APPENDIX B. ESTUARINE HABITAT EVALUATION

I. INTRODUCTION

Since 1991, 12 different ESU's (Evolutionarily Significant Units) 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 (McClure et al. 2003). In recent years, there has been increasing emphasis on the role of the estuary and plume in the decline and recovery of these salmonids. The estuary and plume are the connection between 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). Including the estuary as an element in salmon recovery represents a significant departure from previous management efforts in the system and recognizes that effects of hydroelectric development and other upriver alternations are not localized to "between the dams". Instead these effects can be far reaching, directly affecting estuarine and coastal ocean habitats (e.g., availability of essential habitat in the estuary, size of the plume) of salmon.

Here, we evaluate the effects of a number of factors associated with the Columbia River estuary and plume on the viability of listed, anadromous EUS's in the Columbia River Basin. We only consider those ESU's spawning above Bonneville Dam and lower river chum salmon. Our analysis is aimed primarily at addressing the issue of whether or not there is potential to improve anadromous salmon population status through improvement in conditions in the estuarine and plume environments. It was conducted in support of analyses of the potential to improve anadromous salmon population status through improvement in conditions in tributary environments. Each factor is also considered from the perspective of whether or not its effects on listed populations are directly related to the operation of the Federal Columbia River Hydropower System to help elucidate changes that can be made in the estuary to improve salmon performance beyond those directly related to hydropower operations.

II. THE COLUMBIA RIVER ESTUARY AND PLUME

We defined the Columbia River estuary to encompass the entire habitat continuum (ecotone) upstream of the river mouth to Bonneville Dam where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. The estuary can be divided into different zones based upon various attributes such as geomorphic features, ecological functions, and physical characteristics and each zone can be further subdivided into different habitat types and features (Figure 1). For purposes of this analysis, we divided the estuary into three zones. The first extends from approximately RM-45 to RM-145 and is a long tidal-freshwater zone (referred to as the tidal river zone) where the river is constrained to a simple deep channel and there is only narrow fringe of intertidal habitat. Second, between Tongue Point (RM-18) and upper Puget Island (RM-45) there is a large estuarine mixing zone (referred to as the estuarine mixing zone) where mean salinities range from 0-15 parts per thousand (in deep channels only). Third, from

Tongue Point to the river mouth is a high-energy zone from the river mouth to Tongue Point where the salinity gradient increases to more than 30 parts per thousand at the river entrance (referred to as the lower estuary). A major feature of this zone is the pair of shallow, peripheral bays (Baker Bay and Young's Bay) with expansive intertidal flats that occur along either side of the lower estuary.

Within each of these three zones is a mix of habitats that the juvenile salmon can potentially occupy (Bottom et al. 2001). Habitats can be classified based upon site scale (e.g. depth, temperature, vegetation type, and substrate type) and landscape scale (e.g., connectivity, shape, and size) attributes. The functions of these habitats for juvenile salmon and steelhead depend upon how these attributes, in aggregate, affect the accessibility of the habitat to the fish and its quality (Simenstad and Cordell 2000).

Beyond the semi-enclosed estuary is the Columbia River plume, that salmon must occupy before they are fully entrained in oceanic habitats (Figure 1). The river plume is generally defined by a reduced-salinity contour near the ocean surface of 31 parts per thousand. Its geographic position varies greatly with seasonal changes in river discharge, prevailing nearshore winds and ocean currents. Strong density gradients between ocean and plume waters create relatively stable habitat features where organic matter and organisms are concentrated.

III. APPROACH

A. Defining Life history Type and Life History Strategy

Our overall purpose here is to evaluate and rank selected factors in the estuary and plume with respect to their potential to improve viability of listed populations. Ideally, we would like to link factors in the estuary to their potential to affect the viability of each listed population. However, because we do not have specific, empirical information describing estuarine habitat use by anadromous populations in the Columbia River estuary and plume, we used an alternate approach where effects of candidate factors were linked to viability of an ESU. As each ESU is comprised of a bundle of populations, we can then infer responses of populations based upon what we predict will occur for the ESU.

We first defined each ESU as either stream type of ocean type based upon characteristics of the juvenile outmigrants. While each life history type can potentially produce any life history strategy, ocean type populations are generally (but not exclusively) composed of individuals that migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. Stream type fish generally migrate to sea after rearing for at least a year in freshwater. Thus, ocean type fish tend to spend longer periods in ocean habitats compared to stream type populations. Information used to define life history types came primarily from the species status reviews: chinook salmon (Myers et al. 1998), chum salmon (Johnson et al. 1997), and sockeye and steelhead (Busby et al. 1996). We assumed that all populations in aggregate within an ESU fit a general model of that life history type.

Each life history type is comprised of individual members who employ a variety of alternative spatial and temporal strategies or approaches to using available habitat. We defined a life history strategy as an approach to using available habitats, including the estuary. To define alternate strategies for estuarine use, we used the size at estuarine entry and the time when they arrive in the estuary as defining criteria that are linked back to ESU because numerous studies suggest there is a strong linkage between fish size, habitat use, and residence time (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Carl and Healey 1984; Levings et al. 1986; Bottom et al. 2001; Miller and Sadro 2003). Juvenile salmon are generally distributed along a habitat continuum based upon water depth with the depth of the water occupied by the fish increasing as the size of the fish increases (McCabe 1995).

Based upon patterns of size and time of estuarine entry, we identified six life history strategies based upon historic use (Table 1): 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Fry are defined as fish that enter the estuary at a size < 60 mm with early fry entering in approximately March and April and late fry from May to June. Fingerlings are those fish that enter the estuary at a larger size than fry, which implies there was some period of freshwater rearing, but have yet to begin the physiological transition associated with smolting. Subyearlings rear primarily in freshwater with relatively little time spent in the estuary, and smolt as they outmigrate during their first year of life. Yearlings rear for at least one year in freshwater and then emigrate; these fish generally spend less time in the estuary than fry, fingerlings, or subyearlings. Although some differences between populations within an ESU in the relative proportions life history strategies can be expected but we could not discriminate such differences. Therefore, we assumed that all populations within a life history type/ESU produce a characteristic mix of these strategies when viewed over long time scales.

B. Factors Included in the Analyses

The major estuarine related factors that we believe can potentially affect salmonid population viability include climate and climate change (which control other factors), water flow, access to and quality of habitats, sediment, salinity, temperature, toxics, predators (e.g. terns, cormorants, northern pikeminnow), and hatchery and harvest practices. Although it would be useful to evaluate the role of each of these factors, we limited our analyses to a subset of these nine factors. From the list of nine factors, we selected factors where: 1) a significant change was evident, 2) the factor could potentially affect population viability, and 3) there was quantitative data available that could be used to analyze the effect of the factor within the time we had been allotted. The factors that satisfied these criteria and were included in this analysis are water flow, availability of salmon habitats, toxics, and predation (primarily Caspian terns). For each of these factors we provide a brief analysis as to how this factor could affect population viability. From these analyses, we developed a series of hypotheses or principles about each factor that helped guide how we rated their relative importance for each ESU.

C. Analyzing and Rating the Relative Importance of Limiting 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 drew inferences about how a factor affects an ESU based upon the life history type of that ESU and how we believed the factor would affect the life history strategies that characterized that life history type. Thus, 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 with the context of other non-estuarine factors such as tributary habitat. This is considered elsewhere in this report.

We defined improvement in population status to mean improvement in population viability (McElhaney et al. 2000) as defined by the four VSP performance criteria: abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000). The rating system consisted of two levels. The level 1 screens evaluated ***if*** the factor was likely a concern for an ESU based upon its effects on VSP and change in the factor from historic conditions. The level 2 screens asked ***how*** the factor affected an ESU based upon where the effects occurred.

1. LEVEL 1- What is the effect on each VSP parameter? Clearly, each factor will have some effect on all VSP parameters. We assumed if the factor affected large numbers of individuals in the ESU (again relative to other factors) that there was a significant effect on abundance and productivity. Because most populations in threatened or endangered status are at low levels of abundance, we doubled the score of any factor that affected abundance or productivity. We reasoned that these depressed populations needed short term increases in abundance before long term benefits resulting from increased diversity and structure would be useful. If a factor affected particular life history types or affected specific habitat types more than others, we assumed there was an impact on spatial structure and diversity.

2. LEVEL 1- Has the factor changed from historic conditions and could it be improved relative to the other factors? We considered whether each factor had changed significantly from historic conditions. Because we intentionally selected factors that we believed had changed significantly from historic conditions, this screen did not result in much difference between factors. We also considered from a practical perspective how much change in each factor was possible. A factor could be significantly changed from historic levels but relatively difficult to change relative to other factors.

3. LEVEL 2- Does the factor have a significant effect 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 estuarine shallow water, estuarine deep water and plume habitats. Although there are multiple estuarine zones and habitat types within each zone, knowledge of how different juvenile life history strategies specifically use these habitats and zones is largely absent. Moreover, our present knowledge base is not robust enough to acknowledge differential effects of limiting factors on either habitat within a zone or between zones or to be extensively discriminatory. However, the linkage between life history strategy and use of deep versus shallow water is pronounced. Thus, we collapsed the estuary from Bonneville to the mouth into one zone and the plume as a second major zone. Within the estuary, shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats) were distinguished from medium and deep, higher velocity channel habitats in the analysis because there is strong evidence that habitat use varies between these habitat types; the plume was considered as one habitat unit.

