Study Plan for the Intensively Monitored Watershed Program

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Intensively Monitored Watersheds
Scientific Oversight Committee*
and IMW Partners

Washington State Department of Ecology
Washington State Department of Fish and Wildlife
Lower Elwha Klallam Tribe
Skagit River Systems Cooperative
NOAA-Northwest Fisheries Science Center
Weyerhaeuser

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*The Intensively Monitored Watersheds Scientific Oversight Committee is:
William Ehinger-Washington Department of Ecology
Tim Quinn and Greg Volkhardt-Washington Department of Fish and Wildlife
Mike McHenry-Lower Elwha Klallam Tribe
Eric Beamer-Skagit River Systems Cooperative
Phil Roni-NOAA-Fisheries NWFSC
Robert Bilby-Weyerhaeuser

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INTRODUCTION

Intensive, watershed-scale research and monitoring efforts have generated results that have been very influential in the development of environmental management strategies in North America. Some of the earliest intensive monitoring efforts were instituted by the U.S. Forest Service in the 1950s to better understand hydrologic responses to logging. Efforts at these sites expanded over time to encompass chemical and biological responses as well. Changes in land use practices nationwide have been based on studies conducted at experimental watersheds like the H.J. Andrews Experimental Forest in Oregon, the Hubbard Brook Experimental Forest in New Hampshire and the Coweeta Experimental Forest in North Carolina. The success of these efforts spawned a number of intensive, watershed-level research efforts in the Pacific Northwest to evaluate the response of salmon to forest practices. The Alsea Watershed Study, which was initiated in the 1960s, evaluated the response of coho salmon and cutthroat trout to various logging methods in a series of small watersheds on the Oregon coast. Results from this study provided much of the technical rationale for the measures to protect aquatic habitat incorporated into the forest practice regulations of Oregon and Washington in the early 1970s. In the 1970s an ambitious watershed-level project was initiated at Carnation Creek on Vancouver Island, British Columbia that evaluated the response of coho and chum salmon to the logging of a previously unlogged watershed. The results of this study led to a revision of the forestry code for B.C. and also influenced revisions to forest practice rules in other areas of the Pacific Northwest. Intensive, watershed-level studies such as these form the foundation of our knowledge about the freshwater habitat requirements of salmonid fishes

Millions of dollars have been dedicated to the restoration of freshwater habitat since the listing of many populations of salmon in the Pacific Northwest in the 1990s. Little is known about the efficacy of these efforts. The most effective means of determining the contribution of restoration projects to salmon recovery is to implement experimental, watershed-scale evaluations. Several organizations in the Pacific Northwest have begun to establish such projects. This document describes a series of intensively monitored watersheds that have been established in Washington for the purpose of better understanding how salmon and trout respond to current approaches to restore habitat.

GENERAL CONCEPT

The basic premise of the Intensively Monitored Watersheds (IMW) project is that the complex relationships controlling salmon response to habitat conditions can best be understood by concentrating monitoring and research efforts at a few locations. The data required to evaluate the response of fish populations to management actions that affect habitat quality or quantity are difficult and expensive to collect. Focusing efforts on a relatively few locations enables enough data on physical and biological attributes of a system to be collected to develop a comprehensive understanding of the factors affecting salmon production in freshwater. IWM is an efficient method of achieving the level of sampling intensity necessary to determine the response of salmon to a set of management actions.

Evaluating biological responses is complicated, requiring an understanding of how various management actions interact to affect habitat conditions and how system biology responds to these habitat changes. The habitats required by a species of salmon change through the period of freshwater rearing (Table 1). Therefore, response of the fish to the application of restoration measures depends upon the manner in which the required suite of habitats is affected. Further complicating the issue is the fact that the relative importance of each habitat type in determining fish survival changes from year-to-year due to variations in weather and flow, the abundance of fish spawning within the watershed and other factors. For example, smolt production can be dictated by spawning habitat availability and quality during years when flood flows occur during incubation and greatly decrease egg survival (Seiler et al. 2002). However, during years of more benign flow conditions during egg incubation, population performance may be more influenced by the availability of food during spring and summer or adequate winter habitat. Untangling the various factors that determine performance of salmon and how these factors respond to land use actions or restoration efforts can only be accomplished with an intensive monitoring approach.

The ultimate objective of nearly all efforts intended to improve salmon habitat is to increase the abundance of the fish. Therefore, the most meaningful measurements of the effectiveness of a restoration program are those related to the performance of the fish during their period of freshwater residency; from adult spawning through smolting of their offspring. Because salmon use multiple habitat types during freshwater rearing and may move throughout the watershed to locate these habitats, the spatial scale at which an evaluation is conducted should be large enough to encompass all the habitats required for the salmon to complete this phase of their life history. The size of the area required to capture the full range of habitats needed to complete freshwater rearing will vary by species.

The IMW Program consists of three elements:

- Studies at three complexes of three or four watersheds each focusing on coho salmon and steelhead trout (Figure 1),
- Evaluation of the effects of estuary restoration on juvenile chinook salmon growth and survival on the Skagit River Estuary.
- A Pacific Northwest-wide landscape classification intended to guide the application of IMW results to other watersheds. The classification is based on similarity of physical and biological characteristics to the watersheds included in the IMW project. Watersheds which have biophysical characteristics and patterns of human activities comparable to IMW sites will be locations where IMW results can be extended with the greatest degree of certainty.

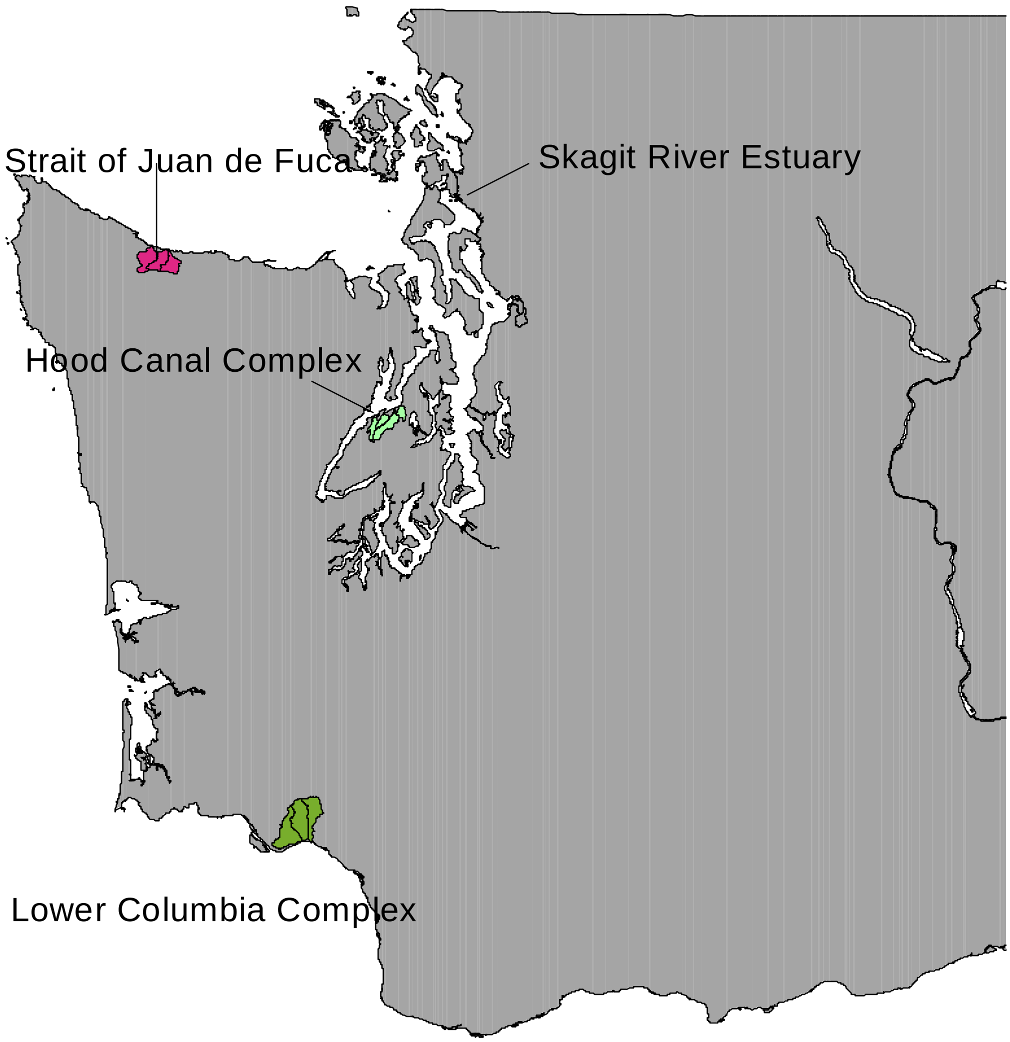

Figure 1. Locations of the four IMW study sites; Straits Juan de Fuca, Hood Canal, Lower Columbia, and the Skagit River Estuary.

Table 1. Habitat requirements of coho salmon during freshwater rearing. The changing requirements of the fish stress the need to develop monitoring designs that evaluate responses at a spatial scale large enough to encompass the full range of habitat types required by the fish to complete freshwater rearing.

Life History Stage	Habitat
Spawning and egg incubation	Gravel bedded riffles and pool tail outs in proximity of cover suitable for adult spawners (e.g., deep pools, undercut banks, debris jams)
Early fry rearing	Low velocity areas with cover in close proximity to food source. Typically associated with shallow, channel margin habitat with cover from wood and overhanging vegetation
Summer rearing	Pool habitat with cover in close proximity to food source. Typically found in low gradient channels with a pool/riffle morphology
Winter rearing	Low velocity areas with cover. Often associated with off-channel habitat on floodplains including low gradient tributaries, secondary channels and ponds

COHO/STEELHEAD IMW COMPLEXES

The three IMW complexes that focus on coho salmon and steelhead trout, Strait of Juan de Fuca, Hood Canal, and Lower Columbia, include a total of ten watersheds. The IMW Complex areas range from 78 km^2 to 206 km^2 (Table 2) with individual watershed areas ranging from 13 km^2 to 75 km^2. Watersheds of this size are sufficiently large to provide all the habitat conditions required for the target species to complete freshwater rearing. We have focused on coho and steelhead in smaller watersheds for four reasons:

1) These species spend more time in freshwater (1-3years) than most other species of anadromous salmonids. Thus, they should be more responsive to changes in the quality and quantity of freshwater habitat than species which only reside in streams and rivers for a short period of time (e.g. ocean-type chinook, chum, pink).
2) Only large changes in fish population metrics will be detectable within the life of this project, given the inherent variability in these populations. In order to cause a detectable change in the fish populations, it is likely that a fairly substantial change in freshwater habitat conditions will need to occur. The relatively small size of the study watersheds will make practicable the application of restoration treatments to a large proportion of the impaired freshwater habitat, increasing the probability of generating a detectable response from the fish.
3) Many of the restoration projects and land use regulations that have been implemented in the region have been based on the habitat requirements of coho salmon. Therefore, this species should be the most likely to respond to many of the restoration actions that are being funded.

4) Because these three species complete freshwater rearing in a small watershed, fish responses to management actions can be assessed using a before-after/control impact design. Use of this type of design should make the responses by the fish easier to detect. Such a design would not be logistically feasible with species requiring a much more extensive area to complete rearing.

Table 2. Characteristics of the three watershed complexes in western Washington.

	Strait of Juan De Fuca	**Hood Canal**	**Lower Columbia**
Watersheds	West Twin East Twin Deep	Stavis Little Anderson Big Beef Seabeck	Germany Abernathy Mill
Focal Species	coho steelhead	coho steelhead	coho steelhead
Land Use	forestry – private, state, and federal	urban, rural residential, forestry – private and state	forestry - private and state agriculture in lower valleys
Complex Area (watershed)	113 km^2 (33, 35, 45 km^2)	78 km^2 (15, 13, 36, 14 km^2)	206 km^2 (57, 73, 75 km^2)
Geology	mixed sedimentary and metamorphic	glacial till	flow basalt w/ interbedded sandstone
Precipitation	190 cm/yr	105 cm/yr	160 cm/yr

Objectives

The goals of the IMW program's coho / steelhead complexes are to determine:

1) Whether freshwater habitat restoration can effect a change in production of outmigrant coho salmon and steelhead trout;
2) What features or processes influenced by the habitat improvements caused the increased production or lack thereof; and
3) Whether the beneficial effects of habitat improvement are maintained over time.

The first question is addressed by measuring smolt/outmigrant production in each treatment basin relative to the reference basin in that complex. However, answering the first question may not provide information about the cause of any increase in outmigrant production. Thus, answering the second and third questions are critical if the results of the IMW effort are to be useful to local restoration advocates to prioritize restoration projects within and among watersheds. However, the data required to answer questions two and three are more complicated to measure, requiring assessment of the fish populations at various stages during freshwater rearing over a period of years. The basic set of monitoring variables described below will provide basin-wide estimates of spawner abundance, egg-to-parr survival, parr-to-smolt survival, smolt production, and habitat. These data will provide the foundation of the monitoring efforts which will be supplemented with additional research to better identify causal mechanisms.

The hypotheses to be tested in all three complexes are listed below.

- The increase in outmigrant production following habitat treatments is greater in treatment watersheds than in reference watersheds.
- The increase in mean parr density is greater in treated reaches than in control reaches.
- The increase in mean parr density is greater in treated watersheds than in control watersheds.
- The increase in mean egg to parr survival is greater in treated watersheds than in control watersheds.
- The increase in mean parr to smolt survival is greater in treated watersheds than in control watersheds.
- The increase in mean smolt length is greater in treated watersheds than control watersheds.

Experimental Design

Long-term monitoring using before-after studies have been recommended to determine biological response to habitat alteration (e.g., Stewart-Oaten et al. 1986; Reeves et al. 1991; Smith et al. 1993). The addition of a control (or controls) to the BA design, commonly called a before-after control-impact (BACI) design, is meant to account for environmental variability and temporal trends found in both the control and treatment areas and, thus, increase the ability to differentiate treatment effects from natural variability (Smith et al. 1993). Additional statistical power to detect treatment effects may be achieved with multiple control sites (spatial replication) and long-term sampling (temporal replication; Underwood 1994). Recent examples of aquatic restoration monitoring using a BACI design include Cederholm et al. (1997) and Solazzi et al. (2000).

Downes et al. (2002), in a thorough review of BACI study designs, identified several types, including one with replication (multiple treatment and controls) that they referred to as the multiple BACI or MBACI. Hicks et al. (1991) referred to this design as an extensive BA study but assumed that sampling intensity would be reduced because of the increased number of treatment and control sites. A replicated BACI design potentially is the most powerful of all study designs because it includes replication in both space and time (monitoring of multiple treatments and controls before and after restoration) but also potentially is more challenging and costly to implement than other designs (Downes et al. 2002). Spatially replicating BA and BACI such as the IMW project addresses many of the problems inherent in these designs and would increase the applicability of our results to other areas. Roni et al. (2005) indicated that while no ideal study design exists to answer all questions, the most powerful design is a BA or BACI design that includes many paired treatments and controls across the landscape that are monitored for many years. Furthermore, they indicated this is the type of monitoring needed to quantify population- and watershed-level responses.

A before-after/control (referred to here as reference) -impact (BACI) design, implemented at different spatial scales depending upon the question being addressed, is the basic design being applied in the IMW studies. However, other approaches may be used depending upon the question addressed, the scale of the assessment, and the data available. For example, the Strait of Juan de Fuca complex has too little pre-restoration smolt production data to use a BACI design at the basin scale or, in other complexes, there may not be a significant relationship between a treatment basin and the reference basin, in which case we will revert to a comparison of trajectories between treatment and control basins post-restoration and a before-after design, respectively.

Each IMW complex includes two or more treatment basins and one watershed serving as a reference site where no restoration projects will be implemented during the study. The pre-treatment data available to assess the relationship between reference and treatment watersheds varies among the complexes and among streams within a complex. Where possible, treatments will be delayed to allow time for more data to be collected to evaluate the relationship of treated to control watersheds prior to applying restoration treatments.

The BACI design will be implemented at multiple spatial scales, the scale dependent on the question being addressed. Some questions are best addressed at a reach scale. Questions that can be addressed at this finer scale include life-history specific biological responses or physical habitat responses to management actions. Reference sites for some reach-level projects can be within the basin designated for treatment. These reference sites consist of a reach in close proximity and comparable in initial habitat condition to the treated section of channel. No habitat manipulation would occur during the period of evaluation in the reference stream reach. For evaluations of effects at the scale of the entire basin, a comparison with the reference watershed in a complex is required. Therefore, the IMW approach does require sufficient influence over management decisions to ensure that reference sites, at all spatial scales, remain untreated through the duration of the study. The IMW project is coordinating restoration plans with the local salmon recovery lead entities for each complex in order to ensure the integrity of the reference sites. We expect human activities will occur in some of the reference watersheds (e.g., logging in those reference watersheds with commercial forest lands). The IMW partners have no ability to control these activities. However, we do not believe these actions will compromise the integrity of the study provided that any effects associated with these activities can be measured and segregated from responses related to restoration actions.

Experimental treatments will vary depending upon the initial condition of the watersheds, the perceived factors limiting fish production, and the feasibility of applying treatments. Selection of treatments will be based on an assessment of current watershed conditions. For example, an assessment of channel conditions in the Straits of Juan de Fuca watersheds revealed very low in-channel wood levels. Therefore, an aggressive program of wood addition has already begun in the treatment watersheds in this complex. Many of the selected watersheds have had some type of watershed assessment already conducted (limiting factors analysis, Washington State watershed analysis, EDT). These analyses are being used in conjunction with supplemental information collected as part of the IMW project to identify the suite of habitat restoration efforts most likely to positively influence the salmon and trout production. Specifics about some of the implemented and planned restoration projects for the SJF and Hood Canal complexes are presented below. Treatment options for the Lower Columbia complex are still being developed.

Variables

The specific parameters measured in each watershed will vary depending on the questions being addressed and the types of treatments being applied. However, a basic set of data will be collected at all of the watersheds (Table 3). These common measures are intended to capture the effect restoration actions are having at a watershed scale and to provide context for interpretation of changes observed following application of treatments. The common parameters include measures of water quantity and quality, habitat characteristics and characteristics of the fish populations.

Water Quantity and Quality

Continuous stage height recorders have been installed near the mouth of each watershed. Discharge is estimated using a relationship between stage height and flow that is being developed for each flow monitoring station. Water samples are collected monthly at the gauge site and analyzed for temperature, dissolved oxygen, pH, specific conductivity, total nitrogen, nitrate+nitrite-N, ammonia-N, total phosphorus, soluble reactive phosphorus, suspended sediment, and dissolved organic carbon. Continuous turbidity monitors have been deployed at each flow gauging site. These instruments collect turbidity data at 15 minute intervals. The turbidity sensor triggers a pump water sampler at high turbidity levels to estimate suspended sediment loads, a method termed Turbidity Threshold Sampling-TTS (Lewis 2003, 1996). In situ water temperature loggers have been deployed throughout each basin at selected locations to characterize changes in water temperature from headwaters to the mouth.

Habitat Conditions

An EMAP (Environmental Monitoring and Assessment Program) based approach, developed by EPA, is being used to provide annual, basin-wide estimates of habitat condition. EMAP uses precise measurements and/or visual estimates of habitat attributes using transects and variable-length samples (Simonson et al. 1994, Angermeier and Smogor 1995) based on stream size (Kaufmann et al. 1999, Peck et al. 2001). These methods have been selected to ensure precise, repeatable measurements because low measurement precision substantially limits the ability to detect spatial differences and temporal trends in habitat attributes (Peterson and Wollrab 1999, Larsen et al. 2004).

Sample sites are randomly selected but the selection process maintains spatial balance among the sites (Stevens and Olsen 2004). Ten sites per watershed per year (30-40 / complex) are being measured. Several years of data will be accumulated prior to application of treatments in most of the watersheds, providing pre-treatment data on 30-40 sites per watershed.

The EMAP sampling approach attempts to allocate sampling effort in a manner that balances the objectives of describing spatial variability in environmental conditions and detecting trends over time. Spatial variation is best captured by maximizing the number of sites sampled while evaluating temporal trends requires re-sampling of sites (Larsen et al. 2001). We have chosen to select new sites each year in order to better describe the current status of habitat prior to restoration rather than revisiting sites (Urquhart et al. 1998, Roper et al. 2003). The duration of the study and temporal periodicity of sampling are the primary determinants of the ability to detect trends in habitat conditions (Larsen et al. 2004), and therefore to assess correlations between changes in habitat conditions and salmon abundance, distribution and production.

Field methods will closely follow those developed in the Western EMAP Pilot Study (see Peck et al. 2001). The measurements and the metrics calculated in the EMAP sampling are listed in Table 4.

Fish Populations

Data on spawning salmon and steelhead, summer parr and emigrating smolts is collected in all watersheds. Abundance of returning salmon and steelhead is assessed at collection fences in two of the watersheds (Big Beef Creek-Hood Canal and Abernathy Creek-Lower Columbia). These fences capture all returning fish. In all watersheds stream adult abundance and distribution surveys are conducted throughout the spawning season. All spawning fish (coho) or redds (steelhead) encountered during the stream surveys are counted and location is noted for later entry into a GIS database. The purpose of the surveys is to generate abundance estimates of spawning fish for the watersheds without counting fences and to assess spawner distribution. In the Hood Canal and Lower Columbia complexes the entire length of stream accessible to anadromous fishes is surveyed at one to two week intervals during the spawning season. In the Strait of Juan de Fuca complex, traditional WDFW index reaches have been supplemented with surveys of randomly selected reaches, stratified by habitat unit.

Table 3. Variables measured in all coho, steelhead, and cutthroat watersheds.

	Frequency	**Data available**
Flow	Continuous	https://fortress.wa.gov/ecy/wrx/wrx/flows/regions/state.asp
Water temperature	Continuous	https://fortress.wa.gov/ecy/wrx/wrx/flows/regions/state.asp
Water chemistry	Monthly	http://www.ecy.wa.gov/programs/eap/fw_riv/rv_main.html
Probabilistic sampling	Annual	http://wdfw.wa.gov/hab/imw/index.htm
Smolt production	Annual	http://wdfw.wa.gov/fish/wild_salmon_monitor/publications.htm
Juvenile abundance	Annual	http://wdfw.wa.gov/hab/imw/index.htm
Spawners	Annual	http://wdfw.wa.gov/hab/imw/index.htm

Parr abundance is determined each summer. Fish are collected at 30-40 randomly selected reaches (site selection based on EMAP protocols) in each complex by one-pass electroshocking surveys. Catch per unit effort (time) is used to provide an indication of parr distribution and relative abundance of age 0 trout. Total watershed abundance of coho and age 1 steelhead parr is estimated using a mark-recapture method. The adipose fin is removed from all coho and age 1 steelhead parr captured in the Hood Canal complex and PIT tags are inserted in fish in the Strait of Juan de Fuca and Lower Columbia complexes. Marks are noted during smolt trapping the following spring, enabling an estimate of the survival of marked fish from summer through smolting. Total parr abundance in each watershed the previous summer is then estimated from the survival rate and the proportion of marked to unmarked fish captured in the smolt trap.

Smolts are collected with a fence on seven of the ten IMW streams (all Hood Canal and Strait of Juan de Fuca streams), providing a complete count of emigrating fish. Partial traps (screw traps) are used in the Lower Columbia streams because of their larger size. A detailed description of smolt monitoring methods is included in Appendix A and the annual reports are available online (Table 3).

Table 4. Calculated metrics procured using the EMAP sampling protocol.

Metrics
width-depth ratio
channel confinement
average pool depth
residual pool depths
Substrate size distribution
bank stability
bank cover
Shading
LWD size and distribution
channel slope
channel sinuosity
water flow profile

Land Use and Land Cover

The IMW experiments assume that human changes to watersheds have effected and will continue to effect salmon production. The most general question addressed by the IMW research is whether restoration treatments can affect greater salmon production given ongoing human disturbances to the systems. In this case measures of human disturbance are simply used to describe changes in watershed conditions and evaluate appropriateness of the reference watersheds in the BACI design.

The IMW experiments also test whether restoration treatments can affect increased production or reduce the decline in production, if further disturbance occurs. To address this question trends in land use and land cover and trends in smolt output will be correlated to infer the strength of the effect and to partial out the variability in smolt production due to land use and land cover changes; facilitating the detection of treatment effects. These analyses may indicate that the treatments were effective, but insufficient given ongoing human disturbance.

Land use and land cover data will be collected for the Hood Canal IMW complex using the 2001 Digital Airborne Imagery Survey (DAIS) of Kitsap County. The DAIS provides 3-m 3-band digital imagery of Kitsap County that facilitates the onscreen delineation of land uses and land covers. DAIS data collected in 2001 will be used to create a base dataset by manually delineating land use and land cover using a geographic information system. Land use and land cover classes will include road length, impervious surfaces, open ground, shrub, and density and 3 age classes of deciduous trees, coniferous trees, and mixed stands. Overlaying the delineated lines and polygons from the detailed base layer on subsequent photographs will be used to rapidly detect changes. We will use a similar approach to interpolate previous land use and land cover conditions using existing, comparable aerial photographs.

The National Agriculture Imagery Program (NAIP) 1:32,000 scale color digital photographs will be available every 3 years, beginning in 2007 (photographs from 2006). As they become available NAIP photographs will be used to update the base layer and to detect changes in land use and land cover classes. When additional aerial photographs are available they will be compared to photographs from previous years to increase the number of years with samples. When necessary and possible, land use and land cover classes will be interpolated between years with aerial photographs to provide estimates for years without photographs. These estimates will be evaluated and adjusted with ancillary data describing zoning provided by Kitsap County. Hydrography layers for IMW watersheds have been validated and corrected using field surveys and GPS for all watersheds and LiDAR data in Hood Canal. Watersheds and subwatersheds have been delineated for all IMW watersheds, allowing the attribution of watersheds, sub-watersheds, and stream reaches with land use and land cover data as they become available.

Attributes calculated will include the spatial density of classes for watersheds, sub-watersheds, and reaches and proportion of riparian zones with each class. Analysis of correlations between changes in land use and land cover attributes, field habitat measurements, and smolt counts will be used to select meaningful covariates for the BACI test of treatment effects and to build alternate models that describe habitat change and predict survival and production. Because the effects of human disturbance are complex and the relatively short study duration will limit the complexity of analytical models, an index of land use and land cover may be calculated from several descriptors using multivariate statistics to account for the most important attributes of disturbance in a relatively simple model. Additionally, if trends in land use and land cover are observed in a watershed those trends will be compared to trends in smolt counts using linear or nonlinear regression and graphical analyses.

Data management

Database management has been largely centralized and integrated into existing databases at WDFW and Ecology (Table 3). This is more efficient and enables easier dissemination of the data. The exceptions to this are special studies where the data require extensive, ongoing manipulation to be meaningful to scientists (e.g. data collected in the PIT-tagging studies of juvenile movement and survival described below in the Straits of Juan de Fuca IMW complex). In these cases the study results will be released and posted in technical reports to the IMW websites.

Treatments

Specific restoration techniques will vary among watershed but most focus on restoration of instream habitat, reconnection with or creation of off channel habitat, or removal of fish passage barriers. Below we recall a review by Roni et al. (2002; 2005) of the response of coho salmon and steelhead, and cutthroat trout reported for LWD, boulder weirs, and off-channel habitat restoration. We focus on recent work from western Oregon and Washington as it is likely to be most applicable to the IMW program. Removal of fish passage barriers can be viewed as creation of new habitat.

LWD Placement

A handful of regional studies have examined the effects of LWD placement on juvenile anadromous fish abundance. Most of these studies have examined reach scale responses. Roni and Quinn (2001) examined juvenile fish response in summer and winter in paired treatment and control reaches in 30 streams in western Washington and Oregon (Table 5). These streams ranged between four and 16m bankfull width and are similar in size to streams being monitored under the IMW project. They found significantly higher numbers of juvenile coho during summer and coho, cutthroat and steelhead in winter. Coho salmon numbers were 1.8 and 3.2 times higher in treatment than control reaches in summer than winter. Winter cutthroat and steelhead numbers (parr and presmolts) were 1.7 times higher in treatment than control reaches, but no significant difference was detected for summer parr.

Cederholm et al. (1997), Reeves et al. 1997, and Solazzi et al. 2000) also examined fish response to LWD placement though only in one or two streams. Solazzi et al. (2000) is most similar to the IMW project and reported 50% increases in summer coho parr and 200 to 300% increases in coho smolts. They found no significant difference in summer steelhead and cutthroat parr numbers, steelhead migrants (smolts) increased from 400 to 900%, while cutthroat trout migrants increased 275 to 400%. Reeves et al. (1997) found inconclusive results in evaluation of coho, Chinook and steelhead response to restoration in Fish Creek, Oregon. Cederholm et al. (1997) reported little response of steelhead and cutthroat to LWD placement in Porter Creek, but a 6 to 20 fold increase in winter coho abundance and 2.6 to 3.2 fold increases in coho smolt production. Koning and Keeley (1997) also reported on various instream techniques from 8 studies in the grey literature and reported 1.8 and 2.3 fold increases for coho and steelhead parr respectively. Most of their studies were in British Columbia or Idaho and examined stream reaches rather than watershed or population level responses.

The range of responses from case studies on individual streams suggests that the estimates from Roni and Quinn (2001) are likely a conservative yet broadly applicable estimate of juvenile coho, steelhead, and cutthroat response to LWD placement

Boulder weirs

Roni et al. (2006a) examined the placement of boulder weirs on summer parr abundance in 13 sites in southwest Oregon. Numbers of coho and age 1+ trout (steelhead and cutthroat) were 1.4 and 1.5 times higher in treatment than control reaches respectively. No difference was detected for young-of-year trout. These results are consistent with previous studies on coho, cutthroat and steelhead trout response to boulder weir and cluster placement for streams in Oregon, California, and British Columbia (e.g., Ward and Slaney, 1981; Moreau, 1984; Fontaine 1987; House *et al*., 1989). We did not sample in the winter, but given results of Roni and Quinn (2001) and other studies, winter response of coho and trout to boulder weir placement is likely much higher than summer.

Off-channel ponds/habitats

Roni et al. (2006b) synthesized smolt trapping data from 30 natural and constructed off-channel habitats and found that an average of 0.37 coho smolts per m^2. The smolt densities in all but one of these sites were well within the range of 0.02 to 3 smolts/m^2 reported in previous studies (Sheng et al. 1990; Blackwell et al. 1999; Giannico and Hinch 2003; Keeley and Koning 1997; Morley et al. 2005). Again the Roni et al. (2006b) estimates likely provide a realistic estimate for the IMW project.

Table 5. Summary of coho, cutthroat and steelhead response to LWD, boulder weir and off-channel habitats from Roni and Quinn (2001)and Roni et al. (2006a,b)

Restoration (reference)	Reference	% change	Fish/m^2
LWD (based on Roni and Quinn 2001) n = 30	Coho parr (summer)	81%*	NA
	Coho winter/presmolts	223%*	NA
	Steelhead parr	19% (not significant)	NA
	Steelhead winter (presmolts & parr)	70%*	NA
	Cutthroat parr	27% (not significant)	NA
	Cutthroat winter (presmolts and parr)	73%*	NA
Boulder weirs (Roni et al. In press) n = 13	Coho parr (summer)	40%*	NA
	Trout > 100 (summer)	50%**	NA
	Trout < 100	-10% (not significant)	
Off-channel habitats	Coho smolts	NA	0.37 (SE=0.0659)

* p <0.05, ** p <0.10

Analysis

While a BACI design is optimal, a BACI with a poorly chosen control can be less powerful than the uncontrolled BA design (Korman and Higgins 1997; Roni et al. 2003). Both BA and BACI designs are subject to a number of potential statistical problems largely due to non-independent measurements or poor selection of the control (Hurlbert 1984; Smith et al. 1993; Conquest 2000; Murtaugh 2000). If the measurements are autocorrelated (i.e., correlated over time or space), the variance will be poorly estimated, leading to incorrect conclusions (inflated confidence) about statistical significance and, potentially, about treatment effects. This can be a particular problem if replicate samples are not spaced adequately in time (Stewart-Oaten et al. 1986). False conclusions also can occur when the pretreatment trends in the parameter of interest are not similar between treatment and control reaches or watersheds (i.e., poor correlation between treatment and control). However, these pitfalls can be avoided through careful implementation of monitoring, selection of control basins, and, if needed, use of alternate analytical techniques. Reeves et al. (1997), Conquest (2000), and McDonald et al. (2000) indicated that interpretation of data from unreplicated BACI studies should include use of graphical analysis and knowledge of ecosystem processes rather than statistical significance to interpret response trajectories.

Changes in smolt production will be assessed using a regression analysis to compare the relationship of treatment to reference stream before and after restoration, where the data allow. For watersheds where the pre-treatment data record is too short to enable the use of regression analysis. Currently, sufficient smolt production data exist in Hood Canal (at least since 1993 on all basins) to allow regression analysis. Depending upon when restoration is implemented, there should be sufficient data from the Lower Columbia Complex streams. Smolt production from the SJF complex streams will be compared using a paired t-test. Other statistical tests, both frequentist and Bayesian, may be employed as needed.

It is clear that habitat restoration projects, properly selected and implemented, can increase fish density. In order for the IMW to test the effects on smolt production, we must ensure that 1) enough projects are implemented to cause an increase in smolt production and 2) the monitoring program is able to detect the anticipated response within a reasonable time frame. The first will be addressed basin by basin as restoration plans are developed. The second, the ability of the monitoring program to detect a change in smolt production, is addressed below through a series of power analyses.

