining a detritus-based food web, centered in the estuarine turbidity maximum (ETM).

Very large freshets before modern flow regulation (i.e., before ≈1970) lasted 30–60 days, with the sharpness of the peak largely governed by the relative timing of snowmelt throughout the basin. Flows in the Columbia River interior subbasin (the flow measured at the The Dalles) are primarily driven by spring snowmelt, although there are rain-on-snow freshets in some winters. Before 1900, the highest flows typically occurred during May–July (Figure 7).

The timing of the maximum spring freshet flow has also changed (Figure 8). Maximum daily spring freshet flow now typically occurs at about water-year Day 242 (29 May), whereas maximum flow occurred in the 19th century at about water-year Day 256 (12 June), a change of about two weeks. In terms of the phase of the annual flow fluctuation, the freshet is about a month earlier. Part of this change is due to climate warming, but a component is also due to pre-release of water for flood control before the spring freshet. Irrigation withdrawal usually peaks in June, which tends to further curtail the freshet.

Another change in water flow, significant to salmon, is the occurrence of overbank flows. The historical bankfull flow level was an estimated 18,000 m^3s^{-1} for the main stem below Vancouver (Jay 2005). Modern bankfull level is set by the standard project flood level of approximately 24,000 m^3s^{-1} for the lower river. According to Jay (2005), some overbank flow occurred in many years before 1900, both in winter and spring (Figure 9), whereas substantial overbank flow (above 24,000 m^3s^{-1}) is now rare, with significant events occurring only five times during the last half century. Historical bankfull levels of 18,000 m^3s^{-1} are now rarely exceeded due to the combined effects of flood control measures and irrigation depletion. The season when

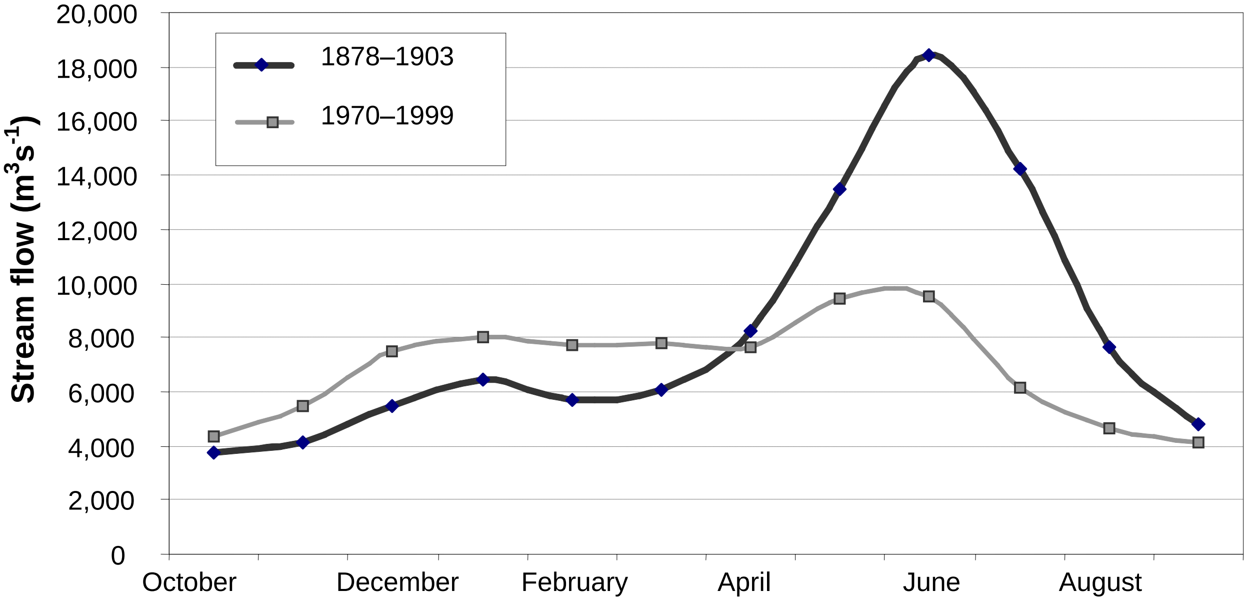

Figure 7. Changes in the annual Columbia River flow cycle at Beaver Army Terminal, near Quincy, Oregon, 1878–1903 versus 1970–1999. (Reprinted from Bottom et al. 2005.)

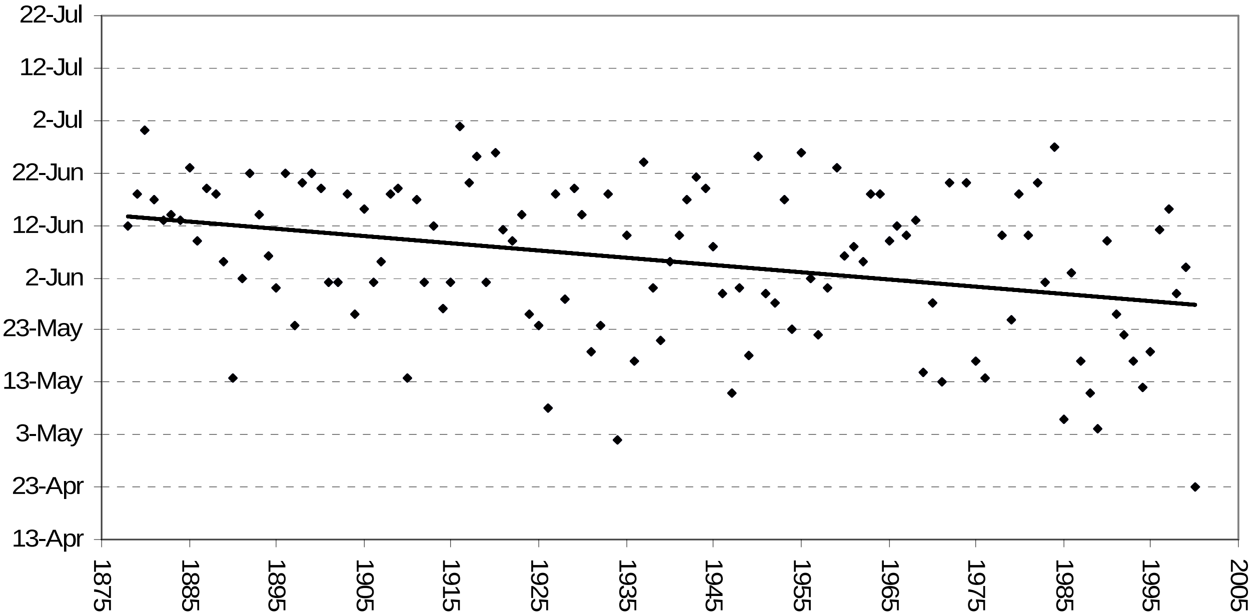

Figure 8. Peak freshet day versus year suggests that the freshet is now about two weeks earlier than in the 19th century in the Columbia River basin. (Reprinted from Bottom et al. 2005.)

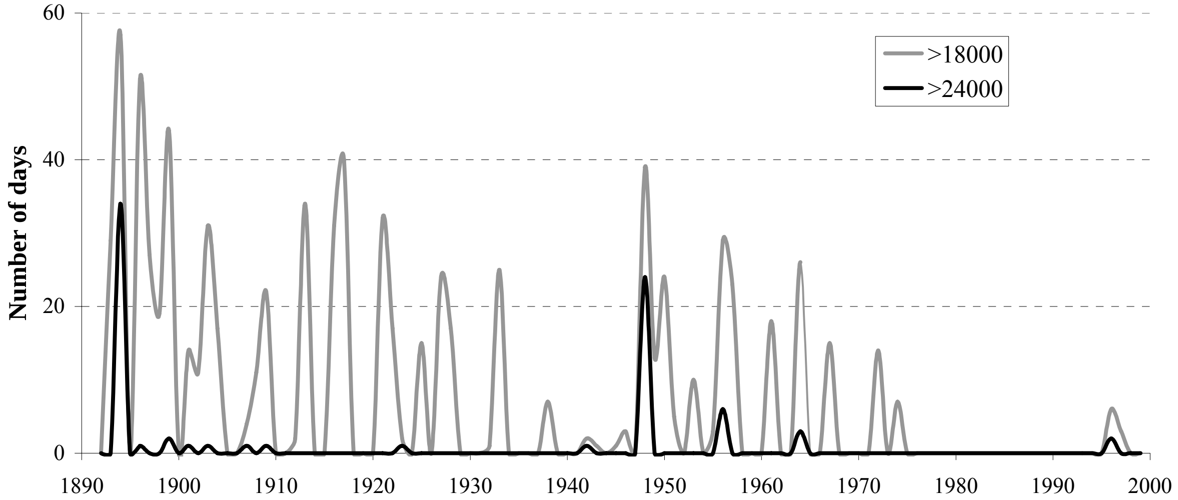

Figure 9. The incidence of flows above 18,000 m^3s^{-1} (the pre-1900 estimated bankfull flow level) and above 24,000 m^3s^{-1} (the present bankfull flow level). The present bankfull flow level has been exceeded only five times (twice in 1956) in four years since 1948. (Reprinted from Bottom et al. 2005.)

overbank flow typically occurs has also shifted from spring to winter, because western subbasin winter floods (not interior subbasin spring freshets) are now the major source of such flows (Jay 2005).

Factors Causing Changes in Flow Attributes

Potentially, flow changes (e.g., decreased spring freshets) can be a result of climate change, flow regulation, and water withdrawal. Jay (2005) apportioned the timing and magnitude of the freshet change to these three factors and found that flow regulation is clearly the source of the largest reduction in spring flow. The total reduction in freshet season (May through July) mean flow due to climate change, irrigation depletion, and flow regulation is 5,870 m^3s^{-1} or 43% of the virgin flow for this period. Overall the present freshet season flow decrease due to flow regulation was an estimated 33.1% (a reduction of 31.6% for May, 32.4% for June, and 19.8% for July, respectively). Jay found that the flow decrease in the freshet period resulting from climate was 5.6%. Similarly, the present decrease in freshet season flow due to water withdrawal was an estimated 10.5% (a reduction of 5.7% for May, 12.5% for June, and 20.8% for July, respectively). The January through July virgin flow average for 1879–1899 was 8,050 m^3s^{-1}, while for 1945–1989 it was 7,850 m^3s^{-1}, a decrease of only 2.5%. Thus most of the loss of freshet flow represents flow that now occurs during other time periods.

Climate was found to be a secondary factor in the incidence of overbank flow (Jay 2005). Overbank flow events were more common during the cold-PDO phase (1945–1977) than during the preceding warm-PDO phase (1921–1944), even though the degree of flow regulation and irrigation depletion grew over time (Figure 10). Jay concluded flood protection, diking, flow regulation, and water withdrawal largely eliminated climate’s influence on overbank flow.

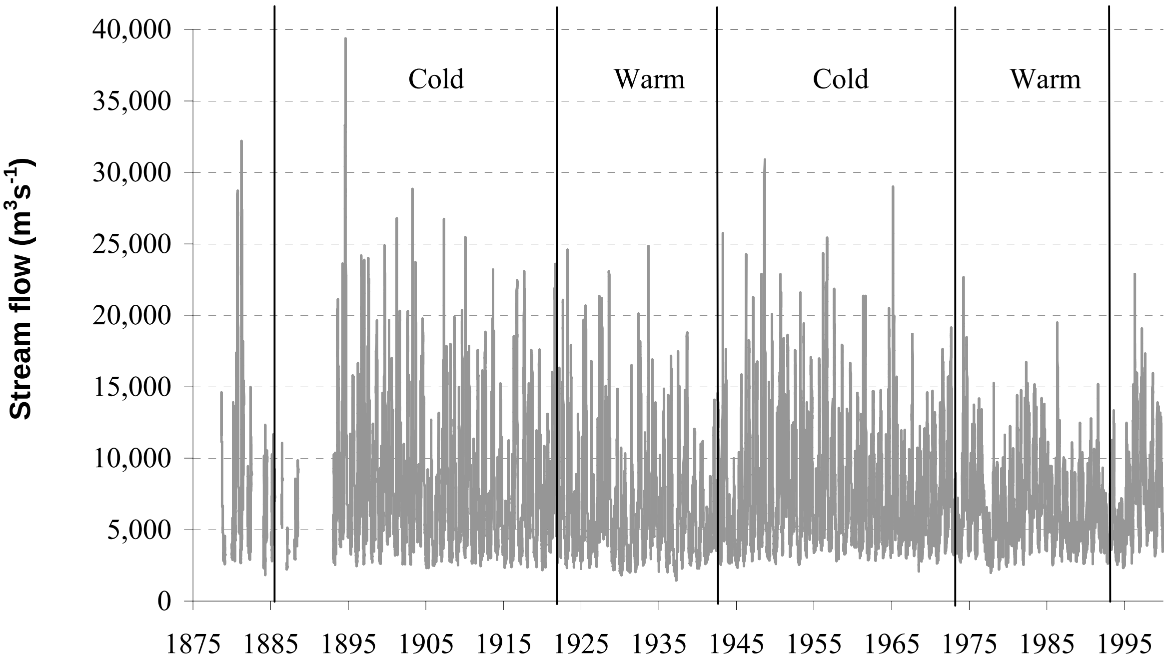

Figure 10. Monthly average flows at Beaver Army Terminal present and historical bankfull flow levels, and warm and cold PDO cycles, 1878–1999. Present bankfull flow levels are 24,000 m^3s^{-1} and historical bankfull flows are 18,000 m^3s^{-1}. Historically there was a major difference between the warm and cold phases of the PDO cycle in disturbance frequency. This has been largely eliminated by flow regulation and diking; overbank flow is now a rare event. (Reprinted from Bottom et al. 2005.)

Flow Effects on Sediment Transport

A significant consequence of altering flow regimes is change in movements of sediment through the Columbia River system. Understanding changes in sediment are important because sediment is a direct part of habitat—it helps form habitat (via sediment deposition and erosion), has associated nutrients from upriver sources (supporting food production in the lower river, estuary, and plume), and contributes to a turbid environment (further enabling the habitat to serve as a refugia from predation). Our ability to evaluate the role of change in flow on supply of sediment through the lower river and estuary is limited, primarily because all the particles comprising sediment are not equally characterized. For example, sand and gravel movement is transport-capacity limited. Material is always available on the bed and will move whenever flow conditions are suitable. Therefore, sand-transport estimates are nearly as accurate as the flow estimates, at least within the limited flow range of the data set used to calibrate sand transport (Jay 2005). In contrast, transport of fine sediments is supply-limited, thus the capacity to move the material is greater than the supply. Since direct measurements are absent from the historical record, characterization of change in fine sediment is not possible.

The effects of climate change on interior subbasin total sediment supply can be assessed by considering historical changes in sediment transport associated with virgin flow at The Dalles (Figure 11). The hindcast total load (sand plus finer fractions) associated with The Dalles virgin flow was 20.5×10^6 t for 1879–1899, whereas hindcast annual average total virgin flow sediment

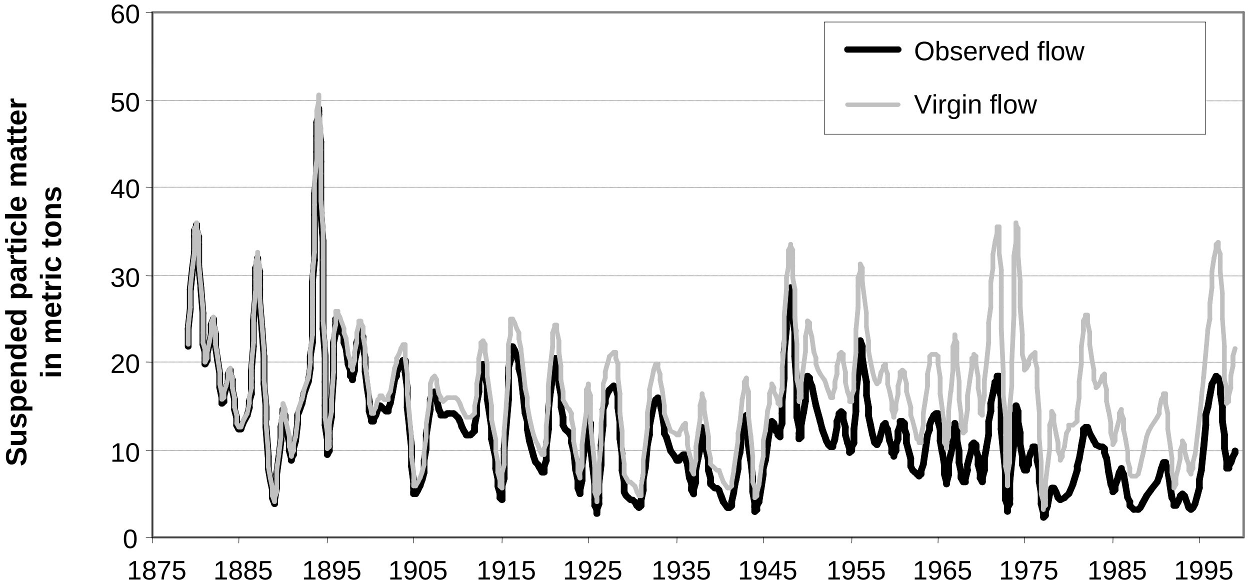

Figure 11. Hindcast total sediment load for the Columbia River eastern subbasin associated with observed flow and estimated virgin flow. The two are very similar before 1900, but the sediment transport hindcast from observed flow after about 1970 is much smaller than virgin flow sediment transport, because of flow regulation and water withdrawal. (Reprinted from Bottom et al. 2005.)

transport was 17.2 × 10^6 t for 1945–1999. The estimated decrease in total sediment transport due to climate change was approximately 17% (Jay 2005). This is considerably larger than the climate-driven change in flow for two reasons: 1) sediment transport varies with a power n of the flow, $n \approx 2.5$, and 2) decreased mean flow also reflects decreased flow variability and a decreased incidence of the very high-flow days that transport most of the total load.

Because there is a nonlinear relationship between sediment transport and flow, it is not possible to precisely apportion the reduction in sediment transport between climate change, water withdrawal, and flow regulation. The largest single factor is, however, reduction in spring freshet flow. This is demonstrated when evaluating the impact of flow on sediment volumes under recent conditions, compared to a more historical period when the spring freshet was less impacted. Jay (2005) reported the average sediment transport associated with the observed 1945–1999 flow was approximately 10.0 × 10^6 t, whereas sediment transport associated with the 1970–1999 observed flow was approximately 8.0 × 10^6 t. Therefore, the difference between annual average sediment transport at Vancouver for the 1858–1899 virgin flow and 1945–1999 observed flow was 10.8 million × 10^6 metric or 52% of the 19th century sediment transport. The difference between annual average sediment transports for the 1879–1899 virgin flow and 1970–1999 observed flow is approximately 12.5 × 10^6 t or 61.5% of the 19th century sediment transport. The sand transport associated with the 1945–1999 observed flow was approximately 3.2 × 10^6 t. Therefore, the difference between annual average sand transport for the 1879–1899 virgin flow and 1945–1999 observed flow was approximately 7.7 × 10^6 t or approximately 71% of the 19th century sand transport. The consequence of the reduced transport of sediment through the lower Columbia River estuary and plume are unknown at this time, but the magnitude of the decrease likely impacts the support of habitat-forming processes as well as increasing predation pressure on juvenile salmon utilizing and migrating through the lower river, estuary, and plume because of increased visibility.

Impacts of Water Flow Changes on Salmonid Habitat in the Estuary

Alterations in the amount and timing of water delivery to the estuary has significantly affected availability of the habitat needed in the estuary to sustain the diverse life history strategies for the various source populations of salmon and steelhead. Several recent analyses provide some empirical evidence on the effects of the altered flow regime described above on habitat in the Columbia River estuary. Baptista (2005) used a hydrologic model developed specifically for the Columbia River and found the estuary during the historical period (late 1800s) was able to sustain habitat features important to salmon (characterized as water velocities less than 30 cm/sec—important to smaller juvenile salmon) to a greater degree than is evident now (Figure 12).

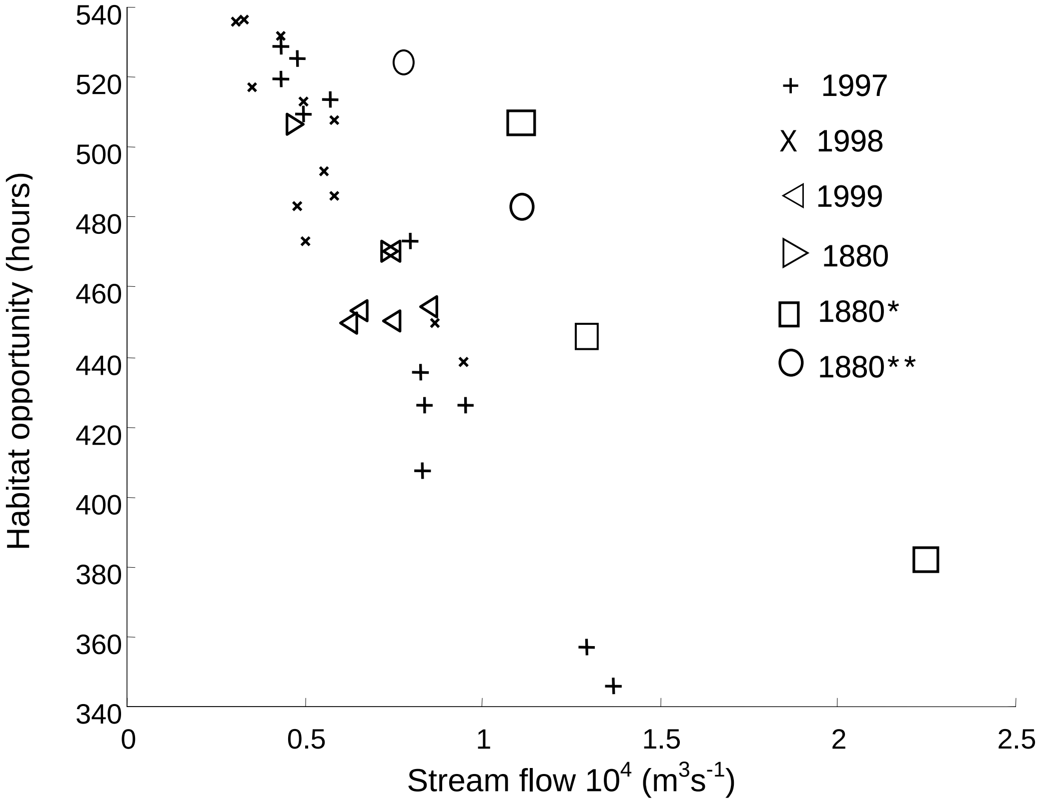

Figure 12. Habitat opportunity in Cathlamet Bay as a function of stream flow. Habitat opportunity, shown in hours normalized to a 30-day month, is defined as a site with water velocity less than 30 cm/sec. There are marked differences between habitat opportunity in the modern (1970 to present) and predevelopment (pre-1900) systems. In the figure, 1880 represents conditions associated with pre-1900 discharge and pre-1900 bathymetry, 1880* represents conditions associated with pre-1900 discharge and modified pre-1900 bathymetry to assess effect of change in bathymetry, and 1880** represents conditions associated with modern discharge and pre-1900 bathymetry to assess change in discharge. (Adapted from Bottom et al. 2005.)

