APPENDIX D: HABITAT VALUATION IN THE LOWER DUWAMISH RIVER AND DETERMINATION OF TIME TO SUSTAINED FUNCTION

(Adapted from Appendix C of March 14, 2002 Hylebos Waterway Natural Resource Damage Settlement Proposal Report)

Habitat Valuation Introduction

For the purposes of the Lower Duwamish River Habitat Equivalency Analysis (HEA), habitats are valued by how well they support juvenile Chinook salmon, four bird assemblages that are representative of avian species occurring in the area, and juvenile English sole. These values are based on a habitat's potential to provide attributes that support feeding and refuge needs of these species and groups. 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

Fifty three species of resident and non-resident fish were captured in recent remedial studies on the Lower Duwamish River (Lower Duwamish Waterway Group, 2010), including eight species of anadromous salmonids (Kerwin & Nelson, 2000). Chinook, coho, chum, and steelhead are common. Pink salmon, sockeye, sea-run cutthroat trout and bull trout are rare. Juvenile Chinook salmon and juvenile English sole are used as representative fish species to assess the value of habitat to fish. This is because juvenile Chinook salmon and juvenile English sole have feeding modes, behavioral characteristics, and habitat requirements that sufficiently overlap those of similar species found in the LDR so as to consider them appropriate surrogates for fish in LDR.

Bird assemblages rather than individual species are used to assess habitat value to birds along the LDR. The bird assemblages are grouped as a function of their foraging behavior and include both resident and migratory species found in the river. Because birds use similar habitat types as juvenile Chinook salmon and are linked with them through their food webs, habitat value for birds is linked to habitat value for juvenile Chinook salmon.

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 in the manner described below.

LDR Species Specific Habitat and Habitat Values

The LDR is an estuarine waterway. There is considerable information on the utilization of estuarine environments by anadromous salmonids, flatfishes, and birds. However, much of the information is qualitative and while it is useful in identifying what constitutes essential habitat, it is of less value in determining how specific habitat attributes relate to habitat value.

Habitat for Chinook salmon

Estuaries are particularly important to juvenile Chinook salmon, which may have the longest estuarine residence time of juvenile salmonids. Estuarine habitats are used as refuge from predators, foraging, and temporary residence during physiological transition for seawater acclimation (Simenstad et al., 1982). There is considerable information regarding the value of estuaries to juvenile salmon but much of it is qualitative, describing generalized relationships and attributes, rather than providing value measurements. 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 Trustees reviewed available information and assigned relative values of habitat for juvenile Chinook salmon based on the reasons set forth below.

Chinook salmon in the LDR consist primarily of summer/fall run fish. Spring Chinook are found occasionally in the Green River, which feeds upstream into the LDR, but it is not known if these fish constitute a self-sustaining run. Chinook in the LDR are a mixture of natural spawning and hatchery fish. Natural spawners are classified as ocean type fish because they typically spend little time in fresh water after emerging from eggs laid in the gravel. It is believed juvenile Chinook migrate from the LDR to the ocean from January through August; however, the complete migratory time period for juvenile Duwamish/Green River fall Chinook is not currently known (Kerwin & Nelson, 2000). Juveniles have been found in the Lower Duwamish through September, and may remain in the estuary even longer. Naturally spawning summer/fall Chinook juveniles generally remain in upstream areas for two to three months following emergence from eggs. They then begin their migration to the estuarine areas of the LDR (Williams et al., 1975, Kerwin & Nelson, 2000). Typically, the Green/Duwamish river basin summer/fall Chinook migrate within their first year of life.

Because of their extended estuarine residence, and the diversity of size classes, juvenile Chinook consume a diversity of prey and use a variety of estuarine habitats, shifting to reflect changes in food habits as they grow (Simenstad et al., 1982). Estuaries provide a diverse array of prey organisms, often in large populations, which allows juvenile salmon to sustain relatively high growth rates while occupying a refuge from predators. Chinook occupy three main zones of the LDR. Smaller individual Chinook occur primarily in the freshwater transition zone in the upper portion of the LDR where they feed on larval and adult aquatic insects, terrestrial insects, and epibenthic organisms. Larger subyearling fish move to tidal flats, gravel-cobble shorelines, and other shallow water habitats where they feed on epibenthic crustaceans such as *gammarid* amphipods, *mysids*, and *cumaceans*. Yearling Chinook occupy the open water habitat of the lower estuary, and may prefer habitats within confined embayments, where they feed on small nekton, insects, mysids, larval fish, and nuestonic drift organisms. Sampling in the LDR in 2005 documented the presence of the various life and transitional stages within the LDR as well as the importance of each of these three estuarine zones for juvenile Chinook salmon (Ruggerone et al, 2006).