Power Analyses

The purpose of these power analyses is to quantify the IMW program's ability to detect a change (e.g. magnitude of change and number of years needed). The detectable change in smolt production should create clear expectations for the IMW program when viewed in the context of the anticipated effects of habitat restoration.

The advantage of the BACI design is that the effect of external drivers of productivity (e.g. weather events and related stream flow) that affect all study streams can be statistically removed, thereby making changes due to habitat restoration easier to detect. The degree to which the ability to detect treatment is improved is a function of the strength of the correlation between the treatment and control basins. The lower regression line in Figure 2 shows the pre-restoration relationship between coho smolt production in Big Beef Creek and Stavis Creek, the reference stream. The assumption is that after restoration smolt production will increase, i.e. the regression line will be displaced upward so that for a given level of production in Stavis Creek, production in Big Beef Creek will be higher. An advantage of the regression model is that additional explanatory variables may be included in the model, further reducing the unaccounted for variability, thereby increasing the power of the test to detect a change in production. However, if there is no significant relationship between the reference stream and a treatment stream, we will use a before-after comparison of smolt production.

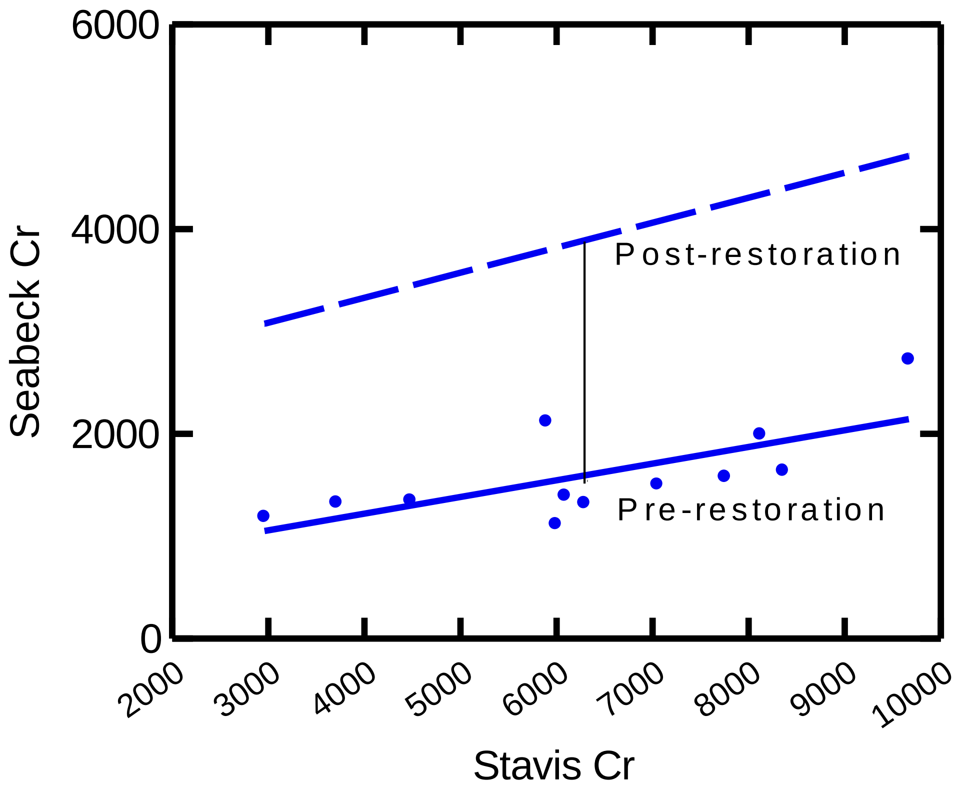

Figure 2. Hypothesized increase in smolt production is shown as a translation of the regression line upward.

The minimum detectable change (assuming a one-tailed, two-sample t-test) is a function of the confidence level (α), power (β), the variance of the data, and the sample size (Equation 1). We have set α=β=0.10 for all analyses.

$$\Delta P = \sqrt{\frac{2s^2(t_{1-\alpha} + t_{1-\beta})^2}{n}} \qquad 1)$$

where ΔP = the detectable change in smolt production,

s^2= variance of the pre-restoration data (for the Before-After case) or the residuals of the treatment vs. reference stream regression (for the BACI design),

$t_{1-\alpha/2}$= $t_{(0.90,\, n)}$ (α= 0.10, one-tailed test)

$t_{1-\beta}$= $t_{(0.90,\, n)}$ (β= 0.10)

n= number of years of pre and post-restoration monitoring (sample size).

We have conducted a series of power analyses using data from the Hood Canal IMW complex. Data from Stavis Creek, Big Beef Creek, and Seabeck Creek in the Hood Canal IMW complex were used because only this complex has sufficient data at this time to estimate the statistical relationship between the treatment and reference basins.   Smolt production has been measured concurrently at all Hood Canal streams since 1993, except that no data were collected in 1996 at Seabeck, Stavis, or Little Anderson Creeks and no estimate is available for Little Anderson Creek in 1998.   Flow data, used as a covariate in the analyses, was available all years except 1996. Little Anderson Creek smolt data are not significantly correlated with Stavis Creek (reference stream) data and so were not used in the power analysis.

The power analyses conducted assumed:

1) a Before-After design, applicable where the relationship between Reference and Treatment was not significant;
2) a BACI design, applicable where there is a statistically significant relationship between the Reference and Treatment basins; and
3) a BACI design using environmental covariates, applicable where there is a statistically significant relationship between the Reference and Treatment basins and explanatory environmental data are available (i.e. habitat, flow, etc.).

The variance of annual, pre-restoration smolt production in the treatment stream (Big Beef or Seabeck) was used in Equation 1 for the Before-After comparison. A simple linear regression model of treatment stream vs. Stavis Creek, the reference stream, was used for the second analysis, (BACI design with no covariates). Maximum November flow and spawner escapement were added to the regression model to estimate the impact of covariates on the detectable change. The results of the analyses are shown in Table 6 and in Figures 3 and 4.

Assuming an equal number of years of monitoring pre and post-restoration, the analysis shows that we could detect an increase in smolt production on Big Beef Creek equal to 65% of mean production (mean production was approximately 26,000 smolts/ year) after six years using a Before-After analysis. At 12 years the detectable difference is reduced to 43%. Use of the BACI design results in a detectable increase in production of 51% and 34% at six and 12 years, respectively. The addition of November flow as a covariate in the BACI model resulted in a detectable increase of 33% and 22% of the mean at six and 12 years respectively.

The results using Seabeck Creek data were similar. Detectable changes of 49% and 33% at six and 12 years, respectively, were calculated using a Before-After design. Use of the BACI design reduced this to 35% and 23% at six and 12 years. The addition of significant covariates to the BACI analysis reduced this further to 27% and 18%.

Table 6. Comparison of detectable change with six and 12 years of post-restoration data based on long-term smolt monitoring data collected in Hood Canal IMW complex. Data indicate that increases in production of approximately 20% and 30% of the mean will be detectable with 12 and six years, respectively, of post-restoration monitoring.

Design	Covariates	R^2	Variance	Detectable change (% mean)	
				6 years	12 years
Big Beef Creek					
Before-After	NA	NA	9.83 x10^7	65%	43%
BACI	None	0.31	6.76 x10^7	61%	34%
BACI	November flow	0.66	2.71 x10^7	33%	22%
Seabeck Creek					
Before-After	NA	NA	227,998	49%	33%
BACI	None	0.42	111,998	35%	23%
BACI	November flow, escapement	0.57	68,590	27%	18%

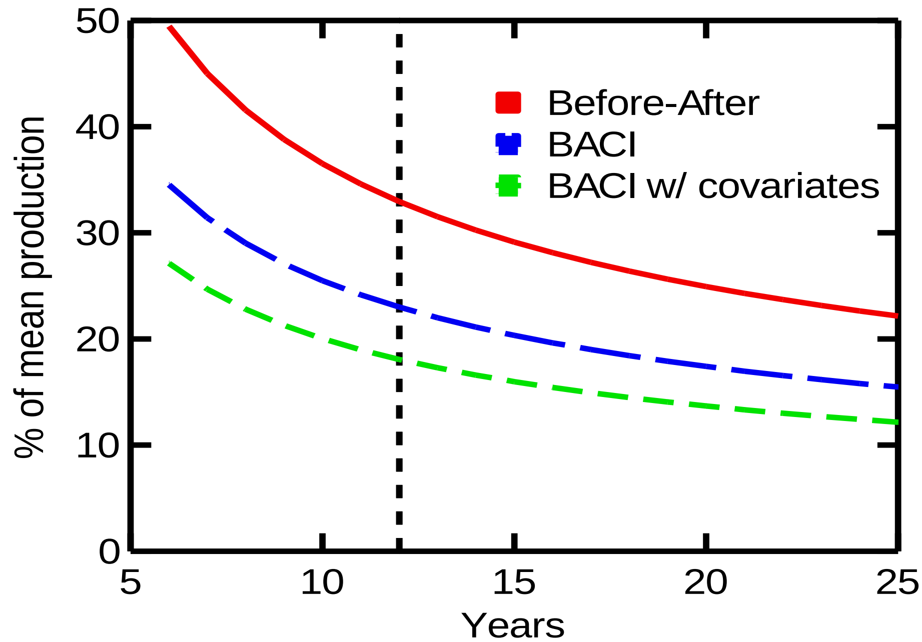

Figure 3. Detectable change in smolt production, presented as a percentage of the mean production, vs number of years needed to monitor for Seabeck Creek. The upper line assumes a Before-After analysis (no significant relationship to the reference stream). The middle line assumes a BACI design with Stavis Creek as the reference. Further improvements are seen in the lower line using a BACI design and flow and escapement as covariates. See Table 6.

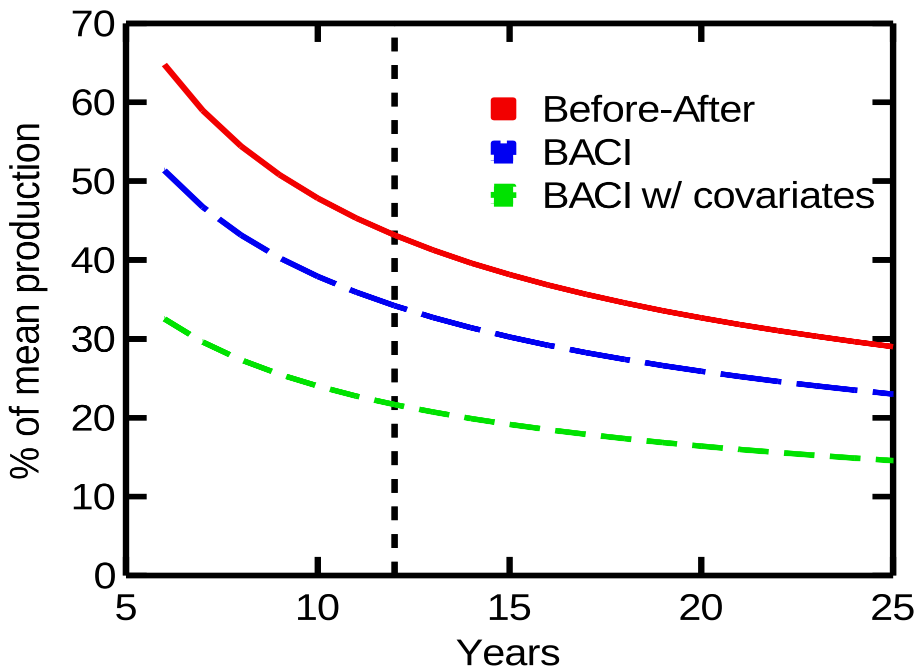

Figure 4. Detectable change in smolt production, presented as a percentage of the mean production, vs number of years needed to monitor for Big Beef Creek. The upper line assumes a Before-After analysis (no significant relationship to the reference stream). the middle line assumes a BACI design with Stavis Creek as the reference. Further improvements are seen in the lower line using a BACI design and flow and escapement as covariates. See Table 6.

Assuming that these relationships are representative of the other IMW basins, there is a relatively high probability that the proposed monitoring will be able to detect the anticipated response to the restoration implemented in the Straits IMW complex, given the additional data collected there. Because of the extensive pre-restoration data and the reference watershed, the anticipated response to Little Anderson restoration is within the range of detectable change. As the restoration plans for the other Hood Canal watersheds and the Lower Columbia IMW complex are developed, we will ensure that the anticipated cumulative effect of all restoration projects on smolt production will be large enough that we have a reasonable probability of detecting it (Table 5).

Complex Descriptions

The watershed complexes differ in physical characteristics, land use patterns and restoration approaches. As a result, there are differences among the complexes in some aspects of the evaluation approach. Descriptions of the complexes, the restoration treatments being applied or planned and some preliminary results are presented in the following sections.

Strait of Juan de Fuca Complex

The SJF complex watersheds are almost completely owned by US Forest Service, Washington Department of Natural Resources, and two private forestry companies. We have the full cooperation of all the dominant owners. Relatively little timber harvest or road construction will occur in these watersheds over the next decade. Therefore, interpreting any responses of the fish to the restoration treatments at the watershed scale will not be complicated by other activities that might affect habitat condition.

Smolt monitoring began in 1998 or 2001, concurrent with the initiation of restoration efforts in Deep and East Twin Creeks. As a result, there is no pre-restoration smolt production data to evaluate the pre-treatment relationship between reference and treatment watersheds. There has been no restoration activity in the West Twin watershed. While the lack of pre-treatment smolt data is not ideal, this complex offers an established restoration and monitoring effort and a high level of certainty about management activities over the next decade. The paucity of pre-treatment data will require that the effects of the restoration on fish populations will have to be quite dramatic in order for us to detect the response. However, the very aggressive restoration effort that has been implemented on Deep and East Twin creeks may enable a sufficiently large response by the fish.

Description

The Deep Creek and East Twin and West Twin Rivers watersheds are located on the northwestern Olympic Peninsula and cover a combined area of approximately 113 km^2 (Figure 5). The Deep Creek, West Twin River, and East Twin River watersheds are of comparable size, 45 km^2, 33 km^2, and 35 km^2, respectively. These watersheds drain directly into the Strait of Juan de Fuca. The headwaters of the stream systems initiate in the Olympic Mountains and flow into gradually broadening river valleys. Stream channels generally flow in a northeasterly direction in the upper watershed areas and then turn northerly to the Strait of Juan de Fuca. Elevations in the watershed range from sea level to 1,142 meters atop Mt. Mueller in the headwaters of the East Twin and West Twin rivers.

Average annual precipitation for the Twin/Deep Creek watersheds is approximately 190 cm. Most precipitation occurs during the autumn and winter months (October through March) with monthly averages ranging from a low of 4 cm in July to a high of around 36 cm in January. Precipitation intensity varies with elevation and some of the higher, headwater areas in these watersheds receive over 250 cm annually. Snowfall typically occurs from November through March and is greatest in December and January. Fog condensation contributes moisture, but the amount of water available for runoff from this process is unknown.

These watersheds are underlain by volcanic rocks of the Crescent Formation, marine sedimentary rocks, and glacial deposits. The oldest rocks (the Crescent Formation) are at higher elevations, while the youngest, the marine sedimentary rocks, are at the lower end of the watershed. Glacial deposits occupy lower valley margins and valley floors toward the upper part of the watershed, and throughout broad terrace areas in the lower parts of the watershed. Recent alluvium is found locally adjacent to higher-order channels, especially at the lower end of the watershed. The area of the watershed underlain by the Crescent Formation is steep and dissected with generally shallow soils. Landslides and resulting debris torrents are most common in this area of the three watersheds. The marine sedimentary rocks include a mixture of siltstones, sandstones, mudstones and conglomerates. Most mass wasting on this geology is associated with steep converging topography and over-steepened channel margin slopes. The low strength, fine-grained nature of these rocks contributes to the generation of fine sediment in these watersheds. Glacial deposits occupy valley bottoms, toe slope areas, and terraces in the lower part of the watershed. Typically they are relatively thick deposits on gentle slopes and not particularly susceptible to erosion. Exceptions are where streams have incised deeply into these deposits, leaving high banks (of relatively weak materials) and may form small inner gorge structures that are susceptible to, and in part created through, erosion and/or mass wasting.

Glaciolacustrine clay overlying dense glacial till is found in some areas along the lower Deep Creek inner gorge and the upper part of the East Fork of the East Twin River, a condition susceptible to deep-seated mass wasting. (Neal and Buss 1992).

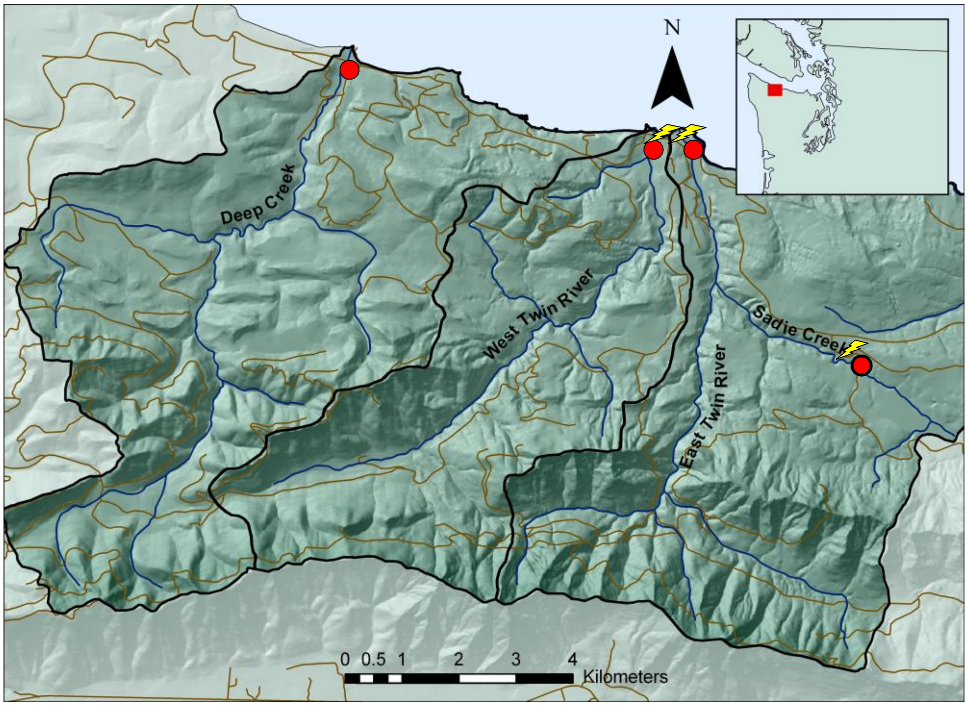

Figure 5. Deep Creek, and West Twin and East Twin Rivers watersheds. Dots represent smolt trap locations arrows (lightening) indicate location of PIT tag readers.

Three vegetation zones occupy the watershed. The Sitka spruce zone is found in the lower valley bottoms, where fog moving inland off the Strait of Juan de Fuca creates mild, moist conditions that allow spruce to compete effectively. This zone occupies about 11.5 percent of the watershed. The western hemlock zone occupies about 77.9 percent of the watershed, in the low to mid elevations throughout the watershed. The silver fir zone occupies about 10.6 percent of the watershed largely in the upper elevations across the southern headwaters of the watershed along the ridge of Kloshe Nanitch/Mt. Mueller.

Early successional stages occupy 27.3 percent of the watershed, mostly on private land while mid successional stages cover 60.8 percent of the watershed. Late successional stands cover 11.0 percent of the watershed, mostly on National Forest land. Only 0.8 percent of the watershed is not forested, primarily wetlands and waterbodies. There are few residences in the three watersheds with essentially no agricultural or urban development.

<u>Land Use</u>

Fires and floods were the primary disturbance mechanisms affecting these watersheds prior to arrival of European-Americans. The pre-European fire regime in these watersheds was characterized by infrequent, intense, large, stand-replacing fires. Large fires that occurred in 1308, 1508 and 1701 likely spread over most of the three watersheds. However, these fires initiated under climatic conditions that were drier and warmer than those that have existed over the last 200 years. Timber harvest began in these watersheds in the 1890s. Introduction of timber harvest and land clearing in the late 19th and early 20th century increased fire frequency. The largest fire events during this period occurred in 1895 and 1939. Private and state lands in the watershed were harvested extensively by 1929. Timber harvest on lands administered by the US Forest Service took place from the 1940s to the 1990s. Second-rotation harvest on State and private lands has occurred in recent years. Impacts to the streams have also resulted from road runoff, failures, and surface erosion. Disturbances associated with logging and road construction have led to an increase in the amount of coarse and fine sediment delivery to fish-bearing streams. Riparian timber harvest has depleted the recruitment source for large woody debris (LWD) and very few large conifer trees are present in the channels of these watersheds today. Increased sediment loading and reduction in LWD size and volume has caused a decline in pool size and frequency and reduced the amount of rearing habitat for juvenile salmonids.

Stream channel characteristics

There is a total of 230 km of stream channels in the Deep Creek (36.8%), West Twin River (32.4%), and East Twin River (30.7%) watersheds (Table 7). Drainage density within the watersheds averages around 2.8 km/km^2. Nearly 80% of the total channel length is relatively steep (>8%). Moderate-gradient (2-4%) and low-gradient (<2%) channel segments accounted for 12.5 percent and 8.6 percent of the channel length, respectively.

Discharge patterns closely follow seasonal precipitation patterns. A US Geological Survey discharge station in the East Twin River collected streamflow data from 1963 though 1978 (USGS gauging station 12043430 - East Twin River near Pysht, Washington). These data indicate the large seasonal differences in discharge. Lowest average monthly discharge occurred in August (0.15 m^3/s) and highest flows occurred in December (4.5 m^3/s). Mean annual flow over the period of record was 1.8 m^3/s.

Table 7. Length of channel segments by gradient and confinement categories.

Gradient Category (percent)	Length by Confinement Category (m)		Total Length (m)	Percent of Watershed Total
	Confined	Unconfined		
< 1	0	5440	5400	2.3
1 – 2	1620	13,160	14,780	6.3
2 – 4	12,140	3320	15,460	6.6
4 – 8	13,820	0	13,820	5.9
8 – 20	47,030	0	47,030	20.3
> 20	136,370	0	136,370	58.6
All	**210,980**	**21,920**	**232,900**	**100**

Fish Communities

Populations of fall chum (*Oncorhynchus keta*), fall coho salmon (*Oncorhynchus kisutch*), winter steelhead (*Oncorhynchus mykiss*), and resident and anadromous cutthroat trout (*Oncorhynchus clarki*) utilize the Deep Creek and Twin Rivers watersheds. Pacific lamprey (*Lampetra tridentata*) and sculpins *(Cottus sp.)* also are present in each drainage. Historical accounts mention Chinook salmon (*Oncorhynchus tshawytscha*) in these watersheds but it is unclear whether these were the results of WDFW hatchery outplants that occurred in the 1970's or a natural population. Chinook salmon have not been observed in recent years.

Historically, Native Americans harvested salmon and steelhead in the Deep/Twins Watershed, as evidenced by a number of archaeological sites around the Pysht River and Deep Creek. Due to chronically low escapements, no terminal salmon fisheries are currently conducted in the watersheds. Tribal fisheries for winter steelhead have been closed in Deep/Twins since 1990. The East Twin River is currently closed to sport steelhead fishing, and all wild steelhead must be released by anglers on Deep Creek and the West Twin River. The status of salmon and steelhead stocks based upon two recent stock reviews is summarized in Table 8 and below.

Table 8. Status of salmonid stocks in the Deep/Twins Watershed.

Species	**Race**	**Production**	**Stock origin**	**Stock status (WDF et al. 1993)**	**Stock status (McHenry et al. 1996)**
Chum	Fall	Wild	Native	Healthy	Critical
Coho	Fall	Wild	Mixed	Depressed	Stable
Steelhead	Winter	Wild	Unresolved	Healthy	Depressed

Strait of Juan de Fuca stocks of coho salmon have been depressed for several decades and likely declined to their all-time lowest levels in the early to mid-1990s. The Pacific Fisheries Management Council reviewed the status of coho populations in the Strait of Juan de Fuca region and concluded that none of the 48 independent drainages in this region supported healthy coho stocks (PFMC 1997). The study concluded that SJF coho populations as a whole are negatively impacted by low freshwater survival, low marine survival rates and high marine interception rates.

Sporadic spawning ground surveys by WDFW in Deep Creek between 1950-1970 reported counts as high as 206 fish/mile (330 fish/km). Repeatable surveys of index areas have been conducted in Deep Creek and Sadie Creek (East Twin tributary) since 1984 by WDFW. These index areas may provide an indication of temporal trends, but cannot be reliably expanded into an estimate of watershed-level spawner abundance. The Deep Creek index reach (river mile 0.0-1.3 /km 0.0-2.1), was established primarily to assess chum salmon and its utility in evaluating coho salmon trends in Deep Creek is unclear. However, these data suggest a decline in fall coho populations in Deep Creek since 1989. Populations in Sadie Creek have varied cyclically with relatively low numbers of spawners (Figure 6).

Significant efforts have been made since 1997 to improve estimates of spawning salmon abundance in Deep Creek and East and West Twin rivers. A stratified random sampling system of available habitat types was initiated in 1997. This new system enables estimation of individual watershed escapement.

Estimates of coho escapement using this system to the Deep/Twins watersheds are depicted in Figure 7. Escapement to each individual watershed has been consistent in four of the five years with Deep Creek supporting the highest number of spawning coho followed by West Twin then the East Twin River.

The status of Winter Steelhead was considered healthy in the early 1990's (as a result of higher escapement to the Pysht River). However, more recent information indicates Deep and Twin River steelhead are in decline (Figure 8). Formal steelhead escapement surveys were only initiated in 1995, limiting the ability to determine long-term trends in watershed escapement. Winter steelhead adults enter the watershed beginning in December and continue through May. Spawning occurs in February through early June. The stock is currently managed for wild production and no hatchery outplants have been released in the Deep/Twin complex since the early 1980's.

<u>Smolt Enumeration</u>
The Elwha Klallam Tribe installed smolt traps in Deep Creek in 1998 and in the East and West Twin Rivers in 2001. Traps, consisting of a fence weir and live box, capture the entire population of emigrating smolts. Trapping begins in late April and continues through mid-June. Peak outmigration occurs in late May. Data collected to date suggest that steelhead smolt production has declined in all three watersheds (Figure 9), while no apparent trend in coho production (Figure 10) has occurred over the relatively short period of record.

Habitat Treatments

<u>Deep Creek</u>
A number of habitat conditions in the Deep Creek watershed, related to historic timber harvest and road construction, were identified during watershed assessments conducted in the 1990s (Table 9). Compromised conditions varied among reaches but generally included alterations in habitat quality, temperature, sediment, large wood, and channel stability.

Table 9. Factors Limiting Smolt Production in SJF complex

Factors limiting smolt production
Excess sediment delivery due to elevated rates of mass wasting
Streambed scour from dam-break floods
Lack of wood in channels and elevated temperatures due to reduction in conifer trees in riparian areas
Loss of off-channel, floodplain habitats (side-channels, alcoves, associated wetlands) from channel incision

In response to declines in both habitat quality and populations of native anadromous fish, the Lower Elwha Klallam Tribe has been actively attempting to restore fish populations within Deep Creek. A restoration strategy was developed with the goal of reestablishing the dominant physical processes that controlled the identified limiting factors. This strategy is outlined in McHenry et al. (1995) and includes the following:

- Reduction in the rate of mass wasting to historical background rates
- Reestablishment of late successional, conifer-dominated riparian forests.
- Reintroduction of large pieces of wood (LWD) to channels.

- Re-creation of off-channel habitats.

Restoration efforts in Deep Creek were initiated in 1997 by the Elwha Klallam Tribe and continue today. Tribal efforts have focused upon the latter three categories. Increased rate of mass wasting in Deep Creek has been caused by poorly constructed roads. In 1999-2001, road maintenance and abandonment were conducted on some hazardous road segments within the watershed. A recently completed NEPA analysis of the entire 3040 road system, which has generated dozens of landslides not only to Deep Creek but also to the East Twin and West Twin rivers, has concluded that significant (~30 miles) portions of this mid-slope road system should be decommissioned. The U.S. Forest Service has funding to achieve approximately half of the proposed decommissioning and the North Olympic Peninsula Lead Entity was awarded SRFB funding in January 2006 to complete remaining treatments. Because this corrective action will be taken simultaneously in all three watersheds in the complex (including the reference watershed) evaluating responses to this treatment is not part of the study.

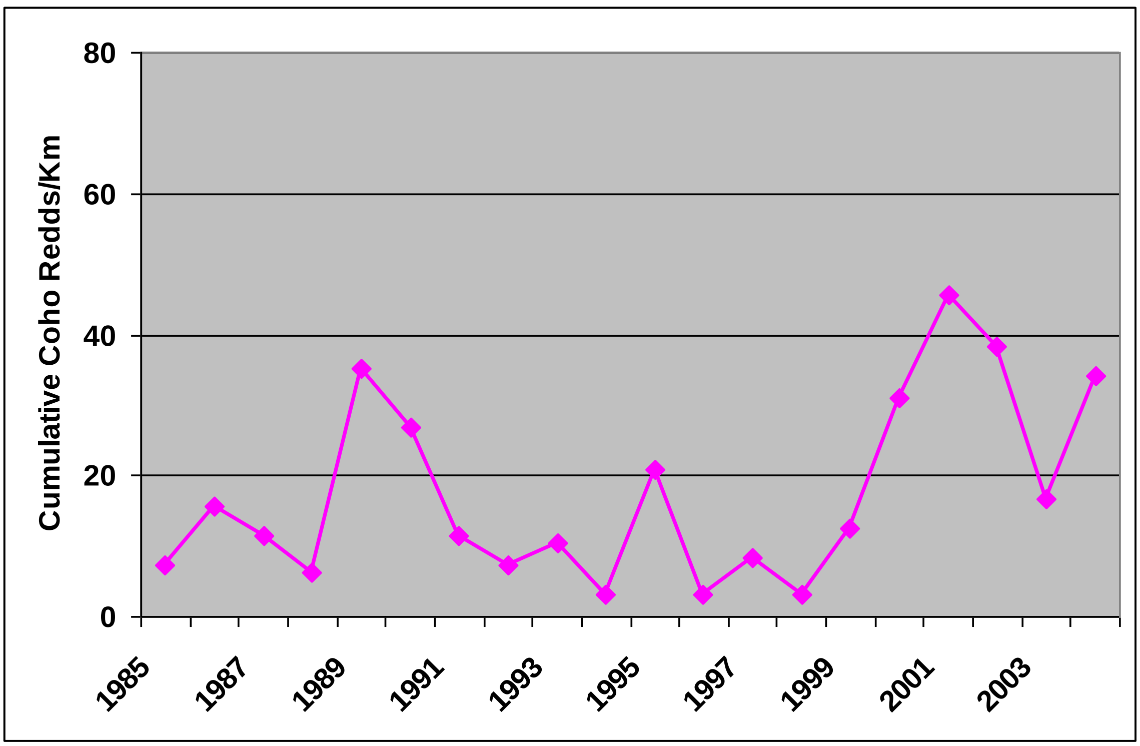

Figure 6. Coho salmon escapement (redds/km) to WDFW index area on Sadie Creek (1984-2004).

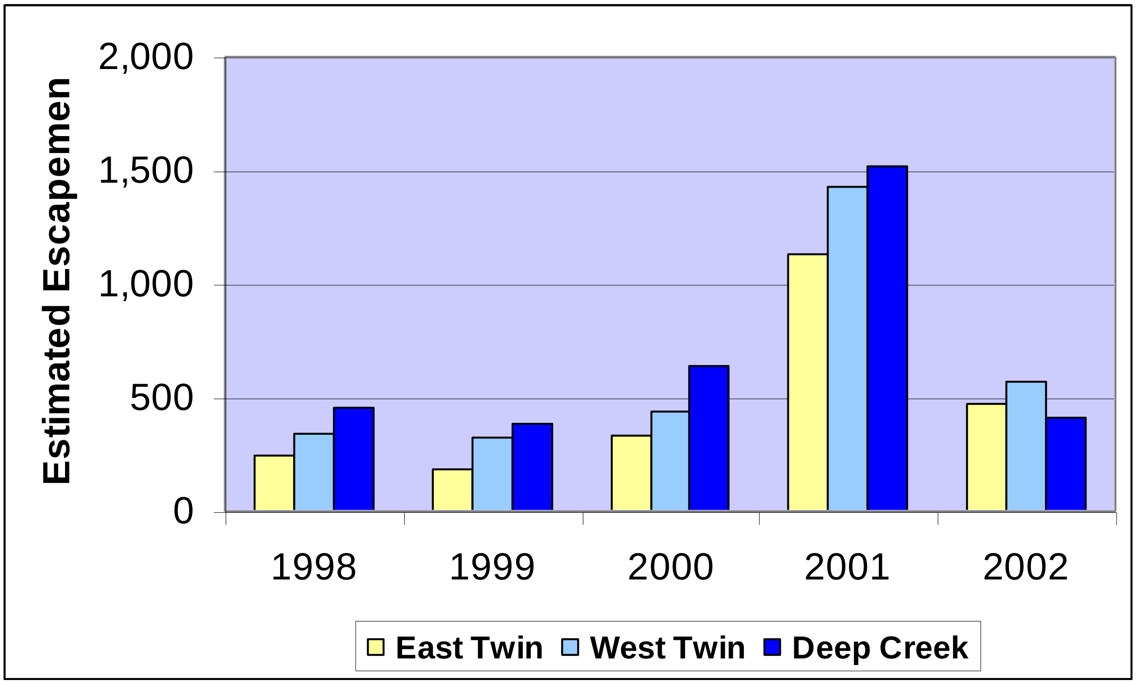

Figure 7. Coho salmon escapement to Deep Creek and East Twin and West Twin Rivers, 1998-2002.

