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CALIFORNIA COASTAL SALMONID POPULATION MONITORING: STRATEGY, DESIGN, AND METHODS

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EXECUTIVE SUMMARY

California's salmon and steelhead populations have experienced marked declines leading to listing of almost all of California's anadromous salmonids under the California Endangered Species Act (CESA) and Federal Endangered Species Act (ESA). Both CESA and ESA listings require recovery plans that call for monitoring to provide some measure of progress toward recovery. In addition, there are related monitoring needs for other management activities such as hatchery operations and fisheries management.

This California Coastal Salmonid Monitoring Plan (CMP) has been developed to meet these monitoring needs, describing the overall strategy, design, and methods used in monitoring salmonid populations. Implementation details of the plan are described in Shaffer (in prep.). The CMP uses the Viable Salmonid Population (VSP; McElhany et. al. 2000) concept as the framework for plan development. The VSP conceptual framework assesses salmonid viability in terms of four key population characteristics: abundance, productivity, spatial structure, and diversity. High abundance buffers a population against both 'normal' and catastrophic variation due to environmental conditions and loss due to anthropogenic factors. High productivity will lead to more certain replacement when populations are placed under either natural or anthropogenic stress. Wide spatial structure reduces extinction risk due to catastrophic events and provides pathways for recolonization. Diversity in life history traits (e.g., time of spawning, juvenile life history, adult fish size, age structure, degree of anadromy, etc.) provides resilience against extinction risk from changing conditions.

The CMP divides California into Northern and Southern areas with a boundary south of Aptos Creek and north of the Pajaro River, based on differences in species composition, levels of abundance, distribution patterns, and habitat differences that necessitate different monitoring approaches.

Both the larger Evolutionarily Significant Units-level scale and the population viability criteria are based on the four VSP parameters. The assessment of viability, however, will be based upon adult population size, and the distribution and connectivity of these populations (Boughton et al. 2007, Spence et al. 2008, and Williams et al. 2008). The CMP provides a sampling framework to collect information at the appropriate life stages and spatial scales to evaluate adult salmonid abundance both at larger regional scales and at the population level. Productivity is calculated as the trend in abundance over

time. CMP design also allows basic assessments of connectivity through the collection of juvenile distribution and relative abundance data. Measurement of diversity will be based on local evaluation of essential life history variants and both broad and focused assessments of genetic diversity patterns.

Adult abundance monitoring will be approached differently between the Northern and Southern areas due to differences in species composition, abundances, and habitat conditions. In the Northern Area, adult numbers will be estimated mostly through expanded redd surveys and in the Southern Area adults will be counted at fixed stations. In the Northern Area, adult abundance estimates will be needed for multiple species over large areas. Surveys will be selected in a random, spatially balanced manner. Spatial balance is important because salmonid numbers from samples near each other tend to be similar, so that more information relevant to a regional scale evaluation is obtained from samples that are spaced out. Redd surveys have generally been shown to be the most reliable means of estimating multi-species populations in California, but will require redd-to-adult corrections to estimate numbers of adults by species from them. Other methods (e.g., live fish counts for Chinook salmon) will be used where necessary.

In the Southern Area, steelhead are the only salmonid present and populations are very small, making abundance difficult to assess. Steelhead arrival is associated with storm events that raise water levels drastically. These species characteristics and environmental features therefore make steelhead in the Southern Area difficult to monitor, and due to the low abundance and difficult sampling conditions, fixed stations will be used to count adult Southern Area steelhead.

Spatial structure refers to the geographical and ecological distribution of salmonids across the landscape. Broad spatial distribution and connectivity among populations are important traits that protect against the effects of catastrophic events and buffer extinction risk, particularly at low abundance. Spatial structure will be monitored using summer and fall juvenile snorkel surveys over reaches selected in a random, spatially balanced manner. A larger number of juvenile surveys can be accomplished in less time and expense than adult surveys because it is simpler and can occur at a more operationally favorable time of the year.

In the Northern Area, spatial structure monitoring will be conducted only for coho salmon since steelhead occur over a wider area. This monitoring will provide estimates of coho salmon spatial structure. Since steelhead occur over a wide area, they will be counted as well as a relative measure of spatial structure. Chinook salmon spawn in only a few well-defined areas and outmigrate in the spring before the juvenile surveys take place, and information on their spatial structure will come from adult monitoring. In the Southern Area, juvenile spatial structure monitoring will be conducted for steelhead, the only salmonid present.