Kukulka and Jay (2003) demonstrated that there was approximately a 62% loss of shallow water habitat (defined by depth between 10 cm and 2 m) that was attributable to diking (physically removing access of water to the tidal floodplains) and the reduction of peak flows by 40% (consistent with the freshet flow reduction discussed earlier) for the region between RM 50 and RM 90 on the Columbia River (Figure 13). The analysis incorporated the spring freshet period, when maximal use of estuarine habitat by different life history types employing a variety of strategies appears to occur (Bottom unpubl. data). Within the freshwater tidally influenced region of the Columbia River estuary, flow reductions and diking have reduced the amount of shallow water habitat by 52% and 29%, respectively (Bottom et al. 2005).

The hydrological changes—particularly those associated with flow regulation and water withdrawals, along with floodplain diking (discussed in the next section)—represent a fundamental shift in the physical state of the Columbia River ecosystem. Such changes can have significant consequences for viable salmonid populations. Major departures from the historical template of an ecosystem can potentially result in the loss of habitat capacity needed to rear fish, resulting in a reduction in abundance of the population. In addition, diversity of salmonid

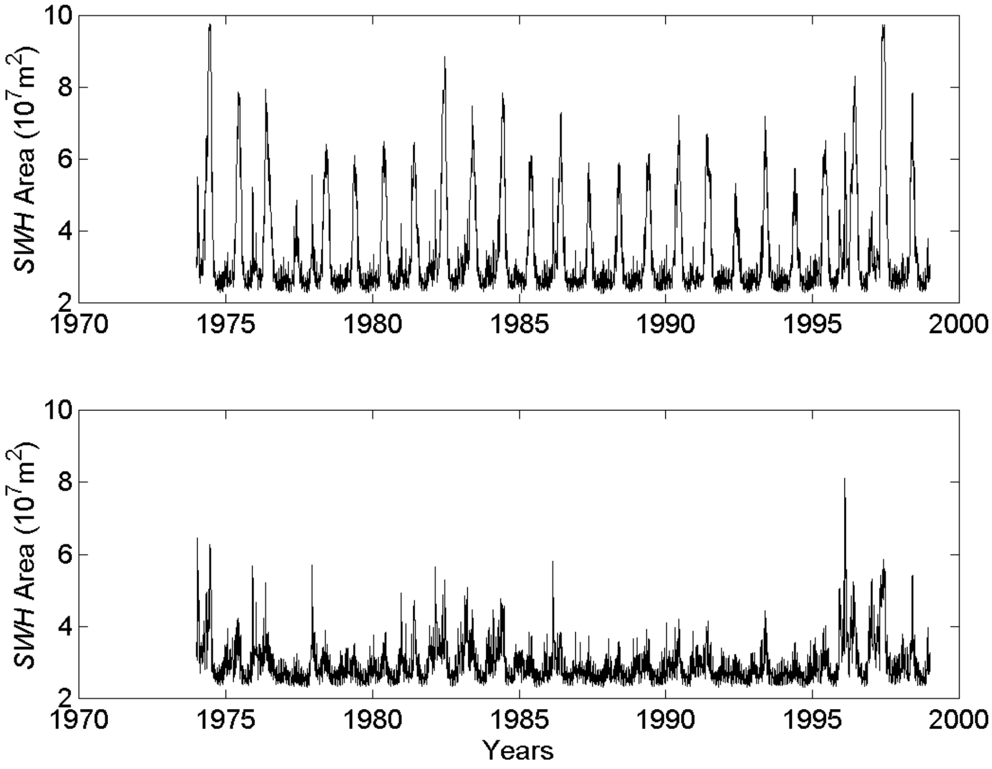

Figure 13. The change in availability of shallow water habitat (SWH) in the tidally influenced region between RM 50 and RM 90 on the Columbia River under unmodified and modified flow conditions only. The top panel represents conditions under virgin flow with no dikes, where extensive inundation of the floodplain occurs for long durations. The bottom panel represents conditions under modern flow conditions with no dikes, where river stage has been lowered and the floodplain for shorter duration innundated. (From Kukulka and Jay 2003.)

rearing and migration behaviors are linked to various habitats and environmental conditions that can support each developmental stage (e.g., egg, fry, smolt). Alteration of the physical environment of the estuary can create mismatches between established salmon behaviors and the physical environment or, similarly, prevent the expression of potential behaviors by eliminating habitat opportunity. The loss of these behaviors can reduce the life history diversity of populations.

The effort to stabilize flows in the Columbia River basin may create less stable conditions for salmonids whose migration and rearing behaviors have adapted to historical patterns of hydrologic variability. Of particular importance is the reduction of the spring freshet, to which the timing of downstream migrations and patterns of habitat use of some subyearling and yearling life history types may have been linked. One potential result of dampening flowvariations in the Columbia River could be a greater uniformity of migration patterns with potential consequences in the timing and sizes of salmon arriving in the estuary or ocean.

The nearly complete elimination of overbank flooding may pose some of the most significant consequences for Columbia River salmonids. One consequence of the loss of overbank flooding is that access to off-channel floodplain habitats and refugia during high flow events is eliminated. If, as we suspect, patterns of extended estuary use by small subyearling migrants are directly linked to the availability of shallow water habitat, the loss of these habitats has significant implications for these estuarine-dependent strategies.

In addition to the lost opportunity to use shallow water habitats on salmon diversity, flow regulation, in conjunction with floodplain diking, may influence the productive capacity of the estuary by regulating "ecological processes" such as food production, competition, and predation. Elimination of overbank flooding can prevent the pulsed delivery of structural and energetic components to the rest of the estuary, including large wood, sediments, detritus, and prey organisms produced in adjacent riparian and floodplain habitats. Floodplain inundation can also greatly increase the surface area of tidal estuarine and riverine habitats available to salmonids, allowing fish to expand their distribution into potentially more productive off-channel. For example, recent studies in a nontidal portion of the lower Sacramento River found that tagged juvenile Chinook salmon released in the seasonally inundated floodplain had better growth, higher consumption rates, and improved survival compared with others released into the main river channel (Sommer et al. 2001).

The effect of flow changes is not restricted to the area traditionally considered the estuary in the Columbia River system (i.e., upstream of the river mouth). Flow from the Columbia River, as well as other physical changes in the estuary (e.g., dredging), can modify the features that define habitat in the plume (Barnes et al. 1972). Attributes of the plume affected by flow changes that define habitat important to salmon include surface area of the plume, the volume of the plume waters, the extent and intensity of frontal features, and the extent and distance offshore of plume waters. Both natural and anthropogenic factors that modify flow can logically be considered to modify habitat used by salmon as they make the transition to marine life.

Clearly any such changes are only relevant if the plume has a role that influences how juveniles make the transition from a freshwater to marine environment. As noted previously, evidence to date suggests the plume serves salmon in multiple ways, such as facilitating primary production during the spring freshet period (Thomas et al. 2003), distributing juvenile salmon in the coastal environment, concentrating food resources such as zooplankton, and providing a refuge from predators in the more turbid low salinity plume waters. Effects of changing plume attributes on juvenile salmon performance are largely speculative. It seems reasonable to hypothesize that a change in plume size would reduce the amount of foraging habitat potentially available to the juvenile salmon. A significant enough reduction in plume size could result in density-dependent interactions that diminish growth. Another affect of reducing the size of the plume would be to increase predation on yearlings occupying this habitat since the fish are occupying a smaller space. If one function of the plume is to transport fish offshore, then having the plume closer to shore could affect the ability of fish to migrate toward oceanic feeding grounds.

Summary of Flow Effects

In summary, flow is a fundamental factor affecting characteristics of salmon and their habitat in the estuary and plume. Large scale effects on flow occur as a result of spatially explicit interactions of short- and long-term climate cycles (ENSO and PDO, respectively) with the watershed. The generation of electricity, flood control, and irrigation have had significant affects on attributes of flow. These include a reduction in the mean annual flow, reductions in the size of the spring freshets, an almost complete loss of overbank flows, changes in timing of ecologically important flow events, and altered sediment processes. The hydrological changes, along with floodplain diking, represent a fundamental shift in the physical characteristics of the Columbia River ecosystem. Such changes potentially have significant consequences for both expression of salmonid diversity and productivity of the populations by affecting accessibility, quantity, and quality of estuarine habitat. In particular, because the changes in habitat are most pronounced in shallow water areas, we conclude that effects on the ESUs and life history strategies (the fry and fingerling strategies) that use these shallow water areas will be most significant. Further, we conclude that flow alterations are an important limiting factor in the plume. Primarily as a result of flow, but also no doubt as a result of physical changes to the estuary (e.g., dredging and diking), the shape, behavior, size, and composition of the plume has been changed.

Habitat

Estuarine Habitat and its Use by Juvenile Salmon

Habitat is the physical, biological, and chemical characteristics of a specific unit of the environment occupied by a specific plant or animal. Thus habitat is unique to specific organisms and basically encompasses all the physiochemical and biological requirements of that organism within a spatial unit. Characterizing habitat and its role in salmon life history and ecology has been a consistent objective in many studies focusing on salmon biology, particularly in freshwater spawning and rearing habitats. Comparatively less is known about estuarine habitat use by juvenile salmonids, including in the Columbia River estuary. The need for more quantitative descriptions of habitat attributes important to salmon in the Columbia River estuary has intensified recently as the calls for habitat restoration actions have increased in response to mandates to reduce risks to endangered salmon populations. Asking what to recover represents the most common response to this call. In lieu of specific knowledge, restoring the entire estuary for all plants and animals becomes the primary option embraced (LCREP 1999). This type of broadscale approach is limiting, however, because estuaries are typically extensively urbanized environments, so any changes that restore historical attributes are costly, particularly those associated with land acquisition. Questions quickly erupt as to what specifically needs to be restored, once real costs for the broad-brush approach become evident.

Although specific studies in the Columbia River estuary are limited, the summary of information from a variety of other sources can be used to provide perspective on what habitat attributes are important to juvenile salmon, how they use this habitat, and benefits of this habitat use. Research in estuarine systems throughout the Pacific Northwest has demonstrated that estuaries are composed of a variety of habitats that are used differently by salmon depending on their life history strategy and population of origin (Healey 1980, 1982, Levy and Northcote 1981, 1982, Simenstad et al. 1982, Levings et al. 1986, Miller and Sadro 2003). For example, the timing of catches of juvenile Chinook salmon in different zones of the Columbia River estuary varies (Figure 14). One of the most important features of life history strategies employed by salmon that affects habitat specific use is juvenile size (Figure 15). Smaller fish tend to be most abundant in shallow water areas. Larger juveniles typically use deeper water habitats within the estuary, located more centrally to mainstem channels; the larger juveniles are associated with yearling and subyearling life history strategies and are often of hatchery origin (Figure 16). The smaller juvenile salmon use the more peripheral side channel areas associated with the more shallow water habitats including tidal emergent marsh and forested marsh habitats (McCabe et al. 1986). These smaller fish are associated with the fry and fingerling strategies (Figure 16) and are naturally produced.

The size structure of Chinook salmon in the estuary demonstrates several attributes of estuarine habitat use (Figure 15). During the early winter months, the sizes of juveniles entering and leaving the estuary are nearly identical, consistent with salmon exhibiting the fry life history strategy. However, as the size of Chinook salmon entering the estuary increases in length as the year proceeds, the corresponding length of juvenile Chinook salmon exiting the estuary during the same period are significantly larger, suggesting growth and thus rearing is occurring in the estuary.

Information currently being collected by NOAA Fisheries Service demonstrates that populations exhibit a variety of approaches in how they use estuarine habitats.[5] For example, although it has been argued by Bottom et al. (2005) that salmon occupying shallow marsh habitats express the range of strategies characteristic of ocean-type salmon, it is now evident that salmon representing many of the endangered ESUs are using these habitats of the Columbia River estuary. Using genetic analysis employing recently developed microsatellite DNA techniques, both ocean and stream-type Chinook salmon from upper and lower basin sources were found in these marsh and forested wetland habitats (Figure 17).

Although the predominant source of fish in marsh habitats was from the lower Columbia River Chinook salmon ESU (Bottom et al. 2005), juvenile Chinook salmon from a variety of other source populations were evident. Confirmation that ESUs express a variety of strategies in their use of estuarine habitats is gained from ongoing studies evaluating the role of the Columbia

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[5] See footnote 3.

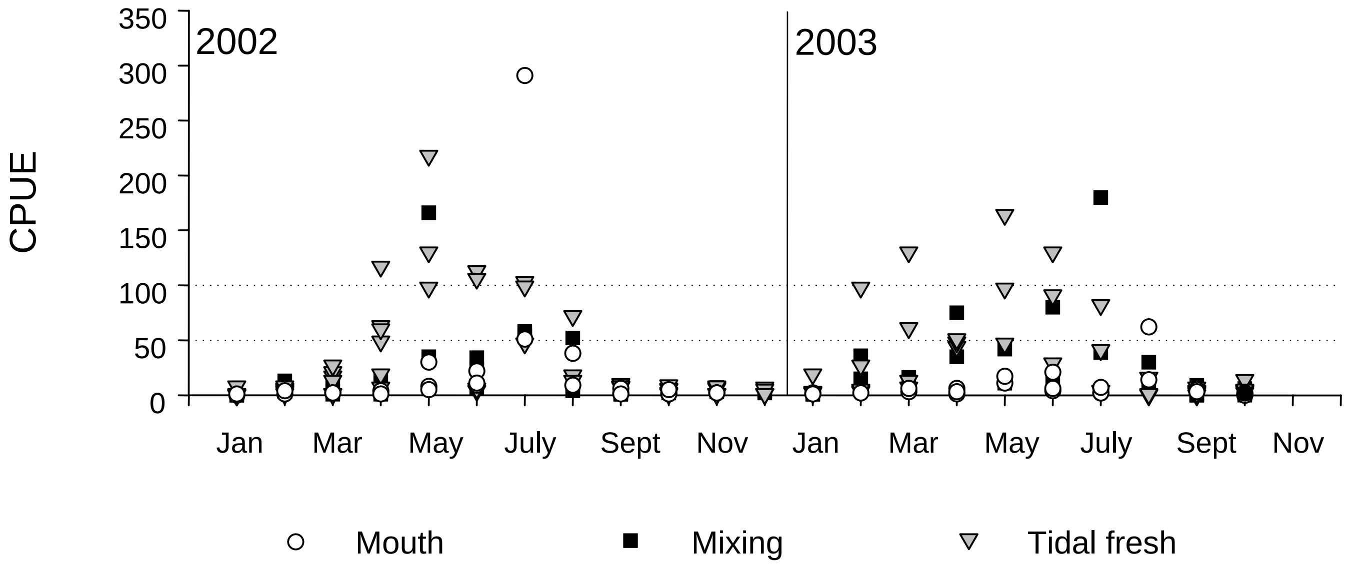

Figure 14. Catch per unit effort (CPUE) for juvenile Chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (triangle). (From Curtis Roegner, NWFSC.)

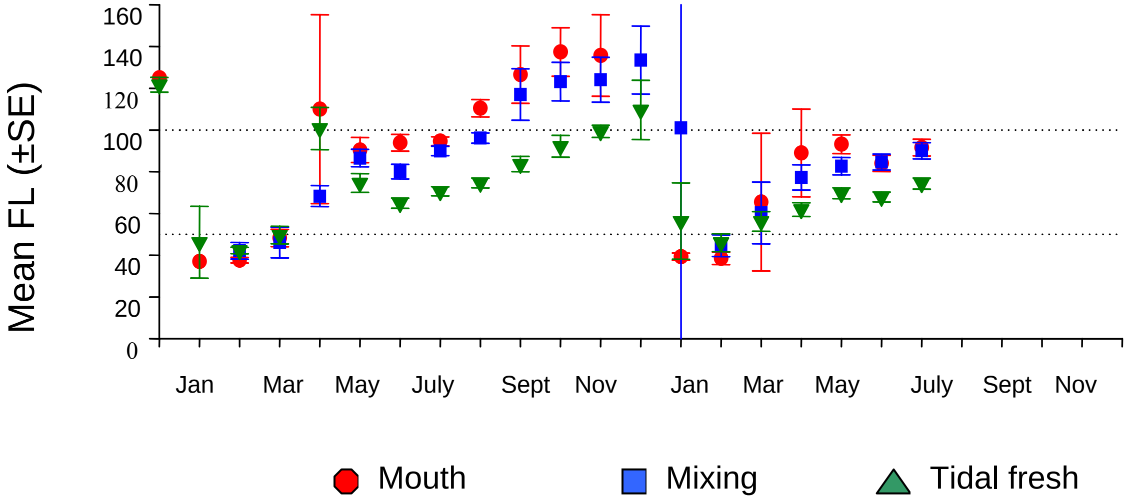

Figure 15. Mean fork length (FL) of juvenile Chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (triangle). (From Curtis Roegner, NWFSC.)

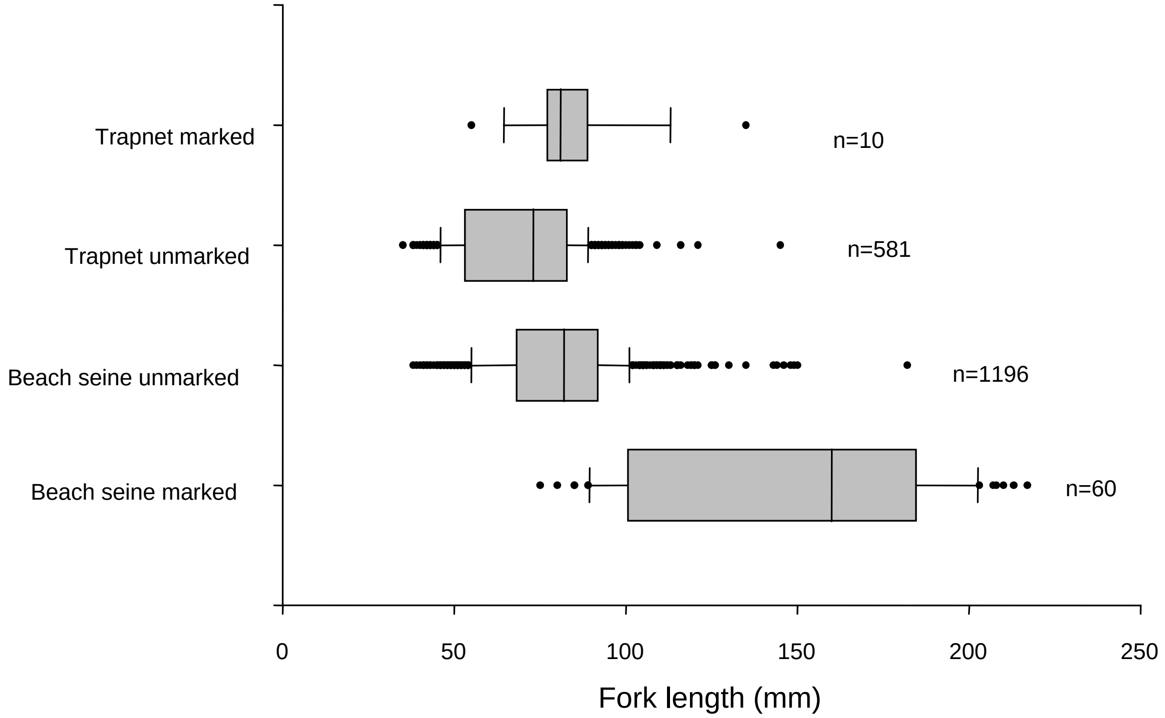

Figure 16. Box plots of size of juvenile salmon captured in peripheral habitats (trapnets) and near the main channels (beach seine) of the Columbia River estuary. Marked salmon (adipose clipped) represent hatchery fish, whereas unmarked fish (adipose fin present) represent an unknown mixture of naturally produced and hatchery released juvenile salmon. (From D. Bottom, NWFSC.)

River plume as habitat for juvenile salmon. Spring Chinook salmon that express both yearling and subyearling strategies have been identified in the plume environment (Figure 18). Clearly, salmon expressing a variety of strategies from source populations originating throughout the basin can use estuarine habitats throughout the year.