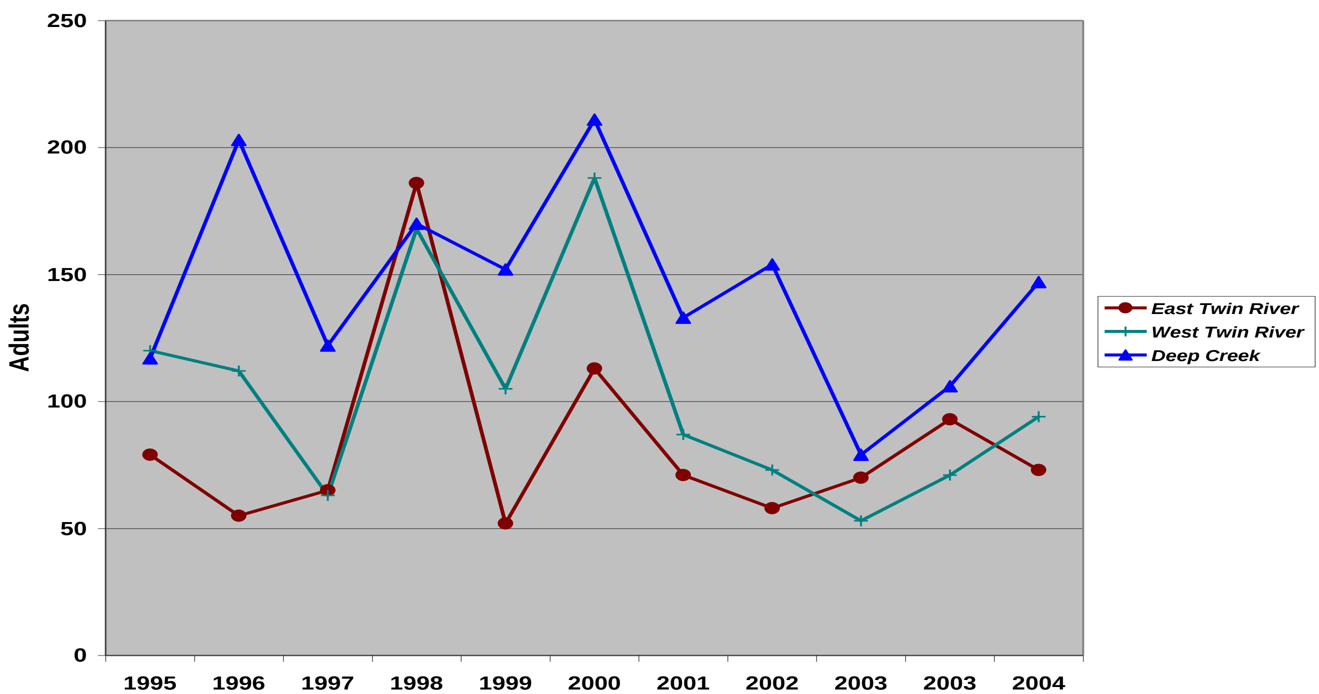

Figure 8. Steelhead escapement to Deep Creek and West Twin and East Twin Rivers, 1995-2004.

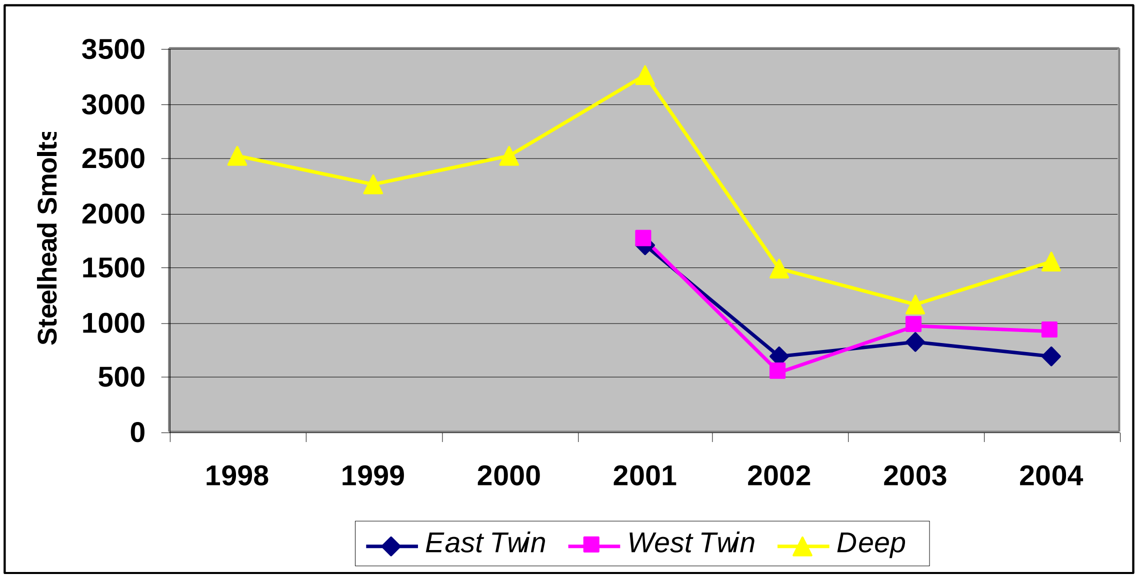

Figure 9. Steelhead smolt production from Deep Creek and West Twin and East Twin Rivers, 1998-2004.

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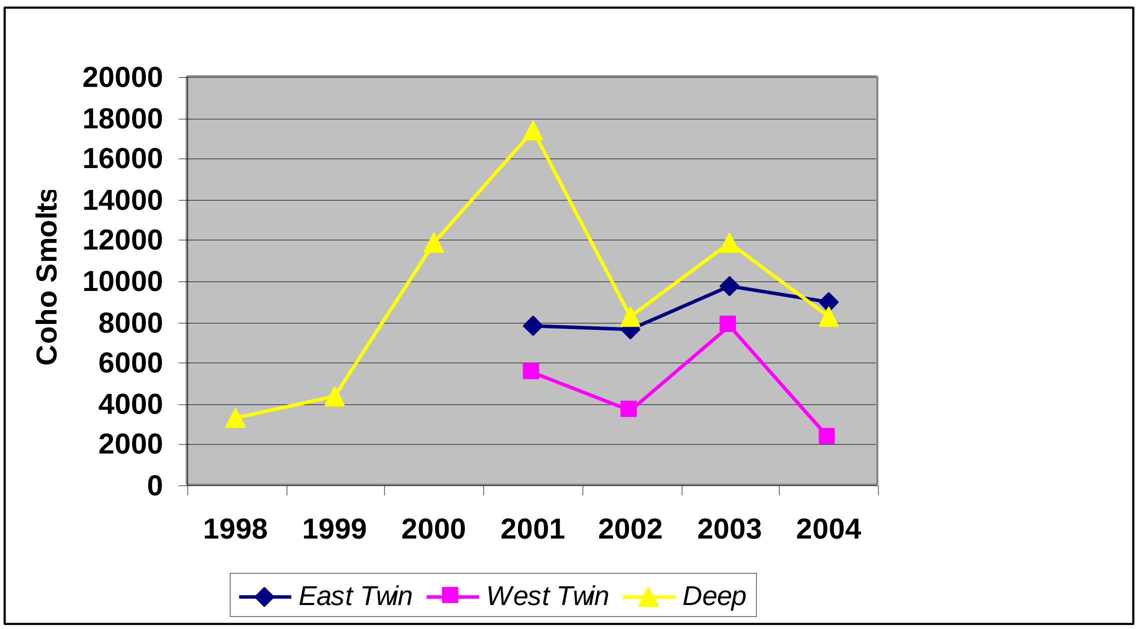

Figure 10. Coho smolt outmigration from Deep/Twin Rivers, 1998-2004.

Channel restoration activities on Deep Creek are focusing on using LWD to accomplish specific goals, depending upon the dominant impact at the reach level (Figure 11). For example, above RM 1.3, the 1990 dam-break flood resulted in severe scour of the bed and the almost complete loss of in-channel LWD. Conversely, below RM 1.3, the impacts were primarily associated with sediment deposition (pool filling, channel widening). Because of the inherent channel instability observed below RM 1.3, restoration activities were initiated above this point (RM 1.0 to 4.0). Between 1997-2002 LWD and rock was placed in an attempt to convert this plane-bed reach into a forced pool-riffle reach. Over 1,500 individual pieces of LWD have been used in the following configurations: log revetments, engineered log jams, constructed log jams, deflectors, log weirs, and rock/log structures. Additionally rock weirs were used in some locations to build channel bed features. In 2004-05 restoration activities focused on the lower reaches of Deep Creek (RM 0 to 1.3) and large, complex logjams (including channel spanning) were constructed at 23 locations. To date, 4.0 miles of Deep Creek, 0.5 miles of Sampson Creek, and 0.4 miles of Gibson Creek (Deep tributaries) have received in-stream restoration treatments, while riparian vegetation improvements have been conducted on 2.5 miles of riparian forest. The riparian vegetation projects included manipulation of existing stands to promote the growth of conifer-dominated riparian stands. Four off-channel, winter rearing habitat projects have been implemented.

East Twin River

A watershed analysis (USFS 2002) conducted in the 1990s identified the same suite of factors affecting habitat condition in East and West Twin rivers as Deep Creek. However, logging related disturbances have been less severe in the Twin Rivers than Deep Creek.

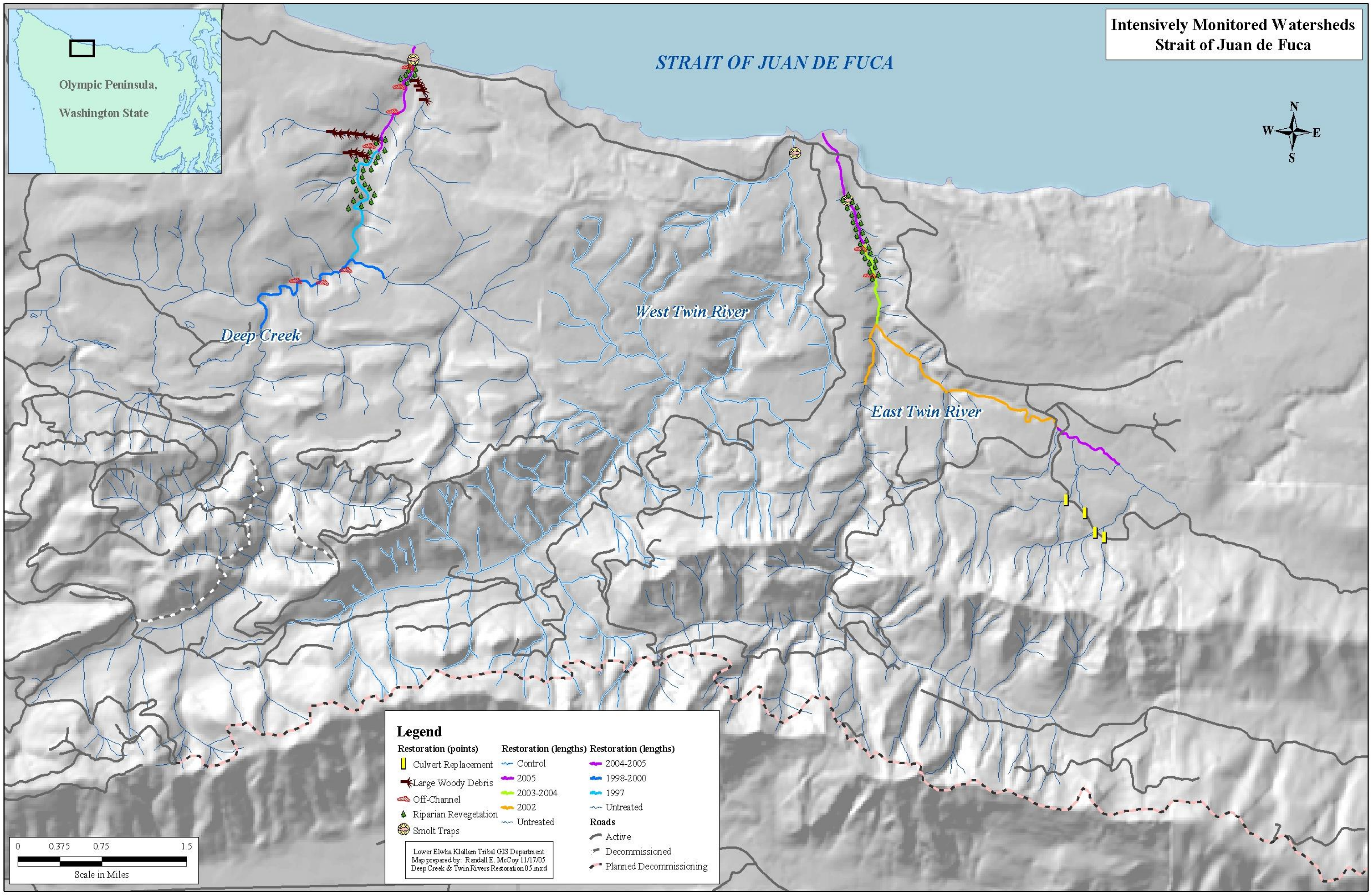

Figure 11. Restoration projects in Deep Creek and East Twin River.

Restoration efforts in the East Twin River were initiated in 1998, when an off-channel rearing pond was constructed on private property near river mile 1.0 (km 1.6). Large scale LWD reintroductions were initiated in 2002 by the Elwha Klallam Tribe when a Salmon Funding Recovery Board awarded a restoration grant to fund these efforts. In the summer of 2002 over 450 metric tons of LWD was placed with a helicopter into Sadie Creek at forty sites in river mile 0-2.0 (km 0.0-3.2) and at 30 sites in the East Twin River in river mile 2.0-3.0 (km 3.2-4.8). These efforts were followed in 2003-04 with ground-based placement at an additional 35 sites in the East Twin at river mile 1.2-2.0 (km 2.0 and 3.2). An estimated 50 year flood occurred in October of 2003, resulting in substantial habitat response to restoration. Additional ground based treatments were completed by the Tribe in 2005 between river mile 0.3-1.0, with the addition of complex LWD structures at 16 sites.

Based on the effects of habitat restoration in Table 5 and the restoration completed to date, the expected increases in smolt production were calculated from existing data collected in Deep Creek and East Twin River (Table 10). Restoration in Deep Creek is expected to result in an increase of 2684 coho smolts, a 24% increase in mean annual production. The increase in East Twin River coho smolt production was calculated at 1855, an increase of 22% over the mean. While these estimates are less than the minimum detectable changes using a Before-After analysis (analogous to the Treatment vs. Reference analysis used in this complex) in Table 6, the research described below will likely improve our ability to detect a change by providing detailed information about life stage specific movement, growth, and survival of juvenile fish.

Table 10. Calculated effects of habitat restoration on coho smolt production in Deep Creek and East Twin River were based on the literature review summarized in Table 5.

Stream	**Wood placement**			**Off channel habitat**		**Total smolts/yr**	**% of mean**
	meters restored	**parr produced**	**smolts produced (estimated)**	**m^2 habitat restored**	**smolts produced (estimated)**		
Deep Creek	5632	3147	1187	4046	1497	**2,684**	**24%**
East Twin R.	6437	3597	1357	1347	498	**1,855**	**22%**

West Twin River

No restoration will be conducted in West Twin River. This watershed will serve as a reference watershed and habitat conditions and fish populations will be compared over time to Deep Creek and East Twin River where active restoration is underway.

Response to Treatments

In addition to the measurements collected on all the IMW watershed complexes (described above) several evaluations specific to the SFJ complex also are being implemented. Prior to beginning the restoration efforts, channels in areas where restoration might be implemented (virtually all areas accessible to anadromous fishes) were mapped. In addition, extensive habitat measurements were made using the TFW

Ambient Monitoring methodology (Schuett-Hames et al. 1994). Repeat surveys of habitat conditions have been conducted at intervals (1992, 1995, 1997, and 2003) as restoration has progressed. Thirty-six permanent cross-section stations have been established throughout the areas where treatments have been applied and at nearby reference sites to measure changes in channel bed elevation and substrate size (Wolman 1954). The cross-sections have been periodically re-surveyed (1998, 1999, 2002, 2005) to assess restoration effects.

The standard juvenile, smolt, and adult monitoring in the Straits IMW complex provides some information on the effects of restoration on fish abundance (methods described above). This information is being augmented through the use of passive integrated transponder (PIT) tags. Prior to the development of PIT tag technology and the more recent development of remote detectors, collecting accurate survival, movement and migration information was difficult. The recent improvements in this technology have enabled us to compare fish abundance, survival, movement and migration timing (life history) among watersheds, reaches, and habitats before and after completion of restoration treatments. Specifically, we are focusing on the following questions:

1. What is the effect of habitat restoration activities throughout the watersheds on survival, growth and migration timing of fish?

2. Do survival, growth and movement differ among tributaries and reach types within the watersheds? Will the application of restoration projects cause differential responses in these variables among locations and reach types?

3. What is the effect of reach-level restoration efforts on local movements and growth of fish?

4. Does survival, growth and movement differ among habitat types (e.g., pools, riffles, glides) and can we improve survival by creating more pool habitats?

Initially, we set out to answer the third question by examining differences in survival and movement between restored (complex habitat with high levels of LWD) and unrestored reaches (simple no LWD placement) in East Twin river. This effort served as a pilot study to assess the capabilities of new PIT tag technologies and provided us with the methodologies required to address the remaining questions.

Stationary multiplex PIT tag readers were installed in East and West Twin Creek in the summer/fall of 2004 allowing for the detection of PIT tagged fish passing the readers (Figure 11). These were located approximately 1000 m and 500 m from saltwater in East and West Twin, respectively. These detectors run throughout the year, although they are occasionally inoperable when damaged by high flows. Generally, the detectors can be repaired within a week after such damage. Deep Creek currently has no PIT tag detector, although we are attempting to identify funding to enable installation.

To address hypothesis three and examine survival and whether fish moved between restored and unrestored reaches, we examined fish movement in two simple and two complex (LWD enhanced) 100-meter-long reaches in East Twin River in 2003 and 2004. During late summer (August and September), about 800 trout and coho were collected by electrofishing, anesthetized, measured, weighed, PIT tagged, and released into their habitat of origin. Movement of the tagged fish was monitored with a hand-held reader used to interrogate fish encountered during periodic snorkel surveys. This work is part of a University of Washington masters thesis (T. Bennett).

Continuous PIT tag detectors were not in place in spring 2004. Surprisingly, only about 5% of the fish PIT tagged in late summer 2003 were captured in smolt traps in spring 2004. The low tag recovery rate in 2003 also suggested that large numbers of fish (>1,500) needed to be tagged. The reason for this low recovery rate could have been high mortality rates or migration of tagged fish from East Twin River prior to the installation of the smolt trap. A stationary PIT tag reader, located near the site of the smolt trap, was installed in 2004 to determine if early emigration of the fish was the cause of the low, spring recovery rate.

In 2004 nearly 3000 fish were tagged in East Twin and West Twin (Table 11). The stationary PIT tag reader indicted that large numbers of coho and trout parr emigrated from the study watersheds in the autumn. As a result of this finding, overwinter survival is being calculated by dividing the number of tagged spring migrants by the total number of tagged fish minus the fall emigrants (survival = spring migrants/ (total tagged – fall migrants)).

Table 11. Number of trout and coho PIT tagged in 2004, 2005, and proposed tagging in 2006.

	2004		**2005**		**2006 (proposed)**	
	E. Twin	**W. Twin**	**E. Twin**	**W. Twin**	**E. Twin**	**W. Twin**
Coho	2,208	189	3,200	2,913	2,500	2,500
Trout	475	92	1477	1710	1,500	1,500
Total	2,964		9,300		8,000	

The number of PIT tags deployed in the study streams was further increased in 2005 and a permanent tag reader was installed on West Twin. We PIT tagged 9,300 juvenile coho and trout in East and West Twin in August and September 2005. About one third of these fish were tagged at randomly selected reaches and the remainder was tagged in the lower few kilometers of the East and West Twin where most of the anadromous fishes are concentrated and we could efficiently collect large numbers of fish. This broad-scale spatial tagging effort in 2005 will not only allow us to compare fish survival, growth, and migration between the treatment (East Twin) and control (West Twin) (question 1), but also allow us to answer questions 2 and 3 as fish were tagged throughout the watersheds and we have information on reach types and habitat types where fish were tagged. We will continue tagging approximately 3,500 juvenile coho and trout in each watershed each year, including Deep Creek in 2007 if funding is available.

Statistical analysis

Differences in survival, growth, and migration among tributaries, reaches and habitats are being compared using an ANOVA or t-tests (survival, growth), graphical analysis (migration timing), and chi-square test or Kolmogorov-Smirnov goodness of fit tests (movement, migration timing). As we collect multiple years of data for each watershed, we will utilize specific metrics such as median migration date and proportion of fall migrants etc. to compare among treatment and control watersheds and among years. These will be compared using parametric statistics such as ANOVA or ANCOVA assuming data are normally distributed. If not, we will apply graphical or nonparametric statistics to examine differences among watersheds and years.

Preliminary analysis and results

<u>Watershed scale</u>

Data from the PIT tag readers and marked fish indicated that unexpectedly large numbers of parr (primarily coho) migrated to sea during fall months (Figure 12). Relatively few parr migrated during winter (January through March) and the largest numbers emigrated as smolts in the spring. A t-test indicated that fall-migrating coho were significantly smaller at tagging than spring coho migrants (64.1 and 74 mm, respectively) (Figure 13). This may suggest that smaller, less fit fish are forced out in the fall or seek other foraging opportunities outside the watershed. The relative contribution of fall and spring-migrating coho to adult returns will be assessed as returning tagged adults are detected at the permanent PIT tag readers and by examining carcasses for tags

Total numbers of tagged smolts captured in the East Twin smolt trap in spring 2005 was 228 coho, 2 cutthroat trout, 32 steelhead, and 7 age 1+ trout, a total of 269 tagged fish. However, 388 passed the PIT reader located a short distance upstream from the smolt trap. Possible explanations for this discrepancy include trap avoidance, predation on tagged fish, and smolts moving to past the trap location during a period when trap panels were pulled for high water between May 23 and May 25, nearly at the height of migration.

After the installation of the West Twin reader in 2004, we recorded four PIT tagged fish moving between East Twin River and West Twin River during the summer. Although this represents a very small proportion of tagged fish, it was surprising that 500 m of saltwater between the two river mouths did not present a barrier to movement of these fish.

As indicated earlier, 9,300 juvenile salmonids were tagged in East Twin and West Twin in the summer of 2005. These data will be analyzed and summarized following the completion of 2006 migration period.

<u>Reach scale</u>

Percentage of fall migrants was not significantly different between treatment and control reaches (t-test, $p > 0.10$; Table 11. Similarly, survival estimates from the complex and simple reaches did not differ (t-test, $p > 0.10$), but complex reaches had higher densities of fish. This suggests that habitat enhancement through LWD placement leads to more smolts not through higher survival, but increased densities.

Table 12. Survival estimates for coho tagged in simple and complex reaches and East Twin versus Sadie Creek (major tributary) in 2004. Fall migrants are the number of tagged fish detected passing the PIT readers before January 1. Overwinter survival was calculated as the # spring migrants/ (total tagged fish - # fall migrants). Number in parenthesis is number of outmigrating fish detected in fall or spring.

	Percentage (#)			
	Simple	**Complex**	**East Twin**	**Sadie Creek**
Fall migrants	20.9 (151)	20.1 (148)	21.6 (351)	2.6 (15)
Overwinter Survival	9.3 (53)	10.2 (60)	10.2 (130)	22.4 (127)

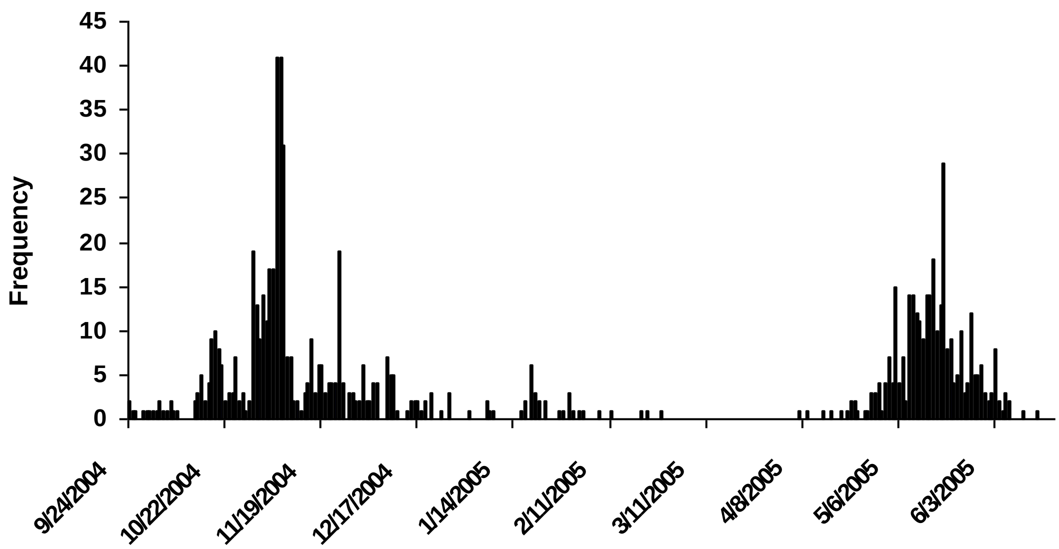

Figure 12. Fall 2004 and winter 2005 migration of East Twin River juvenile coho and trout tagged in August and September of 2004. A total of 459 trout and coho were detected by the PIT tag reader leaving the system between September 24 and December 31, 2004, while 356 fish were detected leaving the system between January 1 and June 20, 2005.

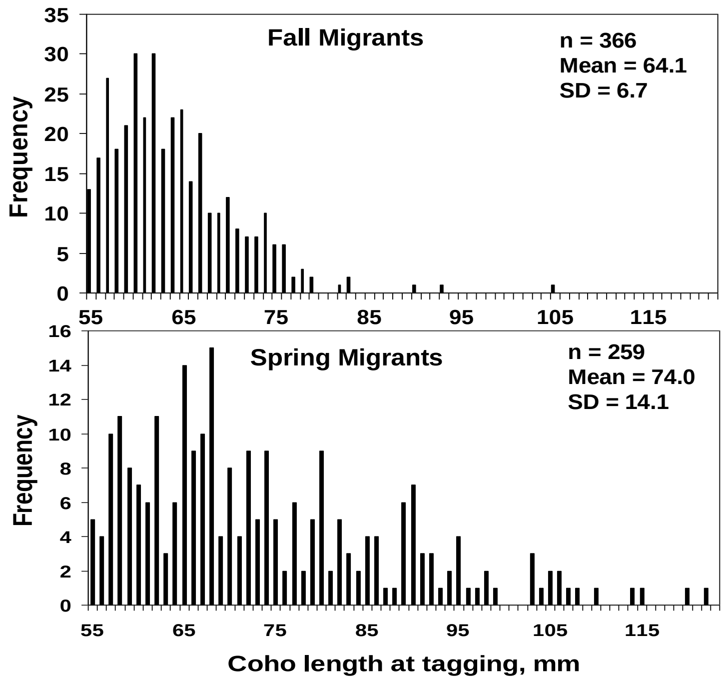

Figure 13. Length at tagging of juvenile coho salmon tagged in 2004 moving past the East Twin Creek PIT tag reader in fall 2004 and spring 2005.

Habitat scale

Data are currently being analyzed and collected to examine differences in survival, growth and migration among pools, riffles, glides and between natural and constructed habitats.

Other observations

Maintaining the permanent PIT tag readers presents some challenges. The readers require a substantial amount of continuous power. Power was initially supplied by eight, 12-volt, car batteries that needed to be replaced/recharged on a weekly basis. The battery system on the East Twin River reader was replaced in June 2004 with a thermoelectric generator powered by liquid propane stored in a 100-gallon propane tank. This power supply is much more reliable and can power the system for 60 days without service. This option is being assessed for the reader on West Twin. The West Twin reader suffered serious damage when it was inundated in a large flood. It was subsequently replaced and the electronics moved to higher ground.

Hood Canal Complex

The Hood Canal complex has the longest smolt production record of the three IMW complexes allowing a more extensive analysis of correlates with smolt production and a longer calibration period of pre-treatment smolt production in treatment vs the reference watershed. Landuse is more diverse in this complex than the other two, including commercial forestry, expanding rural residential development, and even some urban growth from the Silverdale area into the Little Anderson basin. Although the diverse land use patterns complicate our efforts to determine the effects of restoration on fish production, the basins' small size will allow us to treat a substantial portion of the stream network, thereby producing a large change in habitat condition over a short period.

Description

These four basins (Figure 14), on the west side of the Kitsap Peninsula, comprise a large portion of the West Kitsap WAU. This WAU is within the Puget Sound trough which has experienced considerable glacial activity and, as a result, generally has a gently rolling upland of glacial till with steep-sided ravines leading down to the river floodplains. The glacial till of the uplands is fairly resistant to erosion but the loose sandy soil and layers of fine textured material comprising the ravine sideslopes are very erodible. In addition, layers of clay in the ravine walls can transport water laterally and where this intersects a road cut, ground water often flows onto the road.

Commercial logging of lowland areas was underway by 1870 with the establishment of large sawmills. Extensive logging of the uplands began in the 1920s when a railroad network was built to transport the timber and continued into the 1940s until few merchantable trees were left. Although forest practices have improved markedly, legacy effects may exist. Based on early 1990's satellite imagery, over 80% of each basin is forested and the proportion developed is low (Table 13). However, rural residential development has increased continuously since the 1970's and may be degrading habitat through riparian vegetation removal, stormwater runoff, fish passage barriers, and high sediment loads (WA DNR 1995; Seiler et al. 2002).

Naturally produced salmonids from the Hood Canal Complex include coho salmon, fall chum salmon, cutthroat trout, and a small population of steelhead. Efforts are being made to establish a naturally-reproducing population of summer chum in Big Beef Creek. The University of Washington maintains an artificial production facility on Big Beef Creek, where summer chum and chinook are reared. All chinook returning to the creek are sorted at a weir located at the mouth and precluded from migrating upstream to spawn in the wild. All of the releases from this facility occur downstream of the weir and, therefore, do not effect the wild juvenile downstream migrant counts at Big Beef Creek. Hatchery fish are not released in any of the other Hood Canal Complex streams.

Smolt counts began in Big Beef Creek in 1978. Smolt counts in the other three streams date from 1992 or 93 (Table 14). Wild coho salmon from Big Beef Creek have been coded wire tagged since 1976. Historically, a substantial portion of the harvest occurred in outside fisheries (i.e., Vancouver Island Troll Fishery, Washington Troll & Sport Fisheries). As these fisheries became increasingly constrained by weak-stock management and ESA, terminal harvests in the Hood Canal Net Fishery have made up the bulk of the fishing impact on this stock. The terminal Area 12 fishery is centered around Big Beef Creek and extends as far north as Lone Rock and as far south as Stavis Bay. Sampling over the last two years indicated catch rates can be highly variable. In 2002, we estimated a 68% total exploitation rate on tagged, wild Big Beef coho with 98% of the impact occurring in the Area 12 beach seine fishery. Yet in 2003, very few fish were harvested in this fishery as the bulk of the effort was centered in the Areas to the south.

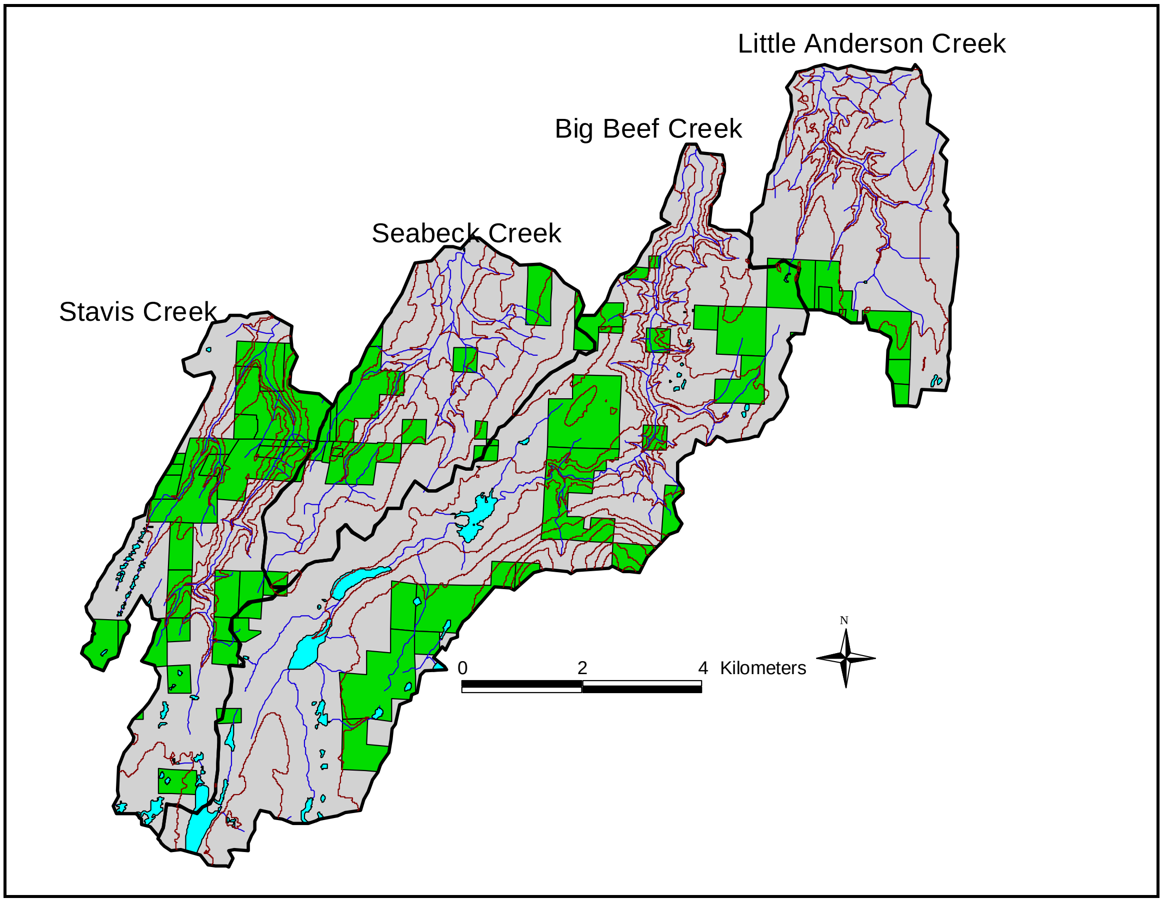

Figure 14. Hood Canal IMW Complex. Washington Department of Natural Resources land is green. Lakes and wetlands are blue. Contour intervals are 100m.