Diversity traits are strongly adaptive for local areas and populations, and these traits allow salmonids to survive in the face of unique local natural and anthropogenic challenges. Higher level diversity traits have been considered in the creation of the listing and stratification units; however, population level diversity traits may be very different from one geographical or population unit to another. Therefore, local diversity traits will need to be surveyed, eventually leading to local diversity monitoring plans. Specific projects targeting both broad and focused levels and patterns of genetic diversity will be developed. Tissue collections for these projects will be coordinated with other CMP activities.

Life Cycle Monitoring (LCM) stations will provide estimates of freshwater and ocean survival, essential to understanding whether changes in salmonid numbers are due to recovery from improvements in freshwater habitat conditions or changes in ocean conditions. An LCM station will include an absolute measure of adult abundance from a counting facility, a spawning survey estimate of adult abundance, and an estimate of outmigrating smolts. The adult counts and outmigrant smolt counts will provide estimates of fish in and fish out, that can be used to provide relative estimates of freshwater and marine survival. The counting station data and adult survey estimates will be used to develop an estimation factor between redds and adults for calibration of adult surveys conducted in other watersheds. The LCM sites are also expected to be magnets for other kinds of recovery-oriented research, particularly studies of fish habitat-productivity relationships and evaluations of habitat restoration effectiveness.

Finally, a data management structure will be created to provide general access to the CMP data. Monitoring is necessary to provide data that will be analyzed to inform management decisions, and those data must be made available in a timely manner to managers in a usable form. The data management structure is one of the most important parts of the CMP, ensuring that consistent data standards and protocols are applied across and within monitoring areas and that data flow is coordinated from the field to a central data collection center. It will also ensure that data reporting necessary for common analytical activities occurs in a timely manner and will provide a data source for other analytical needs.

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INTRODUCTION

Background

Drastic declines in salmon and steelhead populations have led to California Endangered Species Act (CESA) and/or Federal Endangered Species Act (ESA) listings covering all of California's anadromous salmonid waters (CDFG 2002, Good et al. 2005). Both CESA and ESA allow listing of subgroups of vertebrate species called Evolutionarily Significant Units (ESUs) for salmon and Distinct Population Segments (DPSs) for steelhead. These units are collections of populations used for species status assessments. In this document, the term ESU may also include reference to DPS. In California, all coho salmon (*Oncorhynchus kisutch*) are listed under both the CESA and ESA (Figure 1). All steelhead (*O. mykiss*) south of the Klamath River and coastal Chinook salmon (*O. tshawytscha*) south of the Klamath River to the Russian River are federally-listed under the ESA. Although not addressed in this plan, Central Valley winter-run and spring-run Chinook salmon are listed under both CESA and ESA and Central Valley steelhead are listed under the ESA. These listings require that both the California Department of Fish and Game (CDFG) and the National Marine Fisheries Service (NMFS) develop recovery strategies that will conserve, protect, restore, and enhance listed species. The Federal government requires that recovery planning include objective, measurable criteria that, when met, would result in a determination that the species be removed from listing (16 USC 1531, Endangered Species Act 1973). California further requires that listed salmonids be recovered to a level of abundance that would permit commercial use (California Fish and Game Code Sections 2050 to 2097).

Development of recovery goals that would result in delisting and achieving those goals through effective implementation of State and Federal recovery plans are at the core of the two agencies' responsibilities, and the ability to measure progress toward recovery at the ESU and population levels is fundamental to the process. Currently, monitoring of California's adult coastal anadromous salmonid populations is limited to a few adult counting stations, localized carcass surveys of fall Chinook salmon in various reaches (e.g., the Klamath, Mad, and Eel rivers), snorkel surveys of the major spring Chinook salmon and summer-run steelhead populations, and production and harvest monitoring of Klamath-Trinity Basin fall Chinook salmon (Boydstun and McDonald 2005). Limited monitoring of winter-run steelhead and more recently, salmonid moni-