Growth of juvenile Chinook while in the LDR may help increase their survival rates by narrowing the window of vulnerability to predators once they enter the ocean. Residence time in the estuary is related to foraging success and growth in the ocean, increasing marine survival. This suggests that the distribution and abundance of principal habitat types and the availability of prey for Chinook may be a reflection of salmon populations migrating through the system.

Thus, 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 appendix, relative values are assigned to habitat types using available information on the feeding and refuge functions of different habitats for juvenile Chinook salmon, the functional rarity of habitats in the LDR (e.g., tidal marshes, an important habitat for Chinook, are virtually nonexistent), and best professional judgment. References consulted included not only literature on juvenile salmonid habitat utilization and feeding preferences, but also information on the frequency of occurrence of preferred food organisms (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 uncontaminated habitats.

Three estuarine habitat types, based on tidal elevation, are used in the LDR injury assessment: intertidal, shallow subtidal and deep subtidal. In addition, other habitats are identified for use in valuing potential restoration projects: marsh (intertidal habitat containing aquatic vascular plants), vegetated buffer (an upland zone adjacent to the aquatic habitat consisting of native floodplain vegetation with trees and shrubs), upland greenbelt (a vegetated zone landward of the vegetated buffer and outside of the shoreline zone, with trees and shrubs) and rip-rap (rock armor placed along shorelines to protect against erosion).

Chinook Habitat Values

Habitat values are unit-less numbers, based on relative, rather than absolute values, similar to the concept used in the Habitat Suitability Index (HIS) (USFWS 1980). LDR habitat types are assigned values for juvenile Chinook salmon using data from sediment composition and water depth surveys.

Habitat values were assigned to estuarine habitats, ranging from one (optimal conditions) to zero (unsuitable conditions). Each habitat value is relative to the value given to marsh with an associated vegetated buffer which is considered to be the best habitat available for the representative species in the LDR. The reason marsh with a vegetated buffer is considered the optimal habitat, and thus given the highest value is based on several factors. Marsh vegetation provides an environment that increases epibenthic and benthic production and available food for Chinook salmon. It provides an important refuge from predators-and is a scarce habitat type in the LDR estuary.

Habitat values related to elevation (referenced to mean lower low water (MLLW) are assigned, from highest to lowest, to marsh (+ 6 to + 12 ft), intertidal (-4 to +12 ft), shallow subtidal (-4 to -14 ft), and deep subtidal (<-14 ft) (Table D1). These are based on larger number of species and greater populations of food for Chinook and on primary productivity and habitat use (Northcote et al., 1976; Simenstad et al, 1993). With depth, available light decreases, which results in fewer salmonid prey species and hence, a lower habitat value).

Table D 1. Habitat classifications used in the HEA.

Habitat Type	**Elevation ft. (MLLW)**
Marsh (aquatic vascular vegetation)	+6 to +12
Intertidal	-4 to +12
Shallow Subtidal	-14 to -4
Deep Subtidal	< -4

Benthic community structure is affected by a variety of conditions. Different species colonize different substrate types, and mixed substrates (sand, gravel, and cobble) can provide abundant prey species and suitable refuge habitat for juvenile salmonids. In this analysis, habitat classifications are based only on depth and silt, sand, coarse sand, and fine gravel substrates are combined. Shallow, low gradient, unconsolidated sediments are assumed to provide more prey organisms consumed by juvenile salmonids and are thus assigned higher values than structurally complex sediments such as riprap. Deep subtidal habitats (-14 ft and deeper) 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) that feed on benthic organisms.

(2) Waders, which prey on similar, though somewhat deeper benthic organisms than those that prey on the surface and shallow water (e.g. Greater yellowlegs), or prey on small fishes and crustaceans (e.g. Great blue heron).

(3) Surface and diving water birds, which include birds that find prey in deeper waters (e.g. Western grebe, Common merganser, mallard).

(4) Aerial searchers, which include omnivores and carnivores that find prey in all habitats (e.g. Osprey, belted kingfisher, Glaucous-winged gull).

Different types of estuarine birds use different foraging behaviors and thus, require diverse habitats for feeding and resting. Since certain bird species from these assemblages share common habitats and prey items with juvenile salmon, the value of habitat for salmon is related to its value for birds.