We have drawn upon the following data sources in developing our hypotheses of freshwater production constraints in these basins:

- smolt and adult escapement counts at the Big Beef Creek weir since the late 1970s (WDFW);
- Stream discharge has been measured near the mouth of Big Beef Creek by the USGS since 1969 (http://nwis.waterdata.usgs.gov/wa/nwis/discharge/?site_no=12069550) and above Lake Symington by the Department of Ecology since 2000 (http://www.ecy.wa.gov/apps/watersheds/flows/station.asp?sta=15F150) ;
- smolt counts in the other three streams since 1992 or 1993 (WDFW);

- Sporadic coho and chum spawning ground surveys in all four basins (WDFW);
- Habitat surveys in all four streams conducted by Point No Point Treaty Council and US Fish & Wildlife Service in 1993 (USFWS, 1993);
- 1998 Ecosystem Diagnosis and Treatment analysis of Big Beef Creek;
- West Kitsap Watershed Analysis (WA DNR 1995);
- Habitat surveys conducted on all four streams by WDFW in 2000-2002;
- The West Kitsap Limiting Factors Analysis (Kuttel 2003);
- Salmon Index Watershed Monitoring (Seiler et al. 2002); and
- The Kitsap Salmon Refugia Report (May and Peterson 2003).

These data sources were used to develop descriptions for each watershed and analyzed to formulate hypotheses regarding factors constraining salmonid production in each basin. Most of the discussion on production constraints will focus on coho salmon. Cutthroat trout utilize habitats similar to those preferred by coho. Steelhead typically utilize larger, higher gradient channels. Steelhead production in these basins is very low, likely due primarily to the small size and low gradient of most channels in these watersheds.

Table 13. Land cover, land management, and ownership percentages for each trap basin are shown below. Land cover is based on satellite imagery from the early 1990s. Public ownership was based on the Major Public Lands map.

Smolt trap	Basin area (km^2)	Land cover (%)		Ownership (%)	
		Forested	Developed	Public	Private
L. Anderson Cr	13	87	8	12	88
Big Beef Cr	36	90	3	31	69
Seabeck Cr	14	91	2	21	79
Stavis Cr	13	83	2	45	55

Table 14. Period of record and data collected at each smolt trap.

Smolt trap	Juveniles		Adults	
	Since	Species	Since	Species
Anderson Cr	1992	coho	2004	Coho
Big Beef Cr	1978	coho, cutthroat, steelhead	1976	chinook, chum, coho
Seabeck Cr	1993	coho	2004	Coho
Stavis Cr	1993	coho	2004	Coho

Little Anderson Creek

Little Anderson Creek is an independent tributary to Hood Canal located east of Big Beef Creek. The Little Anderson Creek watershed has an area of approximately 13-km^2 and is the smallest of the Hood Canal IMWs (Figure 15). It is bordered on the east by the City of Silverdale and a part of the watershed is within the urban growth boundary of the city.

Little Anderson Creek is used by coho and chum salmon, and cutthroat trout. A few steelhead also spawn in the stream each year. Hypothesized constraints to coho production are listed in Table 15 and discussed below.

Table 15. Factors limiting coho smolt production in Little Anderson Creek.

Factors limiting smolt production
Preferred habitat is limited to the lowest 2.0-km of the mainstem. (High gradient tributaries with little or no summer flow provide little habitat.)
Main channel lacks LWD to control bed movement and create rearing habitat
Steep hillslopes, high channel gradients, and altered hydrology may degrade stream channels upstream of river kilometer (RK) 2.0 and scour and/or bury redds below this point
Fisheries may exert a higher-than-sustainable impact on Little Anderson Creek coho given its current low productivity

Most of Little Anderson Creek and its tributaries are deeply incised. Stream gradient within the fish-bearing portions of Little Anderson Creek is high, averaging 3.1% (WDFW unpublished data). Most suitable habitat is found in the lower 2.0 km of the mainstem, where channel gradient is less than 2%. Upstream of this point, flow is evenly divided between the main channel and the right-bank tributary and the channel steepens to 3-5%. The increased gradient and decreased flow likely limits the use of the stream above this point by anadromous fishes.

Although stream banks are largely intact within the Little Anderson Creek watershed, averaging less than 0.3-m^2 of exposed bank per meter of stream length, bed scour has resulted in the transport of large amounts of sediment downstream. Large quantities of sediment were deposited in the lower reaches of Little Anderson Creek following a 1994 storm when a road fill with an undersized culvert on Anderson Hill Road failed. This incident released large amounts of sediment accumulated above the culvert and resulted in a braided channel below the culvert. Although the culvert was removed and a bridge was installed in its place in 2002, damage to the channel as a result of the 1994 storm were still evident in 2002. Low to moderate levels of LWD were available to retain gravel and create pools resulting in little spawning and rearing habitat (WDFW unpublished data). Recently, beavers have constructed several dams in this reach and the channel has shifted widely across the valley floor in response the dams and high flows.

Little Anderson Creek produces the fewest coho smolts of the four Hood Canal watersheds. Annual coho production has ranged from 45 to 833 smolts while cutthroat production has been much higher than coho in all but two of the ten years of record (Figure 16). In low gradient stream systems, coho smolt production is typically one or more orders of magnitude higher than cutthroat (Seiler et al. 2003b). Coho may be sensitive to peak winter stream flows as their eggs are in the gravel through the winter and thus subject to redd scour and sediment deposition (Figure 17). Cutthroat trout do not spawn until spring and avoid negative impacts associated with winter high flows. Summer low flow, an indicator of the amount of summer rearing habitat available, does not appear to limit Little Anderson Creek coho production (Figure 18). Winter high flows may be exacerbated in Little Anderson Creek by the increases in development and impervious surfaces over the last decade. The relationships shown in Figures 17 and 18 were made using Big Beef Creek flow data, the nearest stream gauge. Given its close proximity, it is expected that flow patterns in the Big Beef drainage is similar to the other Hood Canal IMWs.

Figure 15. Orthophoto of the Little Anderson Creek watershed; the horizontal line indicates the upstream extent of preferred coho and steelhead spawning and rearing habitat

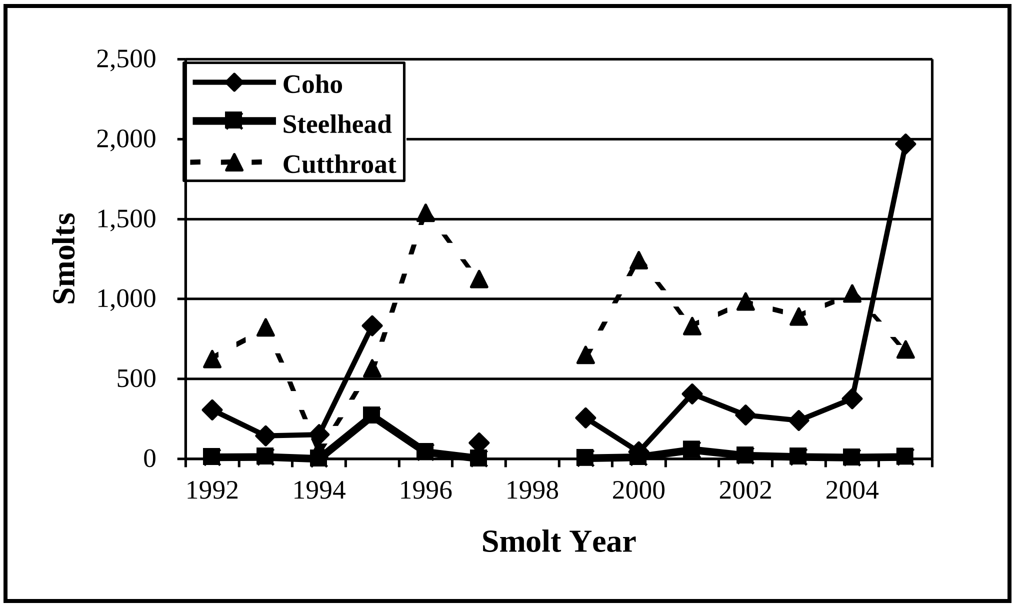

Figure 16. Annual production of coho, steelhead, and cutthroat smolts from Little Anderson Creek.

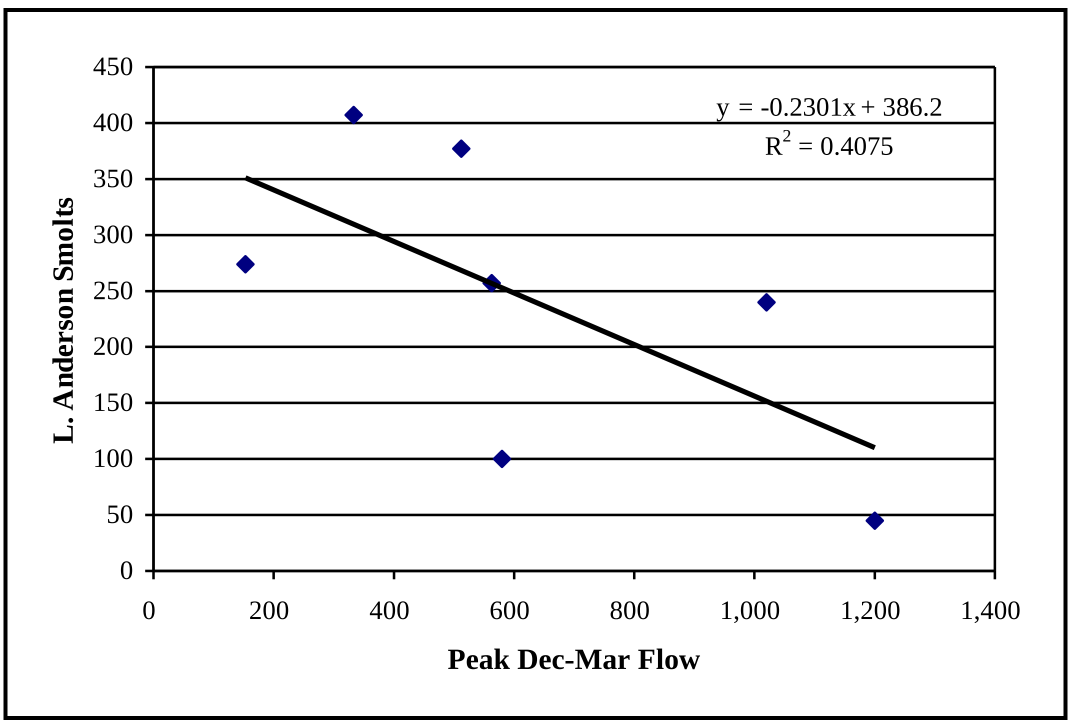

Figure 17. Little Anderson Creek coho production as a function of peak December to March discharge in Big Beef Creek.

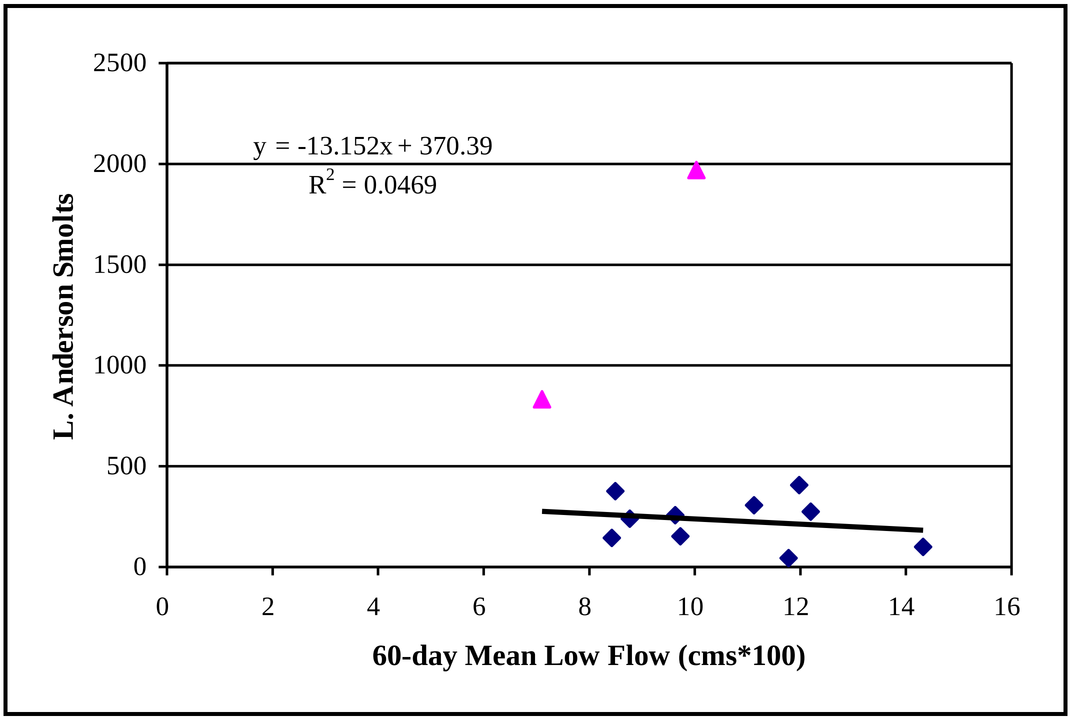

Figure 18. Little Anderson coho production as a function of the lowest 60-day mean flow in Big Beef Creek.

A potential impact is the terminal Area 12 fishery. It is likely that Little Anderson coho, which are not tagged, experience exploitation rates similar to Big Beef coho. Given their low productivity, these relatively high harvest rates on Little Anderson Creek coho may cause very low spawner returns in some years. Therefore, the low coho production observed in Little Anderson Creek may, in part, be due to low escapements. Weekly spawner and redd counts were initiated in 2004 during the coho spawning season to estimate escapement and spawner distribution.

<u>Big Beef Creek</u>

Of the four Hood Canal complex streams, Big Beef Creek is the largest, draining a 36-km^2 basin. The watershed may be divided into four sections. The upper watershed flows through an extensive network of wetlands (Figure 19). Channels in the upper watershed are low gradient and unconfined. Although similar wetlands are also found in the headwaters of Seabeck and Stavis Creeks, they represent a much less prominent feature in these watersheds than in Big Beef Creek. Below the wetland section, Big Beef Creek flows into Lake Symington, a shallow, man-made impoundment surrounded by a housing development. A fishway provides passage for adult and juvenile coho, steelhead, and cutthroat above the dam. Downstream of the reservoir, Big Beef Creek flows through a canyon to Hood Canal. The stream is highly confined through much of this reach. Within three kilometers of Hood Canal the channel becomes much less constrained as the canyon floor widens and the channel is bordered by a floodplain.   At the junction with

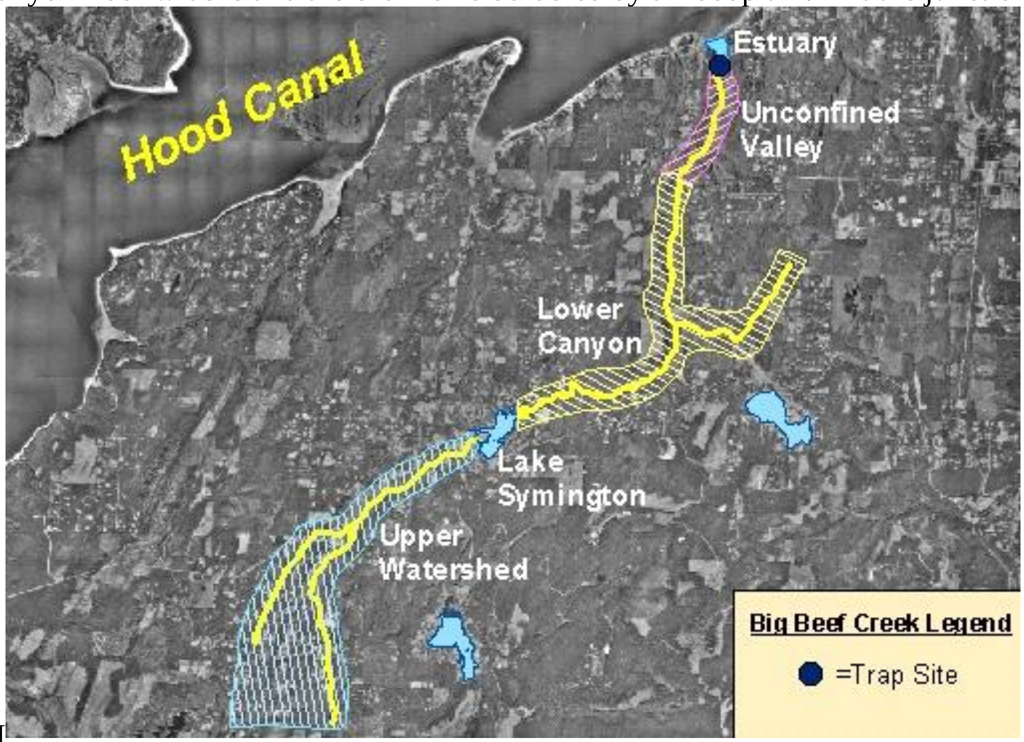

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Figure 19. Primary features in the Big Beef Creek watershed.

Hood Canal, Big Beef Creek forms a small estuary. The estuary connects to Hood Canal through a narrow opening in a causeway carrying the Seabeck Highway. The estuary was spanned by a 200-m wide bridge prior to the 1970s when the old bridge was replaced with the causeway and a 12-m wide bridge. Since then much of the estuary has filled with sediment. Formerly abundant salt marsh habitat that was inundated through all phases of the tidal cycle has largely been replaced by an incised channel meandering across a mudflat at low tide.

The distribution of spawning salmonids varies among these stream segments. Chum salmon spawn almost exclusively downstream of the Lake Symington dam. Coho steelhead and cutthroat access habitat further upstream in the watershed with the majority of coho and steelhead spawning above Lake Symington.

The University of Washington Big Beef Creek Research Station is located at the mouth of the stream. The facility includes a fish counting weir. WDFW built and currently operates this upstream/downstream trapping facility. Both adult salmon entering the watershed and downstream-migrating juveniles are captured at the weir. The trapping facility has been operating since 1976.

Big Beef Creek produces the most smolts of the Hood Canal complex watersheds. Coho smolt production has ranged from 11,500 to 47,000, and averages over 25,000. Over the 26 years trapped (1978 – 2003), coho salmon production has exhibited three short-term trends. Production between 1978 and 1986 showed a lot of inter-annual variation, but little trend, with an average production of 29,000 smolts (Figure 20). Coho production decreased between 1987 and 1996, averaging just over 19,000 smolts. Since 1997, production has returned to the pre-1987 level, averaging 30,000 smolts, and again exhibits considerable interannual variation. Steelhead and cutthroat production are an order of magnitude lower than coho production in Big Beef Creek. Production for both species has been trending slightly upward over the monitoring period. Hypothesized constraints to coho production are listed in Table 16 and discussed below.

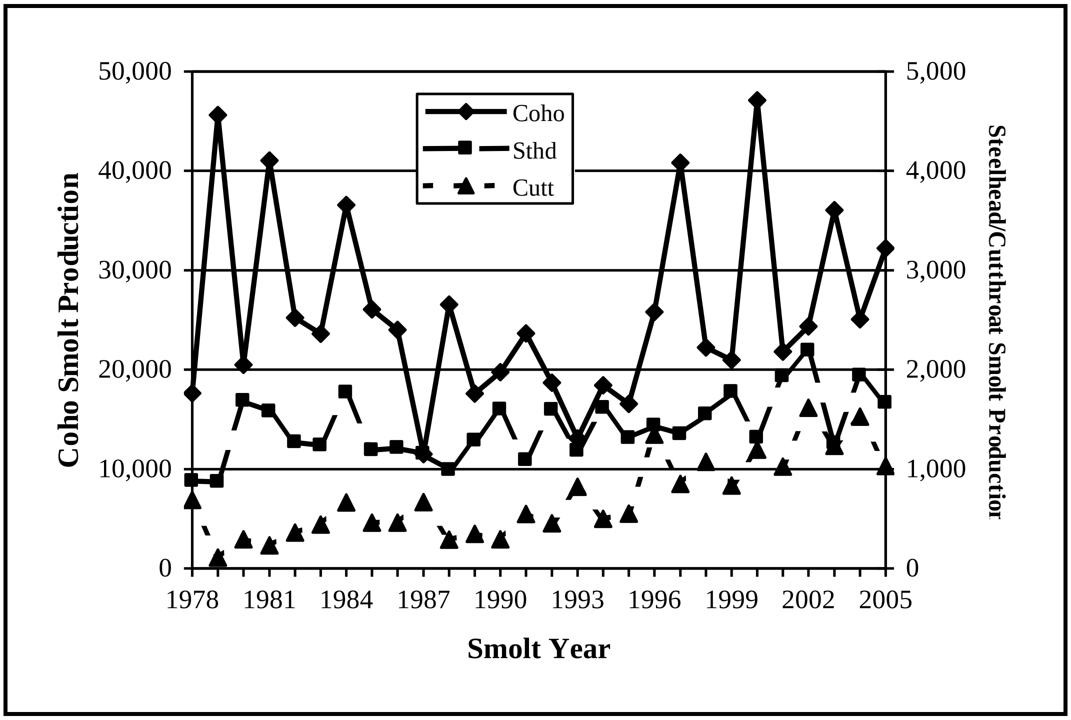

Figure 20. Big Beef Creek coho, steelhead, and cutthroat smolt production.

Table 16. Factors limiting coho smolt production in Big Beef Creek.

Factors limiting smolt production
Extremely low summer base-flow limits the availability of summer rearing habitat in the lower and unconfined valley
Low fall flows limit access to spawning habitat in the upper watershed
Predation by largemouth bass and other exotics on coho salmon over-wintering in or migrating through Lake Symington reduces the survival of offspring produced above the lake
High summer water temperatures reduce available rearing habitat in Lake Symington and a portion of the canyon below the lake
Land use actions have greatly increased coarse sediment inputs from adjacent hill slopes and tributaries in the lower canyon filling pools and widening channels in the lower canyon and unconfined valley sections, thereby reducing rearing habitat and channel stability
Removal of large cedar debris in the lower canyon and unconfined valley in the 1980s has destabilized the channel and reduced rearing habitat

Coho smolt production is positively related to the peak November streamflows (Figure 21). Flow data for Big Beef Creek are only available for nine of the 27 years of coho smolt production record. However, a similar relationship was found using the maximum 3-day November precipitation measured at Bremerton as a surrogate for flow (Figure 22). The relationship between smolt production and high flow during the spawning period suggests that higher fall flows may enable spawners to reach areas of the watershed inaccessible during years of lower flow, thereby increasing the amount of spawning and rearing habitat.

A similar positive relationship was found between smolt production and summer base flow (60 day average flow) (Figure 23). The outlier in the plot coincided with very low November spawning flows, suggesting that limitation on access to spawning habitat may have limited production for this year. Channels in the Hood Canal complex watersheds are prone to dewatering during dry weather. As a result, rearing habitat during summers with low flows may be greatly reduced from habitat area available during higher flow years. Given that some of these watersheds are undergoing fairly rapid development, there is the potential for this situation to be exacerbated as impervious area and water use increases.

A study has been initiated as part of the IMW effort in this complex to better understand the mechanisms causing the observed relationships between flow parameters and smolt production. The specifics of this effort are described later in this plan.

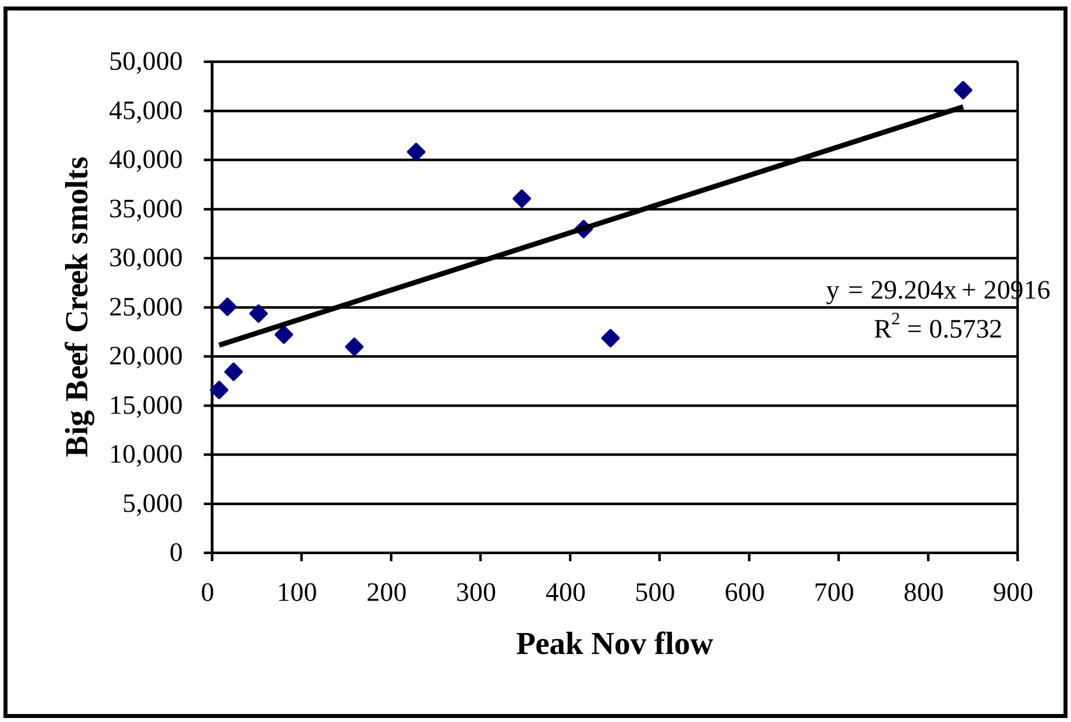

Figure 21. Big Beef Creek coho production as a function of peak November spawner flows.

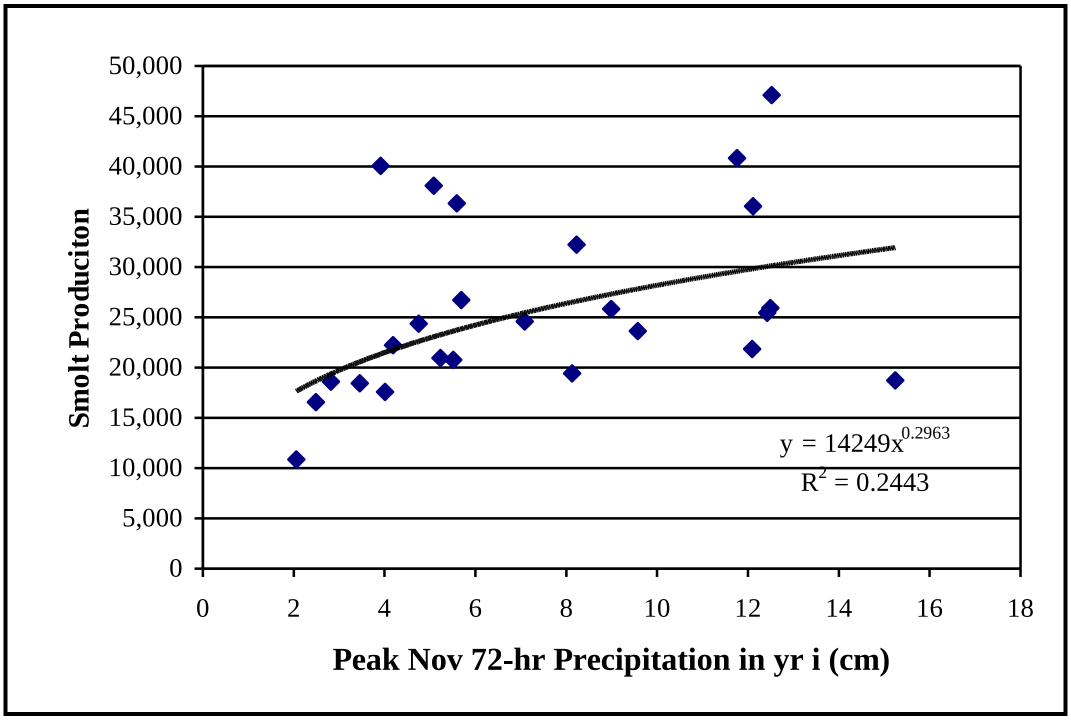

Figure 22. Big Beef Creek coho production as a function of peak November 72-hr precipitation during the parent spawner migration.

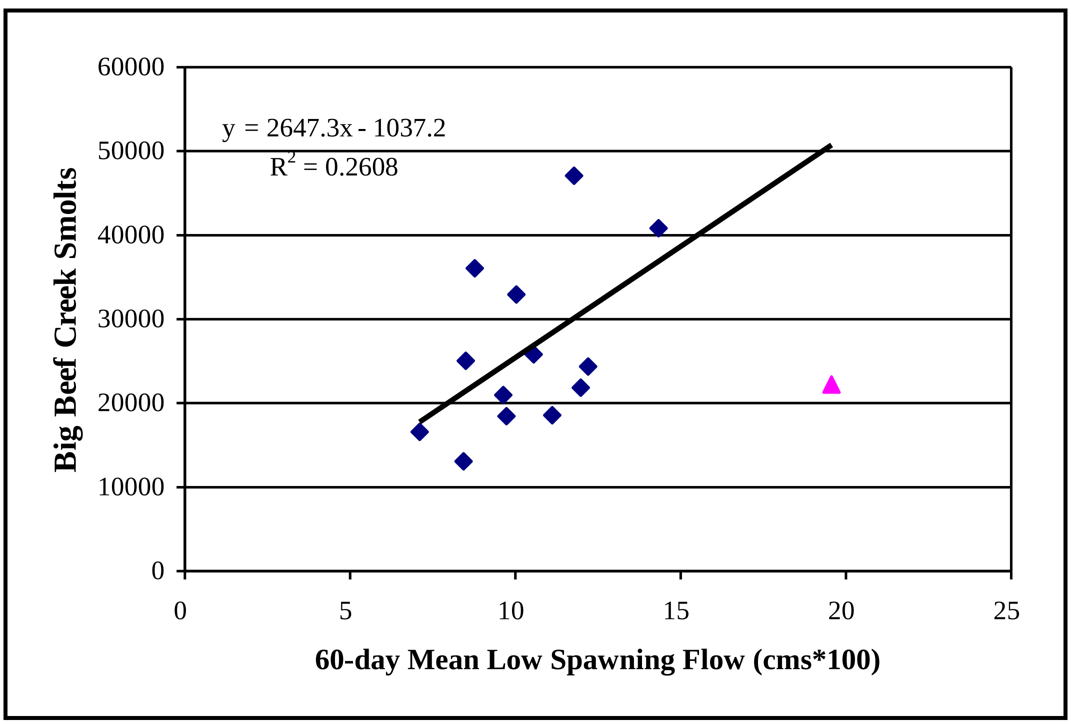

Figure 23. Big Beef Creek coho smolt production as a function of the lowest 60-day mean flow during the period of freshwater rearing for each cohort.

Coho smolt production also is affected by on the size of the parent brood escapement and number of eggs deposited in the gravel, particularly when escapements (egg deposition) are low (Figure 24). Line "A" is fitted to data from years where the peak three-day November rainfall totals and minimum 60-day mean summer flows were above their respective median values (diamonds). Line "C" is fitted to data from years when both were below their respective median values (triangles) and "B" is fitted to data from years when one was above and the other was below the medians (circles). These relationships indicate that survival rates to smolting at any given level of egg deposition are heavily influenced by flow.

Lake Symington may present a challenge to the salmonids of Big Beef Creek not present in other watersheds of this complex. Smolts produced above Lake Symington must pass through the lake to reach saltwater. Largemouth bass predation was estimated to have caused a loss of between 4 and 8% of the total coho smolt production from the watershed (Bonar et al. 2004). These rates likely vary with the annual variations in the abundance of juvenile salmonids and abundance of piscivorous fishes in the lake. However, the available data suggest that a substantial predation impact is occurring in the reservoir.