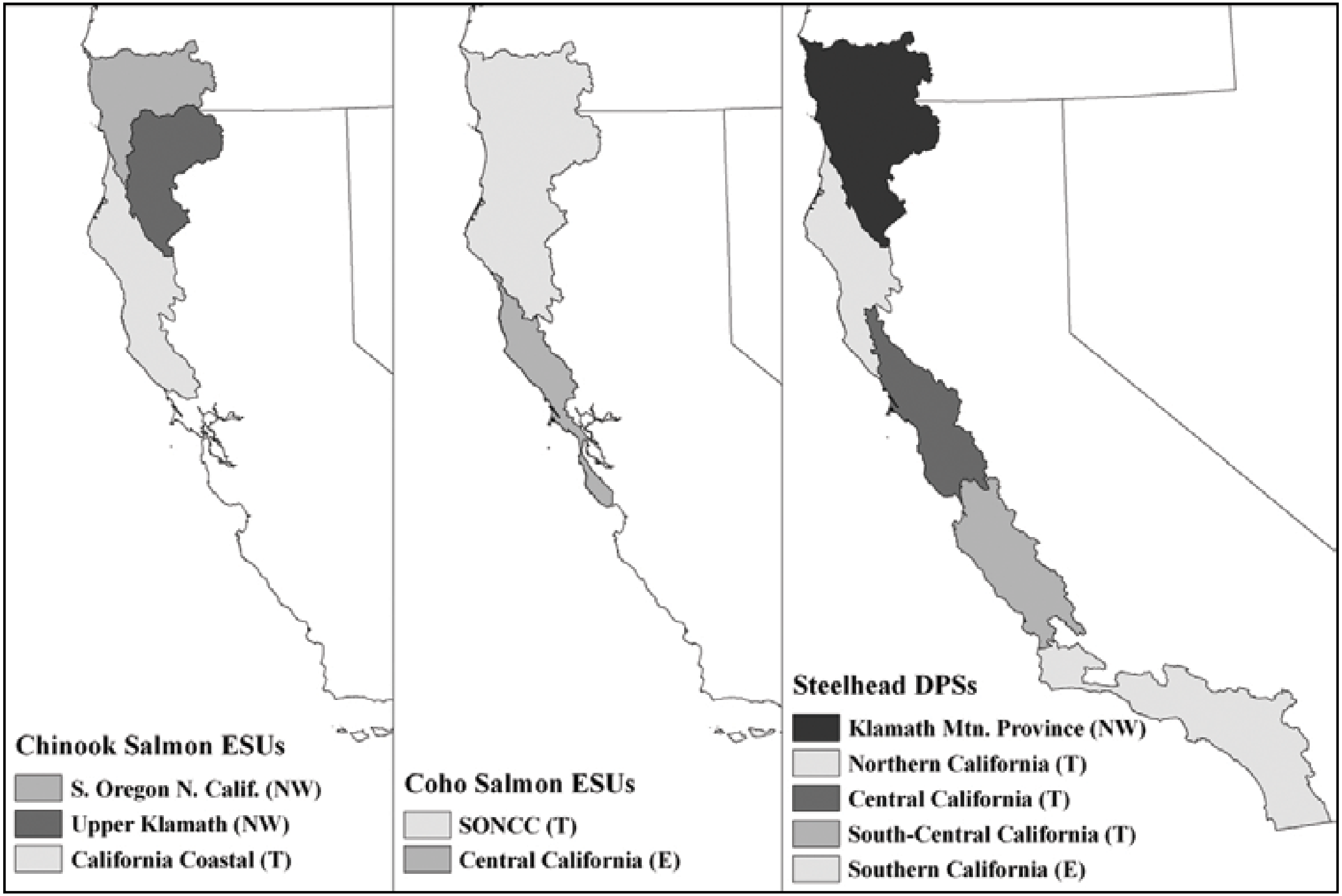

Figure 1. Evolutionarily Significant Units (ESUs) and Distinct Populations Segments (DPSs) for California Coastal Chinook and coho salmon ESUs, and steelhead DPSs. ESU and DPS designations: E for Endangered; T for Threatened; NW for not warranted. Designations by National Marine Fisheries Service, NOAA Fisheries.

toring associated with the development of the California Coastal Salmonid Monitoring Program (CMP) has begun in the Mendocino County coastal area; Freshwater Creek, Humboldt County; Lagunitas Creek, Marin County; and Scott Creek, Santa Cruz County (ibid). Broader and more intensive monitoring efforts are necessary to fulfill the responsibilities to measure progress toward recovery.

