Appendix C

Determining Habitat Value And Time To Sustained Function

Nicholas E. Iadanza

September 11, 2001

Part 1

HABITAT VALUATION

INTRODUCTION

This report presents quality ratings for habitats in the Commencement Bay area for juvenile English sole, juvenile chinook salmon, and four bird assemblages, which are representative of avian species occurring in the area. These values are based on a habitat's potential to provide attributes that support feeding and refuge functions of these species. Habitats are ranked according to their functional importance as relative rather than absolute values, similar to the concept in the Habitat Suitability Indices (HSI) used with the U.S. Fish and Wildlife Service Habitat Evaluation Procedures (USFWS, 1980).

REPRESENTATIVE SPECIES

Juvenile chinook salmon and juvenile English sole are used as representative fish species to assess the value of habitat to fish. Bird assemblages rather than individual species are used to assess habitat value to birds. Birds use similar habitat types as juvenile chinook salmon, and are linked through their food webs, so habitat value for birds is linked to habitat value for juvenile chinook salmon. Although the various fish and wildlife species in the Hylebos Waterway display a variety of life history requirements, juvenile chinook salmon and juvenile English sole have feeding modes, behavioral characteristics, and habitat requirements that sufficiently overlap those of similar sympatric species to consider them appropriate surrogates. The bird assemblages are grouped as a function of their foraging behavior and include both resident and migratory species found in the Commencement Bay area. Scores are assigned to habitat types based on their value to each of these species. These scores are then used to quantify potential injuries to the habitat and to assess the relative value of restoration projects.

HABITAT VALUE

There is considerable information on the utilization of estuarine environments by anadromous salmonids, flatfishes, and birds. Depending on the species, the data vary with regard to their adequacy in predicting what constitutes essential habitat and as to how specific habitat attributes relate to habitat value.

Chinook salmon

Available information indicates that estuarine habitats are important to juvenile salmonids, because salmonids use it for refuge from predators, foraging and temporary residence during physiological transition for seawater acclimation. Estuaries are particularly important to juvenile chinook salmon, which may have the longest estuarine residence time of juvenile salmonids, and use shallow, sublittoral, and neretic habitats (Simenstad et al., 1982). The information, however, is mostly qualitative. The few quantitative data sets that exist are not in formats amenable to developing habitat-species relationships or adequately defining the relative value of different habitat types.

The Puyallup River basin contains both spring and fall run chinook salmon, and naturally produced and hatchery-propagated chinook salmon use the Commencement Bay estuary. Naturally spawning fall chinook juveniles generally remain in their upstream natal areas for about 3 months following emergence, then begin their migration to the estuary as subyearlings (Williams et al., 1975). Juvenile chinook are released from hatcheries as fry, subyearlings, and yearlings. There is little current information about naturally spawning spring chinook salmon, but historical data indicate that they have a variable life history. Studies by Dunstan (1955) showed that 80 percent of White River spring chinook migrate to marine waters in their first year as subyearlings. Williams et al. (1975) reports rearing in fresh water for a year, migrating to salt water early in their second year as yearlings.

Juvenile chinook salmon may have an extended estuarine residence, and because of the diversity of size classes, have a diverse prey spectra and use a variety of estuarine habitats, shifting to reflect changes in food habits as they grow (Simenstad et al., 1982). Smaller individuals occur primarily in freshwater tidal marshes in the upper estuary and in salt marshes, where they feed on larval and adult aquatic insects and epibenthic organisms associated with a detritus based food web. Larger subyearlings move to tidal flats, gravel-cobble shorelines, and other shallow water habitats where they may rear for an extended period, feeding on epibenthic crustaceans such as gammarid amphipods, mysids, and cumaceans. Yearling chinook occupy neretic habitat, and may prefer habitats within confined embayments, where they feed on small nekton, insects, mysids, larval fish, and nuestonic drift organisms. Estuaries provide a diverse array of prey organisms, often in extremely high density, which allows juvenile salmon to sustain relatively high growth rates while occupying a relative refugia from predation.

Juvenile chinook salmon residence times in the Commencement Bay estuary and the Hylebos Waterway are recorded at 9+ and 8+ weeks, respectively (Simenstad et al., 1982). Growth while in the estuary may narrow their window of vulnerability to predation once they enter the marine environment. The relationship between estuarine residence time and foraging success and growth, and the implications on marine survival, suggests that the distribution and abundance of principal habitat types and prey may be a determinant of production of salmon populations migrating through the system.

Estuarine habitat is a critical factor in the life history of chinook salmon, but there are no models available that describe the relationship between habitat types and species utilization. For the purpose of this report, relative values are assigned to habitat types using available information on the feeding and refuge functions of different habitats for juvenile chinook salmon, functional rarity of habitats in Commencement Bay, and best professional judgement. References consulted in this exercise included not only literature regarding juvenile salmonid habitat utilization and feeding preferences, but also information on the frequency of occurrence of preferred food organisms, guidelines on sampling strategies, etc. (e.g. Beauchamp et al., 1983; Northcote et al., 1976; Seliskar et al., 1983; Simenstad et al., 1982; Simenstad, 1982; Simenstad et al., 1985; Simenstad et al., 1991; Simenstad et al., 1993). Values were based on habitats in an uncontaminated condition.

Nine estuarine habitat types are identified to encompass habitats in the Hylebos Waterway. Two additional habitat types are identified to include potential restoration projects. Estuarine habitat types are defined based on tidal elevation (high intertidal, low intertidal, shallow subtidal, and deep subtidal) and substrate (<20% particles >2 mm in diameter is considered sand/silt; >20% particles >2 mm in diameter is considered structurally complex). High Intertidal habitat containing aquatic vascular plants is classified as marsh (including estuarine dendritic marsh and fringing marsh). Other habitat categories include vegetated buffer (a vegetative zone adjacent to the aquatic habitat consisting of native floodplain vegetation, with tree, shrub, and herbaceous layers), and upland greenbelt (a vegetative zone landward of the vegetated buffer outside of the shoreline zone, with tree, shrub, and herbaceous layers).

Hylebos Waterway habitats are classified into habitat types, and habitat values assigned for juvenile chinook salmon (based on uncontaminated conditions), using data from substrate composition and bathymetric surveys (Table 1). Habitat values are a unitless number, based on relative, rather than absolute values, similar to the concept used in the HSI (USFWS 1980). Habitat values (1.0 indicating optimal conditions and 0 indicating unsuitable conditions) were assigned to estuarine habitats relative to the value of 1.0 given to marsh. The high value given to marsh reflects the influence of vegetation and primary and secondary productivity on increased epibenthic and benthic community structure and abundance, the high quality value as refuge habitat, and the rarity of this habitat type in the estuary. Values related to elevation are assigned, from highest to lowest, to marsh (+ 10 to + 13), high intertidal (+ 4 to +13), low intertidal (- 4 to + 4), shallow subtidal (-4 to -14), and deep subtidal (<-14). These values are based on the prey assemblage and abundance (higher taxa richness, species diversity, and abundance in the intertidal areas) (Northcote et al., 1976; Simenstad et al, 1993); the frequency and duration of habitat availability (the availability and duration of use of high intertidal habitats is dependent on tidal elevation), and primary productivity and habitat use (light availability decreases as depth increases, fewer salmonid prey species). Benthic community structure is affected by a variety of conditions. Different species colonize different substrate types and mixed substrates (sand, gravel, cobble) can provide abundant prey species and suitable refuge habitat for juvenile salmonids. However, the coarse screening process done for this analysis does not account for the intermediary mixing of substrates. In this analysis, substrates classified as coarse sand and smaller are included in the sand/silt category, and substrates sized as fine gravel and larger are categorized as structurally complex. In the Hylebos Waterway, this results in a comparison between low gradient, fine unconsolidated substrates conducive to production of benthic epifauna, and high gradient, coarse sediment and vertical hard substrate habitats that do not support comparable benthic species complexity and production. At any given elevation, sand/silt substrates are assumed to provide more prey organisms consumed by juvenile salmonids and assigned higher values than structurally complex ones. Deep subtidal habitats (-14 and deeper), both sand/silt and structurally complex, provide fewer prey organisms and are not preferred habitats of juvenile salmon, and are assigned a minimal value.