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, they are restricted to the high intertidal area and the part of the lower intertidal area exposed at low tide. Tidal fluctuations affect habitat utilization by waders that feed on benthic organisms and by surface and diving waterbirds. 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 the LDR HEA, we assume that the value of a particular habitat type to estuarine birds is the same as the habitat value assigned to salmon (Table D2).

Table D 2. 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
Intertidal	0.67
Shallow Subtidal	0.40
Deep Subtidal	0.05
Rip-rap	0.10
Buffer Habitats	
Vegetated Buffer	0.50
Upland Greenbelt	0.20

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 capacity of the habitat to support the species (Figure D1). 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 one indicates optimal conditions, and a value of zero indicates unsuitable conditions. The HSI is based on the concept of limiting factors. A limiting factor is a component of an organisms' environment that can affect its growth, reproduction, or distribution. The availability of food, shelter or predation pressure, are examples of factors that could be limiting for an organism. Using this concept, the HIS is set at the value of the lowest limiting factor. Habitat values for English sole in the LDR are expressed in terms of HSI. There are two components in the model: water quality, with habitat variables related to bottom salinity, dissolved oxygen, and bottom water temperature; and food, with habitat variables related to the hydrodynamic regime and dominant sediment type.

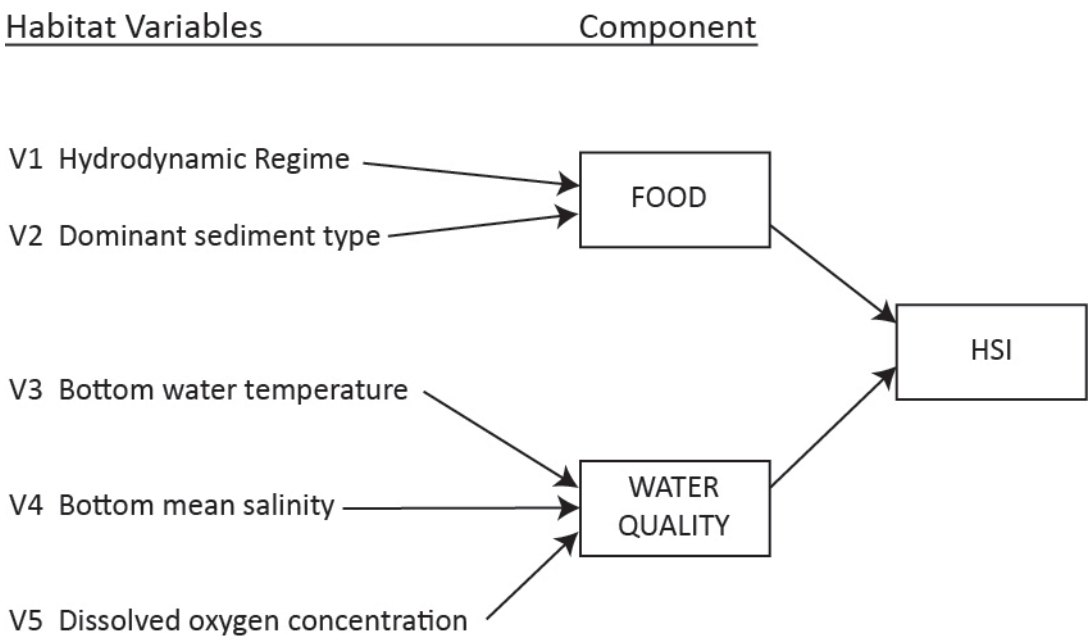

Figure D 1. The relationship of habitat variables, life requisite components, and the habitat suitability index (HSI) for juvenile English sole in estuaries and coastal lagoons. (Toole, C.L., Barnhart, R.A., and C.P. Onuf. 1987. Habitat Suitability Index Models: Juvenile English Sole. U.S. Dept. of the Interior, Fish and Wildlife Service Biological Report 82(10.133) February 1987.)

Because of the river flow and tidal exchange in the LDR, 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 model, 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. The LDR 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 LDR habitats is calculated based on dominant substrate.

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 sediment 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). Sole that have 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 presented in the HSI model, 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 less than 20% particles greater than 2 mm in diameter to 0.15 for substrate with less than 50% particles, smaller than two 2 mm in diameter (Table D3). The predominant substrate in the LDR consists of sand/silt, therefore a value of 1.0 is used to value habitats for English sole living in the LDR.