4. LEVEL 2- 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 effect the factor had in estuarine shallow water, estuarine deep water and plume habitats. We considered effects of the factor on habitat quantity, quality, and opportunity. The concepts of opportunity and quality (or capacity) metrics were proposed by Simenstad and Cordell (2000) and adopted by Bottom et al. (2001) for the Columbia River estuary. Opportunity attributes relate to the accessibility of habitat to juvenile salmon and in general, opportunity metrics are largely physical and chemical in nature such as tidal elevation and location of habitat. In general, 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 deep water estuarine habitat for water borne 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 for each ESU based upon whether they were an ocean or stream life history type. Scoring was done using guidance from the principles/hypotheses developed in the following discussion of limiting factors. Each cell in the 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 water borne and sediment associated toxics. Thus, for flow, habitat and predation, the maximum possible score was 20 whereas the maximum possible toxic score was 28. The final rating was computed as the ratio between the assigned score and maximum possible.

IV. LIMITING FACTORS

A. Flow

Water, interacting with the land, forms the habitat that juvenile salmon occupy. The estuarine habitat features to which salmon have adapted are largely the result of riverine and tidal processes. However, the shaping of estuarine habitats is also controlled by several “external” factors which help establish the physical template for the entire estuary. First, characteristics of the watershed affect such factors as the amount and timing of water arriving in the estuary. For example, because most of the western sub-basin is at too low an elevation to accumulate a large seasonal snow pack, the highest flows are observed in this region during and shortly after winter storms between December and March. In contrast, most of the flow in the interior sub-basin occurs as the result of melting of a seasonal snow pack between April and June.

Second, natural variations in Columbia River flows associated with both short and long term fluctuations in climate have a significant effect on amount of water delivered to the estuary, the connection between the freshwater and marine environments, and which areas are wetted and potentially accessible to juvenile salmon. Because of the vast extent of the Columbia River Basin, the effects of climate vary considerably depending upon location within the basin. Climate-induced variations in Columbia River flow occur on time scales from months to centuries (Chatters and Hoover 1986, 1992). One example of this is the Pacific Decadel Oscillation, commonly known as the PDO (Mantua et al.1997), which alternates between cold and warm phases at approximately 30-year time scales. The cold phases of the PDO (e.g., the 1945-1976 period) are generally considered to benefit salmonid production in the Pacific Northwest. Another climate related feature known to influence weather and conditions in the Pacific Northwest is the phenomena associated with the El Niño-Southern Oscillation (ENSO; typically 3-7 years in duration) index (Redmond and Koch 1991; Kathya and Dracup 1993; Dracup and Kathya 1994; Gershunov et al. 1999; Jay 2001).

Changes in flow attributes, such as when and how much water arrives in the estuary, are an integral measure of changes in a river system. In a recent analysis and review, Jay (as reported in Bottom et al 2001) concluded that there has been approximately a 16.4% reduction in flow over the last approximately 100 years. Seasonal changes, particularly those in spring freshet timing and magnitude, have been much greater than changes in annual average flow. Spring freshets are extremely important for juvenile salmonids in that high flows (especially overbank flows) provide habitat, limit predation by increasing turbidity, maintain favorable water temperatures, and supply organic matter to the detritus-based food web, centered in the estuarine turbidity maximum (ETM). Jay (2001) found that with respect to the phase of the annual flow fluctuation, the timing of the freshet flows is now about a month earlier than historically (Figure 2). And, the maximum daily spring freshet flow is now about two weeks earlier that historically (Water Year Day 242 vs 256). In addition, monthly Columbia River virgin flows at The Dalles were 11,480 m^3s^{-1} (for May), 16,760 m^3s^{-1} (for June), and 12,600 m^3s^{-1} (for July) during 1879-1899. The corresponding figures for 1945-1999 were 13,300 m^3s^{-1}, 15,840 m^3s^{-1}, and 9,420 m^3s^{-1}; these values represent changes of +15.9, -9.5, and -25.2%, respectively.

Flow regulation is clearly the source of the largest reduction in spring flow, with climate change having little effect (Jay 2001). Most of the loss of freshet flow represents flow that now occurs during winter, early spring, or late summer and fall. 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). Finally, the estimated freshet season flow decrease due to flow regulation was overall 33.1% (a reduction of 31.6% for May, 32.4% for June, and 19.8% for July, respectively).

Another feature of water flow that is significant to juvenile salmon is the occurrence of overbank flows. Historical bankfull levels exceeding 18,000 m^3s^{-1} now rarely occur due to effects of flood control measures and irrigation depletion (Jay 2001) while some overbank flow occurred in many years before 1900, both in winter and in spring (Figure 3). The season when overbank flow typically occurs has also shifted from spring to winter (Jay 2001). Flood protection, diking, flow regulation, and water withdrawal largely eliminated climate influence on overbank flow regardless of the PDO phase (Jay 2001).

The effect of flow changes can also be seen in the Columbia River plume. For example, Pearcy (1992) hypothesized that one function of the plume was to distribute juvenile salmon offshore, away from predation pressure closer to the shoreline. In May and June when flows are higher, juveniles are found further offshore, in the low saline waters they appear to prefer, than when flows are lower. During the years when less flow out of the Columbia River is evident during the freshet period, salmon are more localized around the mouth of the Columbia River. The higher turbidity associated with the low salinity plume waters is considered to provide refugia from predators. 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 now considered surrogate physical attributes defining habitat important to salmon.

Evaluating the impact of water flow on habitat is a fundamental part of putting flow changes in the basin into perspective. Baptista (2001) using a hydrologic model developed specifically for the Columbia River found the estuary during the historic period (late 1800s) was able to sustain habitat features defined to be important to salmon (characterized as water velocities less than 30 cm/sec-- important to smaller juvenile salmon) to a greater degree in the face of ever increasing water flows than is evident now. In the tidal freshwater zone of the Columbia River estuarine system (RM 50 to 90), Kukulka and Jay (2003) demonstrated that there was approximately a 62% loss of shallow water habitat (defined by depth between 10cm and 2 m) that was attributable to diking (physically removing access of water to the tidal floodplains) and the reduction of peak flows (a 40% reduction) (Figure 4). Diking and flow reductions have reduced shallow water habitat in the freshwater tidally influenced region of the Columbia River estuary by 52% and 29%, respectively.

The hydrological changes described above, particularly those associated with flow regulation, water withdrawals, and floodplain diking (discussed in the next section), represent a fundamental shift in the physical state of the Columbia River ecosystem. Reductions of the spring freshet can result in a greater uniformity of migration patterns with potential consequences in the timing and sizes of salmon arrival in the estuary and/or ocean. Because the changes in habitat are most pronounced in shallow water areas, effects on the ESU's and life history strategies (the fry and fingerling strategies) that use these shallow water areas is likely most significant. The nearly complete elimination of overbank flooding throughout the expansive tidal freshwater portion of the estuary has almost completely eliminated access to off-channel floodplain habitats during high flow events. The loss of floodplain inundation greatly decreases the surface area of tidal estuarine and riverine habitats available to salmonids. Because fish are prevented from expanding their distribution into productive off-channel areas, competitive interactions may be more severe because fish densities remain high (e.g., Sommer et al. 2001). Flow changes (and diking) also influences the productive capacity of the estuary by regulating so-called "energetic processes" such as fish distribution and density, food production (especially detrius), woody debris recruitment, competition, and predation.

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 effects 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, and changes in 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 ecosystem. Such changes potentially have significant consequences for both expression of salmonid diversity and productivity of the populations by affecting quality of habitat available, its accessibility and quantity. In particular, because the changes in habitat are most pronounced in shallow water areas, effects on the ESU's and life history strategies (the fry and fingerling strategies) that use these and depend upon these shallow water areas is most significant.

B. Habitat

The estuary contains an extensive and diverse array of habitats that are shaped by the interactions of flow and tides with the land. Although quantitative descriptions of habitat attributes important to salmon are limited in the Columbia River, research in estuarine systems throughout the Pacific Northwest has demonstrated that fish size is one of the major factors defining use of estuarine habitats (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). As the size of salmon increases (due either to recruitment of larger individuals or growth in the estuary), salmon shift to deeper habitats. Thus, yearling life history strategies use deeper water habitats within the estuary, located more centrally to mainstem channels whereas smaller, subyearling (all none yearlings) juvenile salmon use the more peripheral side channel areas associated with the more shallow water habitats (McCabe 1986). 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.

Although the abundance of juveniles in the estuary fluctuates, evidence indicates that juvenile salmon currently use the estuary during the entire year (D. Bottom, NOAA

Fisheries, personal communication). This characteristic year long presence is consistent with the historical record. Burke (2001) reconstructed the presence of juveniles from research conducted by Willis Rich in the early 1900s. It is apparent that over the year, juvenile salmon representing different cohorts expressing varying life history strategies were historically using the Columbia River estuary.

The smaller, unmarked chinook salmon are associated with side channel, peripheral tidal marsh and forested marsh habitats and are likely naturally produced wild salmon. The larger chinook salmon, many of which are hatchery produced fish, dominate the deeper, mainstem channel habitats (Figure 5). Salmon occupying shallow water habitats express the range of strategies that are characteristic of most ocean-type life history types. However, it is now evident that salmon representing most of the endangered ESUs are using the peripheral habitats of the Columbia River estuary based upon recent genetic analysis. Both ocean and stream type chinook from upper and lower basin sources were found in these marsh and forested wetland habitats (Figure 6, Paul Moran, NOAA Fisheries, pers. comm.). In addition, spring chinook that express both yearling and subyearling strategies have been identified in the plume environment (Figure 7).