Birds

Birds utilizing estuarine areas may be classified into four "assemblages" based on their foraging behavior (Simenstad, 1983). (1) Shallow-probing and surface searching shorebirds (e.g. sandpiper, dunlin, plover), who are obligate benthivores that feed on benthic infauna and epibenthic zooplankton. (2) Waders, who prey on similar, though somewhat deeper benthic organisms (e.g. Greater yellowlegs); or are predaceous on fishes and motile epifaunal invertebrates that venture into shallow sublittoral areas of their habitat (e.g. Great blue heron). (3) The surface and diving water bird assemblage, which includes several feeding types, including benthivores (e.g. Lesser scaup, Barrow's goldeneye), piscivores (Western grebe, Common merganser), and omnivores (e.g. mallard). (4) Aerial searchers, which include higher level carnivores (e.g. Bald eagle), piscivores (e,g, Belted kingfisher, Osprey) and benthivores (e.g. Glaucous-winged gull).

There are many references about estuarine bird species and their habitats that provide information necessary to identify habitat-species relationships. Estuarine birds, because of their differences in foraging, use a diversity of habitats. Given the dependance on common habitats and food items by some bird species and the utilization of fishes as prey items by others, there is a relationship between the value of habitat for salmon and its value for birds. Certain species from these assemblages share common habitats and prey items with juvenile salmon. Shallow-probing and surface searching shorebirds, some waders, and some surface and diving birds feed on benthic macroinvertebrates in intertidal habitats. Shorebirds feed in exposed areas; however, their use is restricted to the high intertidal area and that part of the lower intertidal area exposed at low tide. Habitat utilization by benthivorous waders and surface and diving waterbirds is affected by tidal fluctuations, but their foraging habits allow greater utilization of habitat. Some waders, surface and diving waterbirds, and aerial searchers feed on juvenile salmon and other fish species with similar habitat requirements. For the purposes of this analysis, we assume that the value of a particular habitat type to estuarine birds is the same as the habitat value assigned to salmon (Table 1).

English sole

The model presented in "Habitat Suitability Index Models: Juvenile English Sole" (Toole et al., 1987) is used to quantify the habitat value for English sole. The model applies to juvenile English sole in estuaries and coastal lagoons year-round. It is based on the assumption that any environmental variable that has an impact on the growth, survival, distribution, or abundance of juvenile English sole can be expected to have an impact on the carrying capacity of its habitat. Habitat Suitability Indices (HSI) are calculated based on Suitability Indices (SI) from either the Food or Water Quality component of the model. An HSI value of 1.0 indicates optimal conditions, and a value of 0 indicates unsuitable conditions. The HSI is determined based on the limiting factor concept, and whichever value is lower is accepted as the HSI. Habitat values for English sole in the Hylebos Waterway are expressed in terms of HSI. Because of the tidal exchange in the Hylebos Waterway, the Water Quality variables (bottom water temperature, mean salinity, and dissolved oxygen concentration) fall primarily in the high suitability value ranges (1.0). Since HSI defaults to the limiting factor, in this situation the SI calculation defaults to use of the Food Component. The Food Component is related to hydrodynamic regime and dominant sediment type; therefore, the HSI value of the habitat is whichever has the lower SI value, that of the hydrodynamic regime or the substrate. Within the hydrodynamic regime there are three SI values: 0.2 for high energy areas of rapid erosion and deposition, 1.0 for areas of intermediate energy with stable substrates, and 0.3 for low energy areas with limited tidal exchange. With the exception of limited areas near the mouth of the Waterway and the outlet of Hylebos Creek, Hylebos Waterway fits best into the category of intermediate energy with stable substrates and was assigned a hydrodynamic regime value of 1.0. Therefore, the HSI value for Hylebos Waterway habitats is calculated based on dominant substrate type.

Substrate composition data from sampling events in the Hylebos Waterway were used to define the categories and HSI values assigned accordingly. Values are based on habitat in an uncontaminated condition. Substrate SI values are based on data relating density and stomach fullness of English sole to substrate type. Fine substrate provides the best habitat for feeding sole, but sediments with as much as 20% gravel (>2 mm in diameter) are suitable. Values are low where gravel and rocks are the dominant substrate type; however, even 100% gravel is assumed to provide some food for English sole. Depth and cover do not affect habitat value. Intertidal, subtidal, and deep water habitats are used by sole at different life stages (Lassuy, 1989). Recently metamorphosed and juveniles in the 50 - 68 mm size range are found in intertidal and shallow subtidal areas where they feed primarily on small epibenthic crustaceans. As they grow, they move into deeper water, where prey items shift to polychaetes, mollusks and other infaunal organisms. The existing literature does not identify cover as an important factor influencing abundance or predation. The variable related to the substrate SI value is dependent upon the percentage of the substrate that is made of particles >2 mm in diameter. Five substrate composition categories are selected for this analysis, based on the percentage of substrate >2 mm. SI values for these categories are interpolated from the substrate composition suitability graph (Toole et al., 1987) with values ranging from 1.0 for substrate with #20% particles greater than 2 mm in diameter to 0.15 for substrate with >50% particles >2 mm in diameter (Table 2).

COMBINED HABITAT VALUES

Combining the elevation and substrate attributes for juvenile chinook salmon and birds with the substrate attributes for juvenile English sole resulted in the identification of 25 habitat types (including fresh water habitats and buffers). The intent is to value habitats not only for injury determination, but also for restoration planning. For restoration, habitats that provide the most benefit to the injured resource may not necessarily be those habitats that are injured, so habitats that do not exist presently in the Hylebos Waterway, but may provide considerable restoration benefit, are included.

All habitats provide some value for all three representative species. In order to reduce some steps in the HEA, a single weighted value combining all three species for each habitat type was used in the calculation, rather than calculating the value for each species individually and adding the individual weighted values (no difference mathematically). The combined value does not weight the species equally. Chinook salmon in Puget Sound are a high profile species, listed as threatened under the Endangered Species Act. There is considerable regional interest in their restoration, and so they were given a higher weighting in the calculation of a combined habitat value. Species weighting in the final combined value was: 50% juvenile chinook salmon, 25% juvenile English sole, and 25% birds. Individual and combined values for the 25 habitat types are shown in Table 3.

SIMPLIFYING ASSUMPTIONS AND ADJUSTMENTS

Reducing the number of habitat types

There are five substrate attributes that affect the relative habitat values for English sole; and one vegetative, four elevation, and two substrate attributes that affect the relative estuarine habitat values for juvenile chinook salmon and birds. An analysis of the relative areas of the 20 habitat types in the high intertidal, low intertidal, shallow subtidal, and deep subtidal elevations by substrate composition showed that approximately 90% of the habitats were in the sand/silt category (highest habitat values within habitat types). The second largest substrate category was structurally complex (21-30%) rock, at approximately 6% (second highest habitat values within habitat type). Within habitat types, sand/silt comprised 82-93% of the area, and 21-30% rock, 4-8% of the area (Table 4). An evaluation of the combined values (salmon + birds + flatfish) associated with the habitat types and the relative amount of each habitat type indicated that some combining within habitats could be done using weighted values without compromising the injury or restoration analysis. The analysis was simplified in a stepwise process. First, the structurally complex habitats with 21-50% rock within each elevation were combined with the sand/silt substrates and incorporated into a single habitat type for that elevation. A weighted habitat value was calculated based on the makeup of the combined habitat type. Combining sand/silt, 21-30% rock, 31-40% rock, and 41-50% rock within each elevation into a single habitat type reduced the number of habitats from twenty to eight. The acreage of the four structurally complex habitats with >50% rock made up a small percentage of the overall habitat, and values for this habitat were similar at all elevations. Ground truthing of habitat areas in the Hylebos Waterway with >50% rock identified it as predominately rip rap, so these habitats were combined into a single habitat type, labeled rip rap. It was decided that rip rap provided little habitat value to any of the representative species and an arbitrary value of 0.1 was applied. Calculation of the weighted values from combining habitats in illustrated in Table 4.