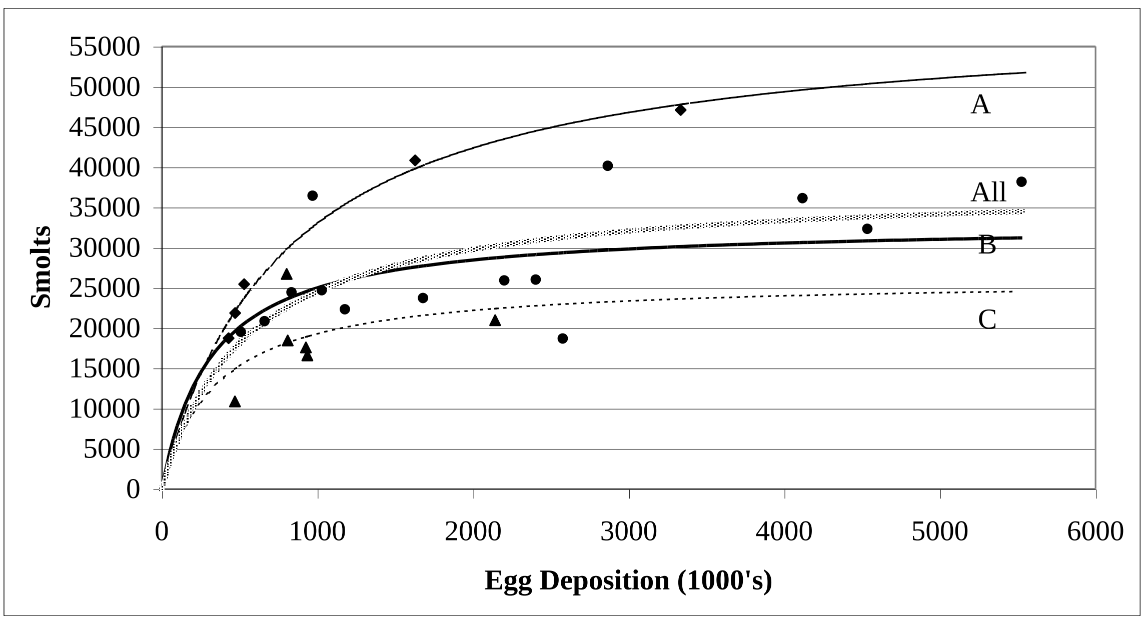

Figure 24. Beverton-Holt coho production functions expressing changes in capacity with changing November and summer flow conditions in Big Beef Creek

Lake Symington also affects summer stream temperatures below the lake (Figure 25). Maximum daily stream temperatures just below Lake Symington exceeded 16^{o}C continuously from May to September 2001 (Seiler et al. 2002). Temperatures exceeded the lethal limit for coho and chum salmon of 25.5^{o}C for brief periods and were well above the preferred range of approximately 12.5^{o} to 14.5^{o} C throughout the summer. Measurement stations above the lake and near the mouth rarely exceeded 16^{o}C, and then only for brief periods. Temperature impacts from Lake Symington continue downstream for an unknown distance. It is very likely that these elevated temperatures negatively affect summer rearing for coho, steelhead and cutthroat. Temperature effects of the lake likely also limit the distribution of adult summer chum in the creek, which migrate and spawn beginning in September.

Below the lake Big Beef Creek flows through an incised gorge whose walls are comprised of a mixture of glacially deposited sediments some of which are very erodible. A number of small tributaries enter Big Beef Creek in this section. Erosion of the valley walls by Big Beef Creek and its tributaries has contributed a tremendous amount of coarse and fine sediments to Big Beef Creek. Land-use activities have intensified sediment contribution rates. The most striking example is along Kid Haven Road. The road was constructed along a small tributary of Big Beef Creek. Material from the road cut was pushed into the stream channel causing the stream to erode the toe of a steep bank on the side opposite the road. Although precise measurements of the sediment generated by this channel realignment were not made, it appears that thousands of cubic meters of material have entered Big Beef Creek as a result. Similar problems exist on other tributaries in this area. As a result, Big Beef Creek moves a large amount of sediment each year and the bed is relatively unstable below the canyon reach, which may influence egg-to-fry survival in this section of the stream. The sediment has also filled pools and reduced rearing habitat, resulting in simplified plane-bed channel morphology in the upper half of the lower canyon section.

Figure 2. Big Beef Creek: Comparison of Daily Maximum Temperatures to WQ Standard, August 2000 to October 2001

Figure 25. Big Beef Creek stream temperatures measured at the inlet and outlet to Lake Symington, and near the smolt weir.

Many large cedar logs were removed from the stream in the early 1980s, primarily to produce shakes and shingles. Following their removal, much of the remaining, smaller wood was flushed from the system within a few years. The loss of LWD has contributed to the degradation of habitat in the lower canyon and unconfined valley reach. These large logs were responsible for stabilizing accumulations of wood in the channel, providing pool habitat and cover, and retaining spawning gravel. Loss of wood may partially explain the reduction in coho smolt production observed from 1987 to 1996. More recently, habitat complexity has been increasing in lower Big Beef Creek with the formation of many log jams below Kid Haven Road. Currently, over 30 jams, composed mainly of alder, have been counted downstream of this point. These are trapping sediment and creating pools and may be contributing to the increase in coho production observed since 1997. The recruitment of wood into Big Beef Creek may be in response to increased bank cutting caused by the input of sediment in the lower canyon. The longevity of these jams may be short given the rapid decay rate for alder. Fewer jams exist between the Kid Haven Road crossing and Lake Symington.

Seabeck Creek

Seabeck Creek is a 14-km^2 watershed located west of Big Beef Creek. The fish-bearing portion of the mainstem is approximately 6.2-km long with the lower 3 km flowing through an unconfined or moderately confined valley (Figure 26). In the upper 3 km, the channel is more confined and is incised within the steep surrounding hills. Seabeck Creek has two right-bank fish bearing tributaries (WDFW unpublished data). The smaller of these, Trib 1, enters Seabeck Creek approximately 150-m upstream of the mouth and the larger, Trib 5, enters the creek approximately 1,600-m upstream of the mouth.

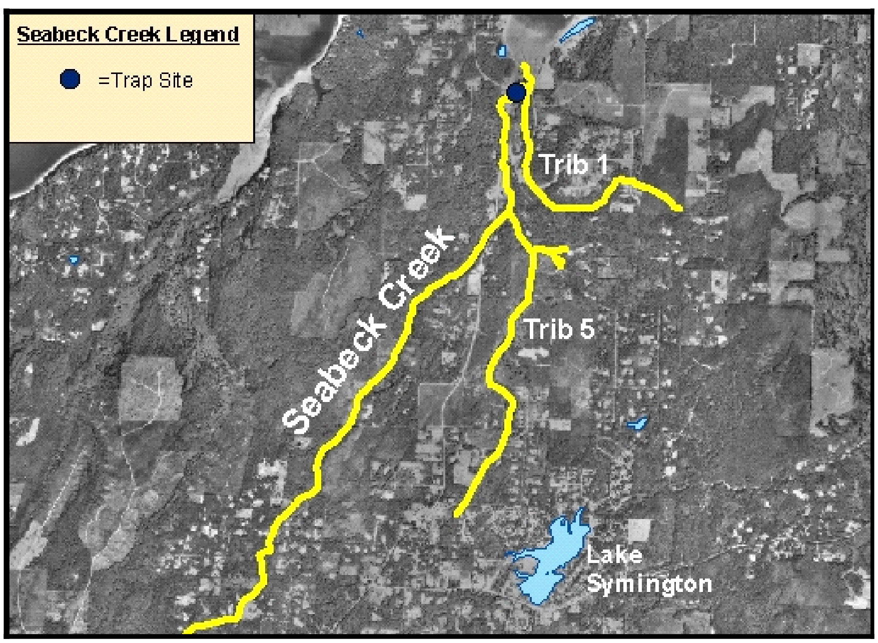

Figure 26. Anadromous fish reaches within the Seabeck Creek watershed

Of the four Hood Canal IMWs, Seabeck Creek has the second lowest smolt production. Since trapping began in 1993, production has averaged approximately 1,400 coho, 300 cutthroat, and fewer than 30 steelhead smolts per year (Figure 27). No trends in production are evident for any species over this eleven-year period. Hypothesized constraints to coho production are listed in Table 17 and discussed below.

Table 17. Factors that are likely limiting coho smolt production in Seabeck Creek.

Factors limiting smolt production
Extremely low summer flows combined with sediment deposition cause much of the accessible habitat to become dry, greatly reducing available summer rearing habitat and fragments much of what remains
Bed erosion in Trib 5 has disconnected the stream and floodplain and degraded habitat downstream in the mainstem.
LWD is scarce in the lower mainstem and Trib 5 reducing rearing habitat

Spawning ground surveys have indicated that approximately 9.6 kilometers of stream habitat are accessible to adult coho salmon. However, about a half of the area accessible to adult salmon exhibits discontinuous flow or is dry during summer. Only 2.4 kilometers at the mouth of the watershed and another 2.3 kilometers upstream , but separated from the wetted reach at the mouth by a long, dewatered reach, flow continuously through the year. The dry reach previously flowed year around (Neuhauser pers. comm.).

Dry and discontinuous portions of the channel occur at low-gradient reaches downstream of obvious sediment sources, suggesting that coarse sediment deposition may cause sub-surface flow. One such reach occurs upstream of a culvert under the Seabeck-Holly Road where the channel gradient decreases downstream from two eroding banks. Downstream of this point, the stream gradient increases and surface flow returns. The culvert appears to be a major contributor to the condition at this site as large amounts of coarse sediment have accumulated above the road.

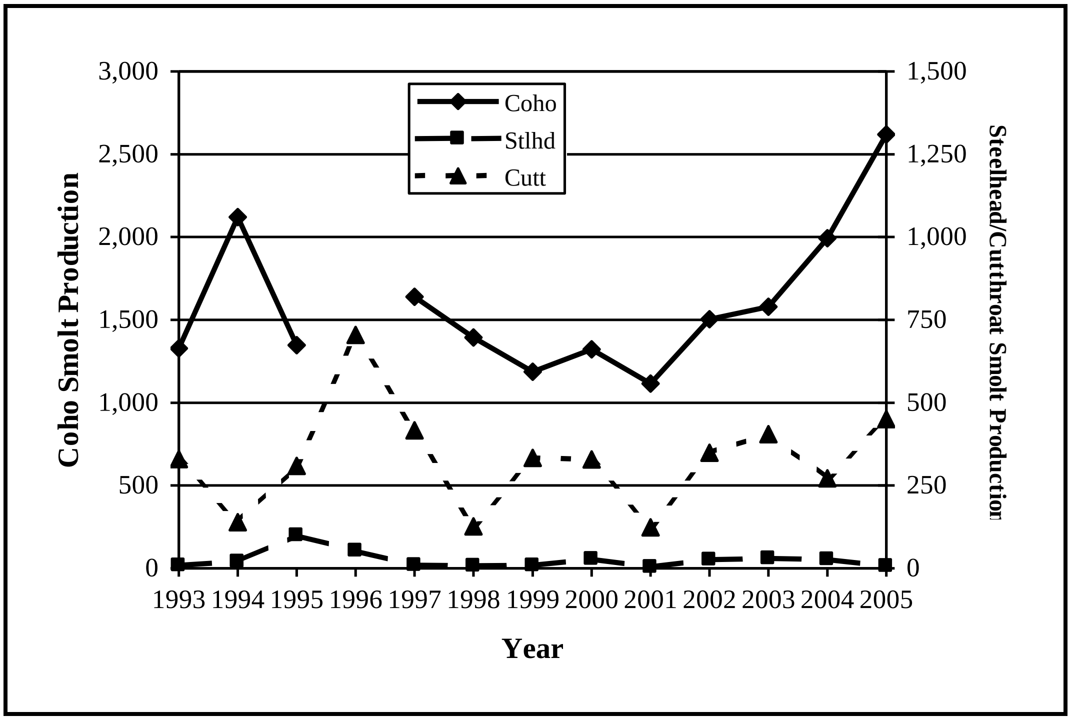

Figure 27. Wild coho, steelhead, and cutthroat smolt production from Seabeck Creek.

The loss of summer rearing habitat resulting from this loss of surface flow may be the principal factor influencing coho, cutthroat, and steelhead production in Seabeck Creek. Furthermore, the fish trapped in isolated pools in areas of discontinuous surface flow are likely much more susceptible to predation. Factors causing the extreme low flow conditions observed in Seabeck Creek and Big Beef Creek will be a principal area of investigation in the near future. The results of these investigations will determine whether or not restoration actions to correct this problem are feasible.

Trib 5 exhibits severe bed erosion, with an average of over 4 m^2 of eroded bank per meter of stream in one 100 m stretch. The stream has incised 2 m or more along this section and banks are eroding in response to the change in bed elevation. Bed and bank erosion continues downstream from this point for approximately 1.7 kilometers, becoming less severe farther downstream. As a result, the channel is highly entrenched and disconnected from its floodplain. An undersized culvert on a forest road may have contributed to the erosion at this site.

Coarse sediment from the bed erosion in Trib 5 has deposited in the mainstem of Seabeck Creek, contributing to the dewatering problem described above. The deposition is especially evident above the Misery Point Road. The bed is currently approximately 0.6 meters below the bottom of the road bridge. Anecdotal reports have indicated that the bed used to be much lower historically (Neuhauser pers. comm.). These reports are supported by the presence of many live cedars along this reach with the base of their trunks buried in sediments. The bridge abutments may constrict flow, encouraging deposition upstream of the bridge. As a result of the sediment generated by bank erosion in Trib 5, areas in the lower mainstem and lower Trib 5 may be more susceptible to scour and sediment deposition than in other portions of the watershed.

Habitat surveys have found functioning LWD to be at very low levels in lower Seabeck Creek and in Trib 5. Consequently, simplified pool-riffle and plane-bed channel morphologies exists in the lower mainstem and Trib 5, respectively; and provide less habitat than would more LWD-rich, complex channel forms. The lack of wood also contributes to bed instability which exacerbates the potential for redd scour and burial, described previously.

Stavis Creek

Stavis Creek is a 15-km^2 watershed adjoining the Seabeck Creek watershed on the west (Figure 28). During summer, fish occupy nearly 8 km on the mainstem, 2 km of South Fork Stavis Creek, and 0.4 km on an unnamed left bank tributary to the mainstem (WDFW unpublished data).

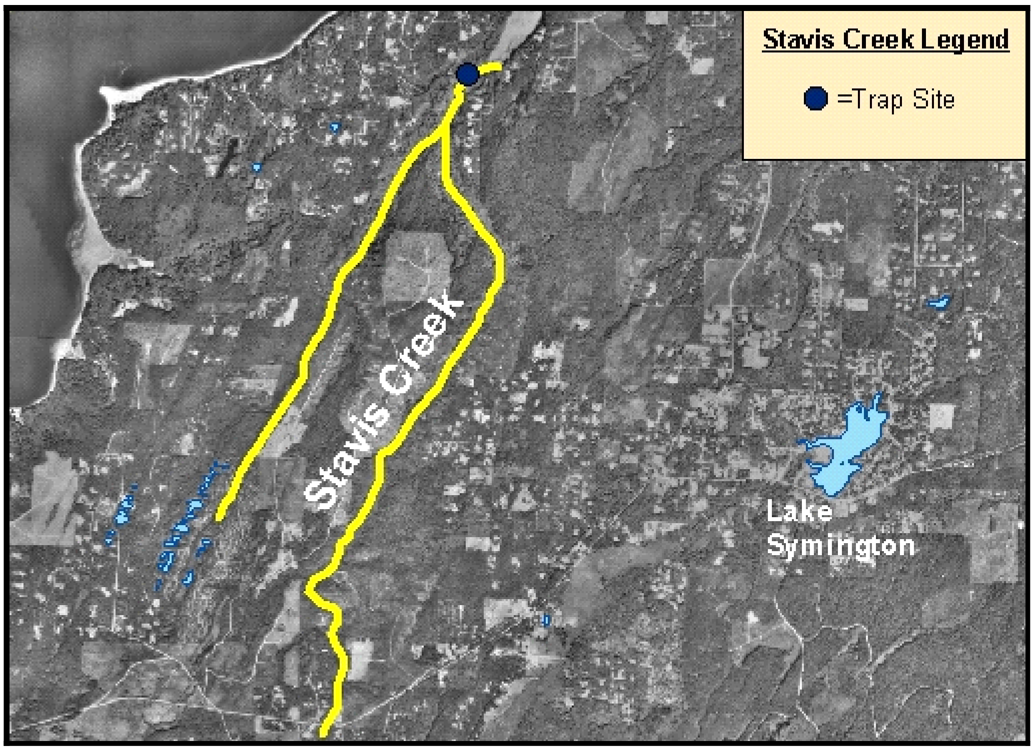

Figure 28. Anadromous fish streams within the Stavis Watershed.

Stavis Creek was selected as the reference watershed for the Hood Canal complex. Of the four watersheds Stavis Creek is the least developed and will likely be subjected to less development pressure during the study than the other three basins. Most of the land within the watershed basin is managed for timber production by the private land owners or by DNR Lands Division. Some rural residential development has occurred along the ridge south of SF Stavis Creek.

Stavis Creek is the second most productive of the Hood Canal IMWs. Production averages approximatlely 6,000 coho, 1,400 cutthroat, and 70 steelhead smolts (Figure 29). As with Big Beef Creek, coho and cutthroat smolt production in Stavis Creek have been increasing since the late 1990s.

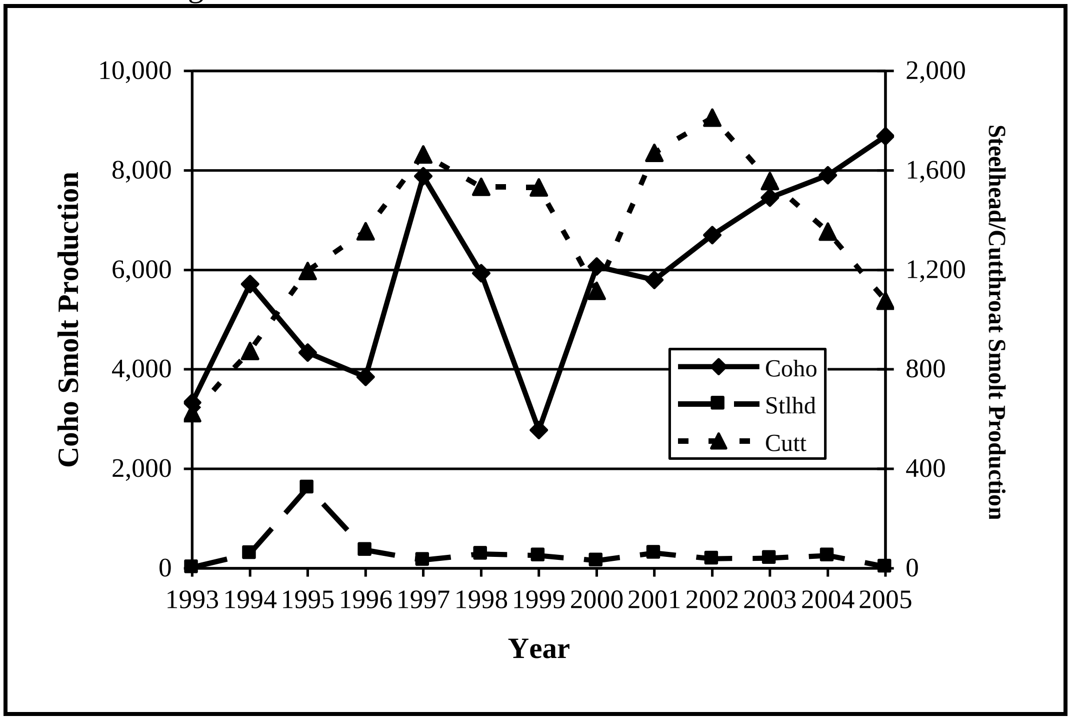

Figure 29. Stavis Creek wild coho, steelhead, and cutthroat production.

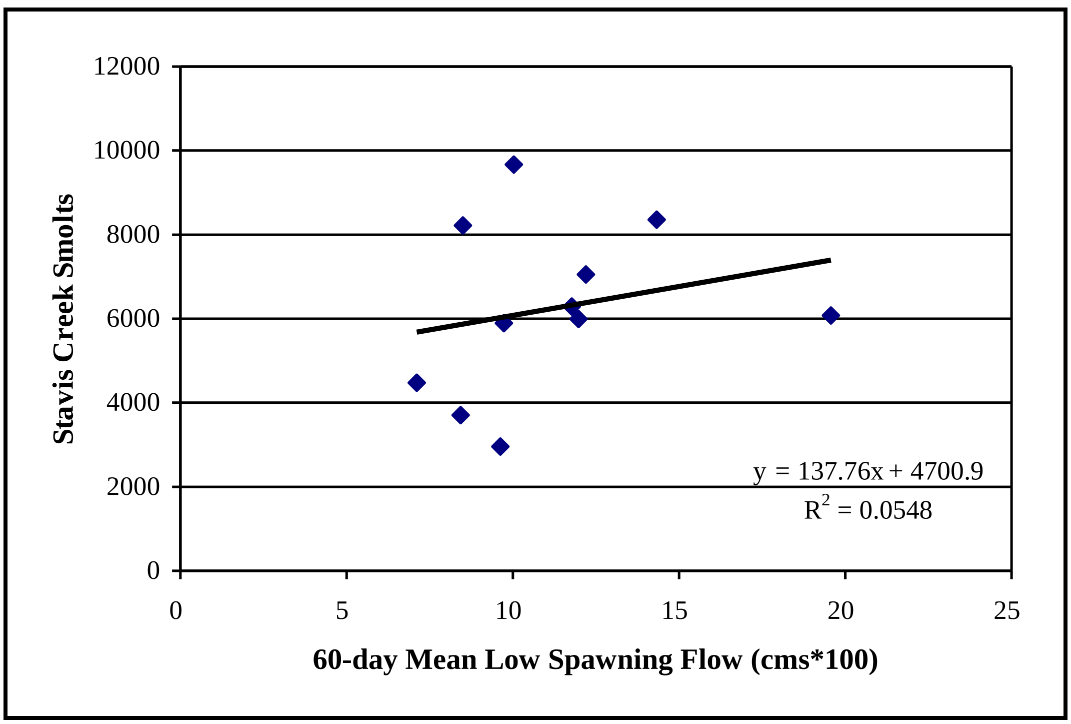

Figure 30. Stavis Creek coho smolt production as a function of the lowest 60-day mean flow from Big Beef Creek.

Unlike Big Beef Creek, summer streamflow does not correlate well with the number of coho smolts produced (Figure 30). Periodic input of large amounts of sediment also may impact smolt production in this system. A large, deep-seated, slope failure, located approximately 600-m upstream of the confluence of SF Stavis Creek, occurred during the winter of 1999 on a steep slope that had been logged about 10-15 years earlier (Neuhauser pers. comm.). The erosion scar

from this slide was estimated at 550 m^2 (WDFW unpublished data). A large amount of fine and coarse sediment was delivered to the channel by this slide impacting habitat down to the mouth of the stream. Coho smolt production during the spring of 1999 was much reduced, possibly due to impacts from the slide on fish that were over-wintering in the lower watershed below the sediment source (Figure 29). Although the greatest impacts to habitat occurred in the first two years following the slide, sediments continue to be transported downstream and may still be affecting smolt production.

Restoration Objectives and Implementation

All Hood Canal complex streams lack LWD and are impacted by current or past sediment inputs (Table 18). These systems also exhibit strong relationships between flow characteristics and smolt production. The impact of low summer flows on rearing habitat and high fall flows on spawner access to upstream habitat are likely the mechanisms responsible for these relationships. Big Beef Creek has the added issue of high water temperature and predation related to Lake Symington.

Table 18. Primary constraints on production are listed by IMW basin.

Constraint	L Anderson	Big Beef	Seabeck	Stavis
Low summer flow	X	X	X	X
Fall spawner flows		X		X
Predation by exotics		X		
High water temp		X		
Sediment input	X	X	X	X
Lack of LWD	X	X	X	X

The restoration objectives for Big Beef, Seabeck, and Little Anderson creeks will attempt to address the deficiencies that have been identified within these watersheds. Restoration actions implemented as part of this study will closely follow the recommendations of the Hood Canal/Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan (HCCC in prep). The HCCC plan is focused on the recovery of summer chum salmon. Although the plan focuses on a single species, the active restoration actions suggested in the plan are likely to benefit multiple species and to address some of the problems identified above. This plan is not yet complete. Restoration actions implemented by the IMW Study prior to the completion of the plan are described below and are based on the information presented above and discussions among WDFW, WDOE, and HCCC staff.

Restoration actions will be implemented sequentially across the Hood Canal complex watersheds, starting with Little Anderson Creek. The physical habitat problems in this watershed are relatively straightforward and there are willing landowners to partner on the restoration projects, including Kitsap County Parks Dept, which owns the lower 0.7 km of the stream.

Large wood is largely absent in the lower Little Anderson Creek watershed. An extensive LWD placement project is being planned for 1600m of Little Anderson Creek above Anderson Hill Road by the HCCC. No action is proposed for the lower 0.7 km located on county property. The channel in this lower reach is actively moving across the valley floor in response to recent beaver dam construction. We are monitoring habitat changes caused by the beaver activity and do not plan to conduct projects in this reach until changes caused by these natural processes are complete.

A suitable control reach for the LWD placement project on Little Anderson Creek is not available. Therefore, we have designated a reach with comparable physical characteristics on Stavis Creek as the project reference for this BACI-design evaluation. The reference reach was selected based on comparability in watershed area, aspect, and reach gradient to the project reach on Little Anderson Creek (Figures 31-32). Pre-project monitoring of large wood placement was initiated in 2004 and continued in 2005. Two of the EMAP habitat sampling sites described earlier fell within project reach on Little Anderson Creek and one within the control reach on Stavis Creek. These sites, spanning 900 meters cumulatively, were sampled in 2004 and 2005 and will be the basis for the project monitoring. The EMAP data have been supplemented with measurements of all channel units throughout the treatment reach. Pools, riffles and glides were classified and measured to obtain channel unit area (Thurow 1994). Additionally, extensive LWD surveys also were conducted using Trimble GPS units.

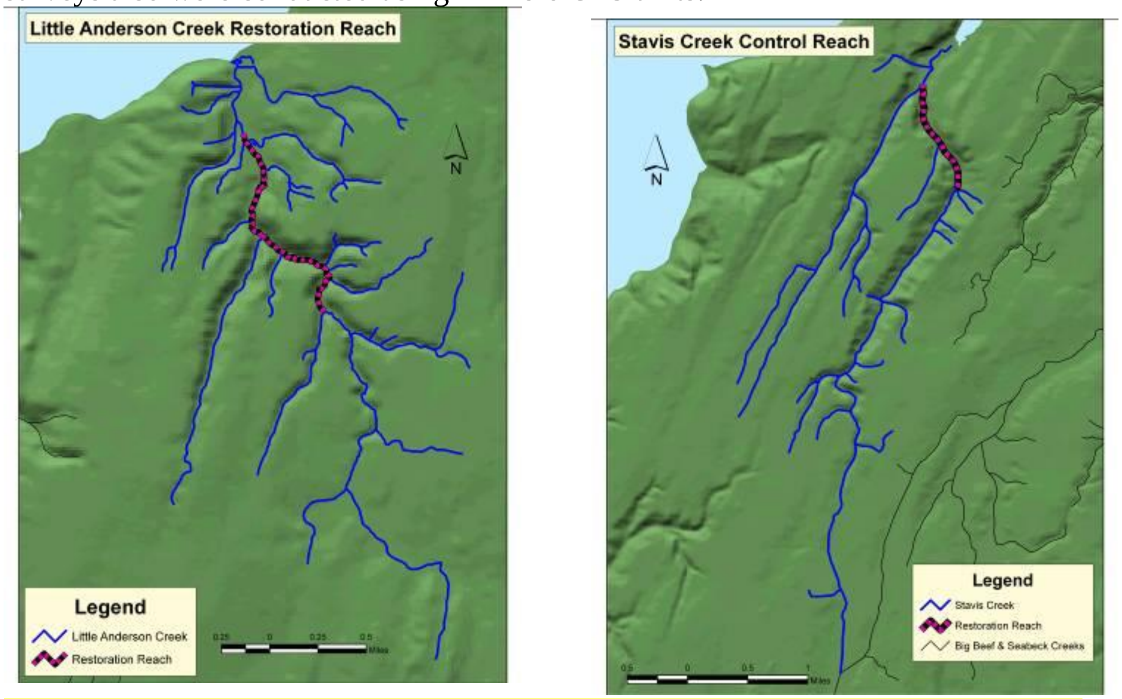

Figure 31. Restoration and control reaches for Little Anderson Creek LWD project.

Biological data at the treatment and reference reaches for this project also have been collected. Fish are being sampled by electrofishing at each of the sampled EMAP sites to provide an index of abundance of fish by species and age class. This sampling will be repeated in each year post-treatment. In addition, we snorkeled all pools and glides, and one sixth of the riffles to obtain salmonid abundance, standardized by channel unit area, throughout the treatment reach (Hankin and Reeves 1988). Sampling will continue for at least two years post-treatment.

We used juvenile Coho density estimates to calculate the likely effect of this project on Coho density and summer juvenile population size. Pre-treatment density estimates were collected by snorkeling 84 randomly selected mesohabitat units in 1.44 km of degraded habitat in Little Anderson Creek in 2005. Approximately 660 m (2033 m^2) of stream was sampled and 851 Coho were counted. We estimated the effect of LWD placement by multiplying the observed density by the multiplier (1.81) developed by Roni and Quinn (2001) to predict the effect of LWD placement. We then multiplied the new density estimate by the area of stream we can treat (1260 m^2). We estimate that LWD placement will increase Coho density in the treatment section from 0.42 to 0.76 Coho/m^2 and increase the number of Coho in the treatment section from 529 to 958. It is as yet unclear whether and how the treatment will affect parr to smolt survival, however if we assume a a survival of 10-20%, as estimated in the Strait of Juan de Fuca complex, this translates into 43-86 additional smolts. This is a 16-32% increase in mean annual production, a change detectable with the existing monitoring program.

Research Objectives and Design

The analyses of the Hood Canal smolt data indicate that flow at certain times of the year is a good predictor of smolt production. Spawning migration flows (Q_{SP}) and summer low flows (Q_{SL}) are both correlated with coho smolt production in Big Beef Creek (the only stream with historic flow data). Additionally, the importance of specific seasonal flows likely varies among years and watersheds.

Given the apparent importance of flow on smolt production, a project is underway in the Hood Canal complex watersheds to better understand the mechanisms by which flow influences fish habitat and smolt production. This information will ultimately be used to identify the most effective locations for habitat restoration and preservation. The ability to accurately forecast smolt production from escapement, seasonal flow, and habitat conditions will provide strong evidence of the functional controls on salmon production. Furthermore, a quantitative understanding of the effect of various factors on smolt production will improve our ability to detect changes in production related to restoration treatments using the BACI design.

Specific hypotheses being addressed are:

1. The geographic distribution of spawners is correlated with maximum November flow, with escapement, and with smolt production.

Current analyses suggest that November high flow (spawning flows-Q_{sp}) affect the geographic distribution of spawning salmon and that greater distribution results in higher production. However, the relationship between geographic distribution and Q_{sp} has not been quantified and alternative mechanisms are possible. For example, Q_{sp} may simply allow access to a few locations of high production (source areas) rather than increasing the overall area accessible to spawners. High Q_{sp} may also reduce redd competition or predation. Quantifying the relationship between Q_{sp} and distribution strengthens mechanistic inferences and promotes better restoration selection and better detection of restoration effects.

Approximately 35 water level loggers were installed at strategically selected (to allow calculation of cumulative sub-watershed flows) EMAP habitat monitoring sites in the Little Anderson, Stavis and Seabeck watersheds. Water level data are recorded every 15 or 30 minutes and collected after winter high flows. The streams are walked at weekly intervals during November and December to record the number and location of spawners, carcasses, and redds. Local water level data and transect-based habitat data (i.e., EMAP-based sample data) are used to estimate accessibility (thalweg depth) across a range of Q_{sp}. Local water level data can be correlated with watershed flow records to estimate probability of access to specific locations at a given flow.

Analysis: Simple correlative analyses are being used to assess the relationship between flow statistics and spawner and redd distribution. Analysis of covariance is used to assess the relationship between flow statistics and geographic distribution while accounting for the possible effects of escapement on geographic distribution. Spatially attributed thalweg and habitat data are correlated with local stage height and watershed flow to assess the effect of local depth on the geographic distribution of spawning salmon.

The effect of Q_{sp} on spawner distribution and subsequent production will be assessed by testing for a significant difference in the length of stream occupied at different Q_{sp} after accounting for differences in escapement using ANCOVA and comparing the prediction accuracy and information content of alternative models that predict spawner distribution and subsequent production using Q_{sp} and escapement as predictors. Hypotheses and models include:

1) Spawner distribution will be positively correlated with Q_{sp}.

2) Spawner distribution will be positively correlated with smolt production.

3) Spawner distribution will be positively correlated with smolt production after accounting for the effect of escapement.

4) Q_{sp} at individual water level loggers will be predict thalweg depth, and reach and sub-watershed spawner distributions.

2. *Summer low flows are correlated with habitat quantity and quality and with parr-smolt survival?*

Summer low flows (Q_{sl}) likely affect the quantity, quality and availability of summer habitat, thereby affecting survival and subsequent smolt production. To evaluate this hypothesis, water level from the water level loggers at 35 EMAP sites in each watershed is being recorded across a range of flows from May through September. The distribution of summer habitat and juvenile salmon are recorded from extensive surveys that map the location of water in a GIS in July through September. The stage height and the habitat data collected for EMAP sites and extensive surveys will be used to estimate available habitat as a function of stream flows (stage height). Juvenile abundance and survival are estimated at the EMAP study reaches as described earlier.

Regression analysis will be used to assess the relationship between water level, habitat quantity, life stage survival, and production. Analysis of the relationship between water level and egg-to-parr or parr-to-smolt survival will require a number of years of data. Hypotheses include:

1) Q_{sl} and habitat quantity are positively correlated.