The major anthropogenic factors affecting the amount and location of estuarine habitat are flow alterations and diking. Dikes are built to prevent over-bank flow and are built for purposes of flood control and conversion of aquatic to terrestrial land (e.g. for farming). The construction of dikes is not a direct result of the operations of the hydropower system, although dikes must be built to accommodate the timing and magnitude of flows that pass below Bonneville Dam. Because dikes affect the connectivity of the river and floodplain (Tetra Tech 1996), the diked floodplain is higher than the historic floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). Given modern bathymetry and the altered flow regime scenario that we described in the previous section, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high. Because the frequency of occurrence of this river discharge is rare, floodplain inundation is uncommon and availability of shallow water habitats is limited (Kukulka and Jay 2003).

Several analyses demonstrate the dramatic changes in the amount and location of shallow water habitat (such as emergent marsh and forested wetland habitat) that have occurred (Thomas 1983; Sherwood et al. 1990) that significantly reduce the estuary's opportunity and capacity to support juvenile salmon. 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. Thomas (1983) and Sherwood et al. (1990) calculated that approximately 121.6 km^2 of tidal marshes (77% decline) and swamps (62% decline) that existed prior to 1870 have been lost (Figure 8). Together with a 12% loss of deep-water habitat, these changes reduced the estuary's tidal prism from 12 to 20%. In addition, the historic surface area of the estuary has decreased by approximately 20% as a result of diking or filling of tidal marshes and swamps. The largest increase of non-estuarine 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).

Tidal swamp is the most impacted habitat type. Almost all the tidal swamp habitat present in 1870 was converted to diked floodplain/non-tidal habitat. The location of tidal marsh habitat within the estuary has changed as a result of modified flow regime, modified tidal action, and/or shipping channel development and maintenance. High elevation tidal marshes have been diked more heavily impacted than lower elevation marshes. For example, almost all of the tidal swamp habitat present in Youngs Bay and Baker Bay in 1870 has been lost. In Grays Bay and Cathlamet Bay, the loss of tidal swamp habitat has been 88.4% and 48.9%, respectively, compared to historic acreage.

The loss of estuarine wetlands has clearly affected the opportunity of salmon to use this type of habitat. Emerging research in the Columbia River and else where demonstrates that these shallow vegetated habitats are important to non-yearling life history strategies, especially fry and fingerlings (D. Bottom, NWFSC, personal communication, Shrefler et al. 1990, 1992; Gray et al. 2002). The degree to which estuary habitat types have been affected by diking is directly proportional to elevation; thus, the highest elevation habitat type (i.e. tidal swamp) has been impacted by diking the most (Thomas 1983).

At the same time that swamps and wetlands have been loss, the total area of non-vegetated estuarine shallows and flats increased 7% between 1870 and 1980. This was independently substantiated by Sherwood et al. (1990), who estimated 68.4 x 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.

In addition to the lost opportunity to use shallow water habitats, estuarine wetland loss has altered the magnitude and character of habitat capacity by causing a decline in wetland primary production. Approximately 15,800 mt carbon $year^{-1}$ (84%) of macrodetritus that historically supported estuarine food webs has been eliminated. 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 implications of this shift in detrital sources are unclear. For example, whereas the macrodetrital food web was historically distributed throughout the lower river and estuary, the contemporary microdetrital food web is concentrated within the localized mid-estuary region of the estuarine turbidity maximum (ETM).

In summary, the location and types of habitats present in the Columbia River Estuary have been substantially changed from historic conditions. Although the entire estuary has not yet been surveyed, the main changes that have been quantified in the estuary have been 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 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 populations that have large numbers of shallow water dependent members.

C. Toxics

Concentrations of toxic contaminants in the Columbia Estuary were historically low. However, beginning in the early 1800's, activities such as agriculture, logging, mining, industrial discharges, and stormwater runoff began to degrade water quality in the Columbia Estuary. Currently, the section from Bonneville Dam to the estuary mouth is the most urbanized section of the river, receiving contaminants from over 100 point sources (Fuhrer et al. 1996), as well as urban and agricultural non-point sources. Contaminants may also be transported to estuary from areas of above Bonneville Dam such as the Yakima River (Fuhrer et al. 1996; Rinella et al. 2000), Lake Roosevelt (Bortleson et al. 1994) and other tributaries (Fuhrer 1989; Roy F. Weston Inc. 1998).

Potentially toxic water-soluble contaminants that have been detected in the Lower Columbia Estuary include a wide range of current-use organophosphate pesticides (OPs; e.g., simazine, atrazine, chlorpyrifos, metolachlor, diazinon, and carbaryl) and trace metals (Fuhrer et al. 1996; Hooper et al. 1997). Contaminants that have been documented in Lower Columbia bed sediments and suspended 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), and polycylic aromatic hydrocarbons (PAHs) (Fuhrer and Rinella 1983; Fuhrer 1986; Harrison et al. 1995; Fuhrer et al. 1996; Tetra Tech Inc 1996; US Army Corps of Engineers 1998; Roy F. Weston, Inc 1999; McCarthy and Gale 2001).

Exposure to these contaminants in the estuary likely varies by life history type and ESU. Stream type populations (e.g, Snake River sockeye), are not likely to accumulate high body burdens of bioaccumulative, sediment-associated contaminants such as PCBs and DDTs. However, they may be affected by short-term exposure to waterborne contaminants such as OPs and dissolved metals. Ocean-type populations (e.g., Lower Columbia River chum), 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 and tendency to rear for longer periods in the estuary. Fine-grained sediments to which toxics adsorb are most likely to be deposited in areas with slower water velocities, including backwater areas in side channels and along the river's margins, so these are areas where elevated concentrations of toxic contaminants are considered most likely to be found (Tetra-Tech 1994).

Although data on contaminant concentrations in listed salmon from the Lower Columbia are limited, available data indicate that bioaccumulative contaminants are present in prey and tissues of juvenile salmon from the Columbia Estuary. Contaminant concentrations were measured in juvenile fall Chinook salmon from several sites in the Columbia Estuary such as near the confluence of the Columbia and Willamette Rivers, near Longview, and at several sites within the Lower Columbia Estuary such as White Island and West Sand Island). The primary contaminants found in whole body samples of Chinook salmon from the lower Columbia were PCBs and DDTs. 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 contaminants were also detected in stomach contents of juvenile fall Chinook salmon from sites within the estuary, indicating they were absorbing some contamination from prey during estuarine residence.

For some contaminants, exposure levels in juvenile salmon from the Lower Columbia are approaching concentrations that could affect their health and survival. For PCBs, Meador et al. (2002) estimated a critical body residue of 2400 ng/g lipid for protection against 95% of effects ranging from enzyme induction to mortality in a fish with 2% lipid, based on a range of sublethal effects observed in salmonids in peer-reviewed studies conducted by NMFS and other researchers. Mean PCB body burdens in juvenile salmon analyzed by the NWFSC were at or above these thresholds at several sites in the Lower Columbia. Of individual fish analyzed from sites within the estuary, ~35% were above the effects threshold. Moreover, in field studies in Puget Sound, at estuarine sites contaminated with PAHs, PCBs, and other OCs also present in the Lower Columbia, juvenile salmon showed immunosuppression, reduced disease resistance, and reduced growth rates, (Arkoosh et al. 1991, 1994, 1998; Varanasi et al. 1993; Casillas et al. 1995a,b, 1998a). Similar results were observed in growth and disease challenge studies with juvenile salmon exposed in the laboratory to PCBs and PAHs (Arkoosh et al. 1994; 1998; 2000; Casillas et al. 1995a,b; 1998).

The likely impact of DDTs on listed salmon is less clear. Most reported effects of are associated with whole body tissue concentrations above those typically found in Lower Columbia Estuary salmon (≥500 ng/g wet wt) (Allison et al. 1962; Burdick et al. 1964; Buhler et al. 1969; Johnson and Pecor 1969; Peterson 1973, Poels et al. 1980; Hose et al. 1989). However, they may affect salmonid prey (Pavlou et al. 1987; Long et al. 1995), and studies in the Columbia Estuary have shown that they also represent a hazard to fish-eating predators through bioaccumulation and bioconcentration (Anthony et al. 1993; U.S. Fish and Wildlife Service 1999; Henny et al. 2003; Thomas and Anthony 1999, 2003).

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, as well as copper at concentrations of 3-6 ug/L, can disrupt olfactory function in salmon after exposures of as little as 30 minutes (Moore and Waring 1996; Waring and Moore 1997; Scholz et al. 2000; Baldwin et al. 2003). In these studies, affected fish could no longer respond normally to test odorants, so predator avoidance, feeding responses, homing, pheromone-triggered sexual behavior were impaired (Moore and Waring 1996; Waring and Moore 1997; Scholz et al. 2000). Concentrations of diazanon in the 1-10 ug/L range have been reported in NASQAN sampling in the Lower Columbia, and other OPs with similar modes of action (e.g., chlorpyriphos, malathion, aldicarb, carbaryl, carbofuran) are detected even more frequently and at higher concentrations. Dissolved copper concentrations at the Lower Columbia sites sampled in the USGS NAQAN survey were within this range (Fuhrer et al. 1996), and copper in suspended sediments was substantially higher (45-120 ug/L).

Available data show that environmental concentrations and tissue burdens of several classes of contaminants are within the range where they could potentially affect two important VSP parameters, abundance and population growth rate, in listed stocks. The true magnitude of the effect is uncertain, but a recent modeling study suggests it could be significant for at least some ESUs. Spromberg and Meador (2004) 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. The results of indicate that after 20 years of continued reductions at the 10% level, population abundance was severely depressed (up to 2 - 3 times lower than non impacted 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 and stream-type stocks in the estuary. Stream-type ESUs are most likely to be affected most 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, and 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 PCBs and DDTs, which they may absorb during their more extended estuarine residence. Consequently, the impact on ESUs exhibiting the ocean life history type may be higher.