There is, of course, an even wider variety of other needs for salmonid monitoring, both to obtain specific information associated with recovery (e.g., hatchery impacts, restoration effectiveness) and for fishery management. Coastal salmonid monitoring has been considered in detail before, and this document summarizes and updates earlier efforts (Boydstun and McDonald 2005, Shaffer, in prep), and was prepared for use in developing a CMP. It should be considered a living document, as it will be modified as new information becomes available. The document has been peer-reviewed to ensure completeness, appropriateness, and use of current approaches to monitor California coastal salmonids.

California Coho Salmon Recovery Strategy

The primary purpose of the Recovery Strategy for California Coho Salmon (CDFG 2004) is to outline actions that will return coho salmon populations to a level of sustained viability and that will allow delisting under CESA. The strategy lists five goals to achieve this purpose: 1) maintain and improve numbers of key populations and cohorts, 2) maintain and increase the number of spawning adults, 3) maintain and enhance the range and maintain and increase the distribution of coho salmon in the State, 4) maintain existing habitat, and 5) enhance and restore habitat within the species' range. The State's recovery strategy identifies an additional objective to reach and maintain coho salmon population levels sufficient to allow resumption of tribal, recreational, and commercial fisheries in California.

Monitoring is essential to assess progress toward (or attainment of) specific regional recovery goals for key populations, such as numbers of spawning adults, range and distribution of fish, brood-year structure (presence), and number of stream kilometers restored or enhanced. The recovery strategy proposes monitoring focused on two essential elements: 1) the status and trends of coho salmon populations, their range and distribution attributes and habitat condition, and 2) the performance of coho salmon recovery efforts. The CMP as described in this document is incorporated into the recovery strategy as the foundation for determining coho salmon status and monitoring of trends. Additional efforts to expand CMP are underway to develop implementation, effectiveness, and validation monitoring of restoration projects to address monitoring the effectiveness of restoring estuarine and freshwater habitat for coastal salmon and steelhead.

Need for Coastal Monitoring Program

Evaluation of species viability

The Viable Salmonid Population (VSP) concept (McElhany et al. 2000, see Boydstun and McDonald 2005, Appendix B) is a conceptual framework for use in assessing salmonid population viability, and by extension, ESU viability. The VSP framework identifies four key characteristics central to attaining and maintaining long-term population viability: abundance, productivity, spatial structure, and diversity. NMFS Technical Recovery Teams (TRTs) have provided localized and detailed strategies for establishing and meeting VSP criteria and the combinations of viable populations that will be necessary to achieve a viable ESU. The CMP is designed to collect data that will allow evaluation of ESU and population viability through assessing VSP parameters. It also provides methods to assess viability at varying spatial scales, from the ESU to the individual population level, and at smaller spatial scales such as individual watercourses.

Evaluation of the effect of ocean conditions on recovery

Work over the last two decades has demonstrated the effects of ocean conditions on salmonid abundance (Ware and Thomson 1991, Francis and Hare 1994, Loggerwell et al. 2003, Botsford et al. 2005, Mueter et al. 2007 and others). Salmonids experience wide variation in marine survival that results from cyclic and non-cyclic changes in ocean conditions. These wide changes in ocean survival can mask both species recovery and declines (Lawson 1993). Effective monitoring should provide an independent measure of ocean survival so that recovery can be accurately assessed. The CMP proposes long-term, intensive monitoring at fixed Life Cycle Monitoring (LCM) stations to evaluate the effects of changing ocean conditions on salmonid populations.

Evaluation of freshwater habitat conditions

Although many factors have contributed to the decline of Pacific salmon (NRC 1995, 1996), the primary cause of imperilment of ESA-listed species overall is degradation and loss of habitat (Wilcove et al. 1998, Gregory and Bisson 1997). An understanding of the relationships among salmonid production, population health, and freshwater habitat condition is essential to developing effective recovery strategies (Holtby and Scrivener 1989, Jones and Moore 1999), for evaluation of progress toward recovery, and to inform listing and delisting decisions under both the CESA and ESA (ESA, 1973, Sec. 4, and CCR Title 14, Sec. 670.1, respectively). Without an understanding of freshwater habitat conditions, meeting State and Federal delisting requirements cannot be accomplished. In addition, State recovery criteria specify that habitat protection and improvement objectives must be met (CDFG 2004).