2) Q_{sl} is correlated with the length of wetted stream.

3) Length of wetted stream is positively correlated with egg-to-parr and parr-to-smolt survival

Lower Columbia Complex

The Lower Columbia complex has a shorter record of smolt production than the other two complexes, virtually no quantitative juvenile abundance data, very little coho escapement data, and little quantitative habitat information. As a result, we are concentrating on the collection of these basic data at this complex to build a pre-treatment data record. A restoration plan for both treatment basins based on the recently approved Lower Columbia Salmon Recovery and Fish and Wildlife Subbasin Plan (LCFRB) 2004 will be developed over the next two years with implementation beginning in 2009.

Description

The Lower Columbia Complex is comprised of Mill, Abernathy, and Germany Creeks, located within the Elochoman subbasin (WRIA 25), in Cowlitz and Wahkiakum Counties, Washington. The watersheds in this complex are larger than those in the other complexes (Table 19). Smolt traps in each creek are located within a kilometer of the stream mouths (Figure 32). Watershed areas above the smolt trap are similar ranging from 5,800 to 7,600 hectares. Abernathy and Germany Creeks drain steep basins with headwater elevations of up to 806 m. Mill Creek is a lower elevation basin with headwater elevations of 555 m.

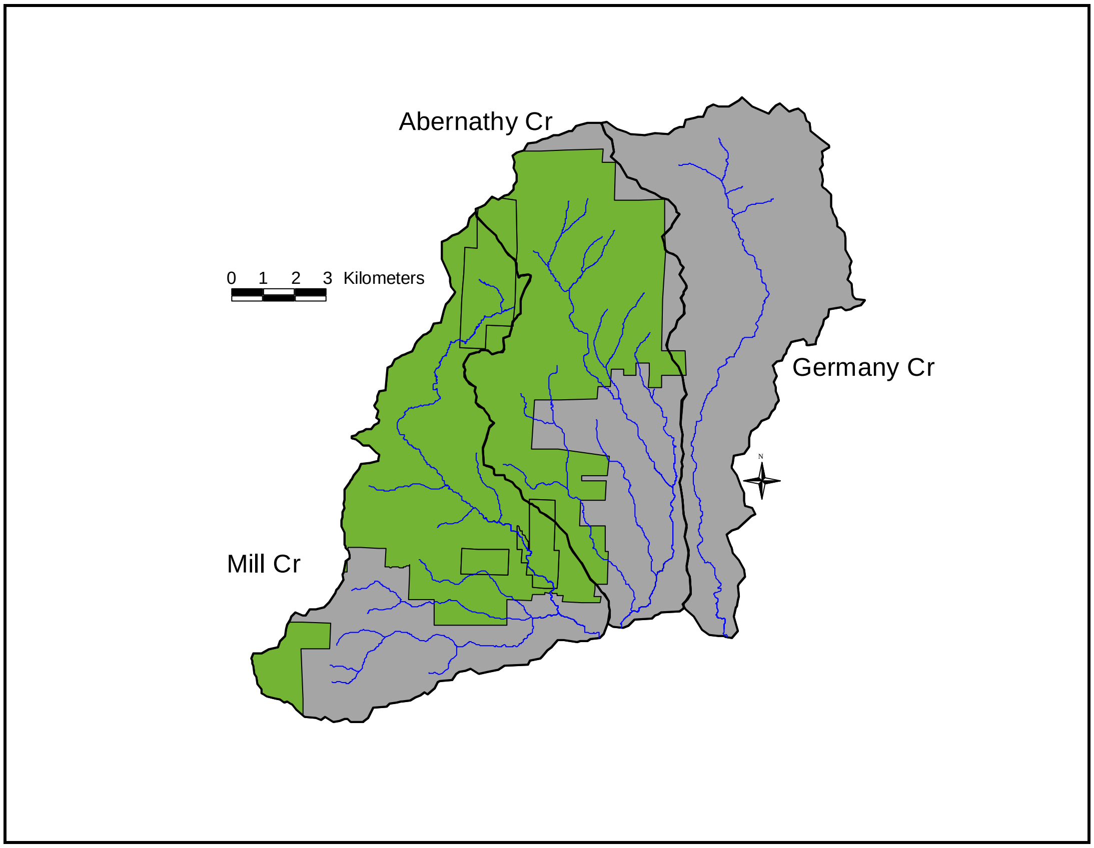

Figure 32. Lower Columbia Complex. Land managed by the Washington Department of Natural Resources is shaded green. Nearly all of the remaining land is privately owned timberland.

Volcanic geology comprises over 95% of both Mill and Abernathy Creeks. Approximately 72% of Germany Creek is comprised of volcanics with 26% in sedimentary rock. The Germany-Olympic soil association dominates (>66%) all basins with 22 and 34% in Bear Prairie-Loper association in Abernathy and Germany Creek, respectively. Soil erodibility was characterized as high for 12, 29, and 23% of Mill, Abernathy, and Germany Creeks, respectively (Wade 2002).

Table 19. Land cover, land management, and ownership percentages for the Lower Columbia Complex basins are shown below. Land cover is based on satellite imagery from the early 1990s. HCP area is based on 2001 maps provided by U.S. Fish and Wildlife Service.

Public ownership was based on the Major Public Lands map, remaining land was assumed to be private.

Smolt trap	Basin area (km^2)	Land cover (%)		Ownership (%)	
		Forested	Developed	Public	Private
Mill Cr	76	94	0	68	32
Abernathy Cr	73	92	0	62	38
Germany Cr	58	85	0	0	100

Road density estimates ranged from 4.2 to 5.8 miles/mi^2 (2.6-3.6 km/km^2) with 11% of road length occurring within 200 feet of a stream. Precipitation within the complex is estimated to range from 140 to 250 cm/year. Nearly all the complex is in a rain-dominated zone. Only Germany Creek, with 27%, has a substantial proportion of area within a rain-on-snow zone.

Much of the complex is forested (Table 19). Most forest land in Germany Creek is privately owned, while Washington Department of Natural Resources (DNR) manages a large share of the Mill Creek and Abernathy Creek watersheds. Some residential development exists in the lower portions of these watersheds, interspersed with agricultural land use in the lower end of Abernathy Creek and Germany Creek.

Chum escapement is currently not monitored in these basins. Chinook, coho, and steelhead escapements are monitored by foot surveys for spawners and redds on the entire stream length accessible to anadromous fishes at 10-14 day intervals throughout the spawning season. Smolt monitoring has been conducted in the Lower Columbia Complex since 2001 (Table 20). Average coho smolt production per square kilometer watershed area in the three Lower Columbia complex streams ranged from 89 in Abernathy Creek to 130 in Germany Creek (Table 21). These levels are substantially lower than those found in the Hood Canal complex. For example, Stavis Creek produced 489 coho smolts/km^2 over the same two years. The low level of coho production in the Lower Columbia Complex may relate to the higher stream gradients of these systems, poor habitat condition, and possibly to low coho escapements, which have only been measured since 2005. Wild steelhead smolt production per square kilometer of watershed averaged 20 in Mill Creek, 108 in Abernathy Creek, and 130 in Germany Creek. The high channel gradient of the Lower Columbia complex streams likely provides habitat conditions more favorable for steelhead than coho salmon.

Because of the short monitoring record, few hypotheses regarding factors constraining production can be drawn directly from smolt production data. There is some qualitative habitat data that has been collected. The following data sets have been assembled and reviewed for the Lower Columbia Complex and

- smolt production estimates since 2001 for Mill, Abernathy, and Germany Creeks,
- Lower Columbia Salmon Recovery and Fish and Wildlife Subbasin Plan (LCFRB 2004)
- Salmon and Steelhead Habitat Limiting Factors: Water Resource Inventory Area 25 (Wade 2002),

- Habitat data collected by the Cowlitz Conservation District in 2000
- Ecosystem Diagnosis and Treatment (EDT): Level II Environmental Attributes: Habitat Surveys (WDFW),

Much of this information have been synthesized in the Lower Columbia Salmon Recovery and Fish and Wildlife Subbasin Plan (LCFRB 2004).

Table 20. Period of record and data collected at each smolt trap and on spawner surveys.

Lower Columbia Complex			
Smolt trap	*Juvenile outmigrants*		*Adults*
	Since	*Species*	*Species*
Mill Cr	2001	chinook, coho, steelhead	chinook, steelhead
Germany Cr	2001	chinook, coho, , steelhead	chinook, steelhead
Abernathy Cr	2001	chinook, coho, , steelhead	chinook, steelhead coho

The available information indicates that habitat impacts related to past clearing of riparian vegetation for agriculture or timber harvest, road construction in the floodplains, sediment input from roads and mass wasting, and direct manipulation of the stream channel have all occurred in these watersheds and may be impacting current production. A restoration plan for both Abernathy Creek and Germany Creek will be developed in coordination with the Lead Entity (Lower Columbia Fish Recovery Board) and local restoration groups based on the information currently available. However, we recognize that the data required to accurately identify production constraints within these watersheds is incomplete. The necessary data is now being collected and as it becomes available, it will be used to update the restoration plan.

Table 21. Average wild coho and steelhead smolt production and productivity for the Hood Canal complex (1992-2002) and Lower Columbia complex (2001-2002).

Stream	Average Smolt Production		Average Smolts/km^2	
	Coho	**Steelhead**	**Coho**	**Steelhead**
Lower Columbia Complex				
Mill Creek	7,912	1,480	105	20
Abernathy Creek	6,596	7,995	89	108
Germany Creek	7,579	7,550	130	130
Hood Canal Complex				
Big Beef Creek	23,443	1,528	651	42
L. Anderson Creek	263	43	22	4
Seabeck Creek	1,313	27	99	2
Stavis Creek	5,239	74	400	6
Note: Coho and steelhead production estimates for the Hood Canal complex, shown here, represent average smolt trap catches. The actual average production is slightly higher due to unaccounted for migration occurring prior to and following trap operation. Estimates for the Lower Columbia complex streams represent the average total migrations of coho and steelhead smolts.				

<u>Germany Creek</u>

Low abundance of large woody debris in the lowest reaches of Germany Creek have resulted in a simplified channel structure (habitat diversity) and is assumed to be a contributing factor to the decline of chinook, chum, and coho salmon production in the watershed (LCFRB 2004). Where wood does exist, it functions to create pools, maintain side channels, and sort fine and coarse sediment. This observation has led to the development of a restoration proposal for this area of Germany Creek. Using a 5th-Round Salmon Recovery Funding Board grant, the Columbia Land Trust (CLT) recently acquired land along approximately 1600m of stream habitat in lower Germany Creek for conservation and restoration purposes.

Recently, Washington Trout proposed to construct between four and seven engineered logjams, consisting of approximately 25 – 30 pieces of large wood each in the CLT-acquired reach. Additionally, up to 14 smaller large wood aggregations consisting of one to four pieces are proposed for the channel margins of this reach. The addition of large wood by Washington Trout is expected to force changes in channel morphology within this reach, improving holding, spawning, and rearing conditions for chinook, chum, and coho salmon. Construction of the engineered logjams and aggregate wood placement could begin in summer of 2007.

Additional restoration actions are under development by Washington Trout in Germany Creek. Washington Trout is currently collecting groundwater data and channel profiles to determine if a rehabilitated chum salmon spawning channel is a suitable restoration activity. Also, three abandoned floodplain gravel pits are being considered for conversion to off-channel rearing habitat for coho salmon. Both the off-channel rearing ponds and chum-spawning channel are located on the newly acquired Columbia Land Trust property, and are being considered for construction in 2006 or 2007. After active restoration activities are completed, native conifer saplings will be planted throughout the CLT property.

These projects are premature relative to the amount of pre-restoration data that has been collected on the watersheds in this complex. We will attempt to work with the project proponents to postpone implementation of these actions until sufficient data are collected. Regardless of the time at which these projects are implemented, we will be able to assess reach scale responses in physical habitat and fish utilization. Subsequent restoration treatments will not occur in Germany Creek until sufficient pre-treatment data has been collected to provide a reasonable probability of detecting a response by the fish.

To assess the habitat response to logjam and large wood placement, we have initiated project monitoring within the lower reach of Germany Creek. We collected pre-treatment data using EMAP protocols after Peck (2001) on the CLT property in 2005. Complimenting the EMAP data, Washington Trout collected extensive cross section data of the channel throughout the reach. Working cooperatively with Washington Trout, we will repeat those surveyed cross sections in subsequent years. To assess the biological response to large wood restoration actions, we collected juvenile fish abundance data by species and life history stage using snorkeling techniques in all pools and glides and in every sixth riffle (Thurow 1994). Data will be standardized by habitat area to obtain density estimates (Hankin and Reeves 1988).

We selected a suitable control reach in Mill Cr as a companion reach in our project effectiveness BACI design because similar reaches in Germany Creek may receive similar treatment in coming years. Monitoring in the Mill Creek control reach will be made with identical protocols as those conducted in the Germany Cr treatment reach. Data collection will be repeated in 2006 prior to treatment, and in subsequent post-treatment years to monitor physical and biological changes in response to large wood placement over time

CHINOOK SALMON

Chinook salmon require a substantially larger watershed to complete their freshwater rearing than coho, steelhead and cutthroat. The larger area required by this species makes it very difficult to use a treatment-reference comparison at the level of an entire watershed, as we are doing for the other species. However, we are working with the Skagit River System Cooperative and the Northwest Fisheries Science Center to evaluate the effect of estuary restoration on chinook salmon growth and survival.

Chinook salmon are well known for utilizing natal river tidal deltas, non-natal “pocket estuaries” (nearshore lagoons and marshes), and other estuarine habitats for rearing during outmigration (Reimers 1973, Healey 1980, Beamer et al 2003). Several studies have linked population responses to availability of estuary habitat, either by examining return rates of groups of fish given access to different habitat zones (Levings et al. 1989) or by comparing survival rates of fish from populations with varying levels of estuary habitat degradation (Magnusson and Hilborn 2003). These studies support the hypothesis that estuarine habitat is vital for juvenile Chinook salmon. However, these necessarily coarse-scale studies have ignored how large-scale estuarine habitat restoration within a watershed contributes to population characteristics. These issues may be critical to understand how to best restore Chinook salmon populations, as many estuaries within Puget Sound and elsewhere have been converted to agriculture and urbanization land uses. For example, the Duwamish River has lost more than 99% of its tidal delta habitat (Simenstad et al 1982), while the Skagit River, which contains the largest tidal delta in Puget Sound, has lost 80-90% of its aquatic habitat area (Collins et al. 2003).

Our goal is to understand changes in population characteristics (e.g., abundance, productivity, survival, and life history diversity) of wild Chinook salmon in response to reconnection and restoration of estuarine habitat. This issue requires us to examine the effects of restoration at a system-wide scale, i.e., the spatial extent that encompasses the entire population of Chinook salmon rearing in the estuary. As the Skagit "estuary" includes wetlands of the tidal delta as well as nearshore and offshore zones of Skagit Bay, this task is by no means easy.

Our goals require long-term monitoring tied to restoration efforts. We are currently monitoring Skagit River Chinook salmon via a long-term interagency program involving sampling of outmigrants at Mt Vernon (Washington Department of Fish and Wildlife, WDFW), fyke trapping of fish rearing in the tidal delta (Skagit River System Cooperative, SRSC), beach seining of nearshore habitats in Skagit Bay (SRSC), and townetting of offshore areas in Skagit Bay (Northwest Fisheries Science Center, NWFSC). This program provides us a system-wide analysis of patterns of abundance and life history diversity across the juvenile salmon migration season.

The benefits of this diverse effort are multifold. First, this program provides adequate redundancy should one element of the monitoring effort fail due to temporary failure of equipment, loss of personnel, or inclement conditions. Second, this plan systematically captures the sequence of habitat types used by juvenile Chinook salmon during migration through the estuary. Third, much of this effort has been in place for 10+ years, and therefore provides a good time series to establish a baseline for evaluating the large-scale effects of restoration. Finally, this program provides important insights into the ecology of Chinook salmon. The outmigrant trapping has documented an important relationship between freshwater survival and incubation flood magnitude (Seiler et al. 2003). This information, combined with fyke trapping in the delta, has provided strong support for density dependence and a habitat area constraint in the tidal delta (Beamer et al. 2005). Systematic beach seining has revealed relationships between nearshore growth rates and residence in the delta (Beamer and Larsen 2004). In addition, analysis of the seasonal distribution of fish caught during townetting indicates that hatchery and wild fish have very different patterns of nearshore residency (Rice et al. 2001).

Despite the success of these current efforts, our program has several weaknesses. First, our consistent use of index sites to monitor juvenile Chinook salmon has resulted in low resolution for assessing spatial variation of the habitats sampled, and complicates assessments of abundance. Second, no studies to date have effectively measured survival of juvenile Chinook salmon in estuarine habitats; leaving open questions how restoration of estuarine habitats improved population productivity[1]. Third, because the current sampling scheme was developed to build an understanding of the actual juvenile life history types using the Skagit estuary and its possible bottlenecks to productivity, it was not explicitly designed for testing effects of restoration at a system-wide scale. Thus, the current sampling design may be ineffective at detecting population responses to restoration.

To effectively evaluate the population response of Chinook salmon to estuary restoration, we need a systematic monitoring program that can detect population changes linked with restoration project implementation. In order to accomplish this goal, we use several study designs, linked to both index monitoring to assess population trends and random sampling to obtain unbiased estimates of population density. In addition, we also conduct several types of studies which should allow us to estimate survival during rearing in the tidal delta. These efforts, in combination with site-specific efforts to examine effectiveness of several large-scale estuary restoration projects, will allow us to evaluate the role of estuary restoration for the recovering Chinook salmon population in the Skagit River. Lessons learned in the Skagit estuary could benefit recovery efforts in other Puget Sound Chinook salmon bearing rivers. This should be true in places that have the same habitat and life history types as the Skagit, although out of system transferability would need to put in a river specific context.

Background

Monitoring Chinook salmon in the Skagit estuary started from several premises: 1) Chinook salmon are federally threatened in the Pacific Northwest, 2) Chinook salmon require estuary habitat for successful rearing and transition to the marine environment, and 3) estuary habitat loss and degradation in the Skagit system has resulted in reduced capacity for salmon. While the first of these premises was supported by other researchers (Myers et al. 1997) at the time we began our monitoring, the other premises had weak (if any) support. Therefore, for the last 10 years, our monitoring goals have been to examine population characteristics and habitat use of the Skagit estuary by different life history types of Chinook salmon, with the goal of identifying their limiting factors.

Our efforts provided strong support for the second two premises. We have documented that the majority of fish use the tidal delta during rearing for up to eight weeks, and may reside in Skagit Bay for several months (Beamer et al. 2000; Beamer and Larsen 2004).

[1] One study currently underway (Skagit Chinook Life History Study) does estimate marine survival (beginning of nearshore residency to returning adult) of juvenile life history types. This study uses otolith microstructure to identify specific juvenile life history types and relatieve survival estimates will be made for two brood years with very different outmigration sizes: 1994 (2.2 million outmigrants) and 1998 (7.1 million outmigrants). While this study does give us a tool to quantify the benefits of different restoration actions that benefit specific life history types, it doesn't directly measure survival at specific juvenile stages. This study is primarily funded by Seattle City Light and Northwest Indian Fisheries Commission. Principle investigators are Eric Beamer (SRSC) and Kim Larsen (USGS WFRC). This study will be concluded in 2006.

Furthermore, we have found that density of fish in the tidal delta peaks at high outmigrations, that body size declines as a function of tidal delta density, and that the frequency of one life history subtype – fry migrants – increases as a function of the abundance of the population entering the tidal delta (Fig. 33). Furthermore, we have found that the return rate of adult salmon is limited by the abundance of juveniles (Greene et al. 2005). All these findings support the third premise, and provide a strong argument for restoration of habitat in the Skagit estuary. The Skagit River System Cooperative and WDFW produced a recovery plan that emphasizes estuary restoration as the centerpiece for recovery of Chinook salmon in the Skagit River (Skagit River System Cooperative and Washington Department of Fish and Wildlife 2005). This plan features several restoration projects already completed or in preparation (Table 22; Appendix B), as well as some that are currently at conceptual stages. The result will be the first large-scale experiment on the effects of estuary restoration on Chinook salmon populations.

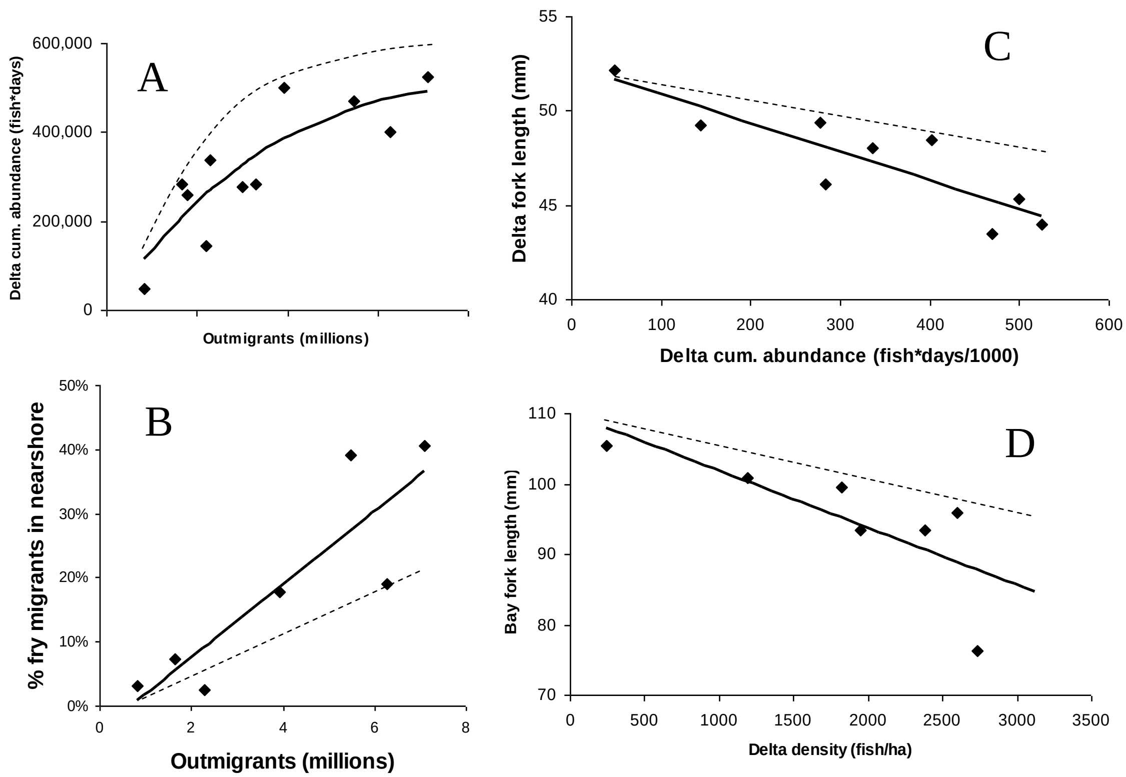

Figure 33. Functional relationships for wild juvenile Chinook salmon from the Skagit River, delta and nearshore (from Beamer et al 2005). Points and solid lines represent the results of a decade of field study while dashed lines illustrate conceptually how these relationships should respond to habitat restoration planned for the Skagit tidal delta. (A) The relationship between freshwater smolt outmigration population size and the density of juvenile Chinook in tidal delta habitat. (B) The relationship between freshwater smolt outmigration population size and the percentage of juvenile Chinook in nearshore habitat that exhibit the fry migrant life history type. (C) The relationship between Chinook salmon density in tidal delta habitat and the size of juvenile Chinook in tidal delta habitat. (D) The relationship between Chinook salmon density in tidal delta habitat and the size of juvenile Chinook in nearshore habitat.

Hypotheses

How will estuary restoration affect Skagit Chinook salmon? If we interpreted the results in Figure 33 strictly and applied it equally to the entire Skagit estuary, we should expect restoration in the Skagit tidal delta to reduce local tidal delta Chinook salmon densities, thereby causing increases in body size and overall population abundance and a decrease in the frequency of fry migrants.

However, because of variation in the accessibility and the current availability of habitat across the estuary, hypotheses should differ in different areas of the estuary. We used a system-scale approach to generate hypotheses about how restoration of tidal delta capacity and connectivity and pocket estuary capacity effect juvenile Chinook abundance, size, and the frequency of life history types (Table 23).

Table 22**.** List of delta restoration projects completed or currently under feasibility/design. See Appendix C for more details.

Site Name	**Sub-delta Polygon affected (Fig. 3)**	**Project type (Area restored to river/tidal hydrology)**	**Year complete**	**First year juvenile Chinook could benefit**
Deepwater Slough	#4	Capacity/Connectivity (221 ac)	2000	2001
Smokehouse Floodplain	#1	Capacity (62 ac)	2005-7	2006-8
Milltown	#4	Capacity (212 ac)	2006/7	2007/8
South Fork Dike Setback	#4	Capacity (40 ac)	2004	2005
Wiley Slough	#4	Capacity/Connectivity (161 ac)	2007	2008
Swinomish Channel Causeway	#1	Connectivity (na)	2008	2009
Fisher Slough	#4	Capacity (68 ac)	2008	2009

We developed sub-delta monitoring hypotheses by thinking how current delta habitat is being utilized by juvenile Chinook salmon (Figure 34) and then by hypothesizing how juvenile Chinook salmon would respond to planned delta restoration (Figure 35). In Figures 34 and 35, the arrow directions depict how juvenile Chinook salmon move through delta habitat and into Skagit Bay. The pathways within the delta are based where delta distributary channels are located or planned to be restored. The pathways for fish moving from delta habitat to Skagit Bay were derived from drift buoy data (see Beamer et al. 2005). Arrow thickness represents the number of juvenile Chinook salmon using each pathway based on the current or restored habitat amount and configuration. Figure 35 shows planned restoration areas in pink. Because of limitations in the migratory pathways that fish can take within delta habitat, we do not expect the entire delta will respond to specific restoration projects in a homogeneous fashion. The sub-delta areas that we do expect to respond similarly are numbered and circled in Figure 35. Monitoring hypotheses are stated for each area in Table 23. All monitoring hypotheses are interpreted as functions to account for varying outmigration population sizes, habitat conditions (e.g.

channels with deep areas with low tide impoundments vs. channels without these features), and environment (e.g., floods, temperature, salinity).

Table 23. Draft monitoring hypotheses for juvenile Chinook salmon abundance in sub-delta polygons shown in Figure 35.

Sub-delta polygon # and name	Potential Restored Area (acres)	Pre-restoration	Post-restoration
#1 Swinomish Channel Corridor	770	Juvenile Chinook density is lowest (along with polygon #5) compared to other sub-delta polygons	Juvenile Chinook density will increase following restoration that improves connectivity with the North Fork Delta Juvenile Chinook salmon population & body size will increase following restoration that increases rearing capacity along the Swinomish Channel Corridor
#2 North Fork Delta	980	Juvenile Chinook density is highest compared to other sub-delta polygons	Juvenile Chinook density will decrease following restoration projects that increase connectivity to other areas within the delta Juvenile Chinook salmon population & body size will increase following restoration that increases rearing capacity within the North Fork Delta
#3 Central Fir Island Delta	470	Juvenile Chinook density is 2nd lowest compared to other sub-delta polygons	Juvenile Chinook density will increase following restoration that increases connectivity to central Fir Island Juvenile Chinook salmon population & body size will increase following restoration that increases rearing capacity within Central Fir Island
#4 South Fork Delta	630	Juvenile Chinook density is intermediate compared to other sub-delta polygons	Juvenile Chinook density will remain intermediate compared to other sub-delta polygons. Juvenile Chinook salmon population & body size will increase following restoration that increases rearing capacity within the South Fork Delta
#5 Stanwood/English Boom Delta Fringe	None Currently Identified	Juvenile Chinook density is lowest (along with polygon #1) compared to other sub-delta polygons	Juvenile Chinook salmon population & body size will increase following restoration that increases source population size originating from Skagit and Stilliguamish Rivers.

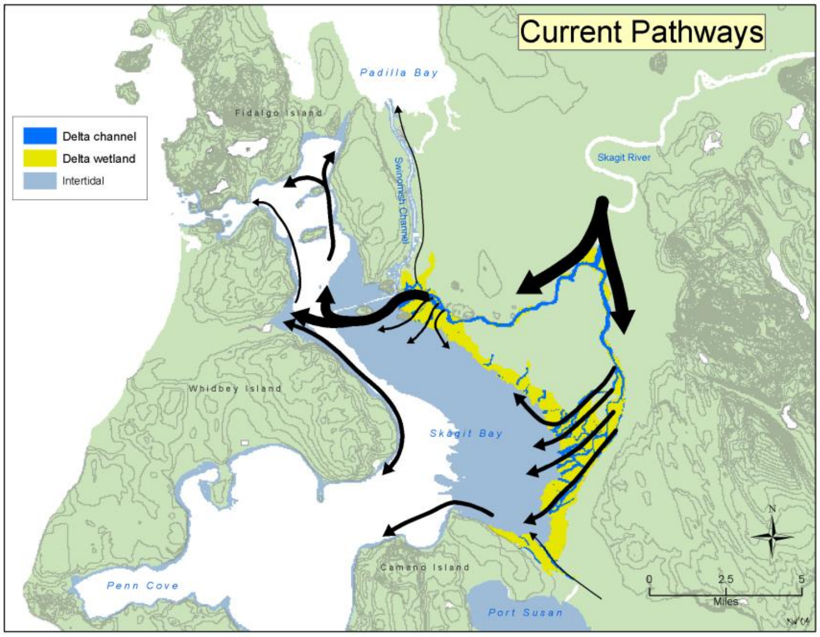

Figure 34. Current juvenile Chinook salmon pathways in the Skagit River estuary. The arrow directions depict how fish move through the tidal delta and into Skagit Bay. Arrow thickness represents the number of Chinook salmon following these pathways, based on current habitat configuration and area.

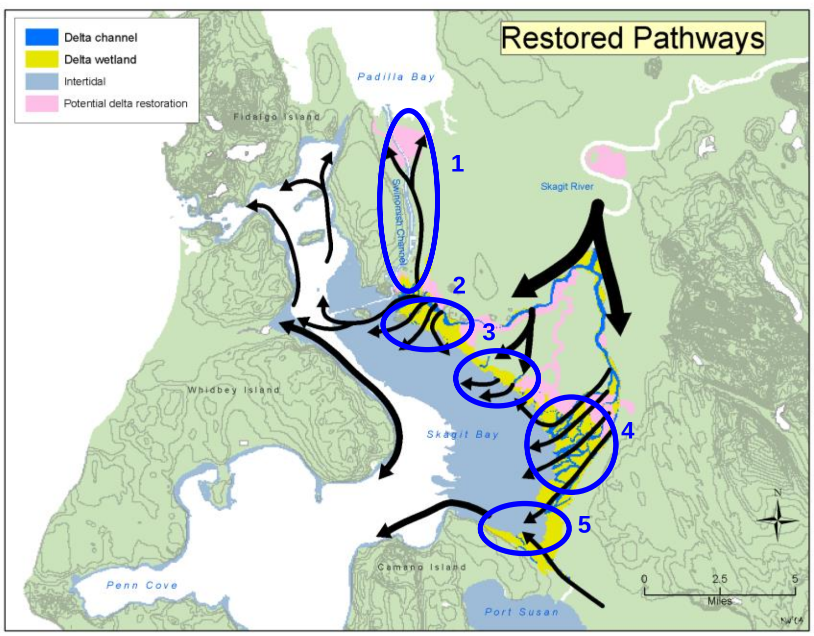

Figure 35. Future juvenile Chinook salmon pathways in the Skagit River estuary after restoration. The arrow directions depict how fish move through the tidal delta and into Skagit Bay. Arrow thickness represents the number of Chinook salmon following these pathways, based on restored habitat area and connectivity. Conceptual habitat restoration areas are shown in pink. Subsets of delta habitat that are expected to respond in similar ways are circled and numbered. Monitoring hypotheses for each area are in Table 23.

The very nature of the system we study presents a set of challenges. First, there is but one natal tidal delta for the Skagit River population, making replication across watersheds impossible. Nor can other river systems in Puget Sound act as comparisons, due to the sheer cost of such a monitoring effort and the highly variable degree to which populations are monitored and supplemented (see IMWSOC, 2004). Second, conditions of fish and habitat in the tidal delta may reflect to some degree conditions upstream. Third, the vast changes in body size and habitat use that Chinook salmon undergo during estuary rearing require us to use several techniques to monitor outmigrants. Despite these challenges, we believe we can provide a system-wide assessment of the effects of estuary restoration on Chinook salmon populations.

Study Designs

We will use different monitoring designs at different spatial scales to evaluate the effects of restoration. At a project scale, we use before-after-control-impact (BACI) designs to examine effectiveness of each restoration project. The Skagit River System Cooperative is employing this technique via treatment and reference reaches to examine whether restoration at Deepwater Slough has successfully increased habitat utilization to match or exceed reference levels (see Appendix C). With the exception of proposed mark-recapture studies (see below), all project-specific monitoring is funded from other sources.

BACI designs may also be possible at larger spatial extents due to the independence of two delta subsystems, the North and South Fork of the Skagit. Because much of the initial restoration has been and will be targeted primarily on the South Fork of the Skagit, we have the opportunity to use a BACI design, where data obtained in the South Fork area can be used as a before-after comparison of a restoration treatment and data obtained from the North Fork acts as a reference during this entire period of time. Data from both beach seine and tidal delta sites contiguous to the North and South Fork will allow us to examine how body size and life history diversity change in response to this restoration. BACI designs will also be possible for two other larger scale analyses related to major restoration projects that improve migration pathways (connectivity) to habitat within Swinomish Channel (polygon #1 in Figure 35, Table 23) and the bayfront along Fir Island (polygon #3 in Figure 35, Table 23). Each of these polygons has multiple sites with ten years of data.