D. Caspian Tern Predation of Juvenile Salmon

The potential for changes in predation on juvenile salmon throughout the Columbia River Basin are significant due to habitat changes and introductions of exotic specie. As an example of this type of change in predator-prey interaction, we consider here Caspian Tern (*Sterna caspia*) predation in the Columbia River estuary. 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 staff. Several 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 1000 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. The Caspian tern colonies in the estuary have since expanded to 9,000-10,000 pairs, the largest ever reported. In 1999, the colony was encouraged to relocate to East Sand Island.

Caspian terns arrive in the Columbia River estuary in April and begin nesting at the end of the month (Roby et al. 1998). 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). Terns are piscivorous (Harrison 1984), requiring about 220 grams (roughly one-third of their body weight) of fish per day during the nesting season. Diet analyses indicated that juvenile salmonids constituted 77.1% of prey items in 1997 and 72.7% of prey items of Caspian terns nesting on Rice Island (Collis et al. 2002). During May when large numbers of salmon are moving to sea, the diet of Caspian terns was consistently over 90% juvenile salmonids (Collis et al. 2002).

Two approaches to evaluate the impact of Caspian tern predation on juvenile salmon were conducted by Good et al. (2003). One approach using bioenergetics modeling, estimated that smolt consumption from 1999 to 2002 ranged from 5.9 to 11.7 million. A second approach used detections of passive integrated transponders (PIT) tags on Caspian tern colonies to estimate salmonid predation rates overall as well as by ESU (Collis et al. 2001a, b; Ryan et al. 2001). 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 if 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. Although the relationship between tern abundance and predation rate is not known with certainty, the estimates (using either bioenergetics modeling or PIT tag data) showed a linear response of predation rate on all salmon to the number of Caspian terns nesting on East Sand Island during the breeding seasons of 1999-2002. The per capita consumption rate in 1999 (mean = 437.5) was equivalent to that of 2000 (mean = 431.1), even though there was an almost five-fold difference in colony size.

Using the CRI model (e.g., Kaervia et al. 2001), 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 ultimately be derived.

The predation rate for 20,000 Caspian terns on all steelhead and spring Chinook salmon was estimated using the regression equations generated using 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 (population growth rate), assuming all steelhead or spring Chinook salmon mortality 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. Predation rates for 20,000 Caspian terns on four of the five ESA-listed steelhead and spring Chinook salmon ESUs were also 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.

When interpreting the results of these calculations, it is important to note that there is no compensatory mortality assumed to occur 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. Thus, 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 also 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 effects primarily salmon and steelhead that exhibit a stream type life history rather than an ocean type life history as they move and utilize the Columbia River estuary. This is primarily because salmon from this life history type move in great numbers at a time when 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, 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, Caspian tern predation has significantly increased due to a recent change in nesting habits of the birds. The main impact of tern predation is on ESU's with stream type life history types, especially steelhead. This is a result of the dominant migratory periods employed by salmonids with a stream type life history. Improvements to λ by managing terns would be expected to benefit these ESUs especially, although benefits to other salmon ESUs in the basin should be evident, albeit to a much lesser degree.

V. IMPACT OF FACTORS ON RELEVANT ESU'S AND POTENTIAL FOR IMPROVEMENT IN ESU CONDITION

A summary of scoring for Level 1 and Level 2 for each life history type/ESU are provided in Table 2 with detailed scoring provided in Tables 3-6. To help guide our scoring, we used the following hypotheses or assumptions about the effects of specific limiting factors which were developed in our analyses of each factor. Cumulative impacts were not considered in the analysis.

1. Tern predation differentially affects the larger yearling strategies, especially steelhead, more than smaller fish such as fingerling chinook (Ryan et al. 2003). Tern predation is assumed to be distributed in the estuary zone but primarily in deep water habitat.
2. The main effect of flow reductions is to affect amount of shallow water habitat available to fish; the main effect of habitat changes is on distribution, quantity and quality of habitat; the main effect of toxics is on habitat quality (capacity).
3. Any reduction in quality or quantity of shallow water habitat affects smaller juvenile salmonids employing strategies such as fry and fingerlings significantly more than subyearlings and yearlings.
4. Subyearling and yearlings primarily use medium and deep channel habitat.
5. Fry and early fingerling life history strategies do not move into the plume, but more likely utilize the surf zone when they exit the estuary proper.
6. Reductions in flow above Bonneville affect the size and shape of the plume.
7. 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 impact occurs.
8. Flow and habitat changes in the estuary are interrelated.

Operation of the Hydropower System affects two of the factors we considered: flow and habitat; we did not consider there to be a direct relationship between operation of the hydropower system and either toxics or tern predation. Changes in flow can permanently eliminate some habitat from use by estuarine dependent 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, the value of some habitat is reduced because it becomes accessible only for a limited time because of the reduction in flow. The non-hydro portion of habitat change involves the reduction in the amount of shallow water habitat due to dikes and levees that permanently isolate this habitat from use.

For stream type ESU's, the primary factors affecting population viability are tern predation and flow (Tables 3,5). Tern predation was ranked in the medium category primarily because abundance of the main life history strategy is affected and there are significant effects upon abundance and productivity. Flow changes were also ranked medium because of effects on the main life history strategies in plume habitat. Toxics and habitat were ranked low for stream type ESUs because the main life history strategies associated with this ESU do not occupy the habitat where the effect occurs.

For ocean type ESU's (only Lower Columbia River chum was included), flow and habitat were rated as having a high ability to affect population viability (Tables 4,6). The dominant life history strategy of ocean type chinook salmon use shallow water habitat which is where the main flow and habitat changes occur. Tern predation has a low effect on this ESU because tern predation does not target fry and fingerling strategies (the dominant ones associated with this ESU). Toxics was scored as a medium factor because both water borne and sediment contaminants can affect these life history strategies in shallow water areas.

In summary, (need to consider both toxics and shallow water habitat).

VI. REFERENCES

Allison, D., B.J. Kallman, O.B. Cope, and C. Van Valin. 1963. Insecticides: Effects on cutthroat trout of repeated exposure to DDT. Science 142:958-961.

Anthony

Arkoosh, M. R, E. Casillas, P. Huffman, E. Clemons, J. Evered, J.E. Stein, and U. Varanasi. 1998. Increased susceptibility of juvenile chinook *salmon (Oncorhynchus tshawytscha)* from a contaminated estuary to the pathogen *Vibrio anguillarum*. Transactions of the American Fisheries Society. 127:360-374.

Arkoosh, M. R., E. Casillas, E. Clemons, B. McCain, and U. Varanasi. 1991. Suppression of immunological memory in juvenile chinook salmon (*Oncorhynchus tshawytscha*) from an urban estuary. Fish & Shellfish Immunology 1:261-277.

Arkoosh, M. R., E. Casillas, E. Clemons, P. Huffman, A. N. Kagley, N. Adams, H.R. Sanborn, T. K. Collier, and J. E. Stein. 2001. Increased susceptibility of juvenile chinook salmon (*Oncorhynchus tshawytscha*) to vibriosis after exposure to chlorinated and aromatic compounds found in contaminated urban estuaries. Journal of Aquatic Animal Health 13:257-268.

Arkoosh, M. R., E. Clemons, M. Myers, and E. Casillas. 1994. Suppression of B-cell mediated immunity in juvenile chinook *salmon (Oncorhynchus tshawytscha*) after exposure to either a polycyclic aromatic hydrocarbon or to polychlorinated biphenyls. Immunopharmacology and Immunotoxicology 16(2):293-314.

Baldwin, D. H., J. F. Sandhal, J. S. Labenia, and N. L. Scholz. 2003. Sublethal effects of copper on coho salmon: Impacts on non-overlapping receptor pathways in the peripheral olfactory nervous system, Environ. Toxicol. Chem 22:2266-2274.

Baptista 2001. Estuarine habitat opportunity. *In* Bottom, D.L., C.A. Simenstad, A.M. Baptista, D.A. Jay, J. Burke, K.K. Jones, E. Casillas, and M.H. Schiewe. 2001. Salmon at river's end: the role of the estuary in the decline and recovery of Columbia River salmon. NMFS, Seattle.

Bonn, BA. 1998. Dioxins and furans in bed sediment and fish tissue of the Willamette Basin, Oregon, 1992-1995. US. Geological Survey Water Resources Investigations Report 97-4082. 12 pp.

Bortleson, GC, Cox SE, Munn MD, Schumaker RJ, Block EK, Bucy LR, and Cornelis SB. 1994. Sediment quality assessment of Franklin D. Roosevelt Lake and the upstream reach of the Columbia River, Washington, 1992. US Geological Survey Open-File report 94-315. 130 pp.

Bottom, D. L., and K. K. Jones. 1990. Species composition, distribution and invertebrate prey of fish assemblages in the Columbia River estuary. Progress in Oceanography 25:243-270.

Bottom, D.L., C. A. Simenstad, A. M. Baptista, D. A. Jay, J. Burke, K.K. Jones, E. Casillas, and M. H. Schiewe. 2001. Salmon at River's end: The role of the estuary in the decline and recovery of Columbia River Salmon. Draft Report, National Marine Fisheries Service.

Buhler, D.R., M.E. Rasmusson, and W.E. Shanks. 1969. Chronic oral DDT toxicity in juvenile coho and chinook salmon. Toxicol. Appl. Pharmacol. 14:535-555.