Another distinguishing feature of salmon that influences their use of estuarine habitat is physiological condition relative to their conversion from parr to smolt. Those animals having made the physiological transformation associated with the endocrine-driven smolting process exhibit negative rheotactic behavior and typically use deeper water main channel habitats (associated with the stronger flow signals). Juvenile salmon that have not entered smoltification, but still are moving from natal rearing areas into the mainstem, estuary, and ocean habitats more frequently use side channel, shallow water habitats within the estuary. However, even for juvenile salmon that have smolted and have been shown to migrate through the estuarine habitat for short periods of time (days), a majority of them are found with prey items in their stomachs, suggesting they are feeding in estuarine habitats.

Although abundance of juveniles in the estuary fluctuates considerably throughout the year, juvenile salmon use the estuary during the entire year (Figures 2 and 14). Obviously existence of sufficient amounts of the appropriate subtypes of habitats allows all salmon species and steelhead in the Columbia River basin to express the appropriate spatial structure and

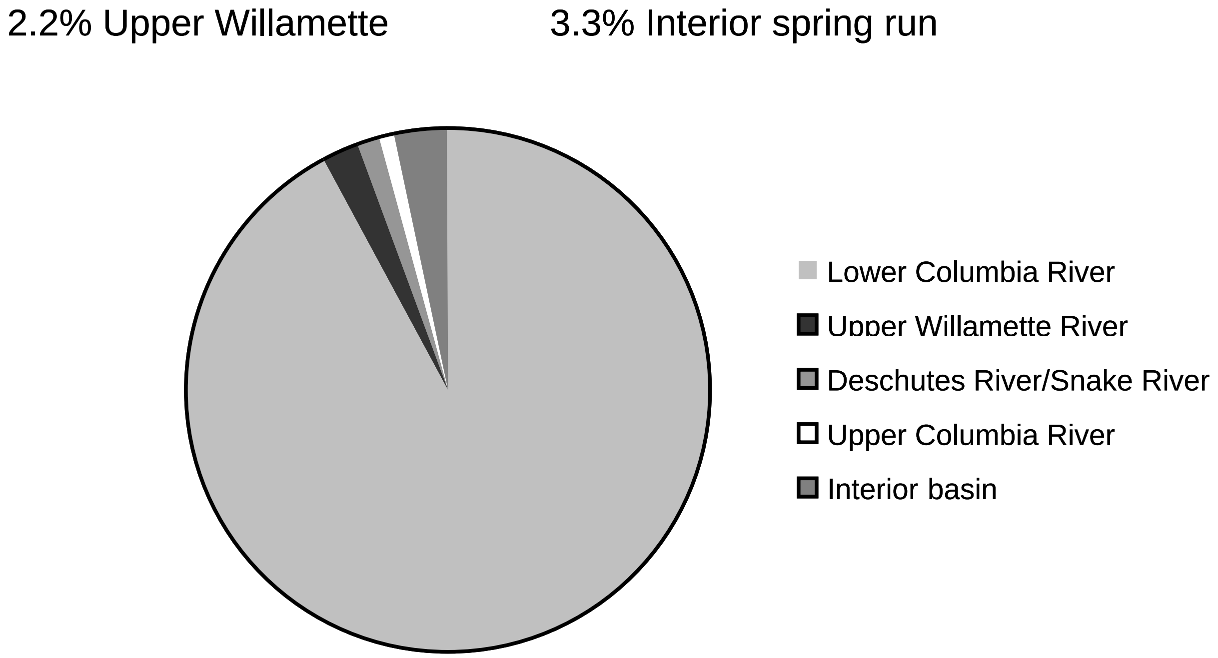

Figure 17. Proportion of Chinook salmon ESUs originating from various parts of the basin identified in samples taken from marsh, forested, and shrub habitat of the lower Columbia River estuary during 2002 from April to August. Lower Columbia, upper Willamette, Deschutes River/Snake River, and upper Columbia River refer to fall type Chinook salmon ESUs whereas interior refer to mid-Columbia River, upper Columbia River and Snake River spring type Chinook salmon ESUs. (From Paul Moran, NWFSC.)

diversity of life history strategies demanded by the environmental and biological conditions the juveniles encounter. This characteristic year long presence is consistent with the historical record reconstructed by Burke (2005) from research conducted by Willis Rich in the early 1900s (Figure 2). It is apparent that over the entire year, juvenile salmon expressing varying life history strategies were using the Columbia River estuary.

The characterization of how salmon employing varying strategies use the Columbia River estuary is comparable to recent evidence developed regarding the role of the Skagit River estuary in Puget Sound, Washington. These studies have suggested that smaller, naturally produced wild salmon are abundant in the estuary for longer periods of time (Figures 19 and 20) and associated with the more peripheral marsh habitats than larger juvenile salmon.[6] These larger fish either are hatchery fish (earlier in the season) or wild fish that have grown larger (mostly later in the season) (Figures 19 and 20).

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[6] C. Rice, NWFSC, Mukilteo Field Station, Mukilteo, WA, and E. Beamer, Skagit River System Cooperative, La Conner, WA. Pers. commun., January 2004.

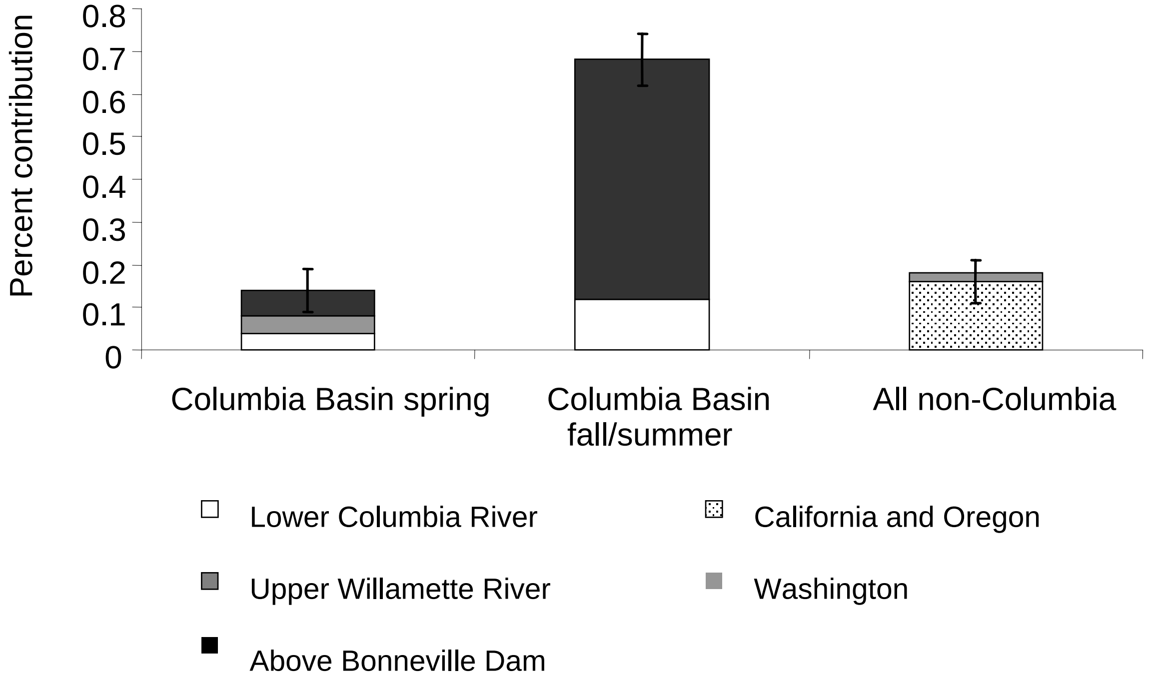

Figure 18. Stock composition of subyearling Chinook salmon in Columbia River plume study area June 1998–2001.

An important question to consider about estuarine habitat is whether habitat availability in the estuary can be a limiting factor to production and expression of a diversity of life history strategies. Although this information is forthcoming for the Columbia River estuary explicitly, studies in the Skagit River system have identified a density-dependent limit to the number of juveniles in the estuary relative to the overall abundance of young salmon in the system (Figure 21). In the face of altered (i.e., reduced) habitat availability, reducing the opportunity to access habitats can be a limiting factor in recovery of salmon and steelhead populations.

How Have Estuarine Habitats Changed

Considering the importance of estuarine habitat to juvenile salmon, what evidence from the historical and more current record can be used to identify changes in the habitat in the system? Thomas (1983) and Sherwood et al. (1990) have calculated losses of emergent marsh and forested wetland habitats in the Columbia River estuary (to approximately RM 42). They characterize the change as substantial and likely a significant factor reducing the estuary's ability to support juvenile salmon. Approximately 121.6 km^2 of tidal marshes (43% decline) and swamps (77% decline) that existed prior to 1870 have been lost (Figure 22). Together with a 12% loss of deep-water habitat, these changes reduced the estuary's tidal prism from 12% to 20%. In addition, the historical surface area of the estuary has decreased by approximately 20% as a result of diking or filling of tidal marshes and swamps.

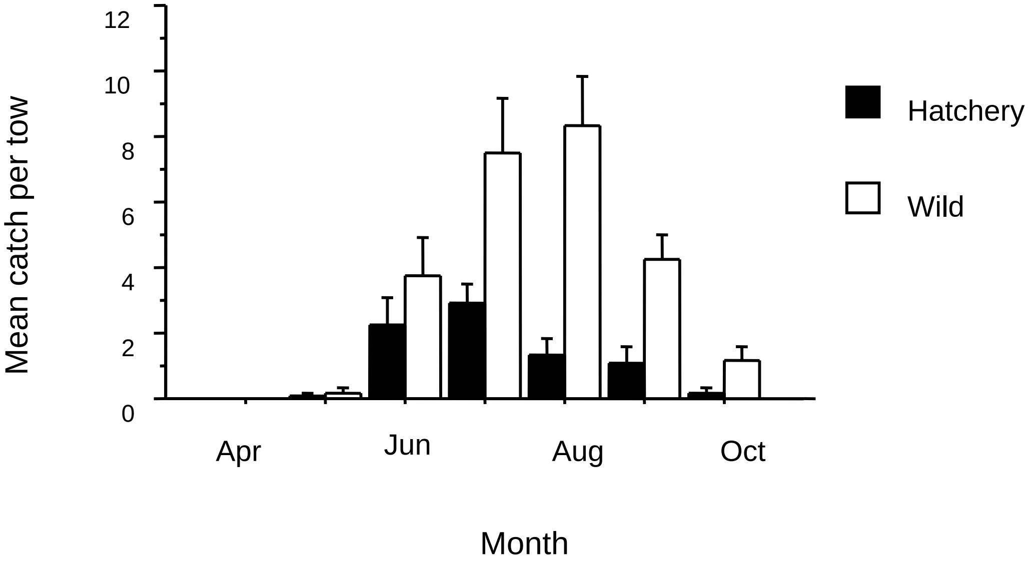

Figure 19. Catch of naturally and hatchery produced juvenile Chinook salmon over time in Skagit Bay, Washington, an enclosed olgohaline region associated with the Skagit River watershed. Fish were caught with a two-boat surface trawl. (From C. Rice, NWFSC.)

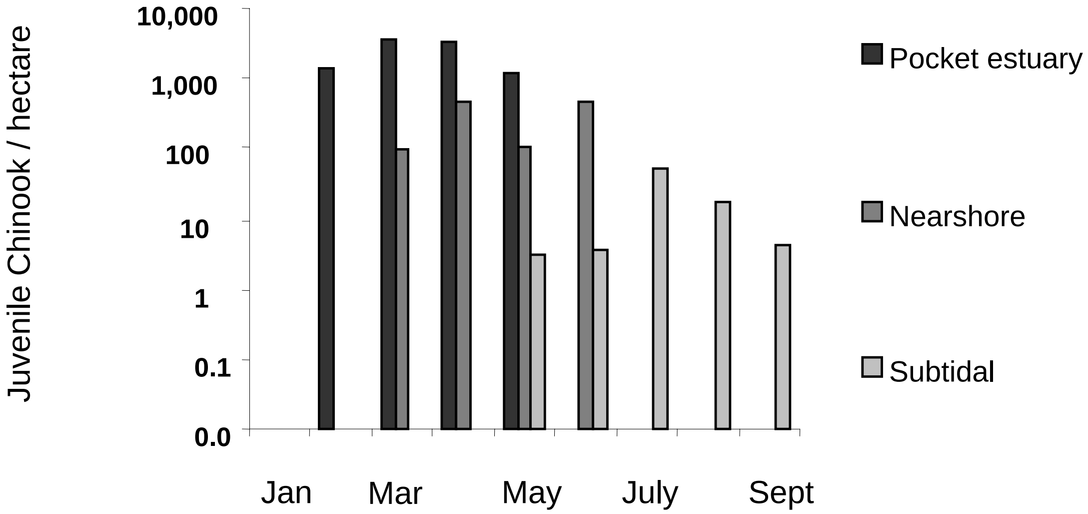

Figure 20. Shifts in distribution of juvenile Chinook salmon in the Skagit River estuary system. Juveniles are initially more prevalent in pocket estuary habitats, then shift to deeper, nearshore habitats later in the year, and more subtidal habitats toward the end of the year. The proportion of wild (unmarked) salmon ranged from 98%, 82%, and 73%. (From E. Beamer, Skagit River System Cooperative, and C. Rice, NWFSC. See also Beamer et al. 2005.)

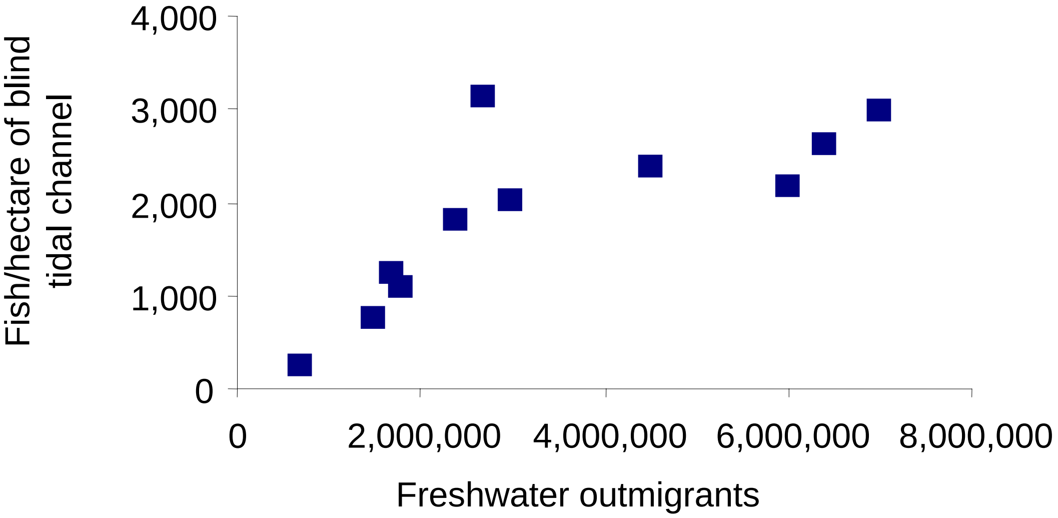

Figure 21. The relationship between freshwater wild Chinook smolt population size and density of juvenile wild Skagit Chinook in Skagit River delta habitat, 1992–2002. The number of Chinook per unit area within the delta levels off as the total number of outmigrants increases, indicating density-dependent use of the delta. (From E. Beamer, Skagit River System Cooperative. See also Beamer et al. 2005.)

As part of developing a restoration plan for the Columbia River estuary, Johnson et al. (2003) refined the losses of different habitat types within particular zones of the estuary using information from Thomas (1983), Graves et al. (1995), USACE (1996), and Garono et al. (2002). To facilitate this comparison, the Columbia River estuary was delineated into eight distinct areas based on physical characteristics. A brief description of each estuary area, a summary of habitat changes, and a qualitative description of changes in select habitat characteristics were extracted from Johnson et al. (2003) and the subbasin review developed by the Lower Columbia Fish Recovery Board as follows:

Entrance—dominated by subtidal habitat; highest salinity in estuary; historically a high-energy area of natural fluvial land forms, a complex of channels, shallow water, and sand bars; supports the Columbia Plume; abrupt changes have resulted from dredging and jetty construction that limit the ocean-fed supply of sediment; deep water habitat (18.9%) and a loss of medium-depth (41.1%) and tidal flat (43.6%) habitat types.

Mixing zone—characterized by a network of mid-channel shoals and flats; highest variation in salinity based on tide cycle and river flow; relatively little change in amount of the five major habitat types.

Youngs Bay—characterized by a broad flood plain and historically abundant in tidal marsh and swamp habitat; diking and flood control structures used to convert land to pasture resulted in 86.4% loss of tidal marsh and 95.7% loss of tidal swamp habitat. Much of the tidal marsh habitat lost throughout the estuary has occurred in this region.

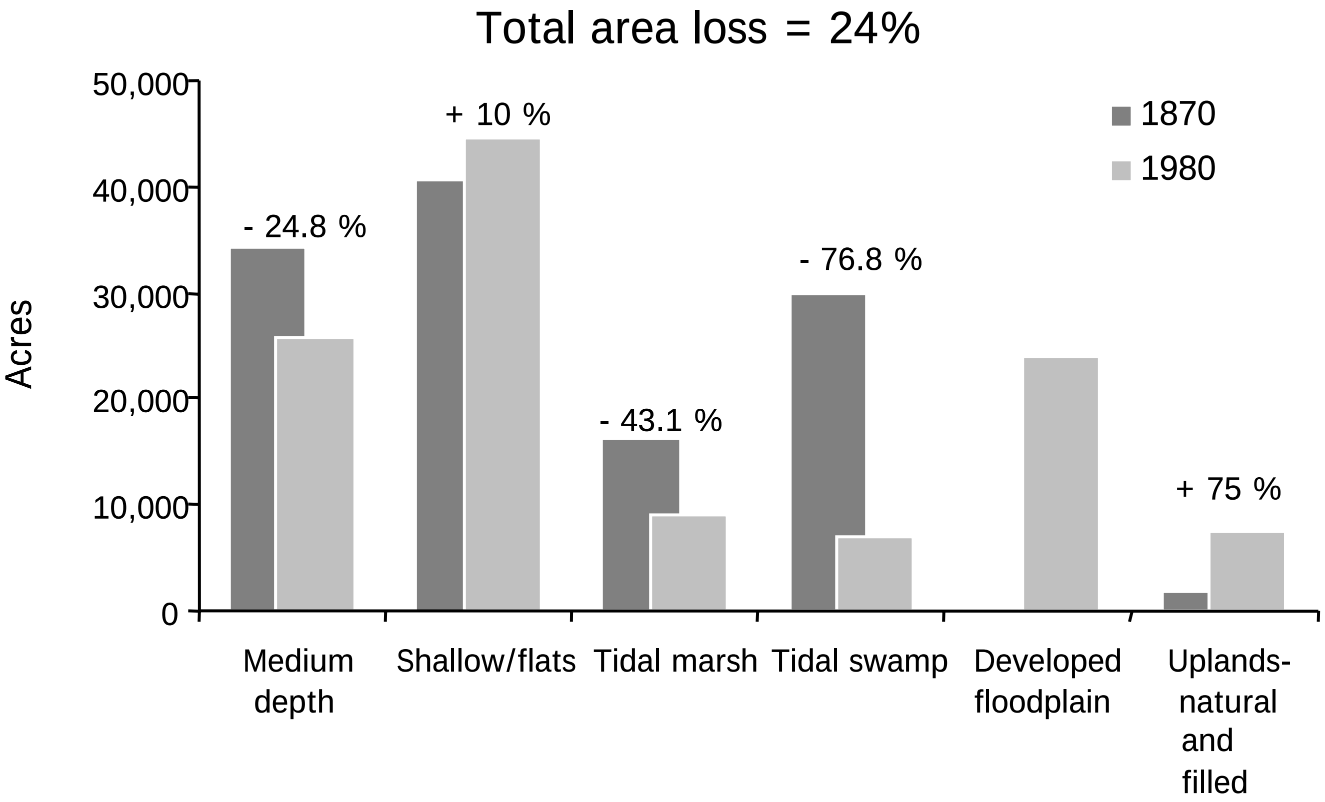

Figure 22. Change in acreage of various habitat types used by juvenile salmon in the Columbia River estuary (to approximately RM 42) from 1870 to 1980 (data source: Thomas 1983).

Baker Bay—historically a high energy area from ocean currents and wave action; migration of mid-channel islands toward the interior of Baker Bay has sheltered the area; some tidal marsh habitat recently started to develop because of decreased wave action (−75.0% deep water, −71.3% medium depth, +74.9% tidal flats, −55.5% tidal marsh, and −100% tidal swamp habitat).

Grays Bay—pile dikes adjacent to the main Columbia River navigation channel have decreased circulation in the bay and caused flooding problems in the valley bottoms; accretion in the bay has led to development of tidal marsh habitat, increasing 145.2% compared to historical conditions; dike construction for pasture conversion has isolated the main channel from its historical floodplain and decreased tidal swamp habitat 88.4% compared to historical levels.

Cathlamet Bay—large portion of area protected by the Lewis and Clark Refuge; other portions are heavily impacted by diking (Brownsmead area and Swenson Island) causing a 48.9% decline in tidal swamp; medium and deep water habitats have decreased (30.4% and 12.5%, respectively) as a result of dredge material disposal; fringe of dredge disposal areas has developed into tidal marsh habitat, resulting in 6.8% increase over historical conditions.

Upper estuary—characterized by deep channels and steep shorelines on both sides of river; typically dominated by freshwater, except during low river flow or large flood tides; extensive diking and clearing has resulted in substantial loss of tidal marsh (64.3%) and tidal swamp (79.9%) habitat compared to historical levels.