The second step taken in reducing the number of habitats was combining the high intertidal and low intertidal into a single habitat type. It is acknowledged that there are differences in the functional value of these habitats to juvenile chinook salmon, juvenile English sole, and birds; related to the distribution of prey items, and availability and utilization of the area. However, the areas (acres) represented by high and low intertidal habitats were equal (Table 4) and the weighted values were similar (0.7226 and 0.7848, respectively, for high and low intertidal). To further simplify HEA parameters, it was decided to combine high and low intertidal habitats into a single habitat category named intertidal habitat, with a weighted value of 0.7537. Final (rounded) values resulting from the combination of habitat types are presented in Table 5. A flow chart illustrating the habitat consolidation is included as Figure 1.

Value adjustments associated with environmental conditions

Habitat values identified for the HEA are used both in quantifying loss of functional value associated with injuries and in assessing benefits (gains in functional value) associated with restoration project development.

Evaluation of the Hylebos Waterway revealed that the current habitat conditions are not quite what is envisioned as functioning juvenile salmon, flatfish, or bird habitats. Value adjustment categories of "fully functional" and "baseline adjusted" were conceived to be applied to marsh, intertidal, and shallow subtidal habitats. The "fully functional" category was based primarily on the premise that the presence of adjacent desirable habitat results in a complex that enhances overall production. Habitats considered "baseline adjusted," have no adjacent habitat to enhance their value. As an example, energy flow in the form of insect and organic matter production from vegetated buffer and marsh habitats provide benefits that increase the value of adjacent habitats. The increased invertebrate recruitment and subsequent juvenile salmonid use of a intertidal area bordered by a marsh or vegetated buffer zone make it more valuable (fully functional) than one that does not have the benefits from these adjacent habitats (baseline adjusted). In restoration planning, adjustments to habitat values are beneficial in identifying habitat mixes to provide maximum benefits (e.g. an intertidal area created in association with a marsh or vegetated buffer area would have more value than one that is created as an isolated habitat).

The Hylebos Waterway is in an urban/industrial/commercial setting, with extensive shoreline development. There are no marshes and few upland areas that could be classified as functioning vegetated buffer habitat. The current condition of intertidal and shallow subtidal habitats in the Waterway classifies all of them as "baseline adjusted," with no adjacent habitat to enhance their value (0.75 for intertidal, 0.55 for shallow subtidal). For purposes of restoration planning, an enhancement of 0.15 is applied to intertidal habitat constructed in association with a vegetated buffer or a fully functioning marsh, and to a shallow subtidal area associated with a fully functioning intertidal area. "Fully functional" values for intertidal and shallow subtidal habitats are 0.9 and 0.7, respectively. The amount of habitat influenced (classified as fully functional) by adjacent habitat includes wetted habitat contained within a polygon formed by (1) the linear shoreline extent of the vegetated buffer or marsh habitat, (2) an imaginary NW-SE (approximate) line drawn down the center of the Waterway, and (3) straight lines drawn from the ends of the shoreline extent of the vegetated buffer or marsh habitat to intersect perpendicularly with the imaginary centerline of the Waterway (Figure 2). Certain site configurations may require adjustments of the delineation method.

The premise for a “fully functional” classification is that habitat complexes are necessary for proper ecosystem functioning. A habitat that is not part of a complex is considered “baseline adjusted.” Marsh habitat was assigned a value of 1.0 (optimal conditions). A marsh associated with a vegetated buffer likely has more ecological value than one that does not. Therefore, for a marsh to be considered “fully functional,” it must have an adjacent vegetated buffer. Marshes without a vegetated buffer are considered “baseline adjusted,” and are assigned a maximum value of 0.85.

In restoration planning, habitat complexes are required for “fully functional” value: a marsh must be associated with an adjacent vegetated buffer habitat; an intertidal habitat must be associated with an adjacent vegetated buffer or an adjacent fully functioning marsh; and a shallow subtidal habitat must be associated with an adjacent fully functioning intertidal habitat.

Development in the Hylebos Waterway has resulted in the facilities and activities that physically degrade habitat quality. The presence of large over-water structures such as piers, aprons, buildings, etc., the occurrence of log rafting in intertidal and shallow subtidal areas, and the presence of concentrations of wood wastes creates conditions that limit the use of affected habitats by species considered in this analysis. This situation called for another category to represent these conditions and a “degraded” classification of reduced value (0.1) was created to value habitats that are severely impacted by physical obstructions.

Potential impacts associated with severe physical habitat degradation (lowered primary and secondary production because of reduced light, disruption of migration/feeding behavior, etc.) warrant application of a low habitat value in some situations. There are gradations of impact from overwater structures, related to their height over the water, piling type and density, orientation, type of structure, water depth and habitat type beneath them, etc. However, there was no attempt to identify sub-classifications based on these gradations to cover the range of impacts. The degraded classification is applied narrowly, only to situations causing severe physical impacts:

<u>Log rafts</u> - Severe habitat degradation results from grounding and abrasion of substrate in intertidal areas, shading, wood debris, prop wash in approaches, etc. (Toews and Brownlee, 1981) - assign degraded habitat value (0.1)

<u>Wood waste</u> - Severe habitat degradation due to debris covering natural substrate, toxic leachates, anoxic conditions during certain times of year (high Biological Oxygen Demand), toxic effects of leachates and H_2S (Toews and Brownlee, 1981) - assign degraded habitat value (0.1)

Overwater structures - This includes permanent and semi-permanent structures such as piers, aprons, buildings, boathouses, houseboats, etc. Because a juvenile salmonid's visual ability to adapt from bright to subdued-light conditions proceeds slowly (Ali, 1959), they are reluctant to pass beneath structures where there is a high contrast between bright and low light levels. Smaller juvenile salmonids are shoreline/shallow water oriented and over-water structures that produce sharp light contrasts may interfere with their feeding and migratory movements. The subdued light conditions found along the periphery of piers are often preferred over bright sunlight, but low light levels interfere with feeding. Structures covering intertidal and shallow subtidal habitat have more impact on epibenthic production than those in deeper water. All other factors being equal, only habitats under structures that extend directly from and are contiguous with the shoreline are assigned the degraded habitat value of 0.1. Piers and docks that have the major pier structure away from the shore, and have a narrow overpass perpendicular to the shore (e.g. T-docks), usually have the major over-water portion in deeper water, and have less shoreline/near shore shading. They have less of an impact than structures extending from the shoreline, and are not placed in the degraded category. Habitat beneath them is included in the baseline adjusted category, based on habitat type. Marinas with docks and boat houses are generally in deeper water, and the shoreline connections are usually narrow. They have an adverse impact, but not enough to be included in the degraded category. Habitat beneath them is assigned the baseline adjusted value, dependent on habitat type.

The foregoing guidelines are not intended to represent acceptance or rejection of particular types of structures or activities. All of the in-water/over-water structures mentioned above can have an adverse impact on aquatic habitat and there are exceptions to each situation that could mitigate or exacerbate the expected impact. However, the decision to include or not include particular over-water structures is made in a general sense based on an evaluation of biological information on potential effects. It is to be used for the sole purpose of classifying habitat values for the Hylebos Waterway HEA in as simple and as equitable of a manner as possible.

Value adjustments associated with environmental conditions are shown in Table 6.