Burdick, G.E., E.J. Harris, H.J. Dean, T.M. Walker, J. Skea, and D. Colby. 1964. The accumulation of DDT in lake trout and the effect on reproduction. Transactions of the American Fisheries Society 93:127-136.

Burke 2001

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, and California. U.S. Dep. Commerce., NOAA Tech. Memo. NMFS- NWFSC-27, 261 p.

Carl, C.M., and M.C. Healey. 1984. Differences in enzyme frequency and body morphology among three juvenile life history types of chinook salmon (Oncorhynchus tshawytscha) in the Nanaimo River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 41:1070-1077.

Casillas E., M.R. Arkoosh, E. Clemons, T. Hom, D. Misitano, T.K. Collier, J.E. Stein, and U. Varanasi. 1995a. Chemical contaminant exposure and physiological effects in out-migrant juvenile chinook salmon from urban estuaries of Puget Sound, Washington. In Puget Sound Research 95; Proceedings. Puget Sound Water Quality Authority, PO Box 40900, Olympia, WA 98504. Pp. 657-665.

Casillas E., M.R. Arkoosh, E. Clemons, T. Hom, D. Misitano, T.K. Collier, J.E. Stein, and U. Varanasi. 1995b. Chemical contaminant exposure and physiological effects in out-migrant juvenile chinook salmon from selected urban estuaries of Puget Sound, Washington. In Salmon Ecosystem Restoration: Myth and Reality: Proceedings of the 1994 Northeast Pacific Chinook and Coho Salmon Workshop, M. Keefe (ed.), American Fisheries Society, Oregon Chapter, Corvallis, OR. Pp. 85-102.

Casillas, E., B-T. L. Eberhart, F.C. Sommers, T.K. Collier, M.M. Krahn, and J.E. Stein. 1998b. Effects of Chemical contaminants from the Hylebos Waterway on growth of juvenile chinook salmon. Interpretive Report prepared for NOAA Damage Assessment Center.

Casillas, E., B-T. L. Eberhart, T.K. Collier, M.M. Krahn, and J.E. Stein. 1998a. Exposure of juvenile chinook salmon to chemical contaminants specific to the Hylebos Waterway: Tissue concentrations and biochemical responses. Interpretive Report prepared for NOAA Damage Assessment Center.

Chatters, J.C., and K.A. Hoover. 1986. Changing late Holocene flooding frequencies on the Columbia River, Washington. Quaternary Research 26:309-320.

Chatters, J.C., and K.A. Hoover. 1992. Response of the Columbia River fluvial system to holocene climate change. Quaternary Research 37:42-59.

Collis, K., D. D. Roby, D. E. Lyons, R. M. Suryan, M. Antolos, S. K. Anderson, A. M. Meyers, and M. Hawbecker. 2001a. Caspian Tern Research on the Lower Columbia River, Final 2001 Summary. Columbia Bird Research, www.columbiabirdresearch.org.

Collis, K., D. D. Roby, D. P. Craig, B. A. Ryan, and R. D. Ledgerwood. 2001b. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the columbia River estuary: vulnerability of different salmonid species, stocks and rearing types. Transactions of the American Fisheries Society 130:385-396.

Collis, K., D. D. Roby, D. P. Craig, S. L. Adamany, J. Y. Adkins, and D. E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: Implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Dracup, J. A., and E. Kathya. 1994. The relationships between U.S. streamflow and La Nina events. Water Resources Research 30:2133-2141.

Fox, D. S., S. Bell, W. Nehlsen, and J. Damron. 1984. The Columbia River Estuary: Atlas of Physical and Biological Characteristics. Columbia River Estuary Data Development Program, Columbia River Estuary Study Taskforce, Astoria, Oregon. 87 pp.

Fuhrer, G.J. 1986. Extractable cadmium, mercury, copper, lead, and zinc in the Lower Columbia River Estuary, Oregon and Washington. U.S Geological Survey Water Resource Investigations Report 98-4052, 67 pp.

Fuhrer, G.J., and F. A. Rinella. 1983. Analysis of elutriates, native water, and bottom material in selected rivers and estuaries in Western Oregon and Washington. US Geological Survey open-file report 82-922, 147 pp.

Fuhrer GJ. 1989. Quality of bottom material and elutriates in the Lower Willamette River, Portland Harbor, Oregon. US Geological Survey Water Resources Invedstigations Report 89-4005, 30 pp.

Fuhrer, G.J., Tanner D.Q., Morace J.L., McKenzie S.W., and Skach, K.A. 1996. Water quality of the Lower Columbia River Basin – Analysis of current and historical water quality data through 1994: U.S. Geological Survey Water Resources Investigations Report 95-4294, 157 pp.

Gershunov, A., T. P. Barret, and D. R. Cayan. 1999. North Pacific interdecadal oscillation seen as factor in ENSO-related North American climate anomalies. EOS, Trans. Am. Geophysical Union 80 (3).

Good, T.P., K. Barnas, D. M. Marsh, B. A. Ryan, B. Meyers, and E. Casillas. 2003. Caspian Tern Predation on Juvenile Salmonid Outmigrants in the Columbia River Estuary

Gray, A., C.A. Simenstad, D. L. Bottom, and T.J. Cornwell. 2002. Contrasting functional performance of juvenile salmon habitat in recovering wetlands of the Salmon River Estuary, Oregon, U.S.A. Restoration Ecology 10:514-526.

Harrison, C. S. 1984. Terns: Family Laridae. *In:* Seabirds of Eastern North Pacific and Arctic Waters (D. Haley, ed.), pp. 146-160. Pacific Search Press, Seattle. WA. 214 p.

Harrison, H.E., Anderson, C.W., Rinella F.A., Gasser T.M., and Pogue T.R., Jr. 1995. Analytical data form phases I and II of the Willamette River Basin Water Quality study 1992-1994. US Geological Survey Open-File Report 95-373. 176 pp.

Healey, M. C. 1982. Juvenile Pacific salmon in estuaries: the life support system. Pages 315-341 in V.S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Healey, M.C. 1980. Utilization of the Nanaimo River estuary by juvenile chinook salmon, Oncorhynchus tshawytscha. Fisheries Bulletin 77:653-668.

Henny

Holtby, L.D., B. C. Anderson, and R. K. Kadowski. 1990. Importance of smolt size and early ocean growth to interannual variability in marine survival of coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 47:2181-2194.

Hooper PR, Goolsby DA, Rickert DA, and McKenzie SW. 1997. NASQUAN – A program to monitor water quality of the Nation's large rivers. US Geological Survey Fact Sheet FS-055-97, 6 pp.

Jay 2001

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California.. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-32, 280 p.

Johnson, H.E. and C. Pecor. 1969. Coho salmon mortality and DDT in Lake Michigan. Transactions of the 34th North American Wildlife Conference.

Kareiva, P., M. Marvier, and M. McClure. 2000. Recovery and management options for spring/summer chinook salmon in the Columbia River Basin Science 290:977-979.

Kathya, E., and J. A. Dracup. 1993. U. S. streamflow patterns in relation to the El Nino/southern oscillation. Water Resources Research 29:2491-2503.

Kukulka, T. and D. A. Jay. 2003. Impacts of Columbia River discharge on salmonid habitat; changes in shallow-water habitat. Journal of Geophysical Research

Levings, C. D., C.D. McAllister, and B.D. Chang. 1986. Differential use of the Campbell River estuary, British Columbia, by wild and hatchery-reared juvenile chinook salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 43:1386-1397.

Levy, D. A. and T.G. Northcote. 1981. The distribution and abundance of juvenile salmon in marsh habitats of the Fraser River estuary. Westwater Research Center University of British Columbia Technical Report 25:117pp.

Levy, D. A. and T.G. Northcote. 1982. Juvenile salmon residency in a marsh area of the Fraser River Estuary. Canadian Journal of Fisheries and Aquatic Sciences 39:270-276.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Manage. 19:81-97.

Mantua, N.J., S.R. Hare, Y.Zhang, J.M Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin American Meteorological Society 78:1069-1079.

McCabe Jr., G. T., R.L. Emmett, W.D. Muir, and T.H. Blahm. 1986. Utilization of the Columbia River estuary by subyearling chinook salmon. Northwest Science 60:113-124.

McCabe Jr., G. T., W.D. Muir, R.L. Emmett, and J.T. Durkin. 1983. Interrelationships between juvenile salmonids and nonsalmonid fish in the Colombia River estuary. Fishery Bulletin 81:815-826.

McCabe, GT Jr, SA Hinton, RL Emmett, and BP Sanford. 1997. Benthic invertebrates and sediment characteristics in main channel habitats in the Lower Columbia River. Northwest Science 71:45-55.

McCarthy, KA; Gale, RW. 2001. Evaluation of persistent hydrophobic organic compounds in the Columbia River Basin using semipermeable-membrane devices. Hydrol. Process. 15:1271-1283.

McClure, M. M., E. E. Holmes, B.L. Sanderson, and C.E. Jordan. 2003. A large-scale, multispecies status assessment: anadromous salmonids in the Columbia River Basin. Ecological Applications 13:964-989.

McElhaney, P., M.H. Ruckelhaus, M.J. Ford, T.C. Wainwright, and E.P. Bjorkstedt. 2000. Viable salmon populations and the recovery of evolutionary significant units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSX-42, Seattle, WA 156pp

Meador, J.P., T. K. Collier, and J. E. Stein. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juveniles salmonids listed under the US Endangered Species Act. Aquatic Conservation, Marine and Freshwater Ecosystems 12:493-516.