Tidal freshwater—although comprising nearly one third of the estuary (based on River Miles), little quantitative information on changes occurring in this region is available. This area is distinct in geology, vegetation, and climate; influenced by major tributaries; contains elongate islands that divide the river and form oxbow lakes, sloughs, and side channels; was historically dominated by a combination of tidal plant communities, ash riparian forests, and marshy lowlands. Changes in this reach of the estuary from RM 46 to 102 include increased upland habitat in the middle reach and substantial loss of nontidal water/wetland, tidal flats, and tidal marsh habitat types; from RM 105 to 146, there has been an increase in nontidal water/wetland and upland habitat and substantial loss of tidal flats and tidal marsh habitat types.

As reported in the Lower Columbia Mainstem and Estuary Subbasin Plan (LCMESP 2004), Thomas (1983) also investigated five categories of nonestuarine habitat (i.e. developed floodplain, natural and filled uplands, nontidal swamps, nontidal marshes, and nontidal water) to identify the fate of floodplain areas that were removed from the estuarine system. Developed floodplain habitat was defined as all diked floodplain converted to agriculture, residential, or other land use. Natural and filled uplands included areas where measurable acreages have been filled, primarily through disposal of dredge material. Nontidal swamps were areas of the diked floodplain that were never cleared or were cleared and converted back to swamp. Nontidal marshes included areas of the diked floodplain that support emergent wetland vegetation; these were typically abandoned pastures dominated by rush and sedge. Nontidal water consisted of former tidal sloughs that were separated from the river by dikes and tidegates. The largest increase, by far, of nonestuarine habitat from 1870 to 1983 was that of developed floodplain habitat. Of the 36,970 total acres of lost estuarine habitat, 64.8% was converted to developed floodplain (Thomas 1983).

As discussed earlier, absolute changes in habitat opportunity alone should not be used to directly infer changes in the ability of the estuary to support salmon. For instance, despite considerable loss of emergent and forested wetlands in the estuary and associated declines in macrodetrital production, the total area of estuarine shallows and flats actually increased 7% between 1870 and 1980. This was independently substantiated by Sherwood et al. (1990), who estimated 68.4×10^6 m^3 net sediment gain within the estuary between 1868 and 1958. Areas of sediment increase include peripheral bays such as Cathlamet Bay and Grays Bay, which had shoaling rates of 0.61 cm yr^{-1} and 0.63 cm $yr^{-1}$and net volumetric increases of 76.2×10^6 m^3 and 19.1×10^6 m^3, respectively.

Loss of estuarine wetlands has not only reduced the total amount of shallow rearing habitat available to young salmon but also altered the magnitude and character of habitat capacity (Sherwood et al. 1990). The resulting decline in wetland primary production eliminated approximately 15,800 mt carbon $year^{-1}$ (84%) of macrodetritus that historically supported estuarine food webs. This macrodetritus originated from the vascular plants and microscopic algae produced within the estuary's wetlands. However, these losses were accompanied by an increase of approximately 31,000 t carbon $year^{-1}$ of microdetritus from upriver sources, originating principally from increased phytoplankton production in the reservoirs behind the mainstem dams (Sherwood et al. 1990). The shifts in the sources and types of detritus available may have altered estuarine food webs, including those leading to salmon. For example, the epibenthic-pelagic food web supported by microdetrital sources favors production of calanoid copepods and other pelagic organisms that typically are not consumed by juvenile salmon (Bottom and Jones 1990, Sherwood, et al. 1990, Simenstad et al. 1990).

As a result of the loss of habitat, altering the spatial distribution of the food web may also be an important determinant of habitat capacity in the estuary. Whereas the macrodetrital food web was historically distributed throughout the lower river and estuary, the contemporary microdetrital food web is most concentrated within the localized mid-estuary region of the ETM.

We have no objective means of quantifying these types of ecological effects of the habitat shift from emergent and forested wetlands to shallows and flats. For example, no historical data are available for salmonid diet composition or stomach fullness within tidal wetlands to compare with other estuarine habitats. Although juvenile salmon may not directly benefit from the microdetrital food web, there is some evidence that they have higher stomach fullness in the mid estuary compared with other estuarine regions (Bottom and Jones 1990). One possible mechanism that has yet to be verified for any increase in feeding rates is that enhanced detrital concentrations within the ETM may also stimulate secondary production in adjacent mid-estuary shallows and flats. However, we do know that prey production and salmon stomach fullness values are relatively high in protected flats compared with many other estuarine habitats. Jones et al. (1990) found that the standing crop of infauna in protected flats of the estuarine mixing region (≈RM 7 to RM 21) was more than an order of magnitude higher (2.058 g m^{-2} AFDW [ash free dry weight]) than benthic fauna standing crop in any of the other channel or unprotected flat habitats (0.098–0.136 g m^{-2} AFDW) within the same estuarine region.

Another recent analysis shows the loss of habitat in the tidal freshwater region of the estuary. This is where we most lack the empirical evidence of change and contribution to expression of spatial structure and salmon life history diversity. Kukulka and Jay (2003) indicated that diking removed nearly 52% of the shallow water flood plain habitat in the tidally influenced freshwater zone of the estuary (Figure 23).

Removing dikes alone would restore accessibility of considerable amounts of shallow water estuary habitats such as tidal floodplains since diking entirely removes habitat from the estuarine system. Other anthropogenic factors change estuary habitats from one type to another (Thomas 1983). Other benefits of removing dikes would include increasing the amount of prey resources available to the juvenile salmon and increasing organic matter inputs. The degree to which estuary habitat types have been affected by diking is directly proportional to elevation; thus the highest elevation habitat types (i.e., tidal swamp) have been most impacted by diking (Thomas 1983).

Mainstem estuarine habitats in the Columbia River have for the most part been reduced to a single channel where floodplains have been reduced in size and off-channel habitat has been lost or disconnected from the main channel. Dikes prevent overbank flow and affect the connectivity of the river and floodplain (Tetra Tech Inc. 1996); thus the diked floodplain is higher than the historical floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). There is a critical level (i.e., the elevation of the diked floodplain) that water level must reach before substantial floodplain habitats are inundated (Kukulka and Jay 2003). Above this critical water level, large amounts of shallow water floodplain habitats become available with small increases in water level up to an optimum threshold (Kukulka and Jay 2003).

Under a modern bathymetry and flow regime scenario, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high, and the frequency of occurrence of this river discharge is rare; thus floodplain inundation is uncommon and availability of shallow water habitats is limited under current conditions (Kukulka and Jay 2003). As is the case in the estuary (Bottom et al. 2005), loss of these vital mainstem floodplain habitats has likely reduced the availability and diversity of habitats in the Columbia River estuary for juvenile salmonids, particularly those juveniles employing life history strategies associated with these peripheral, shallow water habitats.

Many of the changes in the biological production processes of the estuary described above can be attributed directly to physical causes. For example, the apparent shift from macrodetrital to microdetrital food chains in the estuary stems from the diking and filling of intertidal wetlands and the creation of deep reservoirs behind mainstem dams. While changes in the quality and quantity of prey resources could well be a proximal factor affecting the use of the estuary, the ultimate cause is the physical removal of those habitats that supported both macrodetrital production and the diversity of estuarine life histories among salmon.

This is not to say that all biological effects are physically driven. Nonetheless, we find that many of the significant biological changes now observed in the Columbia River estuary are best explained by physical modifications that have altered the habitat landscape to the benefit and detriment of different species and assemblages. These findings have important implications for developing restoration strategies that address the ultimate causes rather than the symptoms of salmon decline.

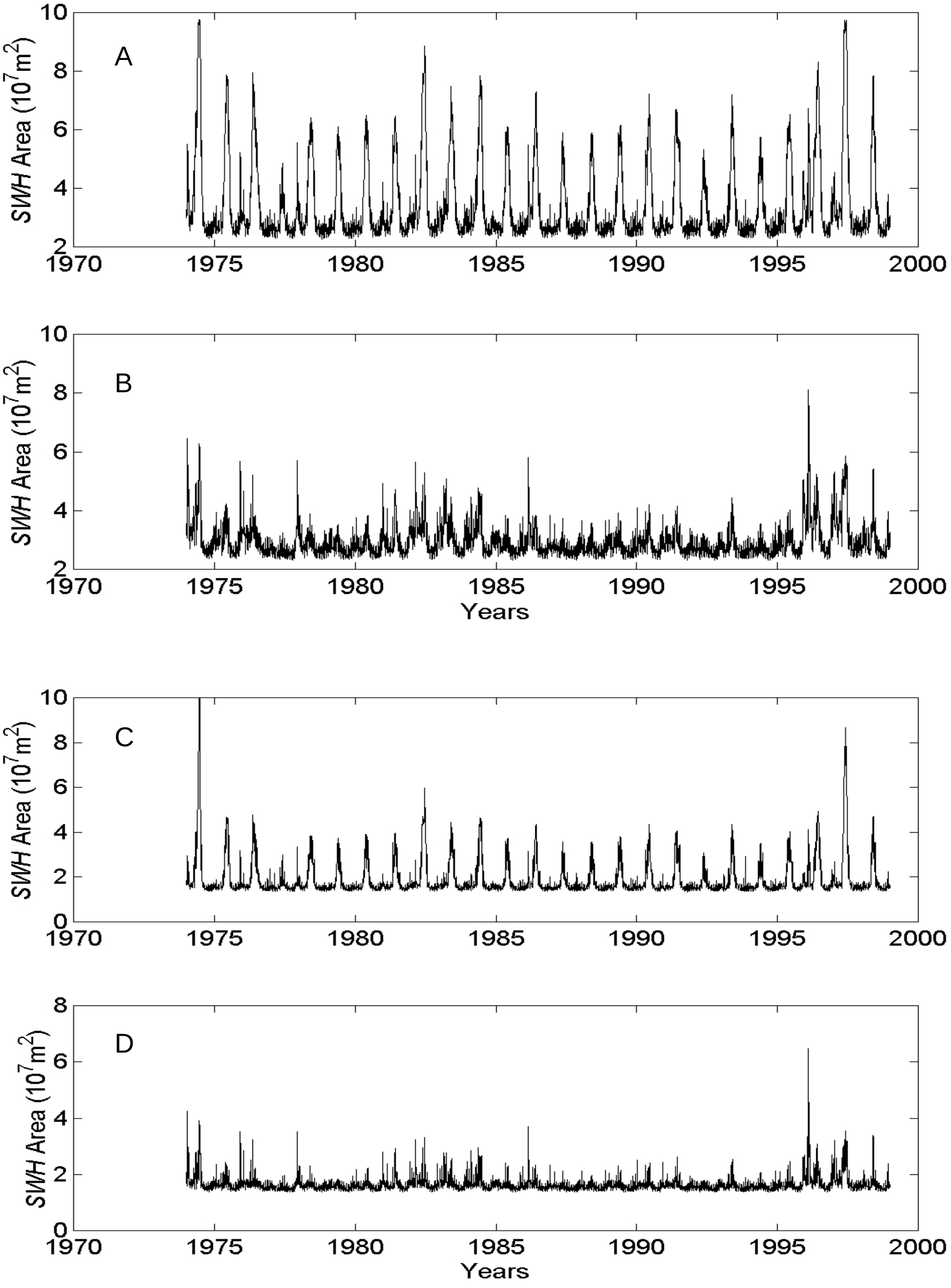

Figure 23. Daily shallow water habitat (SWH) area from 1974 to 1998 for virgin (A) and observed (B) river flows without dikes, and for virgin (C) and observed (D) flows with dikes. (From Kukulka and Jay 2003.)

Habitat Summary

In summary, the location and types of habitats present in the Columbia River estuary have been substantially changed from historical conditions. Although the entire estuary has not yet been surveyed, the main changes that have been quantified in the estuary are a loss of emergent marsh, tidal swamp, and forested wetlands. Shallow water dependent life history strategies (fry and fingerlings) have been most affected by the loss of these vegetated habitat types. Alterations in attributes of flow and diking have caused these changes. Diking is a significant change primarily because it completely isolates habitat from the river and eliminates it from use by juvenile salmon. Further, it has potentially altered estuarine food webs from macrodetrital to microdetrital based. Clearly, restoration of shallow water vegetated habitat by removing dikes is a tactic that can benefit those life history strategies that depend on these shallow water habitats. In the case of populations that have large numbers of shallow water dependent members, restoring these habitats can significantly affect abundance and productivity.

Toxics

In addition to eliminating and physically altering salmon habitat in the estuary, the quality of habitats has been degraded through the release of toxic contaminants. With the exception of some metals and natural products, concentrations of toxic contaminants in the Columbia River estuary were historically low. However, beginning in the early 1800s, activities such as agriculture, logging, mining, industrial discharges, and storm water runoff began to degrade water quality in the estuary. Currently, the Columbia River below Bonneville Dam is the most urbanized section of the river, encompassing the major urban centers of Portland, Oregon, and Vancouver, Washington, and numerous minor cities such as Longview, Washington, and Astoria, Oregon. The estuary receives contaminants from over 100 point sources (Fuhrer et al. 1996), as well as numerous nonpoint sources such as surface and storm water runoff from urban and agricultural areas.

The largest sources of effluent are the Portland and Vancouver sewage treatment plants and associated combined sewer overflows in the Willamette River and the Columbia Slough (LCREP 1999). Contaminants may also be transported downstream to the estuary from areas above the Bonneville Dam such as the Yakima River (Rinella et al. 2000, Fuhrer et al. 1996), Lake Roosevelt (Bortleson et al. 1994), and other tributaries (Fuhrer 1989, Roy F. Weston Inc. 1998). Spills or other accidental releases of toxic substances at Bonneville Dam itself may also contribute to contamination in the estuary (WDOE 2004), although inputs are probably relatively small compared to those from the large urban centers.

A number of potentially toxic water-soluble contaminants have been detected in the estuary. The USGS NASQAN (National Stream Quality Accounting Network) program has reported a wide range of current-use pesticides in the water column at sampling sites at Warrendale at RM 141 near the Bonneville Dam; the confluence of the Willamette and Columbia rivers near Portland at RM 101.5; and the Beaver Army Terminal at RM 53.8 (Fuhrer et al. 1996, Hooper et al. 1997). These water-soluble contaminants include simazine, atrazine, chlorpyrifos, metolachlor, diazinon, and carbaryl. Water concentrations and frequency of detection were highest at the Willamette/Columbia confluence, with detections in 80–100% of samples at concentrations up to 300 ng/L; these compounds were also frequently reported at the Beaver Army Terminal.

Various trace metals have also been monitored as part of this program, revealing high concentrations of iron and manganese, especially near the Willamette/Columbia confluence, and high levels of arsenic in the Columbia River estuary (Fuhrer et al. 1996). These compounds come partly from natural sources, as well as anthropogenic activities, such as the use of lead arsenate as an insecticide for apples. Concentrations of other trace metals were similar to background concentrations in other North American streams (Fuhrer et al. 1996).

Contaminants that have been documented in estuary sediments include trace metals (cadmium, copper, and zinc), dioxins, furans, chlorinated pesticides and other chlorinated compounds (e.g., dieldrin, lindane, chlordane, PCBs, and DDT and its metabolites), polycylic aromatic hydrocarbons (PAHs), and other semivolatile compounds (Fuhrer and Rinella 1983, Fuhrer 1986, Harrison et al. 1995, Tetra Tech Inc. 1996, USACE 1998, Roy F. Weston Inc. 1998). Many of these compounds, particularly lipophilic compounds such as PAHs and organochlorine compounds (OCs), are rarely detected as dissolved material in the water, but rather tend to bind to organic carbon or particulate materials; thus they commonly occur in association with fine-grained materials in the streambed or in suspension (Horowitz 1991, Tetra Tech Inc. 1993).

The fine-grained sediments to which these toxicants adsorb will most likely be deposited in areas with slower water velocities, including backwater areas in side channels and along the river's margins. Elevated concentrations of toxic contaminants are more likely to occur in these areas (Tetra Tech Inc. 1994). In the main navigation channel of the Columbia River, current velocities are generally high, so there is little deposition of fine-grained sediments. Coarse, sandy sediment with relatively low contaminant concentrations typically make up 99% of the bulk bed material in the navigation channel (USACE 1998, 1999, McCabe et al. 1997).

Some contaminants, including PAHs, have been detected at levels that exceed state or federal sediment quality guidelines, or are considered harmful to humans and aquatic life (Tetra Tech Inc. 1996). These contaminants have been detected in sediments from the lower Willamette River in the Portland area (concentrations up to 900 mg/kg wet wt (ODEQ 1994, Harrison et al. 1995, Roy F. Weston Inc. 1998) and in some sediments from other urban and industrial areas in the estuary (Tetra Tech Inc. 1996). Recent sediment data, collected in 2000 by EPA as part of EMAP (the Environmental Monitoring and Assessment Program), identified a few hot spots for the PCBs and DDTs within the estuary, including sites near Longview (total PCBs 860 ng/g dry wt), West Sand Island (total PCBs 965 ng/g dry wt), the Astoria Bridge (DDTs 597 ng/g dry wt), and Vancouver (DDTs 128 ng/g wet wt).

It is noteworthy that PCB and DDT concentrations in the majority of sediments tested in these studies were much lower (below 50 and 5 ng/g dry wt, respectively). To put these values in perspective, the sediment screening guidelines for the protection of marine life are typically in the 20–200 ng/g dry wt range for PCBs (MacDonald 1994, USACE 1998, WAC 1995, Meador et al. 2002, CCME 2002), and in the 3–7 ng/g dry wt range for DDTs (MacDonald 1994, USACE 1998, WAC 1995, Meador et al. 2002, CCME 2002), depending on the organic carbon content of the sediment.

Suspended material may also be an important source of contaminants in the Columbia River estuary (LCREP 1999). This material is predominantly fine-grained, and contains many of the toxic compounds that have been detected in streambed sediments. McCarthy and Gale (2001) analyzed water samples collected with semipermeable membrane devices from nine mainstem and six tributary sites throughout the Columbia River basin (Washington and Oregon) and found dioxins, dibenzofurans, PCBs, organochlorine pesticides, and PAHs throughout the basin, with the highest concentrations of many compounds in the Portland-Vancouver area. Metals including arsenic, lead, chromium, copper, iron, manganese, mercury, and zinc have also been detected in suspended sediments in the estuary (Fuhrer et al. 1996).

Mobilization and transport of suspended sediments during extreme stream flow events can make their adsorbed contaminants available to salmon and other aquatic organisms. For example, during the flood of February 1996, several legacy organic pesticides that are typically associated with the sediment phase, including dieldrin and DDE, were mobilized in the lower Columbia and Willamette rivers and detected in the water column at some sites for the first time (Kelly 1997). During this event, the estimated whole water concentration of p,p'-DDE exceeded the chronic ambient water-quality criterion for the protection of aquatic organisms by at least fivefold. Suspended particulates and associated contaminants may also occur in areas of high turbidity, such as the estuarine turbidity maximum, which may be an important feeding area for salmon (Bottom and Jones 1990). However, currently, the relative contributions of contaminants in water column versus those in bed sediment or body burdens in resident biota are poorly understood (LCREP 1999, SEI 2001).

Exposure to contaminants, and hence the potential effects of these compounds in the estuary, likely varies with life history type or ESU. Stream-type ESUs (e.g., Snake River sockeye, upper Willamette River steelhead, and Snake River spring Chinook) are less likely to accumulate high body burdens of bioaccumulative, sediment-associated contaminants such as PCBs and DDTs because most members of these populations migrate rapidly through the estuary as yearlings or older. However, they may be affected by short-term exposure to waterborne contaminants such as organophospate pesticides (OPs) and dissolved metals.

Ocean-type fish (e.g., lower Columbia River chum salmon and upper Willamette River Chinook salmon), which enter the estuary as fry, fingerlings, or subyearlings, and may rear for an extended period in the estuary, are also at risk for exposure to current-use pesticides and dissolved metals. At the same time, they are more likely than stream-type fish to be affected by bioaccumulative toxicants (DDTs, PCBs) that they may absorb through their diet during estuarine residence. Ocean-type populations may also be more at risk because of their greater use of shallow-water habitats with slower water velocities, including backwater areas in side channels and along the river's margin, where fine-grained sediments to which toxics adsorb are most likely to accumulate. This may be particularly true for naturally produced wild salmon that, as noted earlier in this document, are more commonly found in side-channel, peripheral tidal marsh, and forested marsh habitats than the larger, hatchery size salmon that mostly use deeper mainstem habitat.

Many of the contaminants found in streambed and suspended sediments in the estuary are accumulated by resident biota. A number of studies have identified trace metals, dioxins and furans, chlorinated compounds such as PCBs and DDTs, and PAH metabolites in nonsalmonid fish from the estuary (Tetra Tech Inc. 1993, 1996, Brown et al. 1998, Foster et al. 2001a, 2001b), in some cases at concentrations exceeding health guidelines (LCREP 1999). Although data on contaminant concentrations in listed salmon are more limited, they indicate that chemical contaminants are present in prey and tissues of juvenile salmon from the Columbia River estuary (Figures 24–27).

In studies conducted by the NWFSC in collaboration with the U.S. Army Corps of Engineers (Johnson et al. 2004), contaminant concentrations were measured in juvenile fall Chinook salmon from several sites in the estuary (near the confluence of the Columbia and Willamette rivers, near Longview, White Island, West Sand Island, between East and West Sand islands, Chinook Point, east Trestle Bay, west Trestle Bay, lower Desdemona Sands, and middle Desdemona Sands, Figures 24–27). Fish from the Willamette/Columbia River confluence, Longview, and West Sand Island were collected in shallow water habitats by beach seine, while fish from the other sites within the lower estuary were collected in deeper water by purse seine.