Part 2

TIME TO SUSTAINED VALUE

INTRODUCTION

This document presents development rates for habitats considered for restoration projects in the Commencement Bay area. The assumption that environmental injury or habitat loss can be compensated for by ecological restoration is based on the premise that restored habitat should provide the same values as the natural ecosystem (Pacific Estuarine Research Laboratory 1990). This restoration has been termed ecological equivalence (Kentula et al. 1992), referring to the capacity of a restored, created, or enhanced habitat to reproduce the ecological structures and functions equivalent to an injured or lost habitat. Determining the value of a restoration project depends not only on the level of function expected from the habitat, but also the time it takes for the habitat to reach and sustain this level of function. A created, restored, or enhanced habitat goes through natural successional patterns, gradually increasing in value from its initial condition over a period of time until it reaches some assumed endpoint, where a sustained functional value is reached. There are two components to this function, the shape of the curve and the time to maturation. The shape (rate of increase) of the recovery curve will likely vary with habitat types. It may follow an “S” shaped curve, increasing gradually at first, rapidly approaching a stable maximum, then falling off as the final level of function is achieved; or it may follow some other pattern. An investigation of the use of different curves to describe the increase in wetland functions as created wetlands develop determined that for the purposes of evaluating restoration ratios, the shape of the curve was not important, resulting in differences of only a few percent in the amount of restoration required (King et al., 1993). Growth rate or population dynamics data from existing restoration projects in the Pacific Northwest are not consistent enough to define specific recovery curves, and for the sake of simplicity, it is assumed that function will increase along a linear path until sustained value is achieved.

The number of years after construction that the restoration project is expected to achieve sustained value varies with habitat type. The period of time it takes a habitat to reach a sustained level of function can be short or long, depending on the metric used (Strange et al. 1999). In restored salt marshes on the East coast, vegetative cover was similar to that of a natural marsh within 5 years; however, development of other physical and chemical properties necessary to support fish and shellfish production took 25-30 years. Estimates of times to sustained value for use in this assessment are based on observations made at similar restoration projects in Puget Sound, the scientific literature, unpublished research in the “gray” literature, and best professional judgement of the natural resource trustees. In determining time to sustained value for the various habitat types, the focus is on biological processes that generate and maintain food and habitat for the representative biota, such as benthic and epibenthic invertebrates, number of species present, abundance of individuals, and preferred prey species. Habitats considered are those that may be considered for inclusion in restoration projects: intertidal habitat, shallow subtidal habitat, marsh, vegetated buffer, upland greenbelt, and degraded habitats.

ASSUMPTIONS

The scientific literature suggests that replicating the services provided by a natural habitat with a created one is extremely difficult. Even restoration sites that are essentially identical in physical features to natural habitats may not provide the same ecological functions (Kusler and Kentula, 1990). However, for the purpose of this analysis, a 1:1 productivity ratio is assumed for the level of ecological services provided by created relative to natural habitats. This implies that restored habitats will be as productive as natural habitats in terms of all associated services. There is uncertainty associated with the outcome of restoration projects and different types of habitats carry more risk of failure than others. However, restoration project implementation in the Pacific Northwest commonly incorporates monitoring, success criteria, and mid-course corrective actions to increase the probability of success (Commencement Bay Natural Resource Trustees, 2000; Elliott Bay/Duwamish Restoration Program, 2000). Actions such as soil development and amendment, plant transplantation, weed control, and other “eco-engineering” approaches to supplant important natural processes are often used to promote successful outcomes. For the purposes of this analysis, risk of failure is not incorporated. Habitats are assumed to achieve the expected function within the time identified.

Intertidal and shallow subtidal habitats

The expected sequence of invertebrate recruitment and subsequent juvenile salmon, juvenile English sole, and bird use, is related to the initial condition of the habitat. The further initial conditions are from a mature steady state, the longer it will take a system to approach a state with self-sustaining habitat attributes (Mitsch and Wilson, 1996). Monitoring data from restoration projects in the Puget Sound area indicate that habitat functions associated with intertidal and subtidal sand/silt and gravel/cobble substrates develop rapidly. Many of the projects involved habitat creation, where excavation, regrading, or filling was conducted to create intertidal or shallow subtidal habitats. Some sites showed rapid development of a diverse and abundant assemblage of benthic and epibenthic organisms, achieving within 50-100% of their long term trends within 1 - 2 years after construction (e.g. Milwaukee Habitat Area (Parametrix, 1998)), but, in general, the data indicate that newly placed, newly exposed, and sometimes, newly wetted materials require time for development of the natural processes necessary to support benthic and epibenthic production. The rate of development of a stable community is related to substrate, slope, elevation, exposure, salinity, etc. Although the numbers of epibenthic invertebrates were often highly variable from year to year, by years 3 - 4, benthic and epibenthic production at many restoration sites in the Puget Sound area approached long term production levels and population structure and taxa richness comparable to reference areas. Therefore, for a newly created habitat, 4 years is assumed to be an appropriate time to reach sustained value for baseline adjusted intertidal and shallow subtidal habitats (0.75 and 0.55, respectively). Time to sustained value for fully functional intertidal and shallow subtidal habitats (0.9 and 0.7, respectively) is related to the time to sustained value of the adjacent habitat (8 years for vegetated buffer habitat).[1]

Marsh habitats

Marsh habitat is assumed to include both dendritic marshes and fringing marshes. Success in creating estuarine habitats that support aquatic vascular plants has been limited in Commencement Bay. In other areas, where salt marshes have been created, there are questions regarding how well they actually replicate the ecological functions of natural marshes. Strange et al. (1999) investigated maturity rates and recovery of particular ecological structures and processes in salt marsh restoration and found that conclusions regarding success were dependent upon the metric used to measure it. If vegetative structure alone is assessed, a restoration project may be considered to have achieved equivalence to a natural marsh within 5 years. When the metric is community/ecosystem function, recovery was slower, in excess of 15 years. Development of the physical and chemical properties of soils needed to support infaunal development, and the production of higher order consumers, can take decades to become fully equivalent to a natural salt marsh. There is some thought in the ecological community that, because of the complexity and variation in natural marshes, and the subtle relationships of hydrology, soils, vegetation, nutrients, and animal life; creation of a marsh that duplicates a natural marsh is not possible (Kusler and Kentula, 1990). In this assessment, the marsh habitat is not assumed to duplicate a natural estuarine marsh. However, it is considered a habitat that has the structural characteristics to generate and maintain food and habitat for the representative biota, and a time to sustained value of 15 years is assumed for a fully functional marsh (value of 1.0) and a baseline adjusted marsh (value of 0.85).

In the Commencement Bay area, marsh habitat may be created in sand/silt substrates in the + 10 to + 13 elevation range. This elevation range is included in intertidal habitat (- 4 to + 13). The curve for fully functional marsh habitat is shaped as a stepped function. A newly created habitat intended to reach a marsh endpoint goes through successional stages, first becoming an intertidal mudflat, then gradually transforming into a marsh over a period of years as vegetation develops. The value increases in a straight line from its initial state to the value for a fully functional intertidal habitat (0.9) in years 0 through 8, then increases more gradually to the marsh value between years 8 and 15. A baseline adjusted marsh is valued the same as baseline adjusted intertidal habitat (0.75) through year 4, then increases gradually to its sustained marsh value of 0.85 between years 5 and 15.

Vegetated buffer and upland greenbelt

There is considerable information on the value and size requirements of vegetated (riparian) buffers but much less on rates of development. A literature search did not find published studies

---

[1] See Part 1, Value adjustments associated with environmental conditions, for a description of “baseline adjusted” and “fully functional” habitats.

on development rates in the Puget Sound area. Riparian buffer planting is included in several restoration projects in the Puget Sound area (e.g. Middle Waterway Shore Restoration Project, Sitcum Waterway Remediation Project), but there is, as yet, insufficient data upon which to draw conclusions. However, related information is available from which one can infer how fast a vegetated buffer will develop, and whether development follows a straight line or stepped trajectory. Monitoring guidelines for restoration projects include “success criteria”. Success criteria are defined generally as those measures used to evaluate whether the requirements for functional replacement have been met - if the criteria are met, the project is successful, and functional replacement is achieved.

The supposition used in this assessment is that if these monitoring guidelines are providing a measure of functional replacement, they should provide some determinant of the time frame within which “success,” in terms of functional habitat replacement, may be expected. This is based on guidance on the selection of functional performance objectives indicating that they should be: 1) known or likely benchmarks of success and 2) achievable on the site within the designated monitoring period (Ossinger, 1999).