Miller, B.A. and S. Sadro. 2003. Residence time and seasonal movements of juvenile coho salmon in the ecotone and lower estuary of Winchester Creek, South Slough, Oregon. Transactions of the American Fisheries Society. 132:546-559.

Moore and Waring

Orem, H. M. 1968.Discharge in the lower Columbia River Basin, 1928-1965. Circular 550, U.S. Geological Survey, Washington D.C., 24p.

Pavlou, S., R. Kadeg, A. Turner, and M. Marchlik. 1987. Sediment quality criteria methodology validation: Uncertainty analysis of sediment normalization theory for nonpolar organic contaminants. Work Assignment 45, Task 3, Battelle, Washington, DC.

Pearcy, W.G. 1992. Ocean Ecology of North Pacific Salmon. Washington Sea Grant Program, University of Washington Press. Seattle, WA. 179 pp.

Poels, C.L.M., M.A. van Der Gaag, and J.F. J. van de Kerkhoff. 1980. An investigation into the long-term effect of Rhine water on rainbow trout. Water Res. 14:1029-1033.

Redmond, K. T., and R.W. Koch. 1991. Surface climate and streamflow variability in the western United States and their relationship to large-scale circulation indices. Water Resources Research 27:2381-2399.

Rinella, JF, McKenzie SW, Crawford KJ, Foreman WT, Fuhrer GJ, and Morace JL. 2000. Surface-water quality assessment of the Yakima river Basin, Washington—Distribution of pesticides and other organic compounds in water, sediments, and aquatic biota, 1987-91. U.S. Goelogical Survey Water Supply Paper 2354-B.

Roby, D. D., D. E. Lyons, D. P. Craig, K. Collis, and G. H. Visser. 2003. Quantifying the effects of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Canadian Journal of Zoology 81: 250-265

Roby, D. D., D.P. Craig, K.Collis, and S.L. Adamany. 1998. Avian predation of juvenile salmonids in the lower Columbia River. 1997 Annual Report, Oregon State University and Columbia River Intertribal Fish Commission, Corvallis and Portland, OR

Roy F., Weston, Inc. 1998. Portland Harbor sediment Investigation Report: Portland, Oregon,, prepared for US EPA, Region 10, contract no. 68-W9-0046.

Ryan, B.A., S. G. Smith, J. M. Butzerin, and J. W. Ferguson. 2003. Relative vulnerability to vulnerability to avian predation of juvenile salmonids tagged with Passive Integrated Transponders in the Columbia River Estuary, 1998-2000. Transactions of the American Fisheries Society 132:275-288.

Ryan et al. 2001

Scholz NL, Truelove NK, French BL, Berejikian BA, Quinn TP, Casillas E, Collier TK. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 57: 1911-1918.

Sherwood, C.R., D.A. Jay, R. B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical changes in the Columbia River estuary. Progress in Oceanography 25:299-357.

Shreffler, D.K., Simenstad, C.A., and R.M. Thom. 1990. Temporary residence by juvenile salmon of a restored estuarine wetland. Canadian Journal of Fisheries and Aquatic Sciences 47:2079-2084.

Shreffler, D.K., Simenstad, C.A., and R.M. Thom. 1992. Juvenile salmon foraging in a restored estuarine wetland. Estuaries 15:204-213.

Simenstad, C.A. and J.R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmon habitat in Pacific Northwest estuaries. Ecological Engineering 15:283-302.

Simenstad, C.A., K.L. Fresh, and E.O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pages 343-364 in V.S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58:325-333.

Spomberg and Meador

Tetra Tech. 1996. The health of the river 1990-1996. Integrated Technical Report 0253-01. Prepared for the Lower Columbia River Bi-State Water Quality Program.

TetraTech, 1996. Integrated technical report-Summary and synthesis of study findings, 1990-1996. Prepared for Lower Columbia River Bi-State Water Quality Program.

Thomas, D. W. 1983. Changes in the Columbia Rive estuary habitat types over the past century. Columbia River Estuary Data Development Program. Astoria 51 pp.

Thomas and Anthony 1999, 2000

USACE (United States Army Corps of Engineers). 1998. Dredged Material Evalutaion Framework. Lower Columbia River Management Area. Prepared by the US Army Corps of Engineers, Northwest Division, EPA Region 10, the Oregon Department of Natural Resources, and the Oregon Department of Environmental Quality. November 1998.

USFWS 1998

USFWS (U. S. Fish and Wildlife Service). 2001. Seabird predation and salmon recovery in the Columbia River estuary. U.S. Fish and Wildlife Service. Portland, OR. 10 p.

Varanasi, U., E. Casillas, M. R. Arkoosh, T. Hom, D. A. Misitano, D. W. Brown, S-L Chan, T. L. Collier, B. B. McCain and J. E. Stein. 1993. Contaminant exposure and associated biological effects in juvenile chinook salmon (Oncorhynchus tshawytscha) from urban and nonurban estuaries of Puget Sound, U.S. Dep. Commer., NOAA Tech. Memo. NMFS-NWFSC-8, p. 112.

Waring and Moore

TABLES

Table 1. Description of life history strategies and selected attributes associated with Columbia River anadromous salmonid populations based upon **historic** use of the system. Various sources of information, such as Bottom et al. (2001) and J. Burke (NOAA, Fisheries, personal communication), were used to develop this table. Sizes and times should be considered estimates.

Life History Strategy	Attributes
Early fry	Time of estuarine entry: March- April Size at estuarine entry: <500mm Estuarine residence time: 0-40d Freshwater rearing: 0-60d
Late fry	Time of estuarine entry: May-June, present thru Sept. Size at estuarine entry: <60mm Estuarine residence time- < 50d Freshwater rearing: 20-60d
Early fingerling	Time of estuarine entry: April-May Size at estuarine entry: 60-100mm Estuarine residence time: < 50d Freshwater rearing: 60-126d
Late fingerling	Time of estuarine entry: June-Oct, present thru winter Size at estuarine entry: 60-130d Estuarine residence time: 0-80d Freshwater rearing: 50-180d
Subyearling (smolt)	Time of estuarine entry: April-Oct Size at estuarine entry: 40-130d Estuarine residence time: little Freshwater rearing: 20-180d
Yearling	Time of estuarine entry: Feb-May Size at estuarine entry: >80mm Estuarine residence time: little Freshwater rearing: extended

Table 2. Summary rating table for listed Columbia River Basin ESUs for estuarine and plume factors. Ranks were assigned the following ranges: low (0-0.32), medium (0.33-0.66) 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 Upper Snake River Sockeye				Lower CR Chum Salmon Snake River Fall Chinook			
Rating Level	Factor				Factor			
	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 3. Level 1 ratings for estuarine and plume factors for stream-type ESUs including Snake River spring/summer chinook, upper Columbia River chinook, Snake River Steelhead, upper Columbia River Steelhead, Middle Columbia River Steelhead, and upper Snake River sockeye. An answer to a question of yes equals a 1 other than productivity and abundance which are scored a 2 for yes. An answer of no equals a 0.

Screening Criteria	Factor			
	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		2
Is there a significant effect on productivity (2x)	2	2		2
Is there a significant effect on LH Diversity			1	1
Is there a significant effect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	6	5	4	7
Max Possible Score	8	8	8	8

Table 4. Level 1 ratings for estuarine and plume factors for ocean-type ESUs including lower Columbia River chum salmon 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 0.

Screening Criteria	Factor			
	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	2
Is there a significant effect on productivity (2x)		2	2	2
Is there a significant effect on LH Diversity	1		1	1
Is there a significant effect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	3	5	8	7
Max Possible Score	8	8	8	8

Table 5. Level 2 ratings for estuarine and plume factors for stream-type ESUs including Snake River spring/summer chinook, upper Columbia River chinook, Snake River Steelhead, upper Columbia River Steelhead, Middle Columbia River Steelhead, and upper Snake River sockeye. An answer to a question of yes equals a 1 (however, see toxics footnote). An answer of no equals a 0.

Screening Criteria	Terns			Toxics[1]			Habitat			Flow		
	SW[2]	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	
Max Possible Score		12			20			12			12	

1- Scores for toxics include a value for sediment and water in estuary (ie, the sw quality score can be a 2) and water in the plume.

2- SW=Shallow water estuary, DW=Deep water estuary, Pl=Plume

Table 6. Level 2 ratings for estuarine and plume factors for stream-type ESUs including lower Columbia River chum and Snake River fall chinook. An answer to a question of yes equals a 1 (however, see toxics footnote). An answer of no equals a 0.

Screening Criteria	Terns			Toxics[1]			Habitat			Flow		
	SW[2]	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			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	
Max Possible Score		12			20			12			12	

1- Scores for toxics include a value for both sediment and water in estuary (ie, the sw quality score can be a 2) and water only in the plume.