BACI designs are appropriate as long as restoration does not occur in the North Fork. When this happens (starting in 2009), the effects of restoration will need to be examined at a system-wide scale. To tackle estuary restoration at the system-wide scale, other monitoring designs are required because there is no control possible for the Skagit River tidal delta. For some variables, a before-after (BA) intensive design (Roni et al. 2005) with comparisons among multiple subregions of the Skagit estuary is possible because of 10 years of existing sampling at index sites. Our plan will also provide us with two or three years of randomized data collection before the second major estuary restoration projects are implemented, and should therefore provide the basis for a BA design. However, this design will require some modification because multiple restoration projects will be completed over a span of years, not at one single time which the BA design usually assumes. Consequently, we may use regression designs to examine changes in key variables over time as projects are completed. The dashed lines in Figure 33 illustrate conceptually how different biotic variables might respond to delta habitat restoration.

Sampling methodology

In most systems, a complete census of the entire monitoring area is impossible, and therefore requires some sort of sampling. Status and trends monitoring often follows one of two general sampling approaches: 1) use of "index" sites that are repeatedly sampled over time, or 2) random selection of sites during each sampling event (Larsen et al. 2001). Because site-level environmental variation is largely controlled for by sampling at index sites, they are ideal for detecting trends over time. However, they suffer from the fact that they are not random samples and therefore may produce biased estimates for variables of interest (Larsen et al. 2001). Our monitoring plan shifts some effort from index-site to randomized sampling schemes, while at the same time maintaining some index sites for monitoring trends. This will allow us to obtain unbiased estimates of population density while we continue to monitor trends over time.

<u>Index sites</u>. We propose to continue monitoring index sites at the same temporal frequency that has been conducted over the last decade (Table 24). Current sampling sites include the Skagit River, tidal delta, nearshore and offshore areas of Skagit Bay. The nearshore/offshore study area extends from Deception Pass (north) to Saratoga Passage (south) to be roughly equal in distance from the mouths of North Fork and South Fork Skagit River sloughs. Sites are shown in Figure 36.

Table 24. Current monitoring program related to assessing the effects of restoration in the Skagit River estuary.

Method	Habitat	Sampling regime	# index sites	# years at index sites	Random sites (# per sample trip/ # per year)
Outmigrant trapping	Mainstem	Daily: Feb-Aug	1	12	----
Fyke trapping	Tidal delta & Swinomish Channel	Biweekly: Feb-July Monthly: August	11	12	4/40
Beach seining	Nearshore[1] & Swinomish Channel	Biweekly: Feb-August Monthly: Sept-Oct	18	10	12/192
Townetting	Offshore	Monthly: Mar-Oct	4	4	16/112

[1]Includes 4 pocket estuary sites: Lone Tree Lagoon, Arrowhead Lagoon, Grasser's Lagoon, and Turner's Lagoon. Pocket estuary sampling started in 2002.

<u>Randomized sites</u>. This monitoring proposal will augment the current site-specific monitoring with spatial randomization to test whether our understanding of Chinook salmon populations in index sites is the same throughout the study. We use a stratified

random design to account for large differences in space/connectivity. We will stratify sampling for three habitat types: delta blind channels, nearshore (including pocket estuaries), and offshore (Table 24).

Figure 37 shows the population of potential delta blind channel sampling sites. There are 498 blind channel complexes within the existing tidal delta habitat when you include the delta fringe running from Camano Island to Padilla Bay. We will stratify by the same sub-delta polygons shown in Figure 35 and randomly select sites to sample. We will devote one crew per sampling week to conduct this effort. Approximately 40 sites will be sampled over the season.

Figure 38 shows the population of potential nearshore sampling sites. There are 184 nearshore geomorphic units (includes pocket estuaries) within the Skagit Bay Study area. From these 184 units, we will eliminate those that cannot be sampled via beach seining, and then stratify based on three general divisions of Skagit Bay: south of Strawberry Point (south), Hope Island to Strawberry Point (middle), and Hope Island and points north (north). These areas differ in habitat use patterns, and correspond roughly to areas that would be colonized primarily by fish from the South Fork (south), areas colonized by fish from both North and South Fork (middle), and areas subsequently used by both North and South Fork fish (north). We will include two sites per sampling week to conduct this effort. Approximately 192 sites will be sampled over the season.

We will randomly select offshore sites using the same three strata determined for beach seining (north, middle, and south Skagit Bay), selecting points deeper than 30 feet within these areas based on a uniform grid. Based on the variance determined from previous index sites, this type of design should provide an unbiased estimate of offshore Chinook salmon population density with 27% relative error at $\alpha = 0.90$ if we sample 12-16 sites randomly per month (relative error for our data asymptotes at 25%). The remainder of our effort will be devoted to sampling four index sites each month.

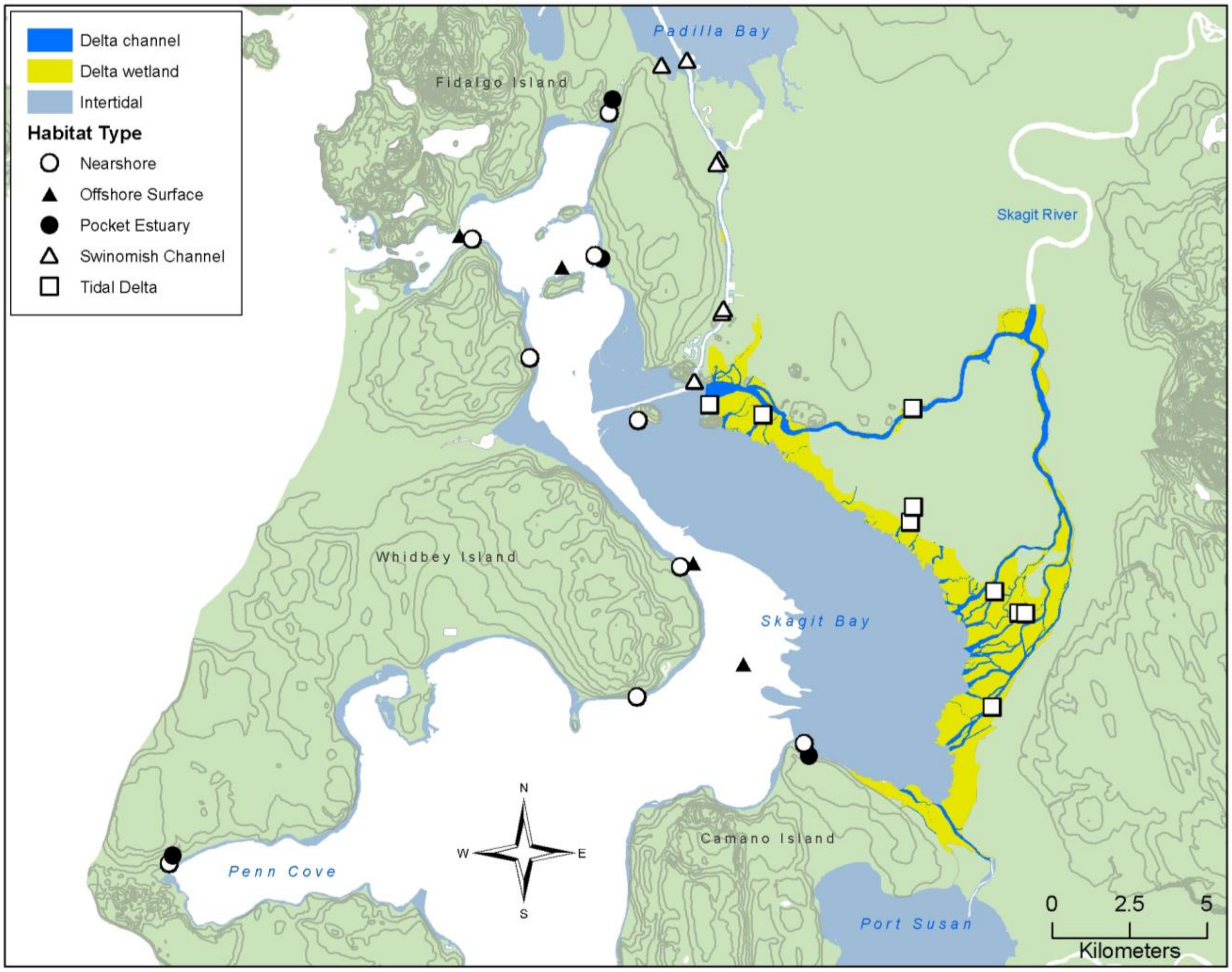

Figure 36. Index sites in the tidal delta, nearshore, and offshore.

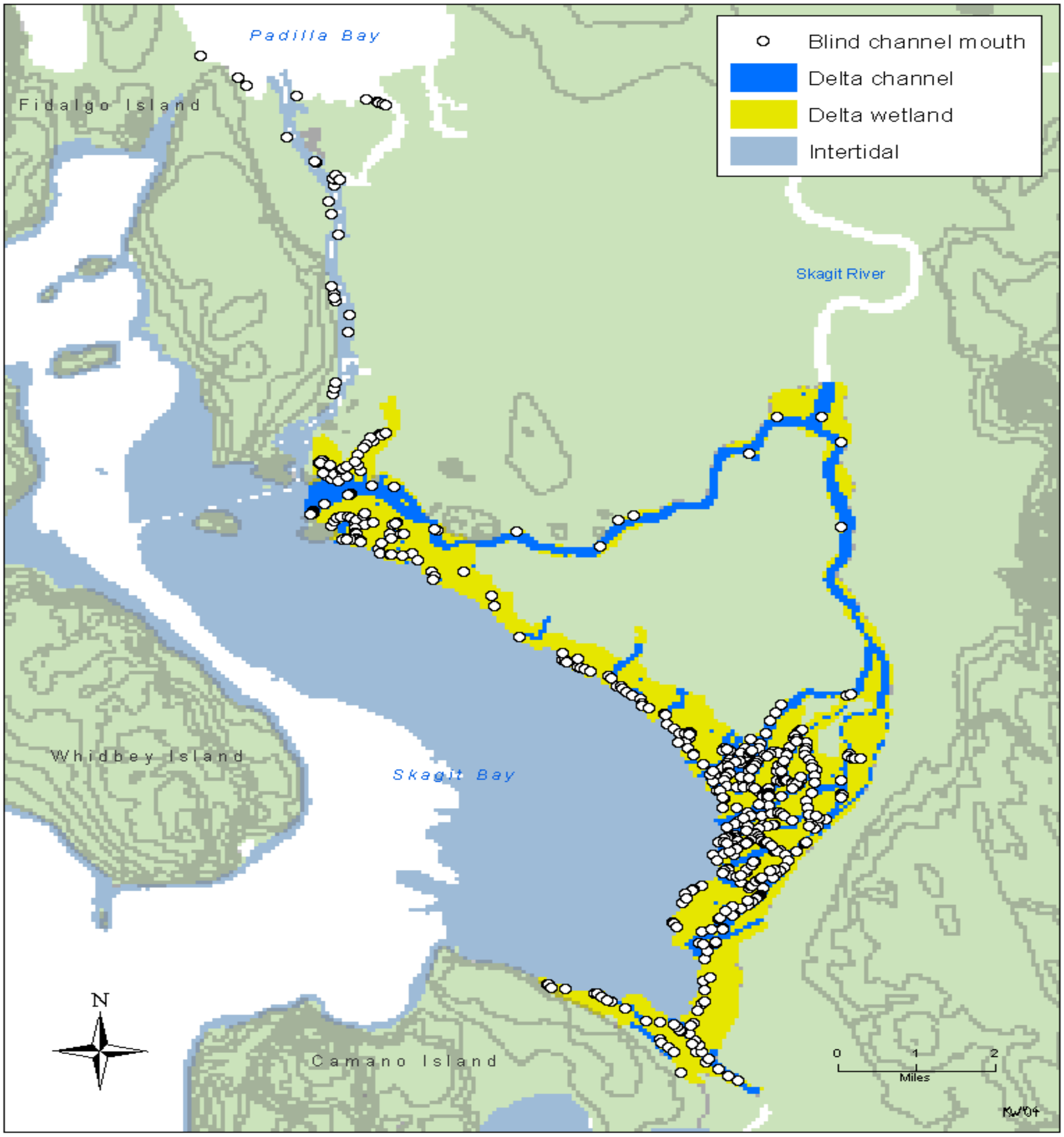

Figure 37. Map of potential blind channel Each point represents an individual blind channel complex that could be fyke trapped or beach seined.

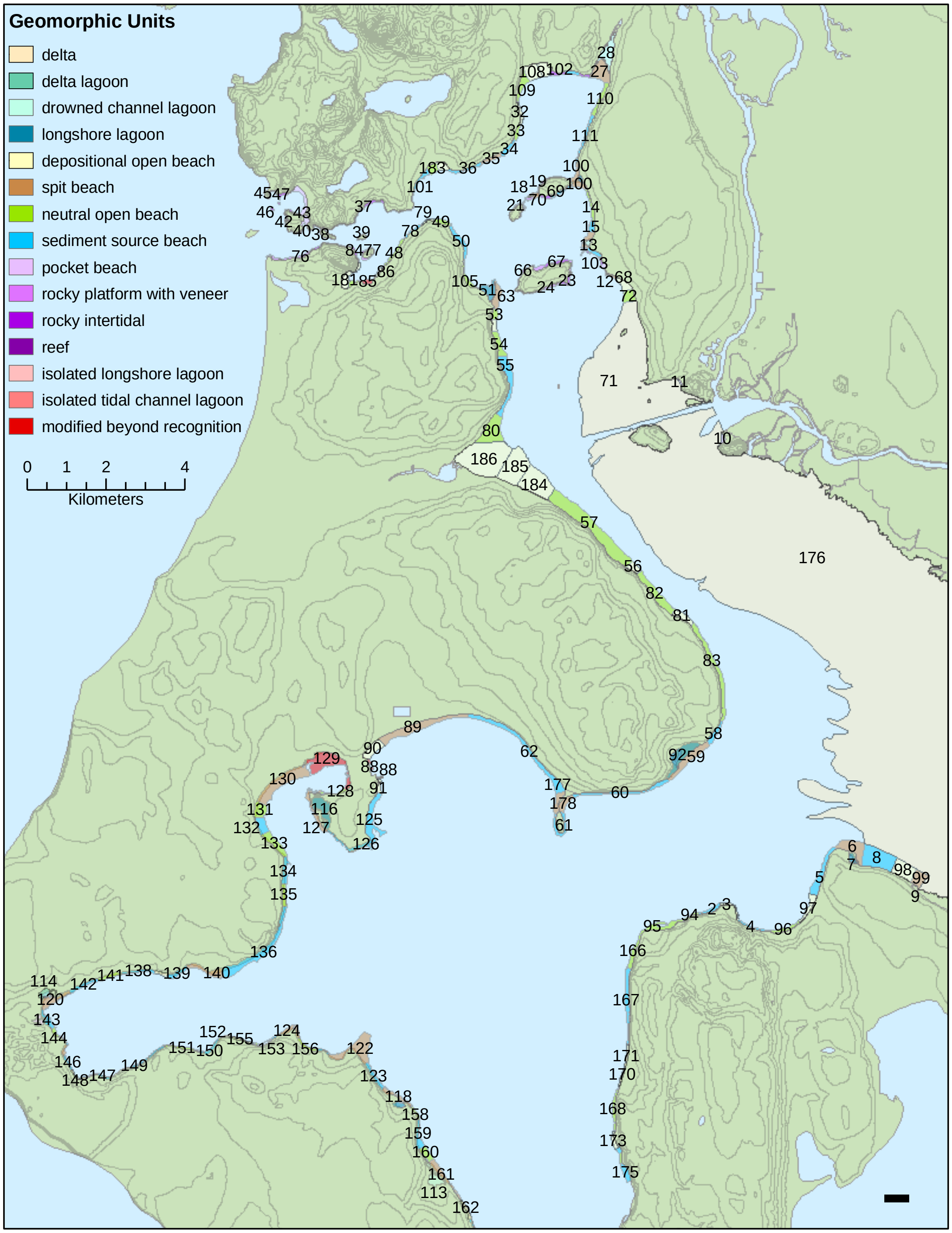

Figure 38. Map of potential nearshore sampling sites. Each number represents a geomorphic unit that could be beach seined.

Variables of interest

Estuary restoration should have a number of system-scale consequences for Chinook salmon. These can be summed up as changes in abundance, changes in spatial distribution, changes in survival, and changes in life history variation (body size, life history types). Our monitoring plan incorporates a number of measures of the juvenile outmigration that can be used to test for effects of restoration. Most of these measures are made at a number of sites (index or randomly sampled), from which we expand to assess population-level changes.

<u>Changes in abundance/density</u>. The most direct measure of the population is an estimate of abundance. The Skagit River watershed benefits from four juvenile abundance estimates, made through the cooperation of three different agencies. The first direct estimate of juvenile population size is made prior to estuary residency, based on the number of outmigrants captured by WDFW in a screw trap at Mount Vernon. This estimate of abundance is critical to our proposed restoration monitoring effort because it provides information about the Chinook salmon population entering the tidal delta and nearshore. The three abundance estimates made in the estuary are via fyke trapping in the tidal delta, beach seining in the Skagit Bay nearshore, and townetting in the offshore. Because all these abundance estimates will depend on the amount of habitat sampled, we convert all abundance measures to densities. In the tidal delta, 36% of the variation in density is explained by environmental variation (temperature, salinity, discharge, and tidal exchange) related to connectivity of sites within the distributary channel network, and 31% is explained by a density-dependent relationship with the number of outmigrants estimated at Mount Vernon (Appendix D.VII in Beamer et al. 2005). Because of these relationships, we can predict the added capacity of any estuary restoration project. This provides our first way to test for system-wide changes due to estuary restoration. The Skagit River Chinook salmon recovery plan did this very exercise to conclude that the proposed number of restoration projects should be sufficient to support recovery of the population (Skagit River System Cooperative and Washington Department of Fish and Wildlife 2005).

<u>Changes in spatial distribution</u>. According to our predictions, tidal delta restoration should restructure the migration pathways of salmon through the tidal delta and into the Skagit Bay nearshore. Testing this hypothesis requires comparisons of Chinook salmon abundance in different regions over time. Therefore, sampling at the site level needs to be stratified across different regions. Changes in spatial distribution over time (years) will be tested using regression designs.

<u>Changes in body size</u>. Our monitoring data support the hypothesis that body size of smolts is most directly related to density in the tidal delta. Therefore, we should be able to test the prediction that restoration should result in an increase in mean body size of fish captured in the nearshore, especially those fish captured near the restoration projects. Our standard monitoring procedure involves taking length and weight of up to 20 fish captured at each site, and as long as the spatial distribution of sampling sites is well

distributed, this design should be sufficient for estimating changes in body size due to restoration using regression designs.

Changes in the frequency of history types. Our monitoring supports the hypothesis that fry migrant subtypes (but not others) are the product of density-dependent migration through the tidal delta (Figure 33B). If so, estuary restoration should decrease the frequency of fry migrants captured in the nearshore. Testing this prediction boils down to our ability to equally detect fry migrants and other life history types in the nearshore. While it is sometimes not possible to distinguish other life history subtypes, the timing and size of fry migrants in the nearshore make this a very distinct population segment. A randomized sampling scheme should allow us to detect decreases in fry migrants using a regression design.

Changes in survival. We are proposing three types of survival studies as part of our future monitoring plan. These consist of mark-recapture studies at project or system-level scales and age structure and population abundance studies at a system scale. Mark-recapture studies can be used to examine survival by individually marking fish and examining the disappearance of these individuals over time. These studies can be logistically challenging because many individuals need to be marked to insure sufficient recaptures and much effort needs to be placed in recapture efforts. However, such problems can be at least partially overcome by using automated tag reading systems. Given the size constraints of juvenile chinook in the estuary, automated systems are limited, but can be used in two circumstances. First, at a project scale, we can PIT tag fish and recapture them using automated pit tag detectors in tidal channels. At a system-wide scale, we can tag large smolts in Skagit Bay or at the Mount Vernon trap and relocate them using linear arrays of nearshore receivers at the exits to Skagit Bay (following Welch et al. 2003).

We are proposing to compare residency and survival of PIT-tagged fish at the Wiley Slough restoration site. This project would involve a BACI design and require the installation of two sets of PIT tag readers, one in a slough undergoing restoration as part of the Wiley Slough project, and one nearby control slough that will not be affected by restoration. This restoration project is slated to be completed in 2008 (see Appendix B), thereby allowing us two years of pre-project data collection.

In 2005, we received funding from the Pacific Salmon Commission for acoustic tagging. During the summer of 2005, we conducted some preliminary field trials, and plan a much more extensive release in 2006. This study would be repeated over the course of the next decade to evaluate the effect of restoration on survival using a regression design.

Another technique for measuring survival is using changes in age structure. These so-called life table approaches have generally been used on populations to estimate annual survival rates in age-structured populations. By extension, these studies can also be applied to study weekly or monthly survival as long as age-structure data exists at this temporal resolution. This type of study should have the benefit of being relatively straightforward to collect, and will be relevant at medium to large spatial scales as long as sufficient data are collected. The disadvantage of this type of study is the investment in otolith preparation and analysis. We now have several years of otolith data from both delta and nearshore life stages with which we can apply this approach and test its utility for estimating survival. Depending upon the results of this preliminary study, we may use this approach based on annual collections of fish in the nearshore to evaluate survival each year, and then examine the effect of restoration using a regression design.

A final approach is to estimate survival through the tidal delta using population estimates of outmigrants made at the Mount Vernon trap and in the Skagit Bay. The second population estimate will incorporate site-specific density information for both nearshore and offshore, expanded to account for availability of these habitats, amount of sampling, and length of residency. We anticipate that these estimates will be unbiased (since they will be based on a randomized design), but may have low precision due to the expansion factors. Even so, they will be useful to compare with the life table approach. These analyses would rely on a regression design to examine changes in survival as additional restoration projects are completed.

Skagit Power Analysis

We examined our ability to detect a response in cumulative abundance to restoration in the tidal delta by performing a power analysis using the available monitoring data in a BACI design. In the case of the Skagit delta, monitoring data from the South Fork acted as a treatment because of the Deepwater Slough restoration, while data from the North Fork acted as a reference because no restoration projects have been implemented yet. We used cumulative abundance over the sampling season from the three index sites on each river fork sampled for 8 years pre-restoration and 3 years post-restoration. Cumulative abundance is based on the density of fish sampled in tidal channels every two weeks between February and June. If restoration works as we hypothesize, density of fish – and by extension cumulative abundance – would decline in the South Fork index due to the increase in rearing habitat capacity in the Deepwater Slough restoration project.

We performed our power analysis by regressing pre-treatment data from the South Fork against that from the North Fork and inserting the variance of the residuals from this regression into Equation 1. We also plotted the post-treatment data as a comparison. As shown by Figure 39, there was a strong correlation (r^2=0.50) in cumulative abundance between North Fork and South Fork pre-restoration data (filled diamonds). This correlation substantially improves our ability to detect a response to restoration. We plotted the minimum detectable percentage change in cumulative abundance based on the number of years of monitoring data post-restoration. As shown in Figure 40, the minimum detectable change improves from approximately 17% change in cumulative abundance detectable with five years of post-restoration data, to an 11% change detectable with 10 years of monitoring data, with diminishing returns thereafter.

Because we already had three years of post-restoration monitoring data, we were able to estimate the actual change in cumulative abundance resulting from the Deepwater Slough restoration project. In Figure 39 the open circles and dashed line represent post-restoration data. Restoration of Deepwater Slough apparently resulted in a reduction in the cumulative abundance by ~164969 fish-days (the difference between the regression lines, assuming their slopes are similar), which is 54% of the average pre-restoration cumulative abundance. This change to date is actually much higher than the minimum detectable change (Figure 40); hence, additional large restoration projects have a very good chance of providing a detectable signal at the spatial scale of the entire tidal delta. As the data become available, we will perform similar power analyses on the other variables of interest described earlier.

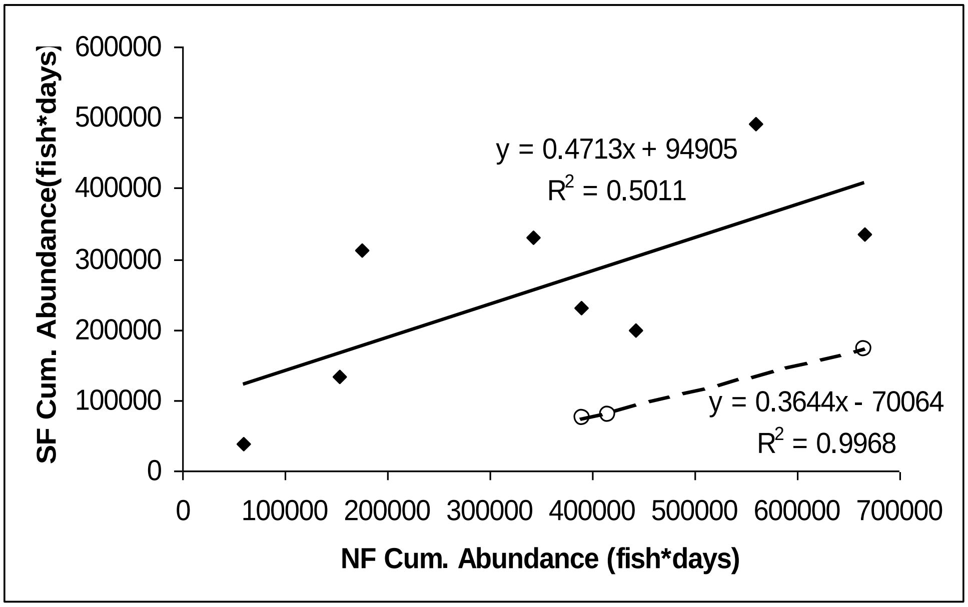

Figure 39. Regression of Cumulative Abundance at index sites in the South Fork (SF) vs. the North Fork (NF) Skagit River. Limited post-Deepwater Slough restoration data indicate a 54% reduction in cumulative fish abundance at the South Fork index sites.

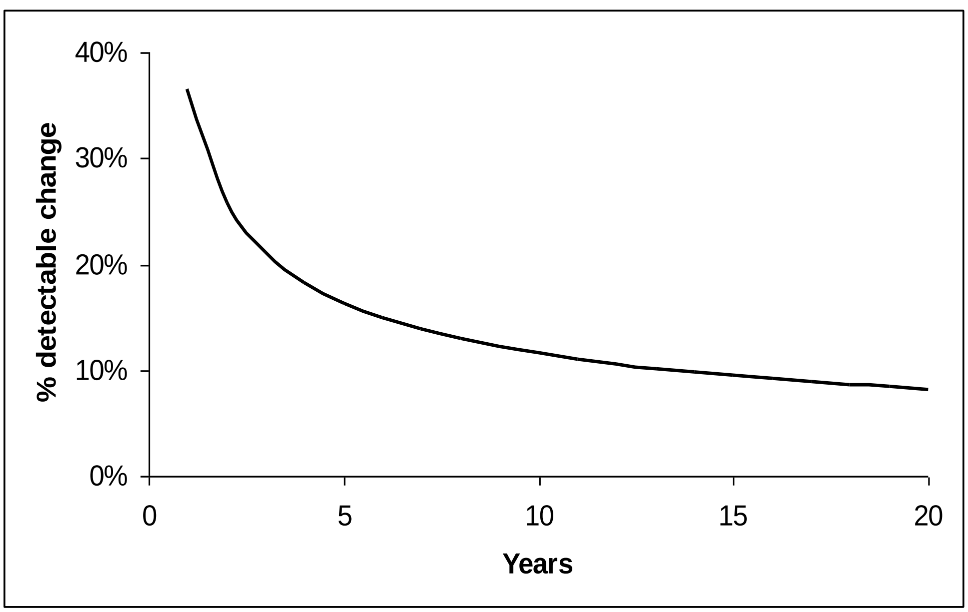

Figure 40. Minimum detectable change in cumulative abundance, shown as a percentage of the mean, vs. number of years of post-treatment data collected based on a BACI

analysis using the North Fork as a reference for the South Fork Deepwater Slough restoration actions (α= 0.10, β= 0.90).

Time line

Table 25 provides an overview of the major changes to the monitoring program resulting from IMW funding over the next five years. Restoration-related activities will include creating a design that can link proposed restoration projects with system-wide monitoring (following Figure 35), and initiating several restoration projects. Changes to monitoring studies includes the initiation in 2005 of the spatial randomization of juvenile sampling and comparing this approach with index sampling strategies after three years, when a sufficient number of randomized sites have been sampled. In addition, we will compare North Fork and South Fork monitoring data pre-and post- restoration in Deepwater Slough for changes in abundance, timing, and body size of juveniles. Survival studies include enclosure studies within the delta habitats and pocket estuaries, acoustic tagging studies in Skagit Bay, and design and implementation of a mark-recapture study to examine population responses to Wylie Slough restoration. Other planned projects relate to data automation.

Table 25. Five-year timeline for IMW-funded projects related to monitoring population responses to estuary restoration.

Year	**Restoration activities**	**Monitoring Studies**	**Survival Studies**	**Other**
2005	Refine design for linking currently planned restoration projects with biological responses	Initiate spatially randomized fyke trapping, beach seining, and townetting Continue index site monitoring	Delta (enclosure) Bay pilot (acoustic mark-recapture) Age structure pilot	Purchase data loggers, automated measuring devices
2006		Compare NF and SF data for restoration signals of Deepwater restoration	Delta (enclosure) Bay (acoustic mark-recapture) Delta mark-recapture feasibility study Skagit Chinook life history study completed	Standardize databases
2007	Smokehouse Floodplain (Fornsby), Milltown, and South Fork Dike Setback Restoration habitat benefits start	Compare randomized and index sampling techniques	Delta (mark-recapture) Pocket estuary (enclosure)	
2008	Wiley Slough Restoration habitat benefit starts		Delta (mark-recapture)	
2009	Swinomish Channel Causeway and Fisher Slough Restoration habitat benefits start		Delta (mark-recapture)	

LANDSCAPE CLASSIFICATION

Background and Motivation

The initial goal of the intensive watershed monitoring (IWM) landscape classification exercise is to classify and group watersheds with similar physical, biological and anthropogenic impact characteristics in relation to the watersheds where intensive watershed monitoring will be conducted. Ultimately, the classification process will support the extrapolation of expected results from restoration projects between monitored and non-monitored watersheds, inform the design and distribution of future restoration and monitoring projects, and support the interpolation or imputation of data across regions of the state not monitored as intensively as the IMWs. To generate landscape classification schemes for this purpose requires choosing biophysical variables that capture most of the information pertinent to salmonid productivity. The choice of these variables is therefore critical to the success of this exercise. Variables are to be chosen based on the current understanding of fish-habitat relationships available in the literature. The two main assumptions underlying this exercise are that the variables used are: 1) some of the most important determinants of the overall characteristic of a watershed, and 2) important determinants of salmonid population processes.

The basic list of variables currently thought to correlate to fish productivity includes climate, geology, watershed topology, vegetation, channel confinement and gradient, land-use/cover, ownership, wetlands. In addition, recent work shows that channel size (e.g., drainage area or some regionally calibrated estimate of discharge) and elevation are also important. A variety of studies have shown empirical correlations between fish numbers and these variables. It is certainly feasible to simply seek correlations between the distribution (histograms, cumulative distributions) of these attributes and fish species and population sizes, which would allow extrapolation to other basins that lack monitoring data. However, it may also be useful to look at how these attributes affect fish directly, which may provide a more powerful means of extrapolation.

Ultimately each attribute included in the extrapolation process somehow affects aquatic habitat and these effects occur point by point through the channel network. Thus, it is the combined suite of variables at each point that is important. For example, the relationship of channel gradient and valley width for a reach is lost when the distribution for each variable is viewed independently. A measure of basin productivity requires a method of assessing the effects and interaction of all variables point by point and then aggregating that information over the basin. A number of recent examples of constructing similar geomorphically based watershed intrinsic potential metrics have been very useful for the management and recovery planning of listed anadromous salmonids.

However, existing approaches to classifying landscapes for the purpose of managing and recovering listed anadromous salmonid populations have not included parallel assessments of immutable characteristics of watersheds and human land-use impacts on the watersheds. Therefore, to extend our current understanding of and approaches towards landscape classification specific to aquatic resources, similar methods must be applied to both the geomorphic and anthropogenic determinants of watershed intrinsic potential. Human activity over the past 100 years in the Pacific Northwest has dramatically altered the region's land- and waterscapes. As such, human activity has impacted the productive potential of most of the region's aquatic systems. In fact, some of the immutable factors used above to describe the inherent potential of aquatic systems have been changed by human activities (e.g., channel confinement, local climate). However, the primary mode by which human activities impact aquatic ecosystems is indirectly through land use practices (e.g., agriculture, urbanization). Therefore, any exercise to characterize broad scale patterns of aquatic productivity would be naïve to ignore the impacts of these activities. Thus, the effect of human activity on the landscape will be assessed through a parallel effort to develop a regional classification of watershed condition as a function solely of human activity. The potential list of human land use practices and activities that have the potential to alter relevant physical and biological processes will include: agricultural activities, forest practices, livestock activities, transportation, channel alteration, mining, urbanization.