The primary contaminants found in whole body samples of both purse seine and beach seine fish from all sites were PCBs and DDTs. Chlordanes, lindane, hexachlorobenzene, dieldrin, and mirex were also detected in fish from the confluence and Longview/Kalama. Average concentrations of PCBs at estuarine sampling sites ranged from 23 to 90 ng/g wet wt, while average DDT concentrations ranged from 32 to 115 ng/g wet wt. In individual fish, DDT levels as high as 270 ng/g wet wt and PCB levels as high as 340 ng/g wet wt were measured. These concentrations were among the highest levels measured by the NWFSC at estuarine sites in Washington and Oregon (Stein et al. 1995, Collier et al. 1998a, Stehr et al. 2000, Arkoosh unpubl. data).

Measurable concentrations of DDTs and PCBs, were also detected in stomach contents of juvenile fall Chinook salmon from West Sand Island, White Island, Longview, and the Willamette/Columbia confluence (at other sites, contaminants in stomach contents were not measured), indicating that fish were absorbing some of these compounds from prey during estuarine residence. Contaminant concentrations in fish from the Willamette/Columbia confluence are shown in Figure 23. Concentrations of PCBs in stomach contents of fish from this area were comparable to those in fish from the heavily industrialized Hylebos Waterway in Puget Sound (Stehr et al. 2000), while DDT concentrations were higher than at any other sampled sites in the Pacific Northwest. Several additional chlorinated pesticides, including lindane, hexachlorobenzene, dieldrin, and certain DDT isomers (o,p-DDD, o,p-DDT, p,p-DDT) were detected only in stomach contents of salmon from the confluence or Longview.

Although the fish used for these analyses were almost entirely unmarked, genetic analyses showed that a high proportion of beach seine collected fish from West Sand Island, Longview, and the confluence were most likely of hatchery origin (40–70%) (Johnson et al. 2004). Consequently, DDTs and PCBs in hatchery feed may have contributed to contaminant body burdens in these fish, since these substances, especially PCBs, have been detected in hatchery feed and in juvenile Chinook salmon collected from Pacific Northwest hatcheries (Johnson et al. 2004, Ylitalo unpubl. data; Figures 24–26). However, no data are available on contaminant concentrations in feed from the specific hatcheries where these fish originated (predominantly hatcheries on the Elokoman, Cowlitz, and Sandy rivers), so this cannot be confirmed. In spite of the potential for hatchery contribution, it is clear from the elevated levels

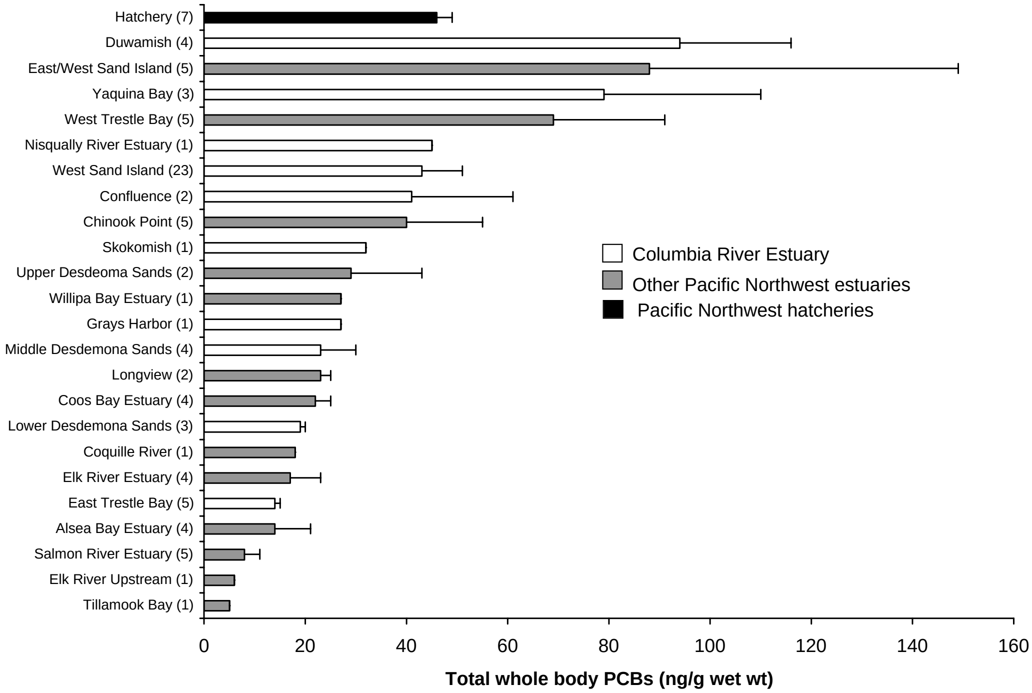

Figure 24. Mean concentrations (±SE) of total PCBs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries.

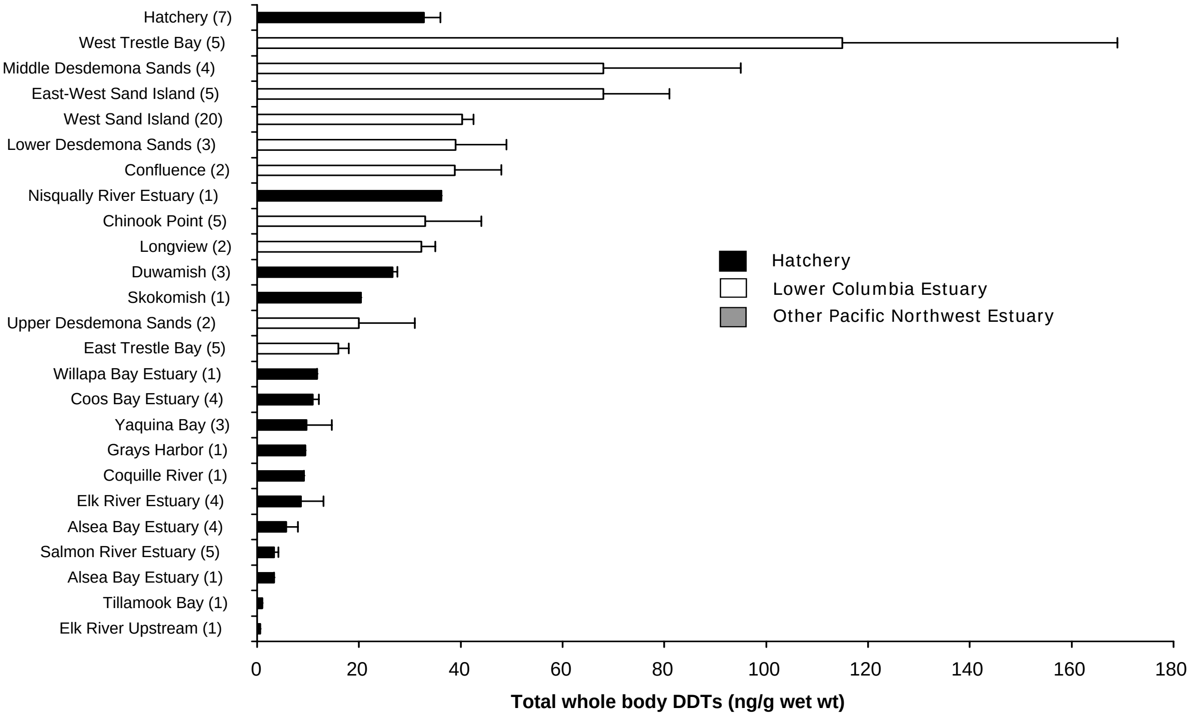

Figure 25. Mean concentrations (±SE) of DDTs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries.

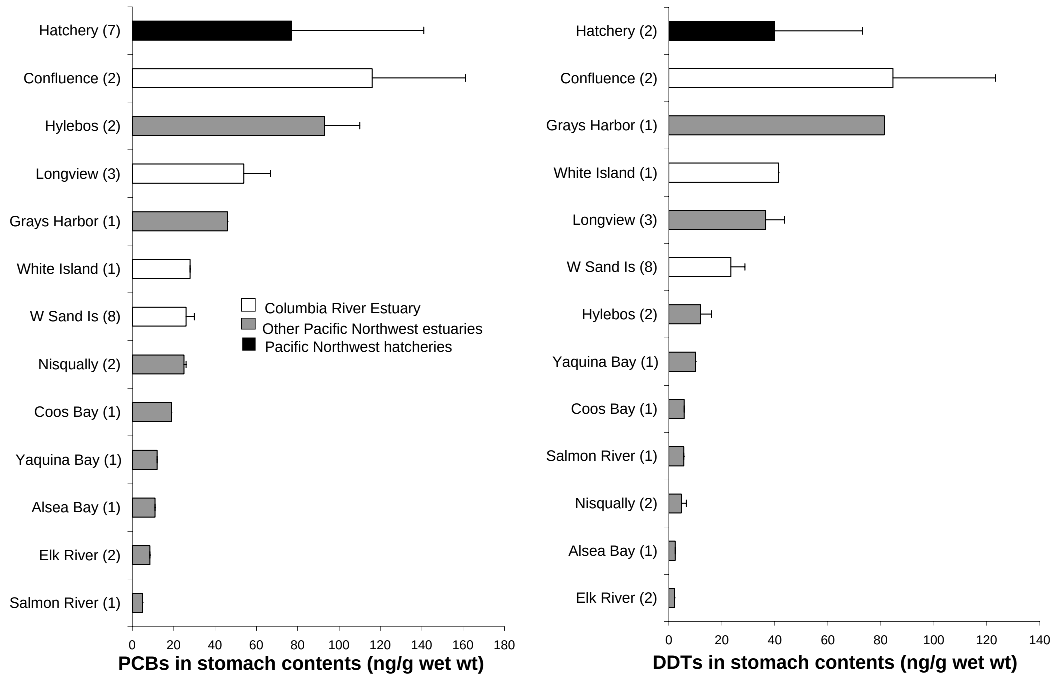

Figure 26. Mean concentrations (±SE) of PCBs and DDTs (ng/g wet wt) in stomach contents of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries.

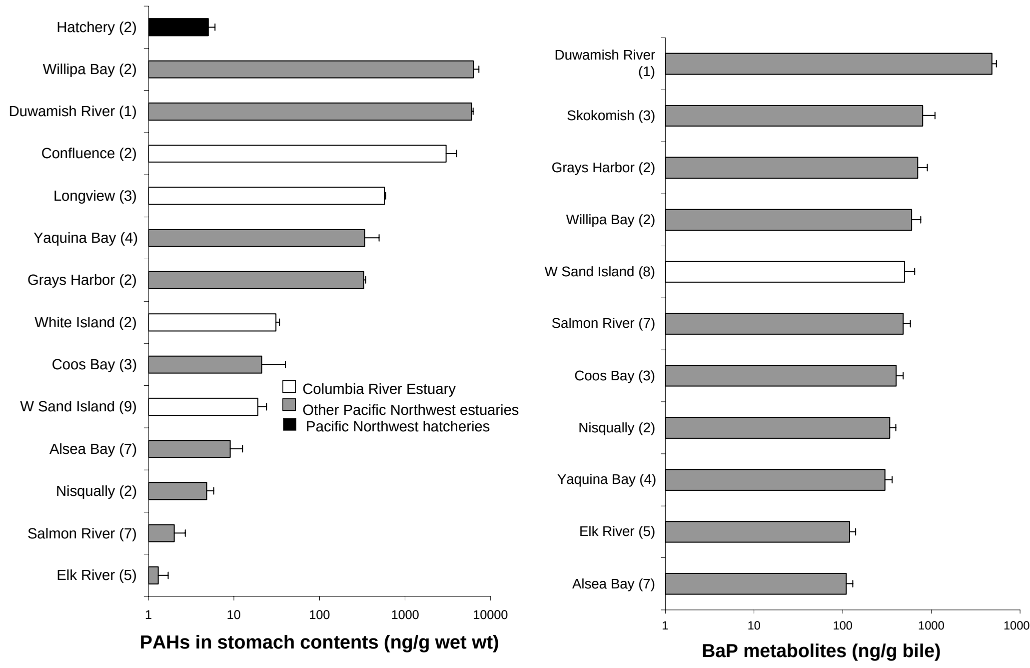

Figure 27. Mean concentrations (±SE) of high molecular weight polycyclic hydrocarbons (PAHs) in stomach contents, and metabolites of PAHs in bile, of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries.

of DDTs and PCBs in stomach contents of fish from sites within the estuary that fish are also being exposed to these contaminants through their natural prey. Additionally, DDT/PCB ratios were several times higher in salmon from Columbia River sites (average 1–4) than in hatchery fish (average 0.7), suggesting uptake of DDTs from the environment.

Less information is available on exposure to PAHs in juvenile salmon from the Columbia River estuary. Data collected by NOAA Fisheries Service between 1998 and 2002 showed that concentrations of PAHs in stomach contents and PAH metabolites in bile were low to moderate in juvenile fall Chinook salmon from West Sand Island in comparison to levels found in fish from other estuaries along the Washington and Oregon Coast (Figure 26). No data were available on metabolites of PAHs in bile of fish from other sites in the estuary. PAHs were measured in stomach contents of juvenile Chinook salmon from several lower Columbia sites (the Willamette/Columbia confluence, West Sand Island, White Island, and Longview). While levels in fish from West Sand Island and White Island were moderate, concentrations in fish from Longview and the confluence were higher than or comparable to concentrations in juvenile salmon from the Hylebos Waterway in Puget Sound (Figure 27).

Data on contaminant concentrations in salmon prey and other benthic invertebrates are limited. Little has been published for contaminants in aquatic macroinvertebrates, although some work has been done on clams, crayfish, and *Corophium*, a benthic amphipod that is important in the diets of several fish species, including salmonids (McCabe et al. 1983, 1986, 1997, Muir and Emmett 1988, LCREP 1999). A small-scale reconnaissance study conducted by NWFSC's Newport Research Station in the mid-1990s measured PAH concentrations in *Corophium salmonis*[7] from several sites in the lower Columbia River, including Longview Bridge and the Multnomah Channel in the Columbia River, and sites at the Willamette River mouth, North Portland Harbor, and Hayden Island in the Willamette River.

At the three sites where sediment PAH concentrations were highest (Hayden Island, Longview Bridge, and Multnomah Channel), no amphipods were present in sediments, suggesting *C. salmonis* has specific habitat requirements, including sediment particle size preferences and possibly concentrations of contaminants, that may prevent it from occupying these sites. PAH concentrations in tissues of *Corophium* were up to 50 ng/g wet wt benzo[a]pyrene (BaP) equivalents or about 150 ng/g wet wt high molecular weight PAs. Average concentrations of high molecular weight PAs in stomach contents from juvenile salmon for the Columbia River estuary ranged from approximately 3,000 ng/g wet wt and approximately 600 ng/g wet wt the Confluence and Longview, respectively, to 20–30 ng/g wet wt at West Sand Island and White Island (Johnson et al. 2004).

In general, these studies show that PCB and PAH concentrations in salmon or their prey from the Columbia River estuary are comparable to those reported in juvenile salmon from other moderately to heavily urbanized sites, while DDT levels are high relative to levels in other Pacific Northwest estuaries. Although concentrations of contaminants were higher in stomach contents of salmon juveniles collected from near the Willamette/Columbia confluence, body

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[7] Data are from preliminary studies conducted by Bruce McCain, NWFSC, Newport, Oregon. Concentrations of PAHs in amphipods were measured using a semiquantitative method that employs high performance liquid chromatography (HPLC) and a photodiode array (PDA) detector (Krahn et al. 1993).

burdens of bioaccumulative OCs were similar throughout the estuary. Sources and pathways of exposure are unclear, and could include contaminated bed sediments, contaminated prey, contaminants in suspended material, and contaminants in hatchery feed for those fish that are of hatchery origin.

Studies suggest that, at least for some contaminants, exposure levels in juvenile salmon from the Columbia River estuary are approaching concentrations that could affect their health and survival. For PCBs, Meador et al. (2002) estimated a critical body residue of 2,400 ng/g lipid for protection against 95% of effects ranging from enzyme induction to mortality. They based this on a range of sublethal effects observed in salmonids in peer-reviewed studies conducted by NMFS and other researchers. On a wet weight basis, the threshold would be 24–48 ng/g for fish with lipid content of 1–2%, typical of juvenile salmon from the Columbia River. Mean PCB body burdens in juvenile salmon analyzed by the NWFSC (Johnson et al. 2004) were at or above the 2,400 ng/g threshold at 7 of 10 sites sampled in the lower Columbia (Figure 28). Of individual fish analyzed from sites within the estuary, approximately 60% had PCB body burdens at or above this threshold.

The likely impact of DDTs, another major contaminant of concern in the lower Columbia, on salmon is less clear. Most reported effects of DDTs on salmonids are associated with whole body tissue concentrations above those typically found in salmon obtained from the Columbia River estuary (≥500 ng/g wet wt) (Allison et al. 1963, Burdick et al. 1964, Johnson

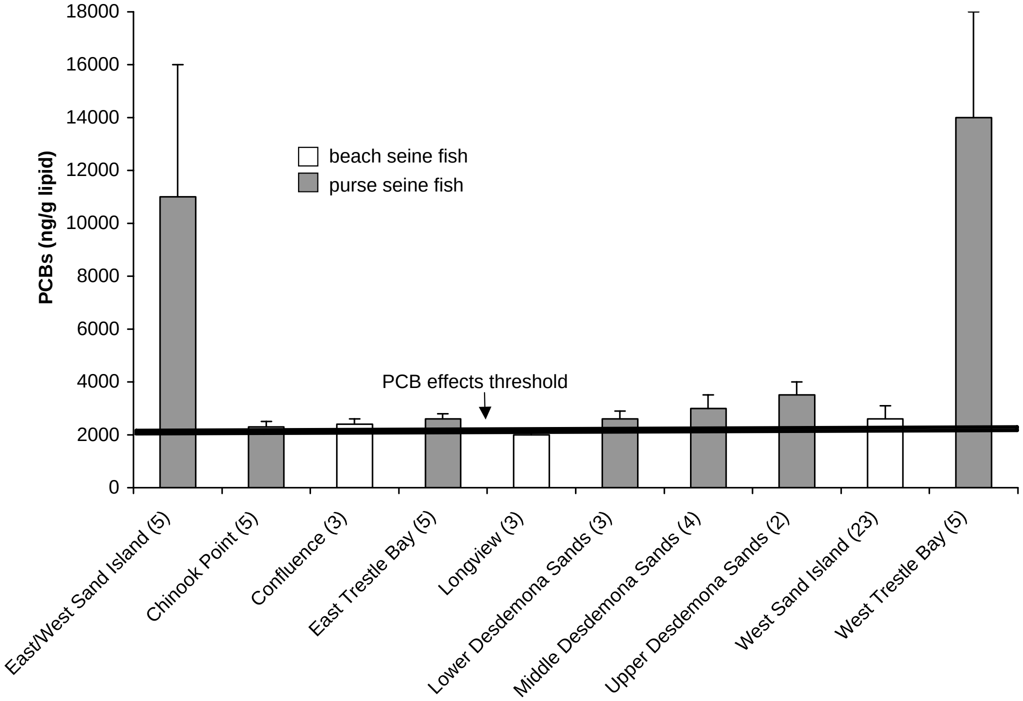

Figure 28. Mean concentrations (±SE) of PCBs (ng/g lipid) in juvenile fall Chinook salmon sampled from the Columbia River estuary as compared to the 2,400 ng/g lipid biological effects threshold of Meador et al. (2002).

and Pecor 1969, Buhler et al. 1969, Peterson 1973, Poels et al. 1980, Hose et al. 1989). More recent studies suggest that certain DDT isomers (e.g., o,p-DDT) may have endocrine-disrupting or immunotoxic effects (Donohoe and Curtis 1996, Celius and Walther 1998, Khan and Thomas 1998, Arukwe et al. 1998, 2000, Christiansen et al. 2000, Zaroogian et al. 2001, Milston et al. 2003, Papoulis et al. 2003).

These effects were typically observed at body burdens or dietary exposure concentrations in the 10–20 ng/g wet range or above, while concentrations of o,p-DDTs in the bodies and stomach contents of juvenile Chinook salmon from the lower Columbia were near or below detection limits (<2 ng/g wet wt) (Johnson et al. 2004). This suggests that o,p-DDT, DDE, and DDD levels are below concentrations likely to be associated with estrogenic activity and related effects. However, these effect thresholds were obtained with laboratory fish, whose lipid content is typically higher than in wild fish (8–10% vs. 1–3%; Meador et al. 2002). This is important, because for a given wet or dry weight tissue concentration, the higher the lipid content, the higher the resistance to the toxicant because a higher proportion of the hydrophobic compound is associated with the lipid and is not available to cause toxicity (Meador et al. 2002). If we assume that test fish had a lipid content of 10%, then the DDT body burden on a lipid weight basis for toxic effects would be 5,000 ng/g lipid. Lipid-adjusted DDT concentrations in or above this range were observed in juvenile Chinook salmon from some sites in the lower Columbia River estuary (e.g., between East and West Sand islands, middle Desdemona Sands, west Trestle Bay). Moreover, some of the DDT isomers could work additively in conjunction with other estrogenic contaminants (e.g., plasticizers, pharmaceuticals, and surfactants) to alter reproductive processes or other physiological functions, if these are also present in the estuary.

DDTs also have the potential to affect salmonid prey, as invertebrate species are generally quite susceptible to impacts of these compounds. Results of laboratory and field investigations, as well as equilibrium partitioning calculations, suggest that thresholds for chronic effects occur at total DDT concentrations in sediments of approximately 10 ng/g dry wt (Pavlou et al. 1987, Long et al. 1995). These DDT concentrations are not uncommon in estuary sediments (Tetra Tech Inc. 1994, Johnson et al. 2004). Moreover, studies in the Columbia River estuary have shown that DDTs represent a hazard to fish-eating predators through bioaccumulation and bioconcentration (Anthony et al. 1993, USFWS 1999, Thomas and Anthony 1999, 2003, Henny et al. 2003).