A review of U.S. Army Corps of Engineers’ (USACOE 1999) “Examples of Performance Standards for Wetland Creation and Restoration in Section 404 Permits and an Approach to Developing Performance Standards” shows most monitoring programs for vegetated buffers (riparian, shrub-scrub, and woody vegetation) extending for 5 years. Specific project information is not provided for the examples in the document, but expectations as a measure of success for shrub-scrub and forested buffers from temperate zone areas are:

California - 75% cover by native riparian species by year 5
Maryland - 85% of site vegetated by planted species and/or naturally regenerated (approved) vegetation by year 5
Maryland - 85% herbaceous cover,75% areal cover by planted woody species by year 2
Alaska - vegetative cover equal to 75% of test plot cover in 5 years
Washington - 60% cover by native shrub species by year 5

An example of Seattle District COE 1994 monitoring guidelines for freshwater wetlands required 80% cover of native shrub/scrub species after 5 years and 40% canopy cover of native species forest vegetation after 20 years (USACOE, 1999). Ossinger et al. 1999, reporting on findings of the “Success Standards Work Group,” a group of wetland professionals from state, federal and private sectors convened to provide practical guidelines for mitigation planning, suggests benchmark values for herbaceous vegetation as 80% cover by year 3, and 90% cover by year 5. For woody cover (wetland buffer/forested zone), 50% cover by year 5 is suggested. Mockler (1998), developing guidelines for King County, suggested that buffers, defined as a dense vegetation that will protect wetland from human encroachment and provide wildlife habitat, should have 60% emergent cover by year 1, 80% by year 3, and 90% by year 5. Shrub or sapling tree cover should be >60% by year 3. A success criterion for riparian vegetation establishment in a recent monitoring program proposal specific to the area (Elliott Bay/Duwamish Restoration

Program, 2000; Commencement Bay Natural Resource Trustees, 2000) specifies native trees and shrubs should cover not less than 90% of the upland vegetated area at the end of 10 years.

Monitoring data from the Puget Sound area are sparse, but there are some that contribute to an understanding of the rate of development of buffer areas and functions provided. The Gog-Li-Hi-Te wetland system, created in 1986, included a mix of upland and wetland habitats. The 5-year monitoring report (Thom et al., 1991) shows that upland trees increased from 725 m^2 to approximately 1500 $m^{2.}$ The data also show that the transitional zone between the intertidal and upland habitats was rapidly colonized by willow and alder, which increased from 0.4% of the area (160 m^2) in 1986 to approximately 4.3% (1,650 m^2) in 1990. The riparian vegetation increases are from natural recovery, as planting of these species does not to appear to have been included in the project design.

Although planting of upland riparian vegetation was included in the Duwamish River Coastal America sites, monitoring conducted 3 years post-construction did not provide data on it (Cordell et al. 1999). However, the authors do report on insect production and juvenile salmon diets. At the T-105 and Turning Basin sites, there was a shift in insect populations captured in fallout traps from aquatic dipterans 1996 to aphids, psyllids, and other homopterans in 1997. The authors report that the shift from insects with aquatic immature stages (shore flies, midges, biting midges) to obligate plant feeders was probably due to the large increase in riparian and emergent vegetation at these sites between 1996 and 1997. This change was also evidenced in the juvenile chinook salmon diets. The makeup of insects consumed was different between 1996, when dipterans predominated, and 1997, when psyllids, wasps, and ants were more important. The findings suggest that within 3 years after construction, the riparian area is developed to the point that riparian plant dependent insects are beginning to be produced, and utilized as a food source by juvenile salmonids.

The current definition for a vegetated buffer is native floodplain vegetation, with tree, shrub, and herbaceous layers. Buffers provide a range of functions, from minimizing human disturbance to filtering sediments from surrounding areas and moderating temperatures. In this assessment, buffers are important not only for the typical benefits they provide, but also for the value they add to adjacent habitats. In that regard, the benefits most important to consider are associated with providing organic matter in the form of leaves and litter, providing insects from riparian vegetation, and providing wildlife habitat. Information from the mitigation monitoring guidelines suggests that significant growth and coverage in vegetated buffer areas can be achieved in 5 years. Data from the Gog-Li-Hi-Te wetland site show significant increases in riparian vegetative growth within 5 years. Data from the Coastal America Sites show development of riparian vegetation-associated insect production within 5 years. Mitigation monitoring guidelines specific to Washington indicate that 90% herbaceous cover may be expected by year 5. Woody vegetation/shrub cover ranges from 50% to 80% by year 5 to 90% by year 10.

Assuming that full coverage equals sustained value, and averaging projections of time to full coverage for woody/shrub cover as the measure, then the time to sustained value for vegetated buffer habitats is 8 yrs (range from 5 to 11 yrs, n = 5). Upland greenbelts may consist of different species mixes but should be predominately native trees, shrubs, grasses and forbs. The time to sustained value for upland greenbelts is assumed to be 8 years.

Degraded habitat classification

Intertidal and shallow subtidal areas adversely effected by log rafts, wood waste deposits, and overwater structures are classified as "degraded," with a value of 0.1 because of the assumed lowered primary and secondary production, and disruption of migration/feeding behavior for juvenile salmonids resulting from reduced light, abrasion of substrate, prop wash, etc. Removal of structures and conditions adversely affecting these habitats could restore their habitat value, making them candidates for restoration projects.

The 4 year time frame selected for time to sustained value for intertidal and shallow subtidal habitats was based on data from restoration projects in Puget Sound. The projects reviewed were habitat creation projects involving excavating, regrading, or filling to create intertidal or shallow subtidal depth habitats. As noted above, in the intertidal habitat discussion, the expected sequence of invertebrate recruitment and subsequent juvenle salmonid utilization are related to initial conditions.

Habitats included in the degraded classification are intertidal or shallow subtidal habitats. Prior to the introduction of the physical impairment, they likely provided the functions associated with the habitat type. Fixed overwater structures result in environmental effects that are different from those of log raft and wood waste areas. Impacts associated with overwater structures are primarily due to shading. Log rafts, in addition to shading, impact habitat because of grounding and associated substrate scouring and compression, wood waste, and erosion from prop wash (Toews and Brownlee, 1981). Wood waste deposit areas can smother natural substrates and produce toxic leachates and anoxic conditions (Toews and Brownlee, 1981).

In general, shading is the physical factor limiting production of habitats impacted by overwater structures; removal of this impact should allow the habitat to return to near natural production quickly. Areas under log rafts that ground and wood debris areas that are cleaned up may take some time to stabilize, and are more akin to the newly placed, newly exposed habitats with regard to the time frame within which natural benthic and epibenthic production may be expected (e.g. 4 years for baseline adjusted, 8 years for full functional).

A literature review did not turn up data addressing the effects of removing overwater structures. However, there are data related to shading effects may allow some inference to be drawn on how to assign a time to sustained value following removal.

The low light environments under overwater structures affect juvenile salmonids by disrupting their behavioral and feeding patterns. Their reluctance to pass beneath piers and aprons and alteration of migratory behavior when encountering piers has been observed (Weitkamp, 1982, Pentec, 1997). Prey visibility and capture efficiency are also reduced in low light situations. Removal of the overwater structure will eliminate this impact.

Evaluation of epibenthic zooplankton production at pier apron sites in Commencement Bay (Parametrix, 1991) showed that in areas having similar substrates, salmonid prey epibenthos at apron stations (shaded) was about 83% of the abundance at non-apron stations. One distinct difference was in the occurrence of the harpacticoid copepods *Harpacticus* and *Tisbe*, which are probably the most important prey zooplankters for small juvenile salmon entering the estuary. *Tisbe* are found where there is abundant detrital vegetation, and there were no significant differences in abundance of *Tisbe* between apron and non-apron stations. However, *Harpacticus* is primarily epiphytic on growing algae and eelgrass, and was rarely found under aprons.