Table D 3. Relative habitat values for English sole based on substrate composition.

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

Combined Habitat Values

Seven habitat types were identified for use in this analysis for injury determination and 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 are included that may not have been injured in the LDR, but may provide considerable restoration benefit.

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. There is 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 habitat types are shown in Table D4.

Table D 4. 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
Intertidal	0.67	0.67	1.00	0.75
Shallow Subtidal	0.40	0.40	1.00	0.55
Deep Subtidal	0.05	0.05	1.00	0.29
Marsh	1.00	1.00	1.00	1.00
Rip-rap	0.10	0.10	0.15	0.11
Vegetated buffer	0.50	0.50	0.00	0.38
Upland Greenbelt	0.20	0.20	0.00	0.15

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 associated with restoration project development. The LDR is in an urban/industrial/commercial setting, with extensive shoreline development. There are only remnant marshes and few upland areas that could be classified as functioning vegetated buffer habitat (TerraLogic GIS, Inc. & Landau Associates, 2004). We created value adjustment categories of "fully functional" and "baseline adjusted" to apply 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 an ecological complex that enhances overall production. Habitats considered "baseline adjusted," have no adjacent habitat to enhance their value. As an example, the presence of insect and organic matter is increased when it is placed adjacent to a vegetated buffer. Also, created marsh habitats provide benefits that increase the value of adjacent habitats. Thus, the creation of a habitat that increases 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).

All of the intertidal and shallow subtidal habitats in the LDR are considered "baseline adjusted," with little to no adjacent habitat to enhance their value. This provides for the values

of 0.75 for intertidal and 0.55 for shallow subtidal. For purposes of restoration planning, an enhancement of 0.15 is added to intertidal and shallow subtidal habitats constructed in association with a vegetated buffer or a fully functioning marsh. Therefore, fully functional values for intertidal and shallow subtidal habitats in LDR are 0.9 and 0.7, respectively (Table D5).

Table D 5. LDR 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

The premise for a fully functional classification in the LDR is that habitat complexes (e.g. a mix of marsh, mudflat and riparian) are necessary for proper ecosystem functioning. Marsh habitat alone and in optimal condition was assigned a value of 1.0. 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 do not receive the 0.15 enhancement and are consequently assigned a maximum value of 0.85 rather than 1.0.

In summary, for restoration planning in the LDR, fully functional value is given to the following:

- 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;
- a shallow subtidal habitat must be associated with an adjacent fully functioning intertidal habitat.

LDR restoration projects involving the creation of each of these habitats will be considered fully functional and valued as such (Table D5). All other types of restoration projects involving less complex habitats will be considered baseline with a lower value relative to the fully functional value.

Development in the LDR has resulted in facilities and activities that physically degrade habitat quality. The presence of large over-water structures such as piers, aprons and buildings 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 decrease the value of habitats that are severely impacted by physical obstructions.

Potential impacts associated with severe physical habitat degradation warrant application of a lower habitat value in certain situations. Examples of physical habitat degradation that result in lower values are reduced light and disruption of migration and feeding behavior. 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. 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 and only to situations causing severe physical impacts.

Overwater structures include permanent and semi-permanent structures such as piers, aprons, buildings, boathouses, and houseboats. Because a juvenile salmonids' 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 and shallow water oriented. 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; however, lower light levels may also interfere with feeding. Moreover, structures covering intertidal and shallow subtidal habitat limit the available light to bottom substrates in the productive near-shore photic zone and have more impact on epibenthic production than those in deeper water. As a result, with 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 and near shore shading. They have less of an impact than structures extending from the shoreline, and are consequently not placed in the degraded category. Habitat beneath them is included in the baseline adjusted value, dependent 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.

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 to representative species selected for the LDR. It is to be used for the sole purpose of classifying habitat values for the LDR HEA in as simple and as equitable of a manner as possible. Value adjustments associated with environmental conditions are shown in Table D5.

TIME TO SUSTAINED VALUE

Introduction

The assumption that environmental injury or habitat loss can be compensated 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, 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 (Kentula et al. 1992). 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, with a sustained functional value. There are two components to this function, the shape of the curve and the time to maturation.

Shape of the Curve

The shape of the curve means how the recovery appears when graphed and allows for a picture of the rate of increase in a habitat's recovery. The shape 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. A study on the use of different curves to describe the increase in wetland functions as created wetlands develop found that, for the purposes of evaluating restoration, the shape of the curve was not important and resulted in minimal percentage differences 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 an ecological function will increase along a linear path until sustained value is achieved.