Specific tasks and steps

1) Describe immutable and human impacts characteristics of watersheds

To classify the watersheds of Washington State based on their potential to support anadromous salmonids both as a function of underlying geomorphic and physiographic characteristics as well as anthropogenic impacts due to land-use practices and activities requires developing a multidimensional (~10) numerical score for each watershed (6^{th} field HUC) based on reducing multiple spatial data layers. For this effort, the input data will be of two types, basic geomorphic descriptions of the landscape and characterizations of human impact. The precise components to be evaluated will be determined during the initial scoping phase of the work. Generating the watershed scale descriptors requires the compilation of existing spatial data layers to generate consistent and complete coverages of biophysical condition of and human impacts on aquatic habitat across the Pacific Northwest. Considerable effort will then be required to standardize and extract watershed descriptions from these layers. To do so, we will (i) use existing or novel numerical algorithms to quantify the geomorphic and physiographic characteristics of watersheds (6^{th} field HUCs) in the Pacific Northwest based on the list of factors determined to be key determinants of physical and biological processes; and (ii) use existing or novel numerical algorithms to quantify the impact and extent of human land-use practices and activities in watersheds (6^{th} field HUCs) in the Pacific Northwest based on the list of factors determined to be key modifiers of critical physical and biological processes.

2) Classification of watersheds based on descriptions

Given the watershed scale description of the Pacific Northwest based on immutable characteristics and human impacts, each 6^{th} field HUC will be scored by reducing the data to a pair of condition vectors for each watershed with respect to immutable biophysical setting and human impacts. This process takes complex continuous data, including multiple data layers that contain significant spatial correlation, and generates a single score for each 6^{th} field HUC. For example, multiple soil or bedrock types could be present within each watershed, thus to score soils or geology, a dominant or most relevant type will be identified and given a numerical score. Alternatively, elevation, precipitation and air temperature within each watershed are continuous variables and are highly correlated, but each contains sufficiently unique information that one could not act as a proxy for all. In this case, watersheds would be classified based on bins of mean elevation (*e.g.,* <100m, 100 – 300m), and classes of temperature and precipitation (*e.g.*, cold-wet, hot-dry)

3) Ordination of classified watersheds

This step revolves around the rigorous quantitative process by which classified watersheds are grouped into clusters of "like" condition independently for immutable characteristics and for human impact scores. The clustering approach most appropriate for these data is a dichotomous ordination and classification procedure that relies on differential characteristics prevalent on one side of a dichotomy. Similar approaches are applied in community ecology analysis (community structure) and phylogenetics. Statistical support for the clusters and branching structure is evaluated by discriminant analysis, cross validation and bootstrapping. There is no preconceived notion of the scale of these clusters, but similar processes have generated groupings of 6^{th} field HUCs that approximate the 4^{th} to 5^{th} field scale, but that are linked by shared condition, not just the hierarchy of stream networks. Separate ordinations processes will be performed for the biophysical classification data and the human impacts data. However, further analysis and assessment may warrant combining some subset of these two classification schemes to construct a single hybrid scheme that represents both the inherent potential of the landscape and the current condition due to human activities. This latter approach would be suggested by testing the classification schemes against field collected data (see below) both separately and combined.

4) Testing and Application of resulting predictive maps

The clustering process generates hypotheses regarding the similarity of watersheds with respect to their physical and biological processes. If correct, then biological and physical monitoring data not used to parameterize the classification and ordination steps can be used to test the maps for consistency and accuracy. Several large-scale monitoring programs have generated data that is appropriate for these tests. These data are in hand, and will be used to evaluate and refine the initial mapping process.

Testing the value of the classification is critically important to its use in extrapolating results from specific watershed scale experiments to more general expectations that can be applied in a broader management context. The internal consistency of the classification scheme will be determined by its ability to explain the spatial variance in broad-scale aquatic monitoring indicators based on benthic macro-invertebrates, fish assemblages and diatoms. In addition, the classification scheme will also be assessed in its ability to describe the spatial variance in physical habitat properties such as flow regime and habitat diversity.

Once sufficient confidence in the initial ordination has been achieved, the maps will be applied in several tests of the overall approach. First, the 10 current IMW watersheds will be assessed for their being representative samples of broad regions of western Washington State. Second, within each cluster of IMW watersheds, individual streams are being considered as replicates and potential reference/controls. The classification/ordination process will allow an assessment of the validity of these assumptions. Third, the intersection of the two classification/ordination maps will be examined to address the issues of how dramatically have human actions altered the landscape, and have these impacts occurred in a manner that is correlated with or independent of watershed immutable characteristics. The last issue is critical to the design and implementation of future monitoring and restoration actions as it supports a landscape scale evaluation and prioritization of efforts across the Pacific Northwest. For example, if human impacts are strongly correlated to watershed characteristics, which they are expected to be since some of the watershed descriptors will be strong determinants of land use practices (*e.g.*, gradient and agriculture), then particular watershed classification clusters must be further subdivided into degree of human impact in order to properly distribute treatments, controls and extrapolation expectations over broad areas.

Of primary importance will be to develop a quantitative method for extrapolating the results from specific management experiments to regions that haven't been treated in this manner. It is probably the case that the SRFB funded IMWs alone do not provide sufficient diversity of landscape setting and management actions to generate the data necessary for quantitative models. However, there are a large number of similarly scaled management experiments underway, or in the planning stages, across the Pacific Northwest such that the larger set of IMWs could provide the data needed to parameterize a regional model of population or habitat benefit from management actions. It is for this reason that the initial spatial scale of the classification approach was increased from Washington state alone to include Oregon and Idaho as well.

5) Review, revision and expansion of approach

The potential broad-scale utility of this work demands a rigorous peer review of its results and methodology. NOAA-Fisheries NWFSC is leading this component of the IMW project, and will make use of its existing peer review process, but will also include the appropriate technical groups specific to Washington State's potential interest in the extrapolation exercise (ISP), and PCSRF's reporting and evaluation needs (SRFB identified technical review group). As a result of the technical review process, necessary modification and improvements will be implemented. In addition, NOAA-Fisheries is interested in applying a similar approach on a region-wide basis. Therefore, when the methodologies have been sufficiently refined, the project will be extended to cover at least the three state area of Oregon, Washington and Idaho.

Time line

Task 1- Compilation of base and more derived layers -- Complete.

Task 2-Classification of base layers at pixel and watershed scale -- Complete

Task 3-Preliminary ordination runs done (Figure 41). Refinements and improvements will be continuously updated, with major reporting of progress on a quarterly basis through calendar year 2006.

Task 4-Test of preliminary ordination runs to be completed, several performance metrics developed. Feedback on design of ordination and classification process will be continuous after initial implementation and testing. Quarterly progress reporting will be done through calendar year 2006.

Task 5-Revised and updated project will be submitted for peer review by Spring, 2006. Expansion of project to additional areas will be implemented following peer review process. Feedback from ISP review will be incorporated into classification validation testing process.

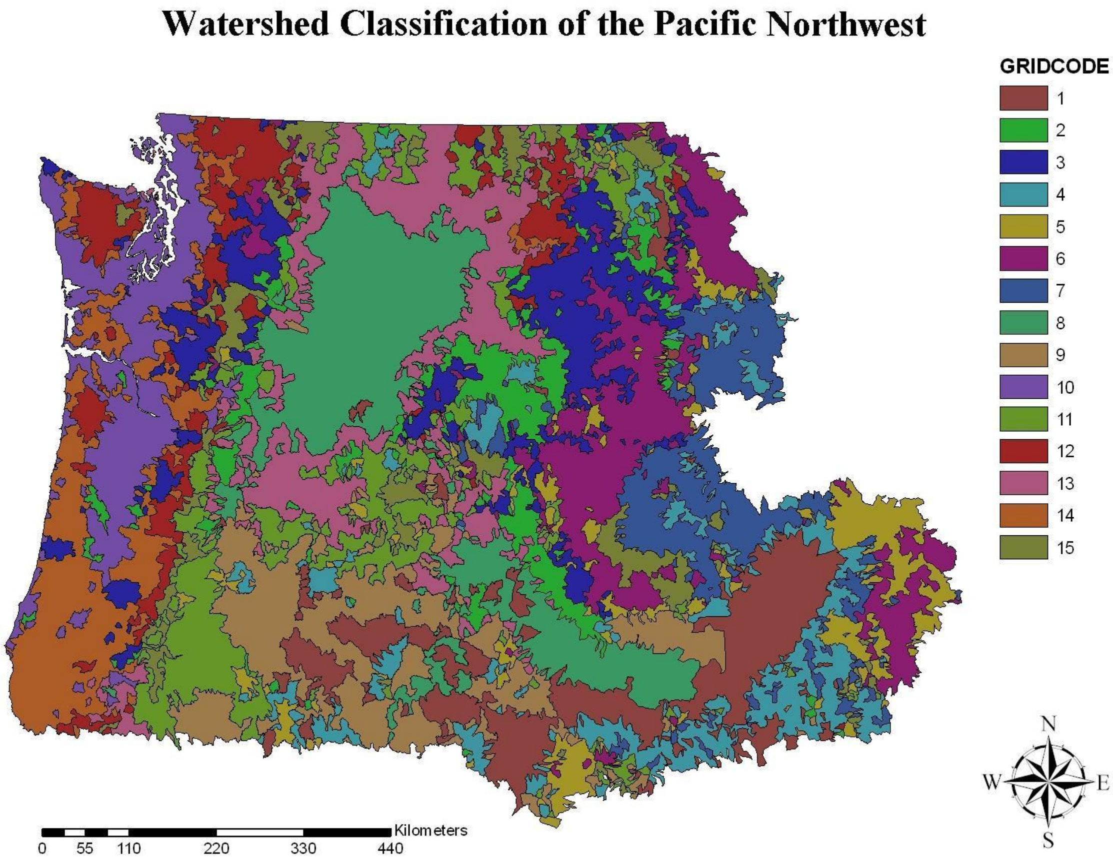

Figure 41. Preliminary ordination of datalayers. In this case, the study area is classified into 15 different "types" of watersheds (groups of 6th field HUCs) based on physical and climatological similarity.

Methods
Eight data layers of immutable landscape characteristics included climatic (temperature range, precipitation, and growing degree day) and influential physical-biological features (geology, elevation, slope, percent response reach, and tributary junction density) (Table 26). These layers were picked because of their importance in shaping catchments, hydrologic features, and fish habitat. Data layers were projected into a common coordinate system, complied in GIS, and summarized in both raster (200m pixel) and vector polygon (HUC6 watershed) space. Raster data sets were summarized to HUC6 watersheds by zonal summaries. The two metrics describing channel characteristics,

response area density and tributary junction density, were calculated at the HUC6 subwatershed scale and then converted to rasters (200m pixels) using ArcView 9.1 Spatial Analyst. All data were transformed for normality and normalized (0-1 scale) in raster space prior to summarizing to HUC6 watersheds. Raster grids were combined into a multilayer stack for classification in Arc/Info GIS software and HUC6 summarized data were classified using MCLUST software.

Table 26. Spatial Data layers constructed for immutable

Layer name	Layer relevance	Source	Source pixel size	HUC summary of 200m raster
Growing Degree day	ecosystem productivity	PRISM	2000m 1.25 arc-minute	mean
Mean annual precipitation	stream power terrestrial vegetation	PRISM	4000m 2.5 arc-minute	mean
Temperature range	Longitude temperature extremes	PRISM	4000m 2.5 arc-minute	mean
Elevation	hydrologic regime terrestrial vegetation	DEM	30m	median
Slope (degrees)	hydrologic complexity	DEM	30m	median
Percent response reach density	sediment delivery hydrologic complexity	NHD	HUC6	
Tributary junction density	hydrologic complexity ecosystem productivity	NHD	HUC6	
Geologic erodibililty	sediment source erodibility	ICBEMP		mean

<u>Data layers</u>

Climate layers

Temperature range, precipitation and growing degree day (base 50) were derived from 2 km grid PRISM data (http://www.ocs.orst.edu/prism/ for temperature range and precipitation; http://www.climatesource.com/products.html for growing degree day) and resampled to 200 m using bilinear interpolation. The median pixel value was used to summarize all datasets to the HUC6 watershed (Figure 42).

Elevation was derived from the USGS 30 m raster digital elevation model (DEM) and resampled to 200 m using bilinear interpolation. The median pixel value was used to summarize data to HUC6 watersheds (Figure 42).

Physical-biological layers

Slope was derived from the USGS 30 m raster digital elevation model (DEM) and resampled to 200 m using bilinear interpolation. The median pixel value was used to summarize data to HUC6 watersheds (Figure 43).

Erodible geology was derived from the Interior Columbia Basin Ecosystem Management

Project (ICBEMP 1999) major lithology data layer, a digital compilation of state geology maps (1:500,000) that has been reclassified into generalized rock categories (http://www.icbemp.gov/). This reclassified vector compilation was attributed using a hardness classification adapted from Dolan et al. (1975) which assigns an ordinal scale value to each rock type based on the relative hardness of minerals comprising the rock (Table 27). Individual classes were discriminated by the relative resistance of each rock type to physical and chemical weathering. The ranking scheme is generalized as erodibility depends upon the specific mineral content, cementation (especially for sedimentary rocks), grain size (for unconsolidated sediments), and presence of planar elements (i.e., bedding, schistosity, cleavage, and fractures) within the rock. Attributed vector polygons were rasterized (200m pixel) and the majority pixel value was used to summarize rasterized geology to HUC6 watersheds (Figure 43).

Response area density and tributary junction density were calculated from the 1:100K National Hydrology Dataset Plus (Dewald, In Press). Response area density was calculated by squaring the length of channel with gradient less than or equal to 4 percent and then dividing by area of the subwatershed. Tributary junction density was determined by counting the number of tributary junctions within each subwatershed and then dividing by the area of the subwatershed. When response area density and tributary junction datasets were converted to raster (200m pixels), all pixels within the same HUC6 watershed were assigned the same value (Figure 43).

Table 27. Geology erodibility based on Dolan et al. 1975.

Lithology	Erodibility
open water	0
alkalic intrusive	130
calc-alkaline intrusive	130
granite	130
mafic intrusive	130
ultramafic	130
mafic meta-volcanic	135
granitic gneiss	140
argillite and slate	150
mafic gneiss	150
mafic schist and greenstone	150
calc-alkaline meta-volcanic	155
meta-sed phyllite & schist	165
mixed meta-sedimentary	170
meta-siltstone	175
meta-sandstone	180
meta-conglomerate	185
meta carbonate & shale	190
shale and mudstone	210
siltstone	220
sandstone	230
conglomerate	240
carbonate	250
quartzite	260
mixed carbonate & shale	270
dune sand	330
glacial drift	350
lake sediment & playa	350
loess	355
alluvium	370
landslide	370
mixed eugeosynclinal	370
mixed miogeosynclinal	370
mafic volcanic flow	410
felsic volcanic flow	420
tuff	420
calc-alkaline volcanoclastic	430
felsic pyroclastic	430
mafic pyroclastic	430

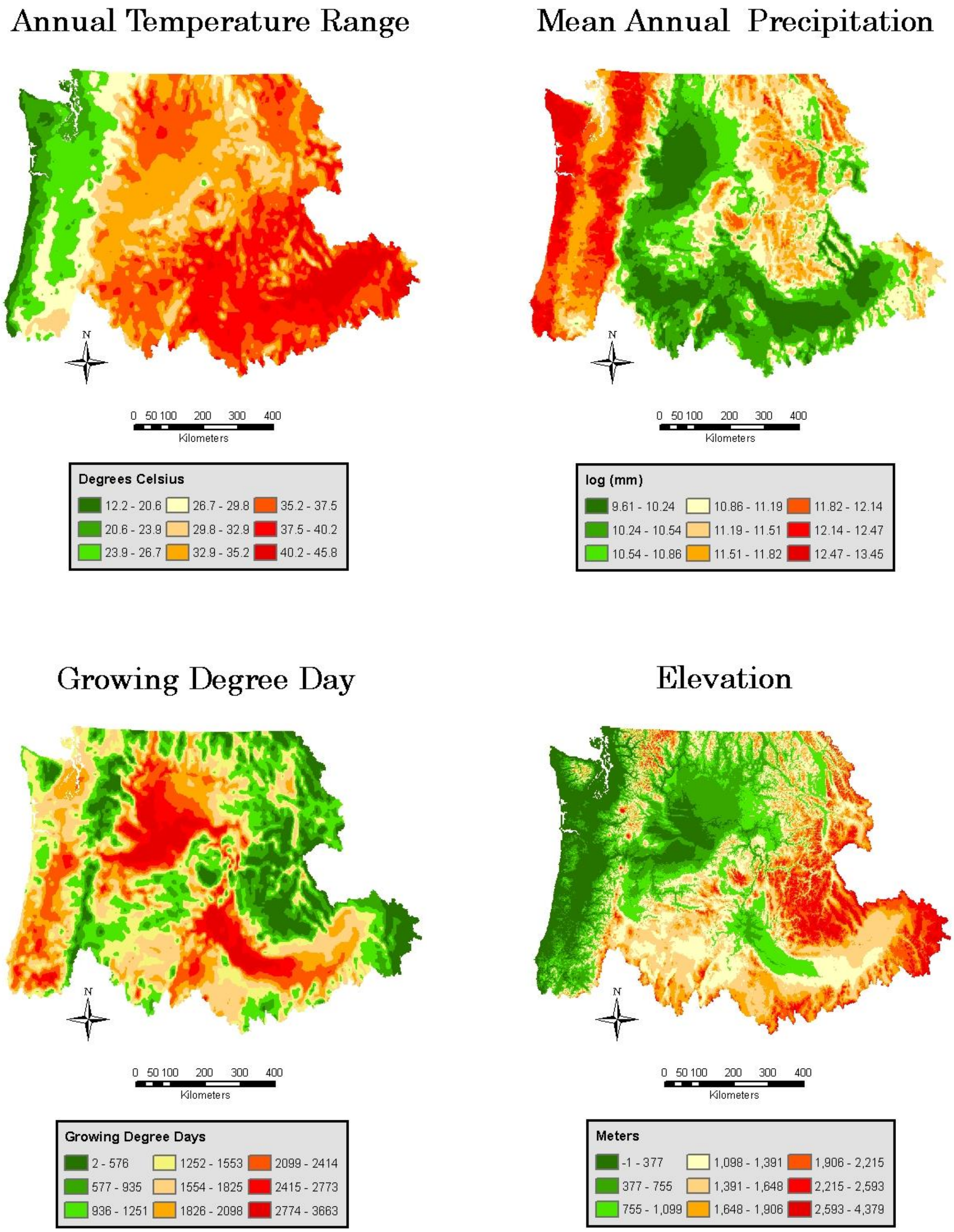

Figure 42. Climate data layers as input to the ordination scheme.

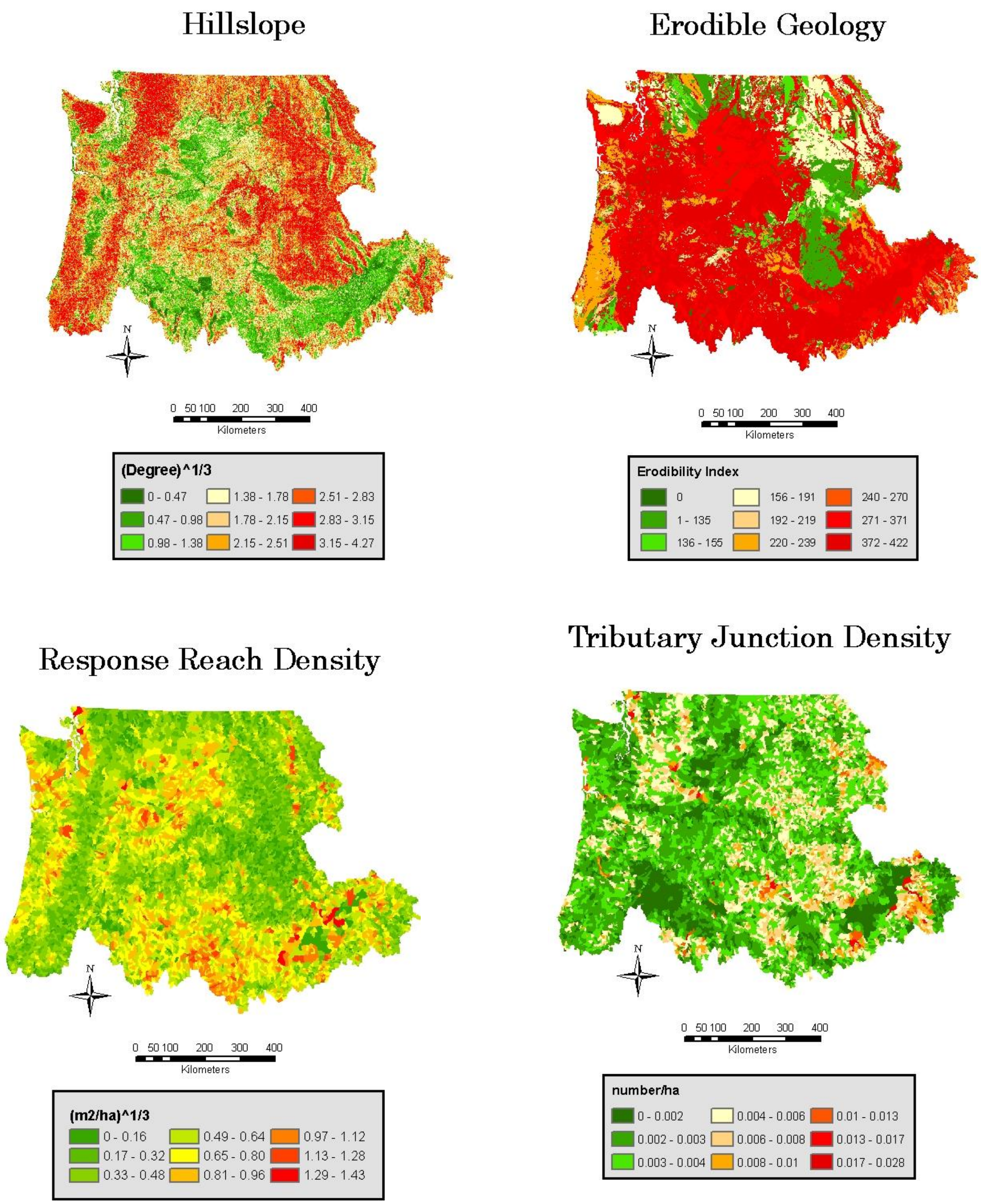

Figure 43. Physical-biological data layers as input to the ordination scheme.

Data preparation

Classification input data should have a roughly Gaussian distribution in order to accurately characterize classes using mean vectors and covariance matrices. Raw data for elevation, growing degree day, temperature range and geologic erodibility exhibited relatively normal distributions (Figures 44-45). Precipitation, response area density and tributary junction density data histograms were slightly skewed and were therefore log transformed to improve their distributions (Figures 44-45). The slope data was strongly left skewed and was transformed by taking the cube root of degree-slope – the resulting distribution is unimodal, but lacks strong tails. Further transformation may be necessary (using a Uniform to Gaussian transformation), but was not done at this point.

Classification clustering is maximized when input data layers have similar data ranges. As a result, our transformed data layers were normalized using the following equation to a range between 0 and 1 prior to running the ISOCLUSTER classification:

$$Z = (X - Xmin)/(Xmax-Xmin) \qquad (1)$$

Where:
Z = output grid with new data range (0-1);
X = input grid with original data range;
xmin = minimum value X;
xmax = maximum value X;

<u>Data Classification/Clustering</u>

200m pixel classification: ISODATA

The main objective of an unsupervised classification is to identify naturally occurring clusters in the data. For this analysis, we applied the ISODATA (Iterative Self-Organizing Data Analysis Technique) unsupervised clustering algorithm (Tou and Gonzalez, 1974) accessed via the ISOCLUSTER function in Arc/Info GIS software. The ISOCLUSTER function uses a modified iterative optimization procedure, also known as the migrating means technique. The process starts with arbitrary means being assigned by the software, one for each cluster (the number of clusters is dictated as a user input). Every cell is then assigned to the closest of these means, all in the multidimensional-attribute space. New means are then recalculated for each cluster based on the attribute distances of the cells that belong to the cluster after the first iteration. The process is repeated enough times to ensure that the migration of cells from one cluster to another is minimal and all the clusters become stable. The user specifies the number of classes, number of iterations, minimum number of cells in a class, and sampling interval. The ISOCLUSTER function returns a signature file, containing class means and covariance matrices, which are then used as input for the maximum likelihood classifier (MLCLASSIFY function in Arc/Info). The classifier uses the mean vector and covariance matrix of each class to compute the statistical probability that a grid cell belongs to a class. Each cell is assigned to the class for which it has the highest probability of being a member.

HUC6 classification: MCLUST

We intend to use the MCLUST software package for watershed classification. The software implements parameterized Gaussian hierarchical clustering algorithms. In MCLUST, 10 distinct models parameterize characteristics of potential clusters. Each model describes the distribution, orientation, volume, and shape of clusters. To initiate the clustering algorithms, initial cluster centers are estimate through discrete classification. For each model, an iterative maximum likelihood procedure determines cluster centers, assigns watersheds to clusters, and reports the Bayesian Information Criterion (BIC). The procedure is iterated for a range in the number of classes. By evaluating the BIC for each model/number of classes combination, the analyst is able assess the best-fit classification. Additionally, we have programmed the software to report a log-likelihood for each iteration. The log-likelihood can be used to calculate alternate best-fit criterion.

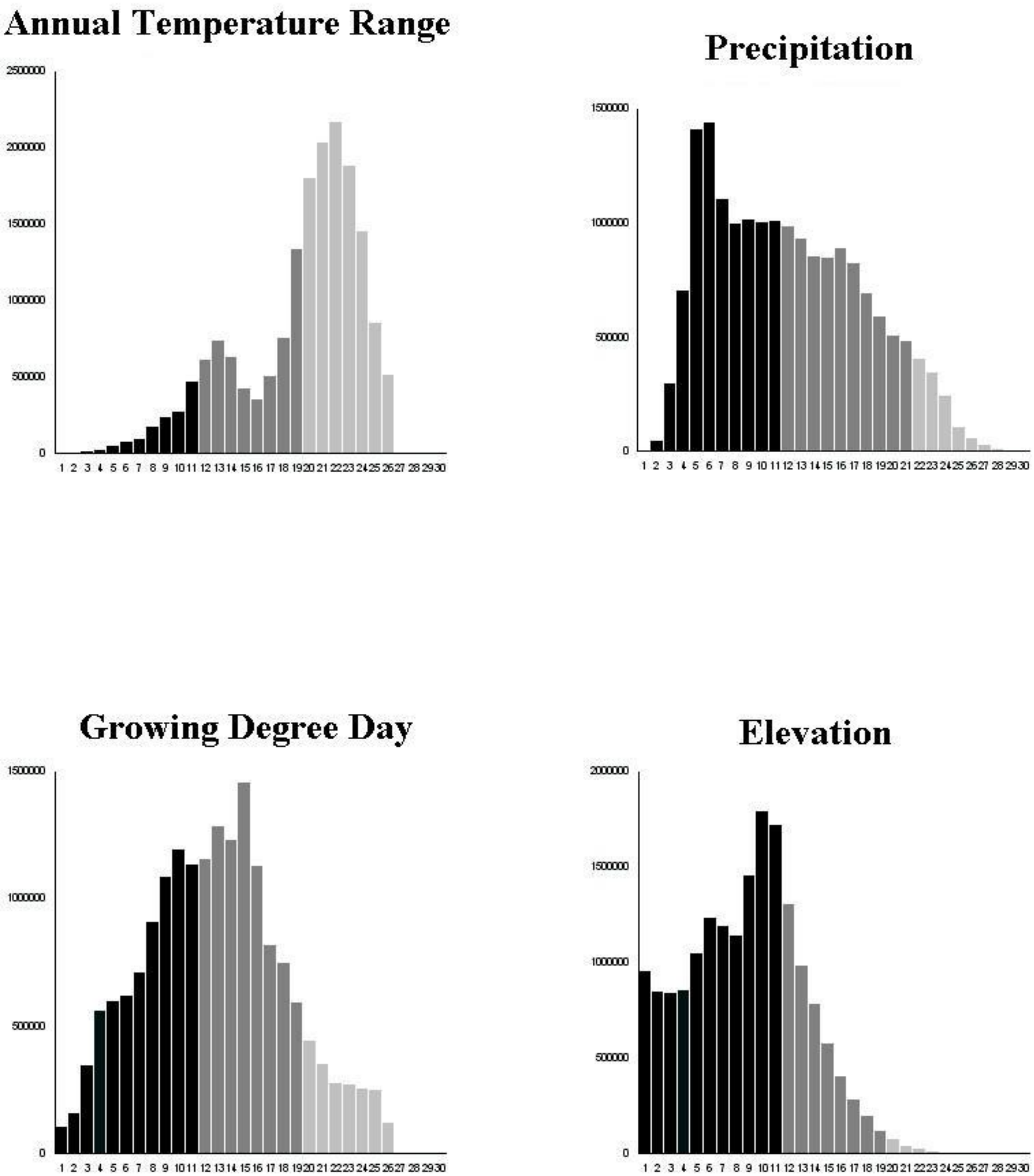

Figure 44. Distribution of Climate data as used for the 6th field HUC based ordination.

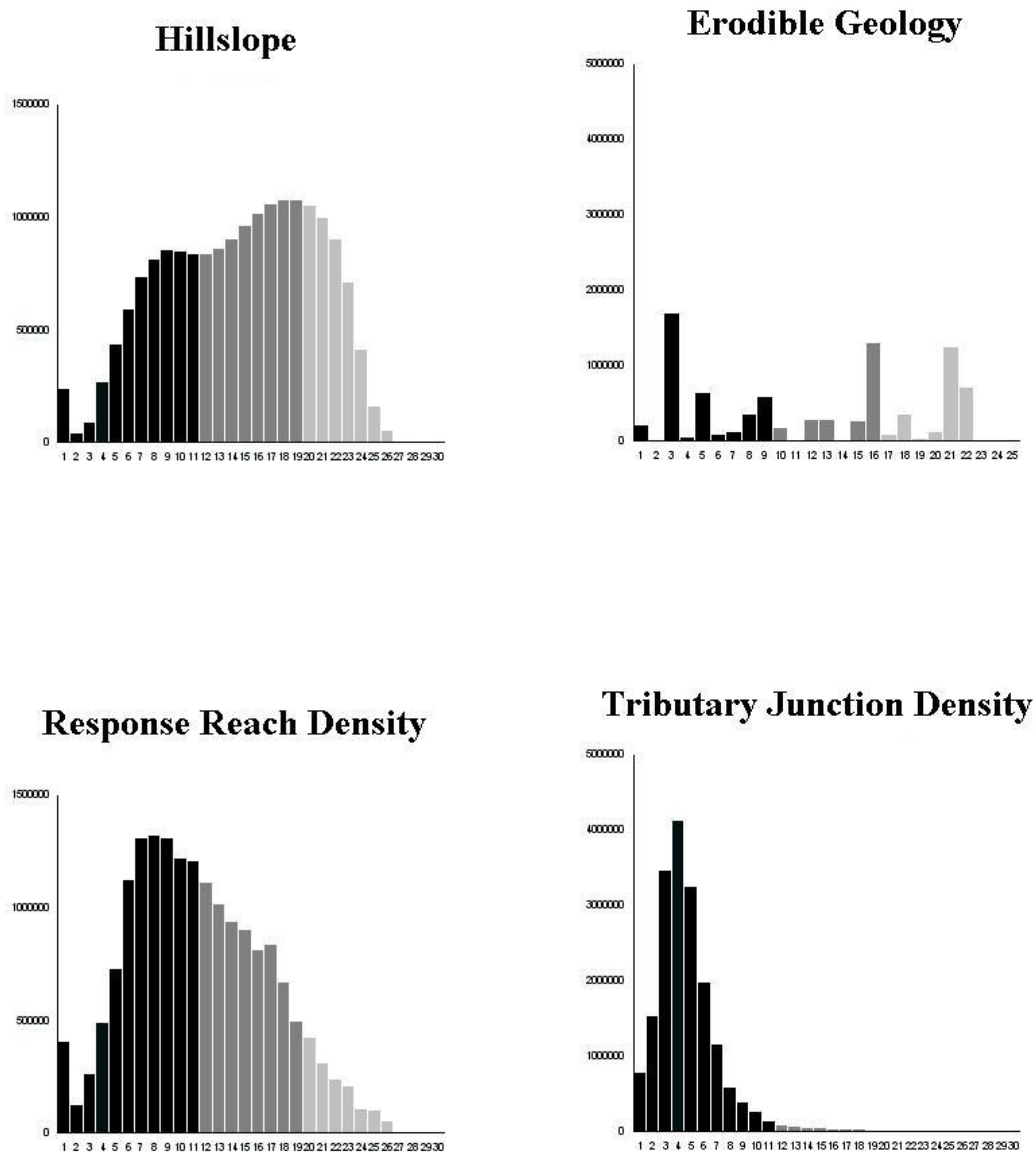

Figure 45. Distribution of Physical-biological data as used for the 6th field HUC based ordination.

BUDGET SUMMARY

The IMW program receives $1.09 million per year from the Salmon Recovery Funding Board to support monitoring activities. This has been supplemented by in kind contributions of staff time to support the monitoring effort and to provide program oversight (Table 28). In addition, the IMW program utilizes existing monitoring efforts where possible. This coordination with existing monitoring and the substantial in kind support comprise a substantial contribution to the IMW program.

Table 28. Estimated in-kind contributions toward oversight and monitoring and cost of the additional monitoring efforts within the IMW complexes with which we are coordinating.

IWM collaborator	In kind FY2006	Existing monitoring
WDOE	$53,000	
WDFW	$87,000	$200,000
NWFSC	$58,000	$200,000
Elwha Klallam	$24,500	$90,000
Weyerhaeuser	$78,900	
Skagit R Sys Coop		$158,000
Total	**$301,400**	**$648,000**

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