2- SW=Shallow water estuary, DW=Deep water estuary, Pl=Plume

FIGURES

Figure 1. The Columbia River estuary extends from the upper extent of tidal influence at Bonneville Dam (RKm 240) through the oligohaline zone of the river mouth into the coastal zone of the plume in the Pacific Ocean. (not completed)

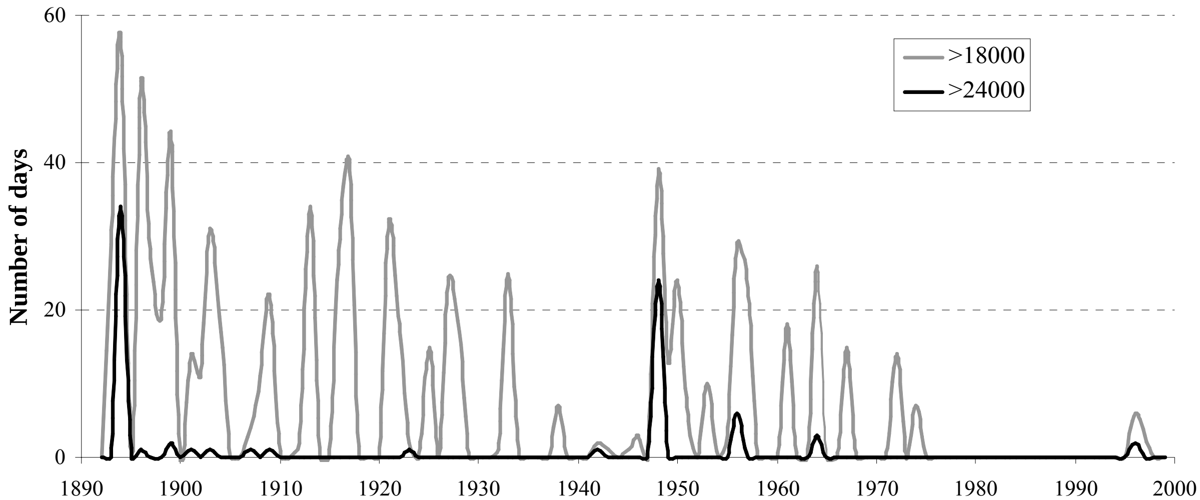


Figure 2. The incidence of flows above 18,000 m^3s^{-1} (the pre-1900 estimated bankfull flow level) and above 24,000 m^3s^{-1} (the present bankfull flow level). The present bankfull flow level has only been exceeded in four years since 1948. (From Bottom et al. 2001).

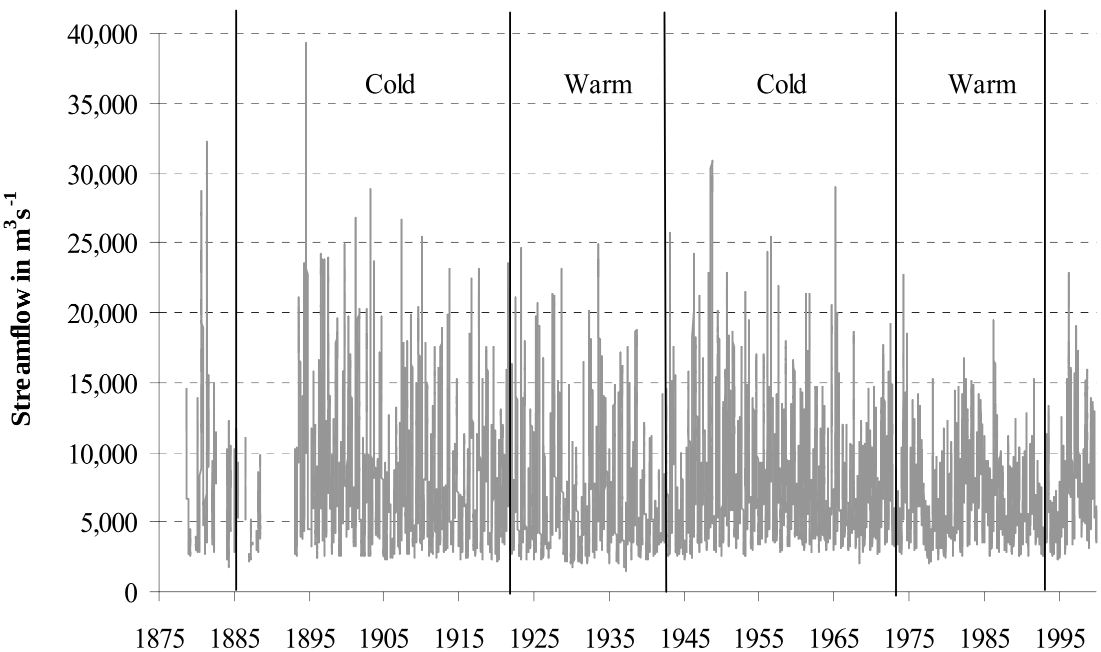


Figure 3. Monthly average flows at Beaver (1878-1999), present and historical bankfull flow levels, and warm and cold-PDO cycles. Historically, there was a major difference between the warm and cold phases of the PDO cycle in disturbance frequency. This has been largely eliminated by flow regulation and diking; overbank flow is now a rare event. (From Bottom et al. 2001).

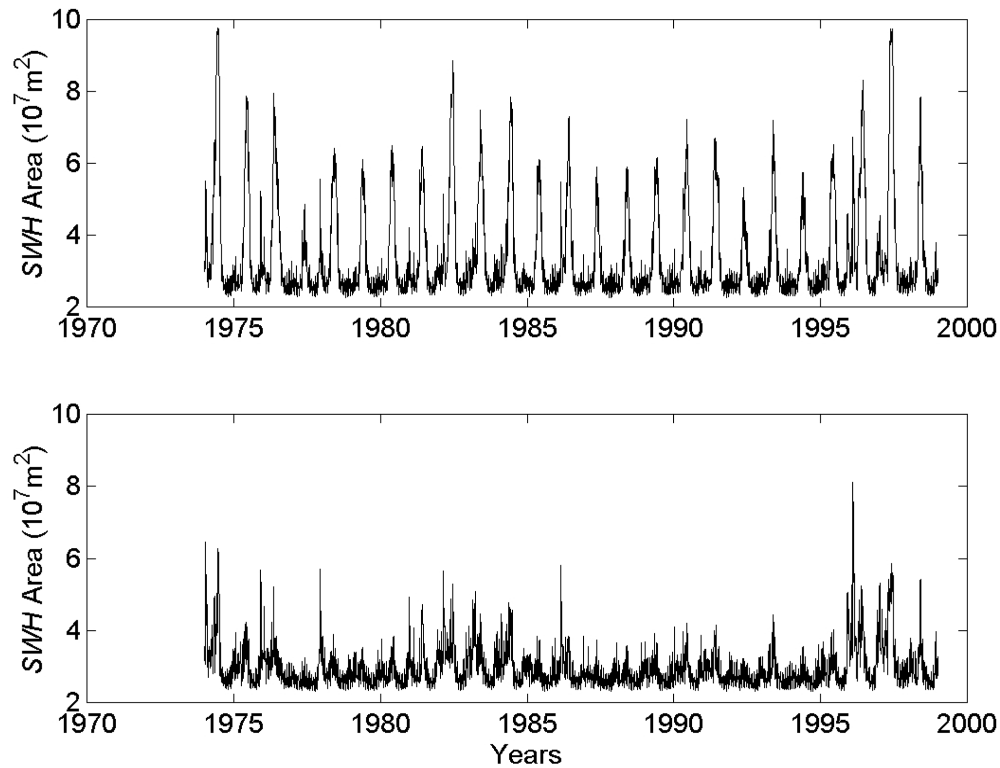


Figure 4. The change in availability of shallow water habitat in the tidally influenced region between RM 50 and RM 90 on the Columbia River under unmodified and modified flow conditions only. The top panel represents condition under virgin flow with no dikes, where extensive inundation of the floodplain occurs for long durations. The bottom panel represents conditions under modern flow conditions with no dikes, where river staged lowered and much less inundation of floodplain for shorter duration occurred. (From Kukulka and Jay 2003).

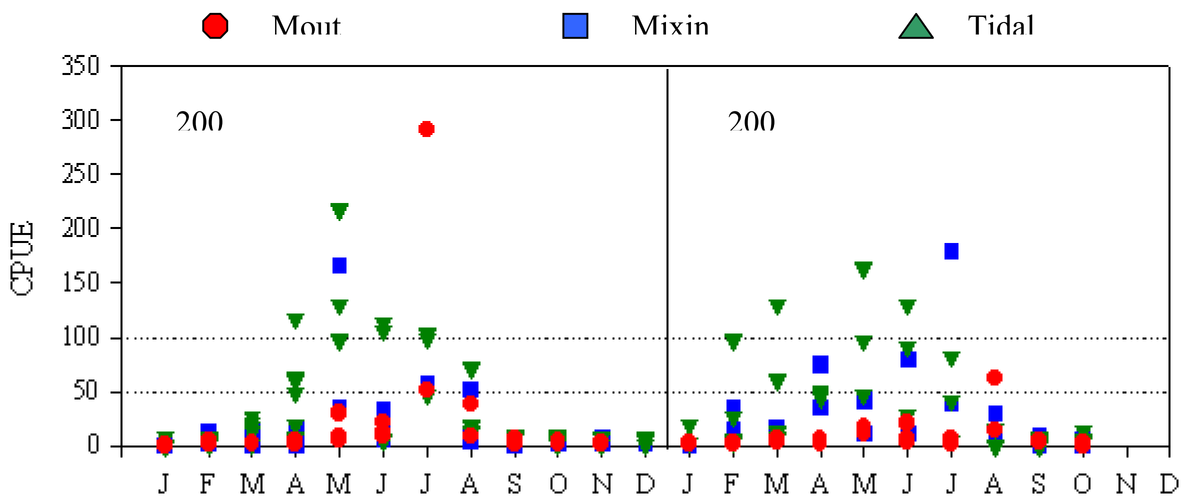


Figure 5. Catch per unit effort for juvenile chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (Curtis Roegner, NOAA Fisheries, pers. comm.)