Investigations on the effect of shading on eelgrass may also be helpful in determining the recovery time associated with removal of overwater structures. Pentilla and Doty (1990) reported that fixed dock structures reduced eelgrass density to zero, even when light attenuation did not approach full darkness. A floating dock site, which moved with the tide and did not cast a continuous shadow over the bottom area did not have negative impacts on eelgrass density. Studies associated with impacts from the Anacortes Ferry terminal showed eelgrass presence related primarily to the height of the docks, which affected the level of shading (Parametrix and Battelle, 1996). Fresh et al. (1995) evaluated dock structures and found measurable declines in eelgrass density under and adjacent to docks in Puget Sound, except for ones that moved up and down and side to side with tidal fluctuations, eliminating constant shading. The investigations all considered sites with similar substrates in areas with homogenous eelgrass coverage, eliminating variables other than shading.

Shading appears to be the primary factor impacting primary and secondary production under overwater structures; therefore, the effect of shading on juvenile salmonid behavior will be eliminated immediately upon removal of the structure. The limited data that exist indicate that epibenthic production occurs under piers but at a level lower than unshaded sites. One study linked the absence of particular epibenthic zooplankers under pier aprons to the absence of eelgrass and algae under the aprons, a condition related to the lack of light. Studies on the effects of shading on eelgrass indicate that within a particular substrate type, eelgrass distribution is limited only by the level of shading. With the foregoing information, it is reasonable to expect that once light becomes available to natural intertidal and shallow subtidal habitats currently shaded by overwater structures, algal and vegetative production necessary to support the functions normally provided by these habitats can be achieved quickly, possibly in as little as 1 year.

Log rafts and wood debris areas have additional environmental effects that may take longer to rectify. The time frame for achieving sustained value after removal/cleanup of log rafts and wood debris should be 4 years for baseline adjusted and 8 years for fully functional, the same as that for areas newly excavated, regraded, or filled to create intertidal or shallow subtidal depth habitats.

Time to sustained value for various habitat types are provided in Table 7.

REFERENCES CITED

334 Ali, M.A. 1959. The ocular structure, retinomotor and photobehavioral responses of juvenile Pacific salmon. Canadian Journal of Zoology. 37:965-996.

335 Beauchamp, D.A., Shepard, M.F., and B.B. Pauley. 1983. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Coast): Chinook Salmon. Prepared for Waterways Experiment Station, U.S. Army Corps of Engineers, Vicksburg, Mississippi. FWS/OBS-82/11.6. 15 pp.

336 Commencement Bay Natural Resource Trustees. 2000. Commencement Bay Natural Resource Damage Assessment Restoration Monitoring Plan. Prepared by the National Oceanic and Atmospheric Administration, the U.S. Dept. of the Interior, and the State of Washington.

337 Cordell, J.R., Tear, L.M., Jensen, K. and H.A. Higgins. 1999. Duwamish River Coastal America Restoration And Reference Sites: Results From 1997 Monitoring Studies. FRI-UW-9903. Fisheries Research Institute, University of Washington, Seattle, WA.

338 Dunstan, W.A. 1955. White River Downstream Migration. Puget Sound Stream Studies (1953-1956). WA Dept. of Fisheries. In: Washington State Department of Ecology. 1998. White River Spring Chinook Habitat Guidance, A Water Quality Management Approach for the Upper White River: Version 1.0. Publication No. 98-10. 87 pp.

339 Elliott Bay/Duwamish Restoration Program. March 2000. Intertidal Habitat Projects Monitoring Program, Panel Publication 23. Prepared for the Elliott Bay Restoration Panel by C.D. Tanner, U.S. Fish and Wildlife Service.

340 Fresh, K.L., B. Williams, and D. Pentilla. 1995. Overwater Structures and Impacts on Eelgrass in Puget Sound, Washington. Puget Sound Research, 1995 Proceedings. (abstract)

341 Kentula, M.E., Brooks, R.P., Gwin, S.E., Holland, C.C., Sherman, A.D., and J.C. Sifneos. 1992. An Approach to Improving Decision Making in Wetland Restoration and Creation. Edited by A.J. Hairston. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

342 King, D.M., Bohlen, C.C., and K. Adler. 1993. Appendix C. Watershed Management And Wetland Mitigation: A Framework For Determining Compensation Ratios. *In* King, D. and C. Bohlen. 1995. The Cost Of Wetland Creation And Restoration. Final Report prepared under contract No. DE-AC22-02MT92006, U.S. Dept. of Energy, Bartlesville, Oklahoma. 107 pp.

343 Kusler, J.A. and M.E. Kentula (eds) 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, D.C.

344 Lassuy, D.R. 1989. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest): English Sole. Prepared for Waterways Experiment Station, U.S. Army Corps of Engineers. Vicksburg, Mississippi. Biological Report 82(11.101). 17 pp.

345 Mitsch, W.J., and Wilson, R.F. 1996. Improving the success of wetland restoration with know-how, time, and self-design, Ecol. Appl. 6, 77-83. *In:* Simenstad, C.A. and J.R. Cordell.2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecological Engineering 15 (2000)283-302.

346 Mockler, A. 1998. Sensitive Area Mitigation Guidelines. King County Dept. of Development and Environmental Services. Renton, WA. 14 pp.

347 Northcote, T.G., N.T. Johnston, and K. Tsumura. 1976. Benthic, epibenthic and drift fauna of the lower Fraser River. Tech. Rep. 11, Westwat. Res. Cent., Univ. Brit. Col., Vancouver, Canada, 227 pp. *In* Simenstad, C.A. 1983. The Ecology of Estuarine Channels of the Pacific Northwest Coast: A Community Profile. U.S. Dept. of the Interior, Fish and Wildlife Service Report FWS/OBS-83/05. 181 pp.

348 Ossinger, M. (principal author) 1999. Success standards for Wetland Mitigation Projects - a Guideline. Washington State Dept. of Transportation. 31 pp.

349 Pacific Estuarine Research Laboratory. 1990. A manual for assessing restored and natural coastal wetlands with examples from southern California. California Sea Grant Report No. T-CSGCP-021. La Jolla, California. 105 pp.

350 Parametrix, Inc. 1991. Epibenthic Zooplankton Production and Fish Distribution At Selected Pier Apron and Adjacent Non-Apron Sites in Commencement Bay, Washington. Report Prepared for Port of Tacoma, Tacoma, Washington.

351 Parametrix, Inc. 1998. Sitcum Waterway Remediation Project: Milwaukee Habitat Area Monitoring Report, 1996. October 1998.

352 Parametrix, Inc. and Battelle Marine Sciences Laboratory. 1996. Anacortes Ferry Terminal eelgrass, macroalgae, and macrofauna survey report. Report for Sverdrup Civil, Inc. and WSDOT. 12 pp.

353 Pentec Environmental, Inc. 1997. Movement of Juvenile Salmon Through Industrialized Everett Harbor. Report to Port of Everett. (abstract)

354 Pentilla, D. and D. Doty. 1990. Progress Report. Results of 1989 Eelgrass Shading Studies in Puget Sound. Washing Dept. of Fisheries, Marine Fish Habitat Investigations Division. (abstract)

355 Seliskar, D.M., and Gallagher, J.L. 1983. The Ecology Of Tidal Marshes Of The Pacific Northwest: A Community Profile. U.S. Dept. of the Interior, Fish and Wildlife Service Report FWS/OBS-82/32. 65 pp.

356 Simenstad, C.A. 1983. The Ecology of Estuarine Channels of the Pacific Northwest Coast: A Community Profile. U.S. Dept. of the Interior, Fish and Wildlife Service Report FWS/OBS-83/05. 181 pp.

357 Simenstad., C.A., Fresh, K.L., and E.O. Salo. 1982. The Role of Puget Sound and Washington Coastal Estuaries in the Life history of Pacific Salmon: An Unappreciated Function. p. 343-364, *In* V.S. Kennedy (ed.), Estuarine Comparisons. Academic Press, New York.

358 Simenstad, C.A., Cordell, J.R., Milward, D.M., and E.O. Salo. 1985. Diet composition of juvenile salmon (*Oncorhynchus spp.)* in an urbanized estuary: results of three years studies in Commencement Bay, Puget Sound, Washington. Final report to the Port of Tacoma, Tacoma, Washington. 71 pp.