The number of years after construction when the restoration project is expected to achieve sustained value varies with habitat type (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 time 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 judgment 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 included 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 habitats 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. Certain types of habitats carry more risk of failure than others. 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 that can assist natural processes to achieve successful restoration projects include: developing and amending soil, transplanting plants, controlling weeds, including invasive and non-native species and other eco-engineering methods. 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

Achieving the expected sequence of invertebrate recruitment and subsequent use by juvenile salmon, juvenile English sole, and birds is related to the initial condition of the habitat. The farther initial conditions are from a mature steady state, the longer a habitat will take to approach a self-sustaining level (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 these projects used excavation, regrading or filling 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). The data indicate that newly placed, newly exposed, and sometimes, newly wetted materials require time to develop 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, and salinity. Although the numbers of epibenthic invertebrates were often highly variable from year to year, by years three to four, 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. For a newly created LDR habitat, four 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. This is generally eight years for vegetated buffer habitat.[1]

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

Marsh habitats

Marsh habitat is assumed to include both dendritic and fringing marshes. Success in creating estuarine habitats that support aquatic vascular plants has been mixed in the Puget Sound area. In other regions where salt marshes have been created, it is still unclear how well they actually replicate the ecological functions of natural marshes. The report 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 five years. When the metric is community and ecosystem function, recovery was slower and was generally 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 creation of a marsh that duplicates a natural marsh is not possible (Kusler and Kentula, 1990). This is because of the complexity and variation in natural marshes, and the subtle relationships among hydrology, soils, vegetation, nutrients, and animal life. 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 within 15 years. Therefore, after this time, it is assumed to be a fully functional marsh with a value of 1.0 or a baseline adjusted marsh with a value of 0.85.

In the LDR, marsh habitat may be created in sand/silt substrates in the + 6 to + 12 ft elevation range. Depending on location, substrate, and salinity, low marsh (+ 6 to + 10 ft) and/or high marsh (+ 10 to + 12 ft) could be expected. This elevation range is included in intertidal habitat (- 4 to + 12 ft). 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 natural 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 (having a value of 0.9) in years zero through eight, then increases more gradually to the marsh value of 1.0 between years eight and 15. A baseline adjusted marsh is valued the same as baseline adjusted intertidal habitat with a value of 0.75 through year four, when it then increases gradually to its sustained marsh value of 0.85 between years five and 15.

Vegetated buffer and upland greenbelt

There is considerable information on the value and size requirements of vegetated buffers but much less on rates of development. Planting riparian buffer is part of several restoration projects in the Puget Sound area, e.g. Middle Waterway Shore Restoration Project in Commencement Bay Sitcum Waterway Remediation Project), but there is, as yet, insufficient data upon which to draw conclusions about how long it takes them to become fully functional. Related information is available to 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)

In the U.S. Army Corps of Engineers' "Examples of Performance Standards for Wetland Creation and Restoration in Section 404 Permits and an Approach to Developing Performance Standards" (USACOE 1999) most monitoring programs for vegetated buffers (riparian, shrub-scrub, and woody vegetation) extend for five 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 five
- Maryland—85% of site vegetated by planted species and/or naturally regenerated vegetation by year five
- Maryland—85% herbaceous cover, 75% areal cover by planted woody species by year two
- Alaska—vegetative cover equal to 75% of test plot cover in five years
- Washington—60% cover by native shrub species by year five

An example of Seattle District ACOE 1994 monitoring guidelines for freshwater wetlands required 80% cover of native shrub/scrub species after five years and 40% canopy cover of native species forest vegetation after 20 years (USACOE, 1999). Ossinger et al. (1999), reported 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. This report suggests benchmark values for herbaceous vegetation as 80% cover by year three, and 90% cover by year five. For woody cover (wetland buffer/forested zone) they suggest 50% cover by year five.

Developing guidelines for King County, Mockler (1998) suggested that buffers, defined as dense vegetation that will protect wetland from human encroachment and provide wildlife habitat, should have 60% emergent cover by year one, 80% by year three, and 90% by year five. Shrub or sapling tree cover should be >60% by year three.

A success criterion for establishing riparian vegetation 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 at the end of year five, the shrub layer is expected to be >50% and the tree layer >40 percent. Both native trees and shrubs should cover at least 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 was not included in the project design.