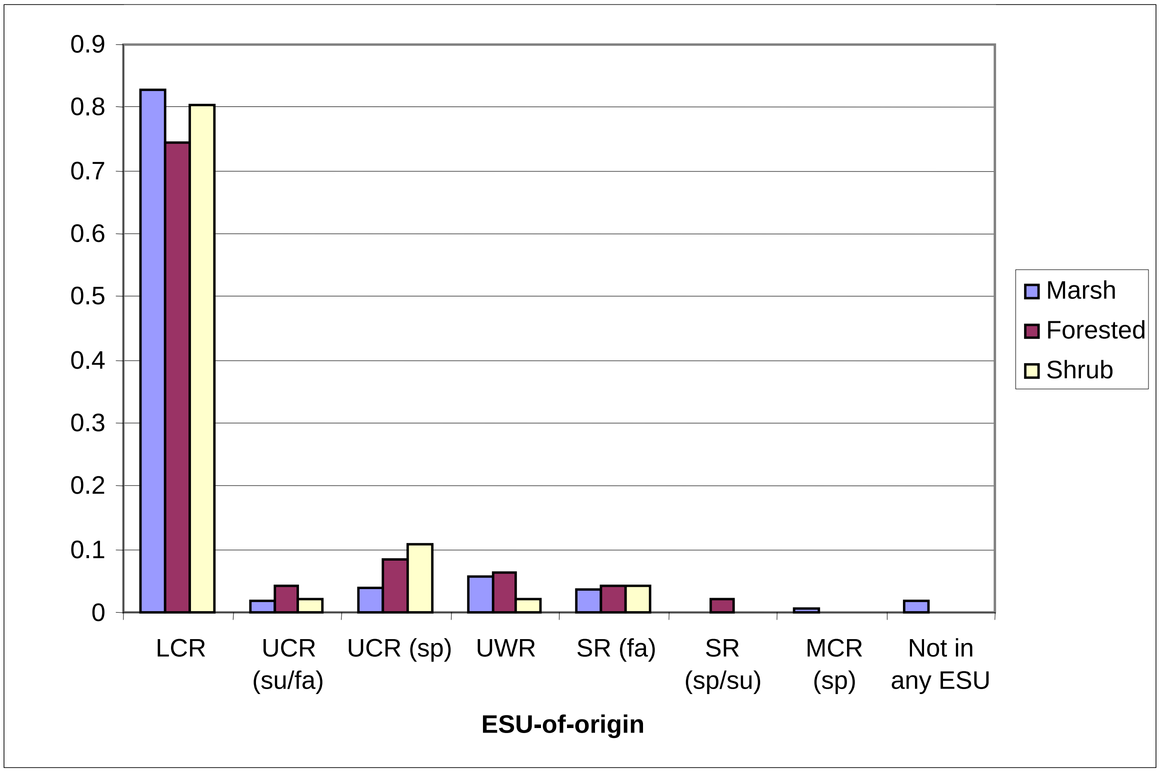


Figure 6. Proportion of Chinook salmon ESU's originating from various parts of the basin identified in samples taken from peripheral habitats of the Columbia River estuary during 2002 between April to August. LCR – Lower Columbia River, UCR – Upper Columbia River, UWR – Upper Willamette River, SR – Snake River, MCR – Middle Columbia River, su/fa – summer/fall run, sp – spring run, sp/su – spring/summer run. From Paul Moran, NOAA Fisheries, pers. comm..)

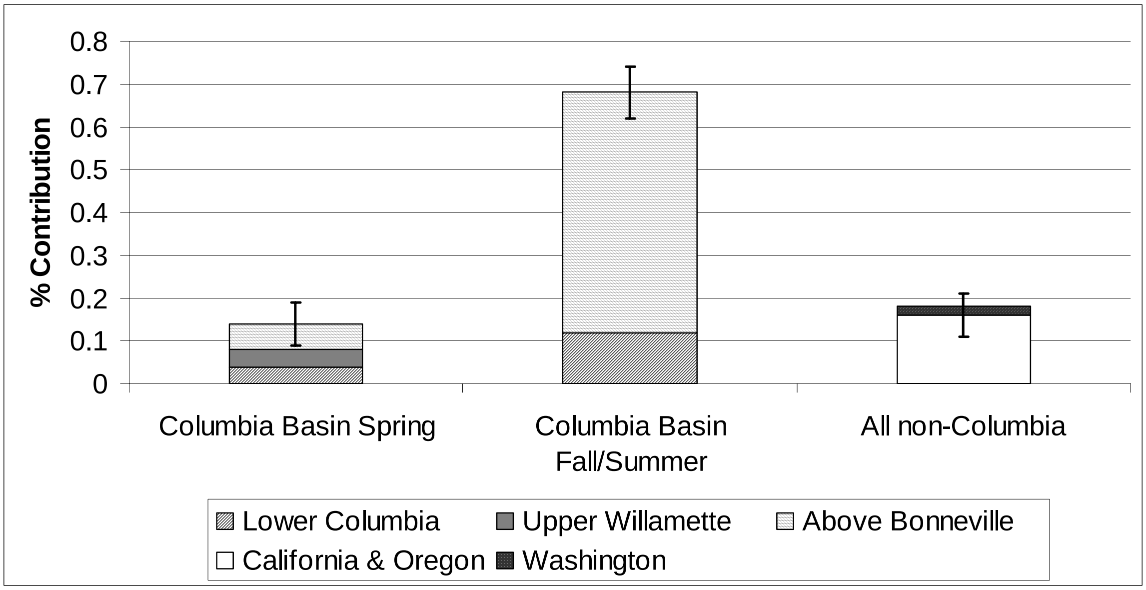
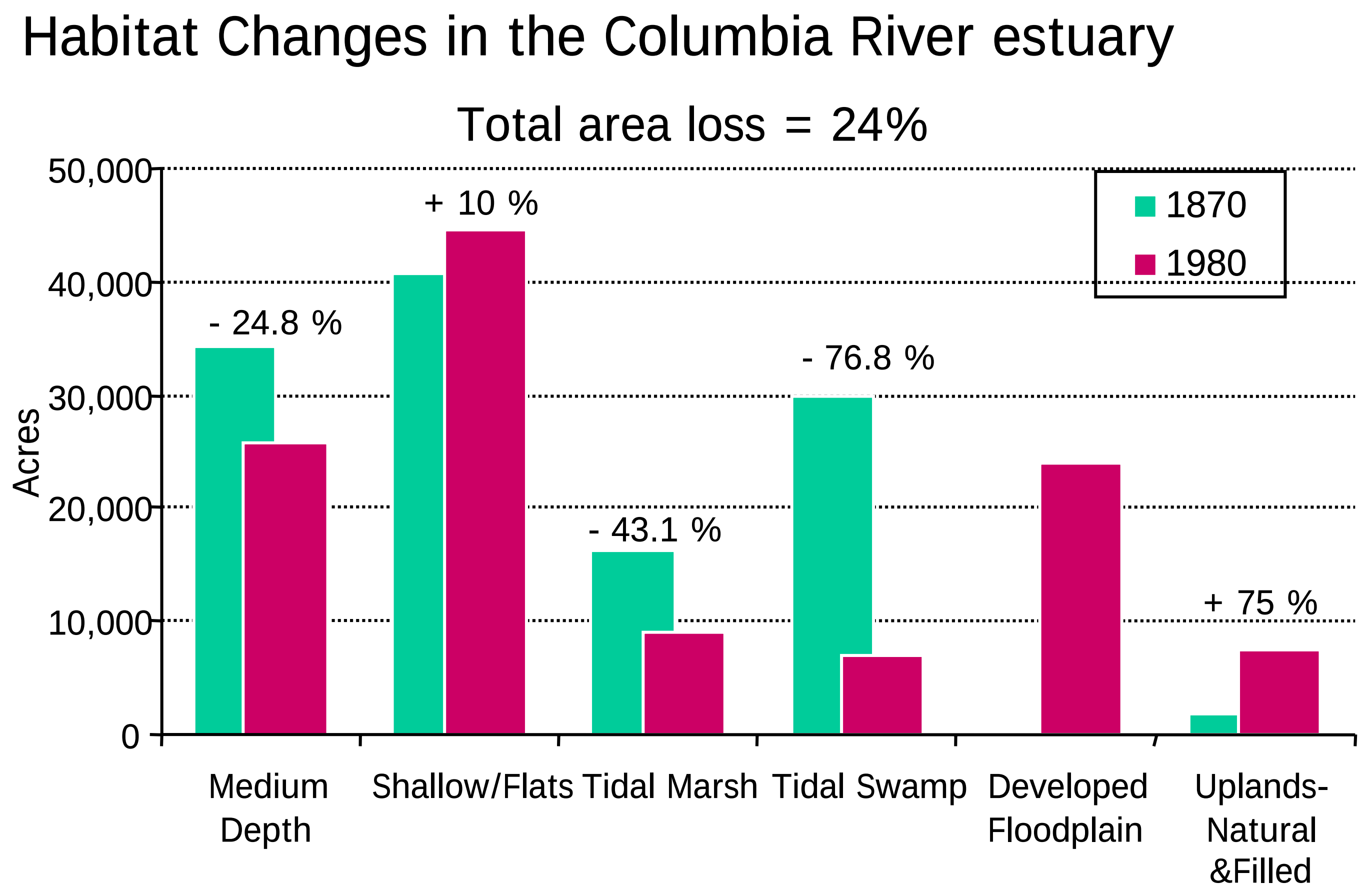


Figure 7. Stock composition of subyearling chinook salmon in Columbia River plume study area June 1998 – 2001.



Data Source:Thomas, T.W.1983. Changes in Columbia River estuary habitat types over
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Figure 8. Change in acreage of various habitat types used by salmon in the Columbia River estuary from 1870 to 1980.