359 Simenstad, C.A., Tanner, C.D., Thom, R.M., and L.L. Conquest. 1991. Estuarine Habitat Assessment Protocol. Report prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA. EPA 910/9-91-037. 201 pp.

360 Simenstad., C.A. H.B. Anderson, J.R. Cordell, and L. Hallum.. 1993. Analysis of Changes in Benthic and Epibenthic Communities in Commencement Bay, Washington. Final Report to U.S. Army Corps of Engineers - Seattle District, Contract No. DACW67-90-D-0008. January 26, 1993. 88 pp + attachments.

361 Strange, E., Galbraith, H., Bickel, S., Mills, D., Beltman, D., and J. Lipton. 1999. Determining Ecological Equivalence In Service-To-Service Scaling Of Salt Marsh Restoration. Stratus Consulting, Inc., Boulder, Colorado. 44 pp (unpublished)

362 Thom, R.M., Simenstad, C.A., Cordell, J.R., and L. Hamilton. 1991. The Gog-Li-Hi-Te Wetland System in the Puyallup River Estuary, Washington: Phase V Report: Five Year Monitoring, January-December 1990. Final Report to Port of Tacoma. Fisheries Research Istitute, University of Washington, Seattle.

363 Toews, D.A., and M.J. Brownlee. 1981. A Handbook For Fish Habitat Protection On Forest Lands In british Columbia. Dept. of Fisheries and Oceans, Vancouver, B.C. 166 pp.

364 Toole, C.L., Barnhart, R.A., and C.P. Onuf. 1987. Habitat Suitability Index Models: Juvenile English Sole. U.S. Dept. of the Interior, Fish and Wildlife Service Biological Report 82(10.133) February 1987.

365 U.S. Fish and Wildlife Service. 1980. Habitat Evaluation Procedures. ES< 102. Fish Wildl. Serv., Div. Ecol., Washington, D.C.

366 U.S. Army Corps of Engineers (1999) "Examples of Performance Standards for Wetland Creation and Restoration in Section 404 Permits and an Approach to Developing Performance Standards" WRP Technical Note WG-RS-3.3, January 1999. 14 pp.

367 Weitkamp, D.E. 1982. Juvenile chum and chinook salmon behavior at Terminal 91, Seattle, Washington. Report by Parametrix Inc. to Port of Seattle, Washington. 21 pp.

368 Williams, R.Q., R.M. Laramie, and J.J. Ames. 1975. A Catalog of Washington Streams and Salmon Utilization - Volume 1, Puget Sound. WA Dept. of Fisheries.

Table 1. Relative habitat values for juvenile chinook salmon (and bird assemblages).

Habitat Type	Relative Habitat Value for Juvenile Chinook Salmon (and Bird Assemblages)
Estuarine habitats	
Marsh	1.0
Low Intertidal Sand/Silt Substrate	0.75
High Intertidal Sand/Silt Substrate	0.67
Low Intertidal Structurally Complex Substrate	0.60
High Intertidal Structurally Complex Substrate	0.50
Shallow Subtidal Sand/Silt Substrate	0.40
Shallow Subtidal Structurally Complex Substrate	0.40
Deep Subtidal Sand/Silt Substrate	0.05
Deep Subtidal Structurally Complex Substrate	0.05
Buffer Habitats	
Vegetated Buffer	0.50
Upland Greenbelt	0.20

Table 2. Relative habitat values for English sole based on substrate composition.

Substrate composition - percentage by weight of substrate particle size >2mm in diameter	Relative habitat value
# 20	1.0
21 - 30	0.86
31 - 40	0.60
41 - 50	0.33
> 50	0.15

Table 3. Relative habitat values for juvenile chinook salmon, birds, and juvenile English sole; and species' combined habitat values.

Habitat Type	Relative Value for Salmon	Relative Value for Birds	Relative Value for English sole	Relative Combined Value for all Species
(High) Intertidal Sand/Silt Substrates	0.67	0.67	1.00	0.75
(High) Intertidal Structurally Complex Substrates (21-30% rock)	0.50	0.50	0.86	0.59
(High)Intertidal Structurally Complex Substrates (31-40% rock)	0.50	0.50	0.60	0.53
(High) Intertidal Structurally Complex Substrates (41-50% rock)	0.50	0.50	0.33	0.46
(High) Intertidal Structurally Complex Substrates (> 50% rock)	0.50	0.50	0.15	0.41
(Low) Intertidal Sand/Silt Substrates	0.75	0.75	1.00	0.81
(Low) Intertidal Structurally Complex Substrates (21-30% rock)	0.60	0.60	0.86	0.67
(Low) Intertidal Structurally Complex Substrates (31-40% rock)	0.60	0.60	0.60	0.60
(Low) Intertidal Structurally Complex Substrates (41-50% rock)	0.60	0.60	0.33	0.53
(Low) Intertidal Structurally Complex Substrates (> 50% rock)	0.60	0.60	0.15	0.49
Shallow Subtidal Sand/Silt Substrates	0.40	0.40	1.00	0.55
Shallow Subtidal Structurally Complex Substrates (21-30% rock)	0.40	0.40	0.86	0.52
Shallow Subtidal Structurally Complex Substrates (31-40% rock)	0.40	0.40	0.60	0.45
Shallow Subtidal Structurally Complex Substrates (41-50% rock)	0.40	0.40	0.33	0.38
Shallow Subtidal Structurally Complex Substrates (> 50% rock)	0.40	0.40	0.15	0.34
Deep Subtidal, Fine Substrate	0.05	0.05	1.00	0.29
Deep Subtidal, Structurally Complex (21-30% rock)	0.05	0.05	0.86	0.25

Deep Subtidal, Structurally Complex (31-40% rock)	0.05	0.05	0.60	0.19
Deep Subtidal, Structurally Complex (41-50% rock)	0.05	0.05	0.33	0.12
Deep Subtidal, Structurally Complex (> 50% rock)	0.05	0.05	0.15	0.08
Estuarine Marsh	1.00	1.00	1.00	1.00
Vegetated buffer	0.50	0.50	0.00	0.38
Upland Greenbelt (buffer areas)	0.20	0.20	0.00	0.15

Table 4. Habitat type acreage and calculations used in combining habitat types. Weighted values are in bold.

Habitat Type	Acres	Value [1]	Percent [2]	High intertidal weighted value [3]	Intertidal weighted value
(High) Intertidal Sand/Silt Substrates	38.501	0.75	0.86552	0.64913	0.324593
(High) Intertidal Structurally Complex Substrates (21-30% rock)	3.1247	0.59	0.07024	0.041445	0.020724
(High)Intertidal Structurally Complex Substrates (31-40% rock)	1.6078	0.53	0.03614	0.019156	0.009579
(High) Intertidal Structurally Complex Substrates (41-50% rock)	1.2494	0.46	0.02808	0.01292	0.00646
Total acres (high intertidal	44.82			**0.722664**	
(High) Intertidal Structurally Complex Substrates (> 50% rock)	0.3491	0.41			
				Low intertidal weighted value	
(Low) Intertidal Sand/Silt Substrates	38.5537	0.81	0.86682	0.702127	0.35104
(Low) Intertidal Structurally Complex Substrates (21-30% rock)	2.9753	0.67	0.06689	0.04482	0.022408
(Low) Intertidal Structurally Complex Substrates (31-40% rock)	1.7785	0.6	0.03998	0.023992	0.011995
(Low) Intertidal Structurally Complex Substrates (41-50% rock)	1.1695	0.53	0.02629	0.013936	0.006968
Total acres (low intertidal)	44.72			**0.784875**	
Total acres (all intertidal)	89.54				**0.753767**
(Low) Intertidal Structurally Complex Substrates (> 50% rock)	0.2484	0.49			
				Shallow subtidal weighted value	
Shallow Subtidal Sand/Silt Substrates	26.5777	0.55	0.81597	0.448786	
Shallow Subtidal Structurally Complex Substrates (21-30% rock)	2.6433	0.52	0.08115	0.0422	
Shallow Subtidal Structurally	2.7511	0.45	0.08446	0.038008	

Complex Substrates (31-40% rock)					
Shallow Subtidal Structurally Complex Substrates (41-50% rock)	0.5996	0.38	0.01840	0.006995	
Total acres (shallow subtidal)	32.63			**0.53599**	
Shallow Subtidal Structurally Complex Substrates (> 50% rock)	0.0621	0.34			
				Deep subtidal weighted value	
Deep Subtidal, Fine Substrate	202.5522	0.29	0.93012	0.26973	
Deep Subtidal, Structurally Complex (21-30% rock)	10.9997	0.25	0.05051	0.01262	
Deep Subtidal, Structurally Complex (31-40% rock)	3.8033	0.19	0.01746	0.00331	
Deep Subtidal, Structurally Complex (41-50% rock)	0.4137	0.12	0.0019	0.00022	
Total acres (deep subtidal)	217.78			**0.2859**	
Deep Subtidal, Structurally Complex (> 50% rock)	0.0161	0.08			

1 - Relative habitat value for juvenile chinook salmon, juvenile English sole, and birds.