Duwamish River Coastal America sites included planted upland riparian vegetation, and monitored three years post-construction. Though there was no data provided on the post-construction monitoring (Cordell et al. 1999), insect production and juvenile salmon diets were reported. At the T-105 and Turning Basin sites, there was a shift in species composition of insect populations captured in fallout traps from 1996 to 1997. Insects with aquatic immature stages (shore flies, midges, biting midges) shifted to terrestrial insects and the authors conclude that this was probably due to the large increase in riparian and emergent vegetation at these sites between 1996 and 1997. This change also occurred in the juvenile Chinook salmon diets. The makeup of insects consumed was different between 1996 and 1997. The findings suggested that within three years after construction, the riparian area developed to the point that insects dependent on riparian plants were beginning to be produced and were utilized as a food source by juvenile salmonids. In 1999, there was a shift back to the insects dominant in 1996, leading the authors to speculate that the vegetation assemblages that support the insects might not yet be stable (Cordell et al, 2001). However, the study reported that although survival and expansion of riparian areas were not monitored, they appeared to have become established successfully.

Monitoring results for riparian vegetation coverage from LDR restoration projects constructed via the Elliott Bay Panel do not provide a good measure of natural succession over time due to complications associated with routine maintenance to remove debris, invasive and non-native species and replanting. The year five goals of >50% tree cover and >40% shrub cover were met at Herring's House, Hamm Creek, and North Winds Weir (USFWS, 2008).

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 most important benefits are providing organic matter in the form of leaves and litter, providing insects from riparian vegetation, and providing wildlife habitat. Mitigation monitoring guidelines suggest that significant growth and plant cover in vegetated buffer areas can be achieved in five years. Data from the Gog-Li-Hi-Te wetland site in Commencement Bay, WA, show significant increases in riparian vegetative growth within five years. Data from the Coastal America Sites on the Duwamish River show development of riparian vegetation and associated insect production within five years. Mitigation monitoring guidelines specific to Washington State indicate that 90% herbaceous cover may be expected by year five. Woody vegetation/shrub cover ranges from 50% to 80% by year five, to 90% by year ten. By assuming that full plant cover equals sustained ecological value, and by averaging projections of time to full plant cover for woody shrubs, then the time to sustained value for vegetated buffer habitats is about eight years. This eight year time frame is based on monitoring guidelines, which determine the time required for "success" in terms of functional replacement; and inferences from two studies (Gog-Li-Hi-Te wetland and Duwamish Coastal America sites). Upland greenbelts may consist of different species mixes but should be predominately native trees, shrubs, grasses and forbs (flowering plants that are not grasses). The time to sustained value for upland greenbelts is also assumed to be eight years.

Degraded habitat classification

As noted above, intertidal and shallow subtidal areas adversely affected by overwater structures are classified as degraded, so removal of structures and conditions adversely affecting these habitats could restore their habitat value, making them candidates for restoration projects.

Time to sustained value for intertidal and shallow subtidal habitats is four years, based on data from restoration projects in Puget Sound. The projects reviewed were habitat creation projects involving excavating, re-grading, or filling to create intertidal or shallow subtidal habitats. The expected sequence of invertebrate recruitment followed by juvenile salmonid use is related to initial conditions at the site. The degraded classification applies only to intertidal or shallow subtidal habitats. Prior to the introduction of the physical impairment, these areas likely provided the functions associated with their habitat type. Overwater structures limit production by shading the habitat; removal of this impact should allow the habitat to return to near natural production quickly. A literature review found no data addressing the effects of removing overwater structures. However, based on inferences drawn from studies on the impacts of shading, a time to sustained value following removal of overwater structures was assigned.

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). The ability of juvenile salmonids to see and capture their prey is 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 shaded apron stations 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 very important prey items 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, in this study, *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, 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. While we do not expect there to be eelgrass in the LDR, it is logical to assume that shading would similarly reduce primary production of benthic diatoms and other algae.

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. A 1991 study (Parametrix, 1991) 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 Pentilla and Doty (1990), Parametrix and Battelle, (1996), Fresh et al. (1995). 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 one year. Time to sustained value for various habitat types is provided in Table D6.

Table D 6. 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.				
Marsh	not applicable	0.825 (82.5%) FF 0.75 (88.2%) 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 0.75 (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 Fully Functional (FF) or Baseline Adjusted (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

Habitat	final value and percent of final value (%) at end of year			
	1	4	8	15
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

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