Recently, Nendza et al. (1997) estimated a no-observable-effect concentration of 22–50 ng/g wet wt for food-chain related impacts for DDTs, based on studies with a number of fish from marine estuaries. Many juvenile salmon sampled from the Columbia River estuary have DDT body burdens at or above this level.

The potential for contaminant-related injury to juvenile salmonids in the estuary is also supported by field studies at Puget Sound sites contaminated with PAHs, PCBs, and other OCs similar to those present in the lower Columbia. In these studies, juvenile salmon from two urbanized waterways, the Hylebos and the Duwamish, showed demonstrable effects, including immunosuppression, reduced disease resistance, and reduced growth rates, due to contaminant exposure during their estuarine residence (Arkoosh et al. 1991, 1994, 1998, Varanasi et al. 1993, Casillas et al. 1995, 1998a). For example, juvenile Chinook salmon from the Duwamish Waterway were not able to produce the normally enhanced secondary immune response

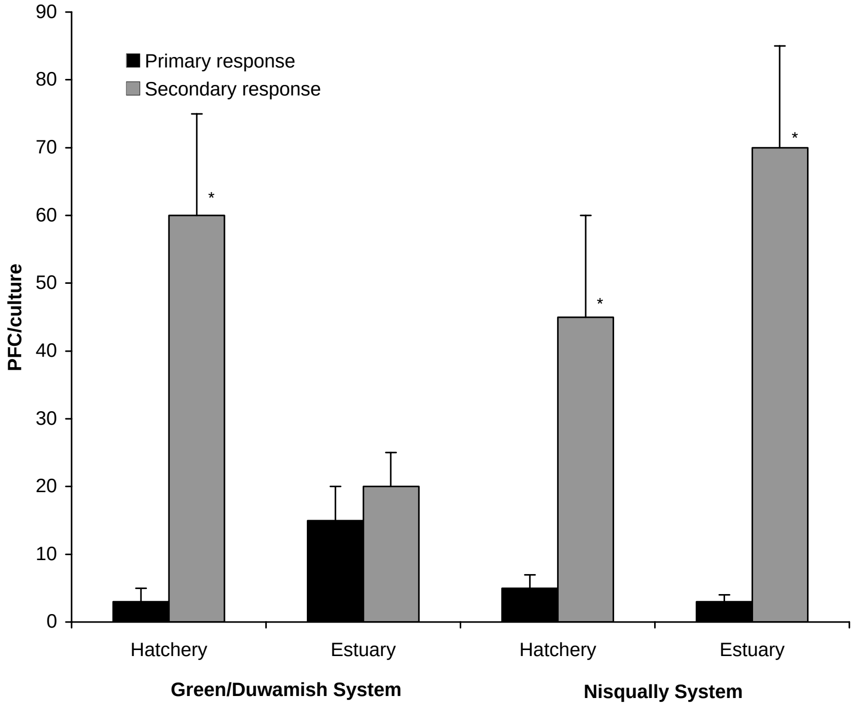

Figure 29. The leucocyte primary and secondary in vitro plaque forming cell response per culture (PFC/culture) against an antigen. The mean (+SD) PFC response was analyzed in Chinook salmon from the Green/Duwamish system (urban) and the Nisqually system (nonurban). The asterisk (*) indicates the secondary PFC/culture is significantly higher ($p < 0.05$) than that observed in the primary response. (Adapted from Arkoosh et al. 1991.)

observed in nonexposed control fish from the Nisqually River estuary and fish from the natal hatcheries (Figure 29) (Arkoosh et al. 1991).

Salmon exposed in the laboratory to PCBs and PAHs were also immunosuppressed (Arkoosh et al. 1994). Additionally, in disease challenge studies with *Vibrio anguillarum*, a marine bacterial pathogen that infects juvenile Chinook salmon from estuaries along the Washington and Oregon coast (Arkoosh et al. 2004), juvenile Chinook salmon from the Duwamish Waterway were more susceptible to disease and exhibited higher cumulative mortality than fish from natal hatcheries on a noncontaminated estuary (Arkoosh et. al. 1998, Figure 30). Similar effects were found in juvenile Chinook salmon injected in the laboratory with extracts of sediments from the Hylebos Waterway in Commencement Bay (Arkoosh et al. 2001).

In related studies designed to assess the effects of contaminants on growth (Casillas et al. 1995), juvenile fall Chinook salmon collected from the Duwamish Waterway and held in the laboratory for up to 90 days did not grow as well as similarly treated fish from the natal hatchery on the Green River. In contrast, juvenile salmon from the nonurban estuaries showed no difference in growth compared to fish from the natal hatcheries. Concentrations of insulin-like growth factor, a plasma hormone involved in the regulation of growth, were also lower in fish

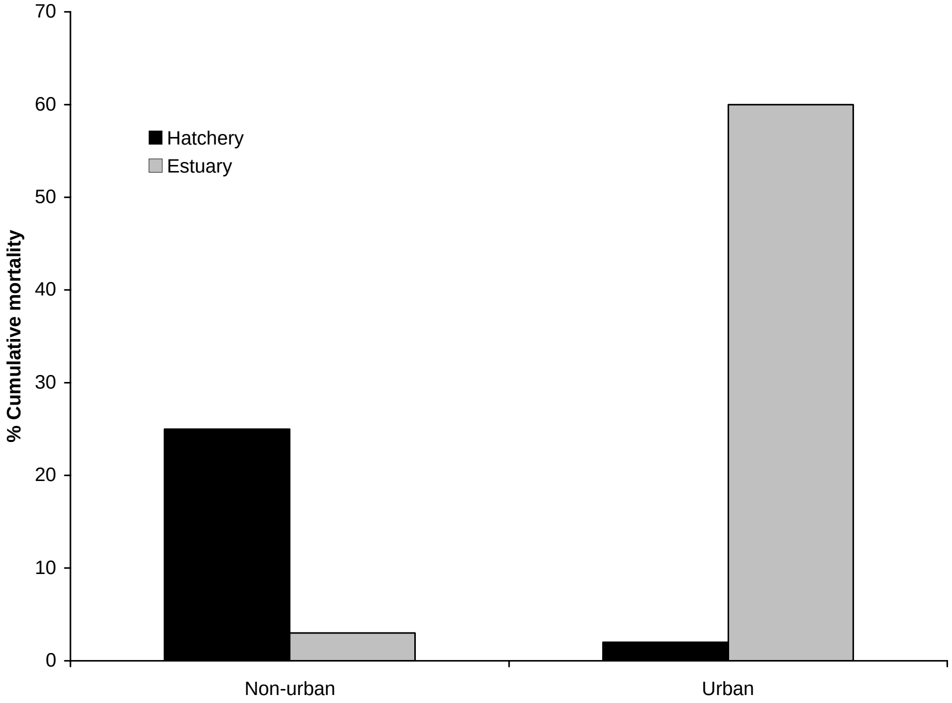

Figure 30. Cumulative mortality of juvenile Chinook salmon collected from the Duwamish Waterway (urban) and the associated hatchery-estuary system, and the Nisqually hatchery estuary system (nonurban) following disease challenge with the marine bacterium *Vibrio anguillarum.* (Adapted from Arkoosh et al. 1991.)

from the urban estuary than in fish from the corresponding hatchery or the nonurban estuaries and hatcheries. In a separate laboratory study (Casillas et al. 1998a), growth was reduced in juvenile Chinook salmon exposed to PCBs and to extracts of sediments from the Hylebos Waterway, another urban estuary. Because growth of salmon during the first year of life appears to be critical to recruitment success (Holtby et al. 1990, Pearcy 1992, Unwin 1997), contaminant-related reductions in juvenile growth rates could increase the risk of salmon mortality.

In other studies in Puget Sound, juvenile salmon from urban estuaries (e.g., the Duwamish and Hylebos Waterways) had significantly higher levels of DNA damage (i.e., PAH-DNA adducts in liver) than salmon from relatively uncontaminated sites. In addition, salmon from the urban estuaries had significantly higher induction of cytochrome P4501A (CYP1A), the enzyme that metabolizes selected contaminants including PAHs, dioxins and furans, and dioxin-like PCBs (Stein et al. 1995, McCain et al. 1990, Varanasi et al. 1993, Collier et al. 1998a, 1998b, Stehr et al. 2000). These biochemical alterations are not necessarily indicative of adverse health effects in themselves, but are associated with reproductive and developmental abnormalities and liver disease (Williams et al. 1998, Whyte et al. 2000, Myers et al. 2003).

At the two contaminated sites where most of the work described above was done, the Duwamish and Hylebos Waterways, average sediment PCB concentrations ranged from 400–500 ng/g dry wt and average PAH concentrations were about 10,000 ng/g dry wt (Collier et al.

1998a, Krahn et al. 1998). These sediment concentrations are within the range of those reported in the estuary (e.g., Tetra Tech Inc. 1996, Johnson et al. 2004), although higher than those typically found in estuary sediments. Total body PCB concentrations in fish collected from the Duwamish and Hylebos sites were in the 250–350 ng/g wet wt range, comparable to some fish sampled from sites within the estuary.

Various health effects have also been documented in nonsalmonid fish and other aquatic biota from the lower Columbia River, including fish with external abnormalities or skeletal deformities (Markle 1995, Tetra Tech Inc. 1995), alteration in endocrine response function (Goodbred et al. 1997, Foster et al. 2001a, 2001b), and pollution-associated liver lesions (Myers et al. 1994). Although the exposure patterns and life histories of listed salmon may differ from those of these fish species, these data raise additional concern about the potential for exposure and health impacts on salmon.

In addition to bioaccumulative contaminants, waterborne contaminants such as dissolved metals and current use pesticides may pose a threat to listed salmon. Various OPs such as diazinon, carbofuran, and chlorpyriphos at concentrations of 1–10 ug/L can disrupt olfactory function in salmon after exposures of as little as a few hours or days (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000, Sandahl et al. 2004). Scholz et al. (2000) reported that diazinon disrupted olfactory function in Chinook salmon at concentrations of 1–10 ug/L; as a result, fish failed to show normal antipredator responses or homing behavior (Figure 31).

Similar responses were observed by Sandahl et al. (2004) with chlorpyrifos at concentrations as low as 0.72 µg/L. Moore and Waring (1996) and Waring and Moore (1997) found that exposure to diazanon and carbofuran in a similar range could desynchronize the reproductive physiology of prespawning Atlantic salmon (*Salmo salar*) by inhibiting the male's ability to detect sex pheromones. Concentrations of diazanon in the 1–10 ug/L range have been reported in USGS NASQAN sampling in the estuary, and other pesticides with related modes of action that would be likely to have the same effects (e.g., chlorpyriphos, malathion, aldicarb, carbaryl, carbofuran) are detected even more frequently and at higher concentrations.

Similarly, Baldwin et al. (2003) and Sandahl et al. (2004) found that exposure to copper at concentrations in the 3–6 µg/L range for as little as 30 minutes affected olfactory function in coho salmon so they could no longer respond normally to test odorants (Figure 32). This could impair the ability of juveniles to find prey and avoid predators, or interfere with homing and reproductive behavior in adults. Dissolved copper concentrations at the estuary sites sampled in the NASQAN survey were within this range (Fuhrer et al. 1996), and copper in suspended sediments was substantially higher (45–120 µg/L). Other contaminants in the water column, including endocrine-disrupting substances such as synthetic hormones, are only beginning to be characterized in this part of the basin, but potentially could also have substantial impacts on salmon.

More research is clearly needed to document exposure and associated effects of chemical contaminants on endangered Columbia River estuary salmon, but the available data show that environmental concentrations and tissue burdens of several classes of contaminants are within the range where they could primarily affect abundance and population growth rate in listed stocks. The true magnitude of the effect is uncertain, but a recent modeling study suggests it

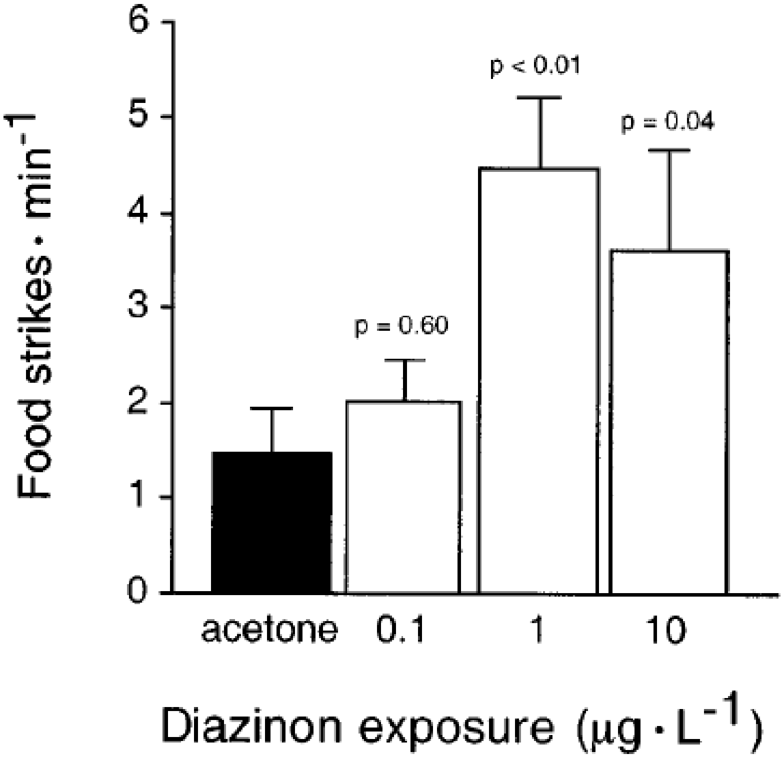

Figure 31. Foraging activity in the presence of olfactory signals of a potential predator in juvenile Chinook salmon exposed to diazinon. Control fish (solid bar) responded to the conspecific skin extracts by reducing their foraging activity and freezing. The magnitude of the antipredator response was reduced in diazinon-exposed fish (2 h at 0.1, 1.0, and 10.0 ug·L^{-1}), and they were more active and fed more often than controls. The effect of diazinon was significant at the 1.0 and 10.0 ug·L^{-1} exposures (*P* = 0.05, Fisher's test). (From Scholz et al. 2000.)

could be significant for at least some ESUs. Spromberg and Meador (in press) used life cycle models to examine the impacts of low-level toxic effects (10–25% response level for mortality, immune suppression, and growth) on the population dynamics of fall run Chinook salmon.

Responses in this general range might be expected in estuary fish exceeding, for example, the tissue benchmark for PCBs developed by Meador et al. (2002). These results indicate that after 20 years of continued reductions at the 10% level, population abundance was severely depressed (up to 2–3 times lower than nonimpacted populations) for several of the endpoints. When the 25% toxicity response was modeled for 20 years, population abundance was between 3 and 20 times lower, depending on the endpoint.

In summary, exposure to chemical contaminants has the potential to affect survival and productivity of both ocean-type and stream-type stocks in the estuary. Stream-type ESUs are most likely to be affected by short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals that may disrupt olfactory function and interfere with associated behaviors, such as capturing prey, avoiding predators, imprinting, and homing. Ocean-type ESUs will also be exposed to these types of contaminants, but will also be affected by persistent, bioaccumulative toxicants such as PCBs and DDTs, which they may absorb during their more extended estuarine residence. Consequently, it is likely that the impact on ESUs exhibiting the ocean life history type is greatest.

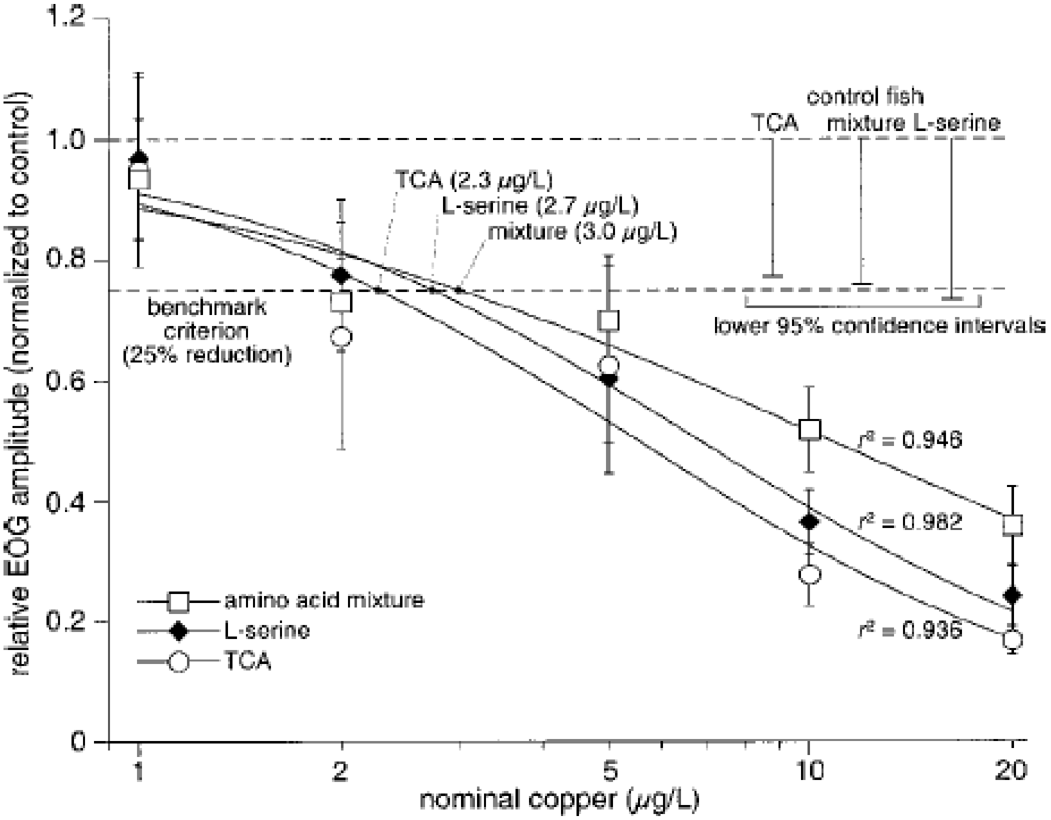

Figure 32. Dose-response curves and threshold determinations for sublethal copper neurotoxicity. Data were obtained from six treatment groups (control and five copper exposures; n = 6 fish per group). The evoked electro-olfactogram (EOG) amplitudes for all copper exposures were normalized to the mean response of the controls and expressed as a group (mean ± 1 standard error). The dashed line indicates a benchmark criterion of 0.75, or a 25% reduction in evoked response (relative to controls). Three vertical lines in the upper right show the lower limits of the 95% confidence interval for the control response to the three different odorants. Filled circles indicate the benchmark concentrations for the different olfactory pathways (Lserine, taurocholic acid [TCA], and the amino acid mixture). Note that the benchmark values are nominal concentrations, or a change (increase) from the copper present at approximately 3 mg/L in the source water for the NWFSC hatchery, Seattle. (From Baldwin et al. 2003.)

Caspian Tern Predation of Juvenile Salmon

In the preceding analyses, we suggested that changes in estuarine habitats affect population viability but that quantifying these changes is problematic. One factor that can be quantitatively linked to VSP criteria is predation. Estimates of life-stage specific predation mortality can be directly linked to changes in population growth rate or productivity. In general, predation by many predators in the Columbia River estuary has potentially increased from historical levels including predation by marine mammals, such as California sea lions (*Zalophus californianus*), smallmouth bass (*Micropterus salmoides*), northern pikeminnow (*Ptychocheilus oregonensis*), and double-crested cormorants (*Phalacrocorax auritus*) (e.g., Beamesderfer et al. 1996, Collis et al. 2001b, Ryan et al. 2003). For most of these predators, we have little quantitative data on historical or current levels of predation and how predation affects population viability.

In this subsection, we consider mortality of juvenile salmon as a result of predation by Caspian terns (*Sterna caspia*). Caspian terns met the criteria we described previously for selection of limiting factors for analysis. For example, levels of predation are greater at present than historically, and increasing numbers of terns nesting on islands in the Columbia River estuary annually consume large numbers of migrating juvenile salmonids (Roby et al. 1998). They are considered one of the factors that may currently limit salmonid stock recovery (Roby et al. 1998, IMST 1998). In the last section of this report, we examine predation by northern pikeminnow. Although much is known about northern pikeminnow predation, we do not consider it as a limiting factor comparable to terns, because of some key information gaps that we were unable to address (e.g., historical levels of predation are unknown).

Caspian terns are highly migratory and are cosmopolitan in distribution (Harrison 1983, Harrison 1984). Nesting has been reported throughout North America and in Australia, New Zealand, South Africa, Asia, and Europe. The numbers of Caspian terns in western North America more than doubled between 1980 and 1999 (Cuthbert and Wires 1999). One reason for the increase is that human-created habitat provides high quality nest sites and is associated with population increases in many parts of North America (Cuthbert and Wires 1999).

In the early 1990s, a substantial increase in the size of newly established Caspian tern nesting colonies on man-made islands in the Columbia River estuary was noted by NOAA Fisheries Service staff. Three estuary islands on which piscivorous birds nest were created from or augmented by materials dredged to maintain the Columbia River Federal Navigation Channel. Before 1984 there were no recorded observations of terns nesting in the Columbia River estuary, when approximately 1,000 pairs apparently moved from Willapa Bay to nest on newly deposited dredge material on East Sand Island. In 1986 those birds moved to Rice Island, a man-made island. The Caspian tern colonies in the estuary have since expanded to 9,000–10,000 pairs. In 1999 the colony was encouraged to relocate to East Sand Island. By 2001 the entire Columbia River estuary population of terns, representing the majority of this species on the West Coast, nested on just four acres on East Sand Island; in 2002 the terns nested on six acres.