2 - Acres of individual habitat type/total acres for category.

3 - Relative habitat value * percent of individual habitat type

Table 5. Hylebos Waterway habitat classifications, area and weighted value.

Number	Habitat Type	Acres	Value
1	Estuarine Marsh	0	1.00
2	Intertidal	88.96	0.75
3	Shallow Subtidal	32.57	0.55
4	Deep Subtidal	217.77	0.30
5	Rip-rap	0.68	0.10

1 - no existing habitat is classified as marsh

2 - the combined acreage and rounded weighted value (calculated value = 0.754) for high and low intertidal habitats (sand/silt through 41-50% rock).

3 - the combined acreage and rounded weighted value (calculated value = 0.536) for shallow subtidal habitats (sand/silt through 41-50% rock).

4 - the combined acreage and rounded weighted value (calculated value = 0.286) for deep subtidal habitats (sand/silt through 41-50% rock).

5 - the combined acreage and assigned value to structurally complex habitats (>50% rock) classified as rip rap.

Table 6. Hylebos Waterway habitat classifications and values applied in the HEA.

Habitat	Value		
	Fully Functioning	Baseline Adjusted	Degraded
Estuarine Marsh	1.0	0.85	NA
Intertidal	0.9	0.75	0.1
Shallow Subtidal	0.7	0.55	0.1
Deep Subtidal	0.3	0.3	0.1
Rip-rap	NA	NA	0.1

Table 7. Restoration project habitat values and time to sustained value for fully functional (FF) and baseline adjusted (BA) habitats.

Habitat	final value and percent of final value (%) at end of year			
	1	4	8	15
Habitats formed through excavation, regrading, or material placement.				
Estuarine Marsh	not applicable	0.825 (82.5%) FF 0.75 (75.0%) BA	0.936 (93.6%) FF 0.786 (92.4%)BA	1.0 (100%) FF 0.85 (100%) BA
Intertidal	not applicable	0.825 (91.6%) FF 0.75 (100%) BA	0.9 (100%) FF no change	no change
Shallow Subtidal	not applicable	0.63 (90.0%) FF 0.55 (100%) BA	0.7 (100%) FF no change	no change
Existing FF or BA habitats restored by over water structure removal				
Intertidal	0.9 (100%) FF 0.7 (100%) BA	no change	no change	no change
Shallow Subtidal	0.7 (100%) FF 0.55 (100%) BA	no change	no change	no change
Existing FF or BA habitats restored by removal or log rafts or wood waste				
Intertidal	not applicable	0.825 (91.6%) FF 0.75 (100%) BA	0.9 (100%) FF no change	no change
Shallow Subtidal	not applicable	0.63 (90.0%) FF 0.55 (100%) BA	0.7 (100%) FF no change	no change
Other				
Vegetated Buffer	not applicable	0.2 (50%)	0.4 (100%)	no change
Upland Greenbelt	not applicable	0.075 (50%)	0.15 (100%)	no change

Figure 1. Consolidation of habitat types.

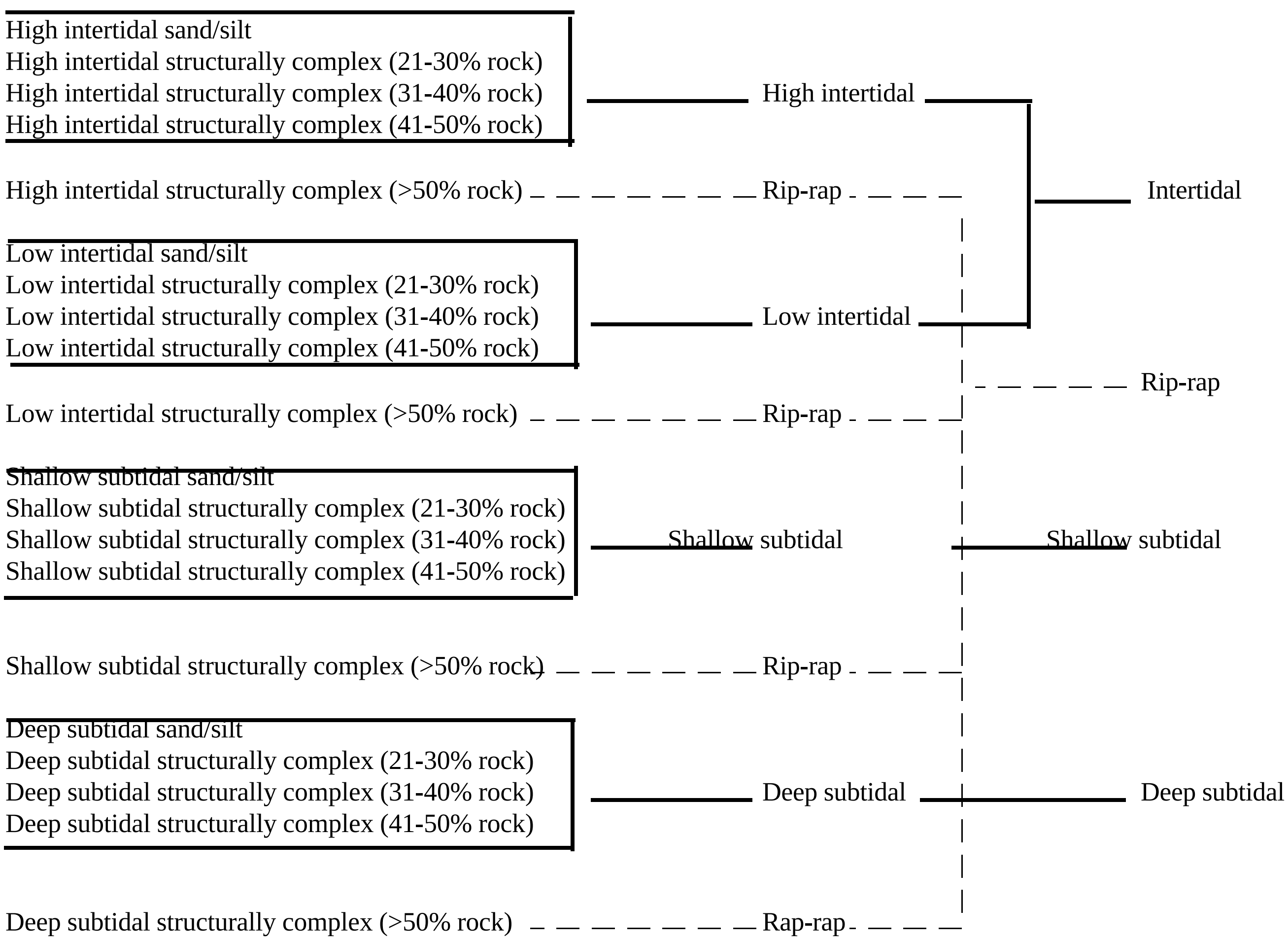

Figure 2. Representation of habitat classified as "fully functional" based on influence of adjacent vegetated buffer.