Caspian terns arrive in the Columbia River estuary in March through April and begin nesting soon thereafter (Roby et al. 1998). To avoid mammal and avian predators, terns construct their nests on islands (Harrison 1984) and show a preference for barren sand. They are piscivorous in nature (Harrison 1984), requiring about 220 grams (roughly one-third of their body weight) of fish per day during the nesting season. The timing of courtship, nesting, and chick rearing corresponds with the outmigration of many of the salmonid stocks in the basin (Collis et al. 2002).

Salmon and steelhead constitute a major portion of tern diets, particularly when the birds nested on Rice Island. Based on diet analyses conducted in 1997–1998, juvenile salmonids constituted 77.1% of prey items and 76.7% of prey diet mass of Caspian terns nesting on Rice Island (Collis et al. 2002). During the May peak in the smolt outmigration of steelhead, yearling Chinook salmon, and coho salmon through the estuary, the diet of Caspian terns on Rice Island was over 80% juvenile salmonids (Collis et al. 2002). In early May, steelhead were the primary salmon eaten while coho salmon were the primary salmonid prey eaten in late May.

Efforts to relocate the terns to East Sand Island from Rice Island that began in 1999 have succeeded in reducing consumption of smolts without decreasing tern productivity. East Sand Island is a site lower in the estuary with abundant alternate prey sources. Over the last few years, studies suggest that consumption of salmonids in the estuary has been lower than previous levels while consumption of alternative prey species has increased. Relocating the colony to East Sand Island, which is lower in the estuary and closer to periodically abundant Pacific herring (*Clupea harengus pallasi*) and anchovies (*Engraulis mordax*), has contributed to the reduction. Observed diets, which consisted almost exclusively of salmonids at Rice Island (77% in 1999 and 90% in 2000), shifted to 46%, 47%, and 33% salmonids at East Sand Island in 1999, 2000, and 2001, respectively (Collis et al. 2001a, Roby et al. 2003).

These data suggest that substantial declines in juvenile salmonid mortalities from Caspian tern predation occurred from relocating the colony. This is supported by studies estimating tern consumption of salmonids. In 2000 salmonid consumption by terns was estimated at 7.3 million smolts, which is 4.4 million less than in 1999—the last time a substantial number of terns nested on Rice Island (Collis et al. 2001a, USFWS 2001). In 2001 salmonid consumption was estimated at 5.9 million smolts, which is 5.8 million less than in 1999 (Collis et al. 2001a). These data were substantiated by PIT (passive integrated transponder) tag detections on the two islands in 1999 and 2002. Approximately two to four times fewer tags per pair of terns were detected per nest on East Sand Island in 1999 and 2000 than were detected on Rice Island in 1999 and 2000.

In a recent evaluation of the impact of Caspian tern predation on juvenile salmon (Good et al. 2003), two approaches were recognized as providing the types of predation rate estimates that are needed for salmon life cycle models that are used to assess the effects of various factors on risk of extinction in the Columbia River basin (Kareiva et al. 2000). One approach uses bioenergetics modeling. Since 1997 biologists with the Bonneville Power Administration-funded research project, Avian Predation on Juvenile Salmonids in the lower Columbia River, (a joint project of Oregon State University, the U.S. Geological Survey, the Columbia River Inter-Tribal Fish Commission, and Real Time Research Consultants) have used observed salmonid consumption at tern colonies in a bioenergetics model (Roby et al. 1998) to estimate the consumption of salmonids in the Columbia River estuary. Smolt consumption estimates from 1999–2002 using this approach ranged from a low of 5.9 to a high of 11.7 million smolts eaten.

Another approach uses detections of PIT tags on Caspian tern colonies to estimate salmonid predation rates overall, as well as by ESU (Collis et al. 2001b, Ryan et al. 2001). Since 1987, researchers in the Columbia River basin have placed over five million PIT tags in juvenile salmonids for a variety of studies (Ryan et al. 2001). Identifying PIT tags on bird colonies can provide a minimum estimate of proportion of the stocks that were consumed by terns in these colonies. In recent years, approximately one million juvenile salmonids have been PIT-tagged annually (Collis et al. 2001b), the vast majority of which are steelhead and Chinook salmon from the Snake River basin. Over 115,000 PIT tags were detected on Rice Island in 1998 and 1999 (Ryan et al. 2001).

Of the PIT tags placed in steelhead smolts in 1997 that were detected at Bonneville Dam, 2.8% of wild smolts and 5.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For steelhead that were PIT tagged in 1998 and detected at Bonneville Dam, 11.7% of wild smolts and 13.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For yearling Chinook salmon that were PIT tagged in 1998 and detected at Bonneville Dam, 0.5% of wild smolts and 1.6% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b).

Ryan et al. (2003) analyzed PIT-tag data from 1998 to 2000 on Rice Island and East Sand Island and determined that steelhead experienced higher predation rates (0.6% to 8.1% on East Sand Island, and 1.3% to 9.4% on Rice Island) than Chinook salmon (0.2% to 2.0% on East Sand Island, and 0.6% to 1.6% on Rice Island). Overall, Caspian terns consumed approximately 6% to 14% of the estimated outmigrating population of juvenile salmonids originating from the Columbia River basin.

In a recent analysis of the impact of Caspian tern predation on salmon recovery, efforts focused on determining whether a unique predation rate could be identified. The effort focused on the Caspian tern colonies on East Sand Island in the lower estuary of the Columbia River because the colony currently represents the majority of the West Coast Caspian tern population. The focus period was 1999–2002 because this represents the time period after relocation from Rice Island during which this colony has dominated Caspian tern predation activity in the Columbia River estuary. Bioenergetics modeling was used to calculate predation rates (estimated number of salmon consumed/estimated number of salmon available in the estuary) using updated and refined estimates of the number of outmigrating salmon that migrate through the river or are transported and released below Bonneville Dam. PIT-tag detections were also used to generate estimates of predation rates on salmon.

The relationship between tern abundance and mortality rate due to tern predation is not known with certainty. The estimated predation rate based on either bioenergetics modeling or PIT-tag data was linearly related to the number of Caspian terns nesting on East Sand Island during the breeding seasons of 1999–2002 (Figure 33). Moreover, PIT-tag detection also allowed ESU-specific predation rate estimates to be derived. Per capita consumption rates (number of smolts consumed per adult tern) have been relatively constant throughout the range of colony sizes on East Sand Island from 1999 to 2003. The per capita consumption rate in 1999 (mean = 437.5) was nearly equivalent to that of 2000 (mean = 431.1), even though there was an almost fivefold difference in colony size (Figure 34).

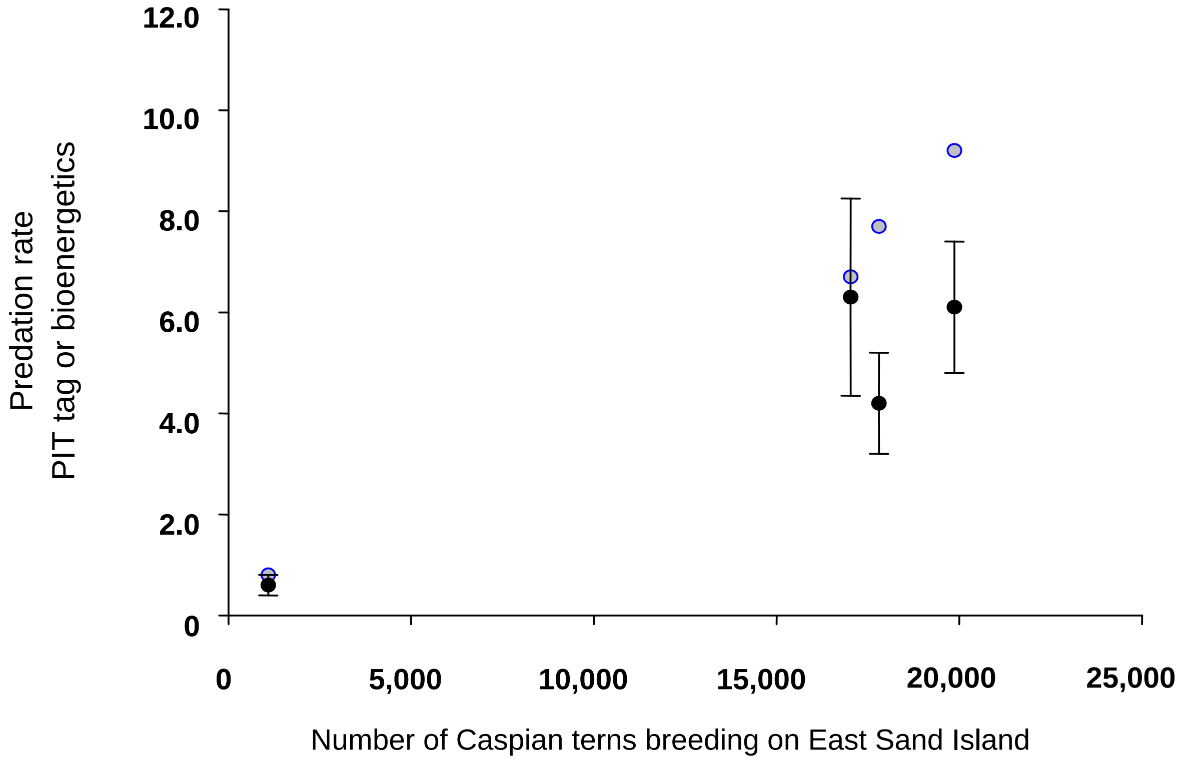

Figure 33. Estimated predation rates on all steelhead in the Columbia River estuary by Caspian terns, 1999–2002, using bioenergetics modeling (black symbols) and recovery of PIT tags (grey symbols). Error bars on bioenergetics estimates represent 95% confidence limits.

NWFSC has developed a matrix model to assess salmonid population trends and the impact of anthropogenic activity on those trends. This model has application when mortality rates can be constructed and attributed to a particular source. The value of life cycle models derive from providing an objective outcome for comparing the influence of various factors influencing population growth rates, rather than attempting to estimate real gains from any management action. Assessing the impact of predation by Caspian terns on juvenile salmonids during a particular life history phase was amenable to such evaluation.

Using the Cumulative Risk Initiative (CRI) model, Good et al. (2003) estimated the impact of Caspian tern predation on the population growth rate (λ) of all steelhead and spring Chinook salmon in the basin using predation rate estimates derived from bioenergetics modeling and PIT-tag detections. Because of the similarity in the results between the two approaches, we present information only from estimates derived from PIT-tag detections, as ESU-specific impacts can also be derived with this information.

The predation rate for 20,000 Caspian terns feeding on all steelhead and spring Chinook salmon was estimated using the regression equations generated from PIT-tag detections. This number of terns represents the maximum number observed to date on East Sand Island. Reductions in predation rate corresponding to reduced tern population sizes were used to model the potential increase in λ, assuming all steelhead or spring Chinook salmon mortality

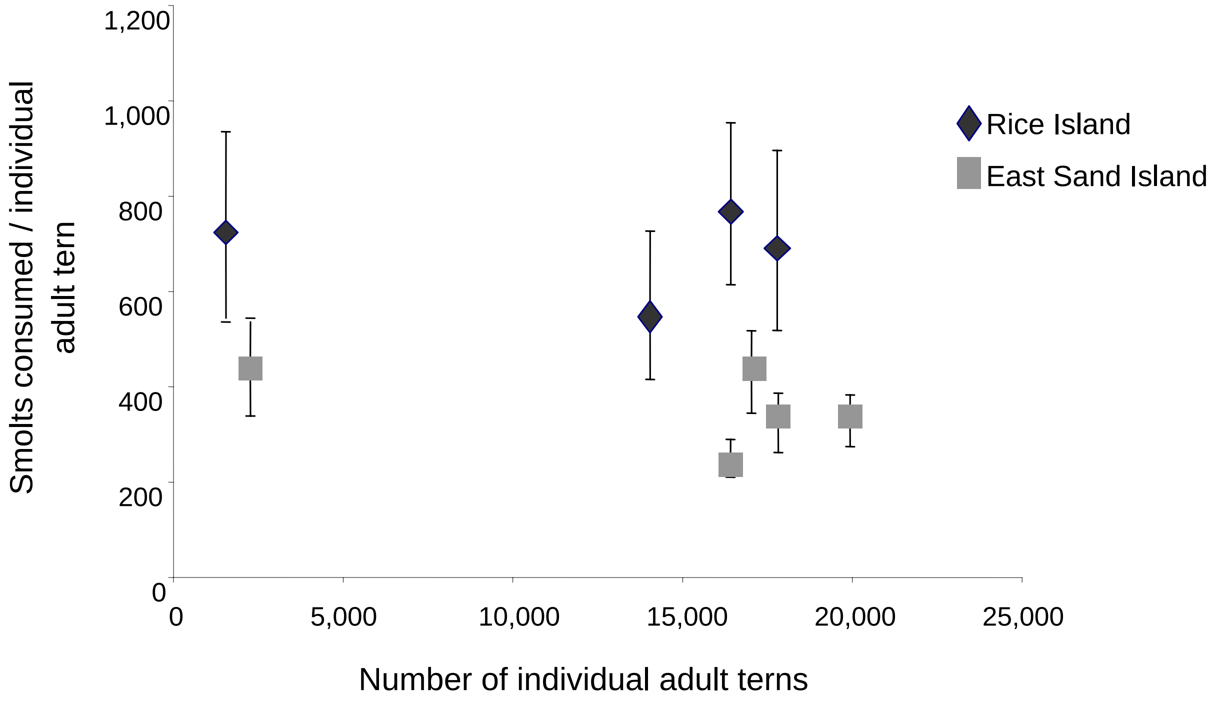

Figure 34. Per capita smolt consumption by Columbia River estuary Caspian terns on all salmonids, 1997–2003 (with 95% CI). (Courtesy of D. Lyons and D. Roby, Oregon Fish Cooperative Unit, Oregon State University.)

attributable to terns is not compensated for by mortality due to other sources. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation (i.e. removal of all terns from the estuary) was 1.9% and 0.8% for steelhead and spring Chinook salmon, respectively, using the PIT-tag estimate of predation rate.

The PIT-tag detection approach enables the calculation of ESU-specific estimates of predation rate (and hence proportion increase in λ). Good et al. (2003) used the CRI model to estimate impact of Caspian tern predation on the population growth rate (λ) of steelhead and spring Chinook salmon ESUs. They used predation rates estimated from PIT tag detections for which reasonable estimates of the number of smolts available to be consumed could be generated. Predation rates for 20,000 Caspian terns on four of the five ESA-listed steelhead and spring Chinook salmon ESUs were estimated using linear regression. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation ranged from 1.9% to 4.9% for steelhead ESUs.

Several factors must be considered when interpreting the results of these calculations. Perhaps the most important is that this type of calculation assumes that there is no compensatory mortality later in the life cycle, and that any reduction in tern predation is fully realized. In their assessment of predation impact by Rice Island terns on salmonids in 1997–1998, Roby et al. (2003) hypothesized that tern predation was 50% additive. Given these limitations and uncertainties, the estimates of percent change in population growth rates should be viewed as maximum potential improvements. Realized improvements in population growth would likely be lower from any management action that reduces Caspian tern predation impacts on salmonid ESUs.

These results may not be as easy to achieve as they are to calculate. It is important to recognize that other factors, such as ocean conditions, may also influence population growth rate to a greater degree than the potential gains that may be realized from reducing predation by one species of avian predator on one island located in the lower estuary of the Columbia River basin.

Overall, it is evident that Caspian tern predation primarily affects salmon and steelhead that exhibit a stream-type life history rather than an ocean-type life history. This is primarily a result of the dominant migratory periods employed by salmonids with a stream-type life history. Salmon from this life history type move in great numbers at the time Caspian terns begin nesting (May through June) and have the greatest energetic needs for chick production.

Although there are some impacts to juvenile salmon exhibiting an ocean-type life history, characteristic of fall Chinook in the Columbia River basin, the impact is less than for the stream-type salmonids (Roby et al. 2003). Good et al. (2003) concluded that gains in λ for steelhead ESUs were comparable to gains that could be derived from additional improvements to the FCRPS to increase survival, but much less than can be achieved by harvest modifications. Because steelhead ESUs were most strongly affected by Caspian tern predation, improvements to λ by managing terns were considered to benefit other salmon ESUs in the basin, albeit to a much lesser degree.

In summary, predation is a major source of mortality of all salmonid populations. Although many predator-prey interactions in the Columbia River estuary have probably changed from historical conditions (e.g., northern pikeminnow), we have little quantitative data on most predator-prey interactions. One exception is the Caspian tern predation which has significantly increased recently due to a change in nesting habits of the birds in the Columbia River estuary. The main impact of tern predation is on ESUs with stream-type life history types, especially steelhead. This is primarily because the dominant migratory periods employed by salmonids with a stream-type life history most overlap with the nesting period of the terns. Improvements to productivity of populations by managing terns would be expected to benefit stream-type ESUs especially, although lesser benefits to other salmonid ESUs in the basin should also occur.

Effects of Estuarine Factors on Recovery Potential of ESUs

In this section, ESUs, life history type, life history strategy, limiting factor impacts, habitat attributes, and population viability are integrated in order to compare and contrast the importance of each limiting factor on population viability. We consider each factor separately from the others and do not consider cumulative impacts. All listed ESUs in the Columbia River basin are included in addition to lower Columbia River coho salmon because of the possibility that this ESU will get listed in the future. The focus of this analysis is on effects of estuarine factors on population viability, not on the importance of the estuary relative to other factors operating at other life stages. That broader analysis is outside the scope of this review.

We evaluated effects of factors within two zones of the estuary. Bonneville Dam to the river's mouth was defined as one zone and the plume was considered a second major zone. Within the first zone, we also differentiated shallow, low velocity, vegetated habitats from medium and deep, higher velocity habitats generally associated with the main channels. We were limited to this broadscale analysis because a finer scale evaluation of how juvenile life history strategies use habitats and zones is not possible with existing information.

For example, we know fry and fingerling strategies are more closely associated with shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats), while the opposite pattern exists for larger size classes such as yearlings, which are more abundant in deeper, higher velocity channel habitats (Healey 1980, 1982, Levy and Northcote 1981, 1982, Simenstad et al. 1982, Levings et al. 1986, Miller and Sadro 2003). But we do not know how different types of shallow water habitats are used (e.g., emergent marsh vs. swamp vs. mudflat) within a zone. While differences are likely in use of different habitats between zones, we lacked the ability to make this type of discrimination other than between the confined portion of the estuary and the plume.

To guide our evaluation, we used the following hypotheses about use of estuarine habitats by juvenile salmon and the effects of specific limiting factors that were developed in previous sections:

1. Tern predation differentially affects the larger yearling strategies, especially steelhead, more than smaller life history strategies, such as fingerling Chinook (Ryan et al. 2003). Tern predation is assumed to occur in the estuary zone but primarily in medium and deep water channel habitat, rather than shallow water areas (Tables 5 and 6). Tern predation is assumed to be minimal in the plume.
2. Mortality of juvenile salmon in the estuary is occurring due to predation by species other than terns. We did not consider these other predators in this analysis.

Table 5. Linkages between limiting factors associated with the estuary upstream of the river's mouth, life history strategies, life history type (ocean-type and stream-type), and ESU (for a complete list see Table 3). Only two general habitat types in the estuary were considered: shallow, low velocity, and medium/deep channel higher velocity. Factors were ranked as having a high, medium, or low ability to affect the relative abundance of particular life history strategies.

ESU*	**Life history type**	**Life history strategy**	**Shallow, low velocity**				**Deep channel**			
			Flow	Habitat	Terns	Toxics	Flow	Habitat	Terns	Toxics
SR Fall, LCRC,	Ocean-type	Early fry	High	High	Low	Medium	Low	Low	Low	Low
		Late fry	High	High	Low	Medium	Low	Low	Low	Low
		Early fingerling	High	High	Low	Medium	Low	Low	Low	Low
		Late fingerling	High	High	Low	Medium	Low	Low	Low	Low
		Subyearling	Low	Low	Medium	Low	Medium	Low	Medium	Low
		Yearling	Low	Low	Medium	Low	Medium	Low	Medium	Low
LCRS, UCRSC	Stream-type	Early fry	Medium	Medium	Low	Low	Low	Low	Low	Low
		Late fry	Medium	Medium	Low	Low	Low	Low	Low	Low
		Early fingerling	Medium	Medium	Low	Low	Low	Low	Medium	Low
		Late fingerling	Medium	Medium	Low	Low	Low	Low	Medium	Low
		Subyearling	Low	Low	Medium	Medium	Medium	Low	Medium	Low
		Yearling	Low	Low	Medium	Medium	Medium	Low	Medium	Low

* Examples: SR Fall = Snake River fall Chinook salmon, LCRC = Columbia River chum salmon, LCRS = lower Columbia River steelhead, UCRSC = upper Columbia River spring Chinook.

Table 6. Linkages between limiting factors associated with the plume, life history strategies, life history type (ocean-type and stream-type), and ESU (for a complete list, see Table 3). The plume was considered as one habitat zone. Factors were ranked as having a high, medium, or low ability to affect the relative abundance of particular life history strategies using the plume.

			Plume			
ESU*	**Life history type**	**Life history strategy**	Flow	Habitat	Terns	Toxics
SR Fall, LCRC,	Ocean-type	Early fry	Low	Low	Low	Low
		Late fry	Low	Low	Low	Low
		Early fingerling	Medium	Low	Low	Low
		Late fingerling	Medium	Low	Low	Low
		Subyearling	Medium	Low	Medium	Low
		Yearling	Medium	Low	Medium	Low
LCRS, UCRSC	Stream-type	Early fry	Low	Low	Low	Low
		Late fry	Low	Low	Low	Low
		Early fingerling	Medium	Low	Medium	Low
		Late fingerling	Medium	Low	Medium	Low
		Subyearling	Medium	Low	Medium	Low
		Yearling	Medium	Low	Medium	Low

* Examples: SR Fall = Snake River fall Chinook salmon, LCRC = lower Columbia River chum salmon, LCRS = lower Columbia River steelhead, UCRSC = upper Columbia River spring Chinook.

3. The main effects of flow reductions are on the amount of shallow water habitat available to fish and the opportunity for the fish to use the habitat. The main effect of habitat changes is on distribution, quantity, and quality of habitat. The main effect of toxics is on habitat quality (capacity).
4. Any reduction in quality or quantity of shallow water habitat affects smaller juvenile salmonids employing strategies such as fry and fingerlings significantly more than yearlings (Tables 5 and 6). Changes in the quantity and quality of shallow water habitats most impacts viability of ocean-type populations.
5. Subyearlings use shallow, medium, and deep water habitats, whereas yearlings primarily use medium and deep channel habitat.
6. Fry and early fingerling life history strategies do not make extensive use of the plume, but more likely utilize the surf zone when they exit the estuary proper.
7. Reductions in flow above Bonneville Dam affect the size and shape of the plume. Primarily as a result of flow, but also no doubt as a result of physical changes to the estuary (eg., dredging and diking), the shape, behavior, size, and composition of the plume has been changed.
8. Toxics impact the quality of habitat, but consequences of toxics can occur downstream of where the burden was acquired. The impact, though, is assumed to be associated with the habitat where the exposure occurs.

9. Flow and habitat changes in the estuary are interrelated.

For stream-type ESUs (e.g., Snake River spring/summer Chinook salmon and mid-Columbia River steelhead), the primary estuarine factors affecting population viability are tern predation and flow (Tables 7, 8, and 9). Tern predation was ranked in the medium category for several reasons. First, tern predation is primarily directed at subyearling and yearling size fish, which are the dominant strategies in stream-type ESUs such as Snake River steelhead. Second, these larger fish occur in habitats (deeper water channel habitats) where they are most vulnerable to the terns. Third, these larger fish migrate at a time when they are most susceptible to tern predation. Fourth, tern predation significantly affects abundance and productivity; scores for these parameters were doubled if we concluded there was an affect. Based on anecdotal observations of NOAA Fisheries Service working in the Columbia River plume, we assumed that most tern predation occurred upstream of the river mouth. If significant predation did occur in the plume, then the score for this factor would increase.

Flow changes were also ranked medium for stream-type ESUs because both abundance and productivity were affected. Further, the main life history strategies for these ESUs are abundant in the plume, where they are most vulnerable to flow-related changes in plume habitat.

Table 7. Summary rating table for listed Columbia River basin ESUs for estuary factors. Ranks were assigned based on the following ranges: low (<0.34), medium (0.34–0.66), and high (0.67–1.00).

Life history type	**Stream-type**	**Ocean-type**
ESUs	Snake River spring/summer Chinook Upper Columbia River Chinook Snake River steelhead Upper Columbia River steelhead Middle Columbia River steelhead Lower Columbia River steelhead Upper Willamette River steelhead Upper Snake River sockeye Lower Columbia River coho salmon	Columbia River chum salmon Snake River fall Chinook Upper Willamette River Chinook Lower Columbia River fall Chinook

	Factor for stream-type				**Factor for ocean-type**			
Rating level	Tern predation	Toxics	Habitat	Flow	Tern predation	Toxics	Habitat	Flow
Level 1	6	5	4	7	3	5	8	7
Level 2	6	2	0	3	2	6	6	7
Total score	12	7	4	10	5	11	14	14
Total possible	20	28	20	20	20	28	20	20
Ratio	0.60	0.25	0.20	0.50	0.25	0.39	0.70	0.70
Rank	Medium	Low	Low	Medium	Low	Medium	High	High

Table 8. Level 1 ratings of estuary factors for stream-type ESUs (Snake River spring/summer Chinook, upper Columbia River Chinook, Snake River steelhead, upper Columbia River steelhead, middle Columbia River steelhead, lower Columbia River steelhead, upper Willamette River steelhead, upper Snake River sockeye, and lower Columbia River coho salmon). An answer to a question of yes equals a 1 other than for the VSP criteria of productivity and abundance, which are scored a 2 for yes. An answer of no equals a zero.

	Factor			
Screening criteria	Tern predation	Toxics	Habitat	Flow
Level 1—Is the factor of concern for the ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected? (2x)	2	2	0	2
Is there a significant affect on productivity? (2x)	2	2	0	2
Is there a significant affect on LH diversity?	0	0	1	1
Is there a significant affect on spatial structure?	0	0	1	1
What is the level of change possible in factor?				
Is there a significant change from historical levels?	1	1	1	1
Is the amount of improvement possible substantial?	1	0	1	0
Score	6	5	4	7
Maximum Possible Score	8	8	8	8

Toxics and habitat were ranked low for stream-type ESUs because the main life history strategies associated with them do not occupy the habitat where the main effects occurs, although there may be some risk associated with short-term exposure to waterborne contaminants for these fish.

For ocean-type ESUs (lower Columbia River fall Chinook salmon and Columbia River chum), flow and habitat were rated as having a high ability to affect population viability (Tables 7, 10, and 11). Flow and habitat affects are most significant in shallow water areas; both the quantity of habitat and the opportunity to use this habitat are affected. Finally, both the flow and habitat limiting factors affect all VSP parameters for ocean-type populations. The loss of shallow water habitat and changes in its distribution and quality caused by flow and habitat changes will reduce the capacity of estuarine habitats to support ocean-type populations; this will reduce abundance and productivity of these populations. Further, loss and degradation of shallow water habitat will also diminish the spatial structure and number of life history pathways available to the fish. This has the potential to make these populations more vulnerable to effects of extreme events such as severe droughts or strong El Niño events. As we have noted, the losses of shallow water habitats due to the combined effects of flow and habitat changes are dramatic while losses of deeper water habitats appear to be minimal.

Flow and habitat also have a significant affect on ocean-type populations because these populations are dominated by small size classes (fry and fingerlings) that make extensive use of shallow water habitats and have the longest residence time in the estuary (i.e., they are most dependent on shallow water habitats) (e.g., Bottom et al. 2005). A major function of these shallow water habitats for these small size classes is to support their feeding and growth; high

Table 9. Level 2 ratings of estuary factors for stream-type ESUs (Snake River spring/summer Chinook, upper Columbia River Chinook, Snake River steelhead, upper Columbia River steelhead, middle Columbia River steelhead, lower Columbia River steelhead, upper Willamette River steelhead, upper Snake River sockeye, and lower Columbia River coho salmon). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a zero.

	Terns			**Toxics**[a]			**Habitat**			**Flow**		
Screening criteria	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
Level 2—Significance of factor												
For the dominate LHS, is the relative impact on numbers by habitat type significant?	1	1	1		1							1
For the dominate LHS, does the factor significantly affect habitat?												
1. Quantity												
2. Quality					1							1
3. Opportunity	1	1	1									1
Score	2	2	2	0	2	0	0	0	0	0	0	3
Total factor score		6			2			0			3	
Maximum possible score		12			20			12			12	

[a] Scores for toxics include a value for sediment and water in estuary (i.e., the SW quality score can be a 2 and water in the plume.

[b] SW = Shallow water from the river's mouth to Bonneville Dam; DW = Deep water from the river's mouth to Bonneville Dam; Pl = Plume.

Table 10. Level 1 ratings of estuary factors for ocean-type ESUs (lower Columbia River chum salmon, upper Willamette River Chinook, lower Columbia River fall Chinook, and Snake River fall Chinook). An answer to a question of yes equals a 1 other than productivity and abundance, which are scored a 2 for a yes. An answer of no equals a zero.

	Factor			
Screening criteria	Tern predation	Toxics	Habitat	Flow
Level 1—Is the factor of concern for the ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected? (2x)	0	2	2	2
Is there a significant affect on productivity? (2x)	0	2	2	2
Is there a significant affect on LH diversity?	1	0	1	1
Is there a significant affect on spatial structure?	0	0	1	1
What is the level of change possible in factor?				
Is there a significant change from historical levels?	1	1	1	1
Is the amount of improvement possible substantial?	1	0	1	0
Score	3	5	8	7
Maximum Possible Score	8	8	8	8

growth rates experienced here can help population members avoid (i.e., outgrow) some of the predation that these small fish experience (Simenstad et al. 1982). Yearling and subyearling fish are generally not abundant in the habitats where they are most vulnerable to the effects of these two factors. Because relatively few stream-type fish use shallow water habitats, we predict that flow and habitat will have less of an effect on capacity of these habitats to rear and support stream-type populations, but a more significant affect on diversity and spatial structure of these populations (Table 7). Because of the loss of shallow water, estuarine dependent strategies (i.e., fry and fingerlings), the number and quality of the life history trajectories expressed by these populations will decline.

Effects of toxic contamination on ocean-type ESUs was rated medium. Both waterborne and sediment contaminants can affect these life history strategies in shallow water areas where the dominant life history strategies are most abundant. We assumed that toxics impact the quality of habitat upstream of the river mouth and that there was not significant affects in the plume. The consequences of the uptake of toxics can occur downstream of where the burden was acquired. We assumed the impact was associated with the habitat where the exposure occurred. Tern predation has a low effect on these ESUs because terns do not target fry and fingerling strategies (the dominant ones associated with these ESUs).

Table 11. Level 2 ratings of estuary factors for ocean-type ESUs (lower Columbia River chum salmon, upper Willamette River Chinook, lower Columbia River fall Chinook, and Snake River fall Chinook). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a zero.

	Terns			**Toxics**[a]			**Habitat**			**Flow**		
Screening criteria	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
Level 2—Significance of factor												
For the dominate LHS, is the relative impact on numbers significant by habitat type?	1			2	1		1			1		1
For the dominate LHS, does the factor significantly affect habitat?												
1. Quantity							1		1	1		1
2. Quality	1			2	1		1		1	1		
3. Opportunity							1			1		1
Score	2	0	0	4	2	0	4	0	2	4	0	3
Total factor score		2			6			6			7	
Maximum possible score		12			20			12			12	

[a] Scores for toxics include a value for both sediment and water in estuary (i.e., the SW quality score can be a 2 and water only in the plume.

[b] SW = Shallow water from the river's mouth to Bonneville Dam; DW = Deep water from the river's mouth to BonnevilleDam; Pl = Plume.

Other Considerations in Evaluating the Role of the Estuary in Recovery of Anadromous Salmonids

Data Gaps

While our intent in this report was not to define all major information needs (this is done more comprehensively by Bottom et al. 2005), there were several data gaps that notably constrained our ability to analyze effects of factors on population viability, that are important to highlight. One of these is the lack of information on juvenile salmon use of the estuary by different populations, especially how salmon use different geographic zones and different habitats within each zone. This is particularly true of the upper estuary above Jones Beach where sampling has been very limited. Because of this lack of habitat and zone-specific data, we had to assume that all shallow water areas are similarly used between the river's mouth and Bonneville Dam. We expect that the results of ongoing research conducted by NOAA Fisheries Service will significantly increase our understanding of the use of different estuarine habitats and zones.

A second major data gap was the lack of knowledge regarding adult salmon use of the estuary. Our entire analysis was based on juvenile salmon use of the estuary. Although it seems reasonable to hypothesize that some of the factors we considered could impact adult salmon in the estuary, we had no information on adult salmon and their use of estuarine habitats. For example, adult salmon are probably exposed to toxics in the estuary with unknown effects, while changes in flow attributes may alter the timing of adult salmon migrations and expose them to predators such as marine mammals for longer periods.

A third major data gap was that we were only able to consider the effects of four factors on population viability. The selection of these four factors was based on the relative effects of each limiting factor within the estuary and plume and the availability of empirical information to include in the evaluation. These four factors were not selected because we felt that they would have the most significant impact; some of the factors not included may have a considerable affect on population viability of some ESUs. For example, we expect that estuarine water temperatures have warmed from historical levels, which could affect metabolic processes of both salmon and their predators. These changes could increase mortality rates of salmon while in the estuary. Further, warmer water temperatures may exclude some habitats from use by juveniles during part of the year.

A fourth important data gap is our lack of knowledge of habitat changes in the upper estuary (above RM 46). Quantitative analyses by Thomas (1983) and others have been limited to the lower estuary and have not quantified the nature and extent of habitat changes in this large portion of the estuary. Finally, with the exception of tern predation, we were unable to quantify the effects of other predators.

Hatchery Fish Effects

We did not consider any potential effects of the large numbers of anadromous salmonids released from hatcheries throughout the Columbia River basin that pass through the estuary. There are several possible implications of these releases. First, because our ability to distinguish wild and hatchery fish captured in the estuary has been limited, and remains so even at present, many of the spatial and temporal patterns of habitat use suggested by historical data sets may apply to hatchery fish rather than wild fish (Dawley et al. 1985, 1986). For example, the large peak in abundance of juveniles in the estuary reported by Burke (2005) in July is probably comprised largely of hatchery fish.

Second, hatchery salmonids can potentially affect the growth and survival of wild salmonids in a variety of ways (Fresh 1997). Large numbers of hatchery fish co-occurring with wild fish might actually increase survival of wild fish by buffering them from predators such as Caspian terns. In addition, hatchery fish can reduce growth and survival of wild fish by competing for food resources. Some of the larger hatchery salmonids could prey on wild fish.

Northern Pikeminnow Predation on Juvenile Salmonids

As we noted previously, many predator-prey interactions in the Columbia River estuary involving juvenile salmon have potentially changed from historical conditions. We treated predation by Caspian terns on juvenile salmonids as a “full” limiting factor because it met our criteria for inclusion. While other predators were not included because they did not meet our criterion, one predator that nearly had enough information to include as a limiting factor was northern pikeminnow. We include a discussion here of predation by northern pikeminnow as another example of the effects of a predator.

Northern pikeminnow are relatively large, long-lived, slow-growing, predaceous minnows native to the Columbia River basin and other parts of the Pacific Northwest. Maximum fork length, weight, and age are approximately 600 mm, 2.5 kg, and at least 16 years in the Columbia River (Parker et al. 1995). Large, old individuals typically dominate unexploited populations of northern pikeminnow. Juvenile salmonids are generally an important diet component only for these large, old northern pikeminnow (Vigg et al. 1991, Zimmerman 1999), and consumption rates of juvenile salmonids by northern pikeminnow increase as size increases. Zimmerman (1999) found that fish consumed by northern pikeminnow in all reaches of the Columbia River were overwhelmingly juvenile salmonids, and that numerical frequency of Chinook salmon greatly exceeded that of steelhead. Daily consumption of juvenile salmonids was greater in summer than in spring.

Development of the hydropower system in the Columbia River basin has resulted in increased losses of juvenile salmonids to northern pikeminnow. At dams, migrating juvenile salmonids are concentrated in forebays and tailraces, causing increased predation and salmonid loss (Poe et al. 1991, Vigg et al. 1991, Ward et al. 1995). Migration past dams also causes injury and physiological stress, which may increase the vulnerability of salmonids to predators (Mesa

1994). Impoundments increase travel time for migrating juvenile salmonids, prolonging their exposure to predators (Raymond 1988, Poe et al. 1991).

Significant losses of juvenile salmonids occur downstream of Bonneville Dam. Beamesderfer et al. (1996) estimated that about 10 million juvenile salmonids were consumed annually by northern pikeminnow in the Columbia River downstream from Bonneville Dam prior to implementation of a Northern Pikeminnow Management Program (NPMP). This was about 5% of the approximately 200 million juvenile salmonid migrants in the Columbia River basin, but a much higher percentage of salmonids that reached the river downstream from Bonneville Dam. This estimated loss exceeded the total estimate for the remainder of the lower Columbia and Snake rivers. Estimates of predation losses were relatively unbiased by consumption of juvenile salmonids killed by dam passage (Gadomski and Hall-Griswold 1992, Petersen et al. 1994). Abundance of northern pikeminnow downstream from Bonneville Dam greater than 250 mm fork length likely ranged from 600,000 to 800,000 individuals (Beamesderfer et al. 1996).

Unlike other resident fish predators, population dynamics and behavior of northern pikeminnow indicate that reductions in predation through a removal program are feasible. The NPMP began in 1990, based on findings from the earlier work conducted in John Day Reservoir (Rieman and Beamesderfer 1990, Poe et al. 1991, Vigg et al. 1991, Beamesderfer and Rieman 1991, Rieman et al. 1991). Since 1990 over 750,000 northern pikeminnow have been removed from the Columbia River downstream from Bonneville Dam. Abundance of large northern pikeminnow downstream from Bonneville Dam has been reduced to about 500,000 individuals. Based on changes in the population structure of northern pikeminnow resulting from this harvest, Friesen and Ward (1999) estimated that predation on juvenile salmonids by northern pikeminnow was reduced by approximately 25% annually.

Empirically derived estimates of predation indicate that reductions since implementation of the NPMP may be more substantial. Zimmerman and Ward (1999) estimated that predation from 1994 to 1996 was about 50% of predation in 1992. Zimmerman et al. (2000) reported similar results for 1999.

The major uncertainties that precluded our including northern pikeminnow predation as a limiting factor were:

1. Uncertainty about historical levels of predation in this reach of the river. While the high levels of predation in the tailrace of Bonneville Dam is a change from historical levels, we could not determine whether predation downstream of this point had increased from historical levels.
2. Uncertainty about where predation occurs. While predation is generally regarded as most significant in littoral areas this is defined as depth less than 13m. We could not determine more specifically where predation in the littoral areas was occurring.
3. Uncertainty about what size classes were affected in the portion of the river. While Zimmerman (1999) provides data on size of salmonids eaten by northern pikeminnow throughout the basin, we could not determine what sizes of salmonids were being eaten in the river below Bonneville Dam.

Direct Affects of the Hydropower System on the Estuary

One of the outcomes of this analysis is that we identified a clear linkage between hydropower operations and two of the factors we considered: flow and habitat. We did not consider the operation of the hydropower system to have a direct affect on either toxics or tern predation other than by perhaps increasing travel time and hence exposure of the fish to both predators and toxics. We concluded that most of the flow changes and some of the changes in habitat in the estuary could be directly attributed to affects (e.g., reduction in magnitude of the spring freshets) of the FCRPS.

For example, reductions in flow can permanently eliminate some habitats from use by estuarine dependent life history strategies. Even though the habitat may not be diked, it becomes functionally "too high" in elevation for the fish to use because of reductions in flow. In addition, because there is a relationship between flow and habitat, the value of some habitats can be diminished by a reduction in flow because it becomes accessible for less time than under historical conditions. Many attributes of the plume environment are also directly affected by the hydropower system including the size, shape, and seasonal movements.

While some of the affects of the hydropower system in the estuary could potentially be mitigated by altering flow regimes (e.g., increasing flows in spring and summer), there are clearly a variety of other consequences that would need to be considered with such changes (e.g., an increase in gas bubble disease). Some habitat changes not directly linked to affects of the hydropower system are possible, however. This involves restoring some of the shallow water habitat used by ocean-type ESUs and the fry and fingerling portions of stream-type ESUs that have been isolated from the system by levees and dikes. Dikes permanently isolate this habitat and make either direct use by the juvenile salmonids (access) or indirect use (organic matter transport) impossible. Breaching or leveling dikes is clearly a strategy that can be used to restore some of the shallow water habitat important to these shallow water dependent strategies.

How Much Change Is Necessary to Affect Population Status

From the perspective of the estuary, population viability of stream-type ESUs is most affected by tern predation and flow, while ocean-type ESUs are most affected by flow and habitat. At this time, we do not know how much of a change in each factor is required to affect viability of relevant ESUs. Probably the greatest opportunity to affect ESUs by manipulating one of these factors is by restoring lost, shallow water, low velocity, vegetated habitat (e.g., emergent marsh). This is because there is a strong linkage between dominant life history strategies of ocean-type ESUs and shallow water habitat. A large amount of that habitat type has been lost due to diking. Clearly, restoration of some shallow water habitat can be done without changing hydropower operations.

While more shallow water habitat could be made available with flow changes, this would have to be very carefully considered because it could have other, unintended consequences such as an increase in gas bubble disease. We must also recognize that any questions about how much restoration is needed must address the question of how much change is possible. There are now constraints on the system such as climate change and "permanent" changes in the landscape such as from urbanization that clearly constrain how much change is possible; a return to presettlement conditions is unreasonable to expect. We expect that studies now underway will provide more insight into how much change in shallow water habitat is both possible and needed.

Our analyses did not attempt to compare the ability of estuary factors and nonestuary factors to affect viability. The analysis by Kareiva et al. (2000) and more recent analyses suggest that the estuary and plume environment are generally important to the productivity of anadromous populations. In general, recovery is a life cycle process that requires strategies that focus on crafting and evaluating alternative scenarios involving all life stages of the animals.

Thus addressing estuarine factors can potentially improve population status and help recovery and should be included in any comprehensive plans for recovering populations in the basin. A variety of factors (such as landscape connectivity within and between habitat zones) will need to be considered in deciding how to distribute recovery efforts directed at any suite of limiting factors. Any recovery actions directed at shallow water habitats will also need to consider the cumulative effects of all factors affecting ocean-type populations in these habitats. For example, some recovery actions directed at habitat restoration in shallow water areas may need to simultaneously reduce toxic contamination if sites targeted for restoration are found to be contaminated.

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