Specific links between fish production and freshwater habitat condition are difficult to determine, and have not been well established (Smokorowski et al. 1998, Roni et al. 2002, Feist et al. 2003). Current thinking tends toward the view that population viability is more dependent on a complicated collection of spatial features and processes at the landscape level (Dunning et al. 1992, Bond and Lake 2003, Williams and Reeves 2003). Habitat monitoring is not included as part of this plan, but will be dealt with in a separate document.

Plan Goals and Objectives

The goals and objectives of the CMP are to develop broad and intensive monitoring strategies and techniques that:

1) Create a monitoring framework that includes all coho salmon, Chinook salmon and steelhead in coastal California;
2) Provide regional (ESU-level) and population abundance estimates for both status and trend of salmonid populations;
3) Estimate productivity trends from status abundance data;
4) Provide estimates of regional and population level spatial structure of coastal salmonids;
5) Consider the diversity of life history and ecological differences in the three species of interest; and
6) Create permanent LCM stations that will allow deeper evaluation of both freshwater and marine fish-habitat relationships and provide long-term index monitoring.

This document is intended to provide a concise technical description of the overall strategy, design, and methods of the CMP elements for the purpose of technical peer review. Larger, more detailed description of the CMP is presented by Boydstun and McDonald (2005) and the implementation strategy can be found in Shaffer (in prep.).

Finally, it should be pointed out that the CMP does not provide for collecting all salmonid information necessary for a comprehensive evaluation of fisheries, hatchery impacts, and habitat condition. Nor does the CMP contain implementation logistics necessary to execute field operations. These aspects of monitoring are in some cases already available (Johnson et al. 2007) or will be detailed in separate documents.

Plan Development Approach

The CMP is the result of a Salmon Restoration Grant to CDFG and NMFS. The first task of the grant was to hire knowledgeable individuals to write the CMP and develop the proposed statistical methods. The second task was to conduct two workshops to gain scientific consensus on the CMP goals and monitoring priorities by species and life history form. The first workshop provided the general outline for the CMP and the second workshop provided more specific recommendations for geographical areas and for habitat monitoring. The attendees were experts on salmon ecology, sampling, and fisheries and habitat management from NMFS, CDFG, other State and Federal agencies, and various academic institutions. The participants provided input on: 1) technical feasibility of implementing the recommendations in the field, and 2) technical suitability of the resulting data sets for assessing extinction risk under the CESA and the ESA. Scientists involved in developing State and Federal extinction risk criteria and policy standards, as well as in conducting State and Federal status assessments, were centrally involved in the workshop and CMP writing processes. The CMP development process is covered in more detail in the two companion documents by Boydstun and McDonald (2005) and Shaffer (in prep.).

Geographical Areas

The CMP divides California geographically into Northern and Southern monitoring areas due to differences in species composition, abundances, and habitat conditions. Adult abundance monitoring will be approached differently in the Northern and Southern areas so that similar types of sampling are grouped together for operational efficiency. The boundary between the

Northern and Southern monitoring areas is between Aptos Creek, Santa Cruz County and the Pajaro River, Monterey County. Creation of separate Northern and Southern monitoring areas will group areas that have similar monitoring conditions and needs. The sampling efficiencies will be to some extent financial, but to a larger extent, these efficiencies will be due to the ability to apply region-specific operational knowledge of sampling procedures and gear. Also, the Northern and Southern division follows CESA and ESA listing boundaries, CDFG Regional boundaries, and the southern boundary of the Environmental Protection Agency's (EPA) Marine West Coast Forest Ecoregion (EPA 2008). Of course, the differences in these operational efficiencies are not absolute and some amount of knowledge and gear will be transferable across both areas.

The three target species in the Northern Area (Chinook salmon, coho salmon and steelhead) complicates the monitoring design. Species-specific differences in distribution, age at maturity, run-timing, and other life history features will preclude selection of sampling locations that fulfill the monitoring requirements of all three species simultaneously. Chinook salmon from south of the Klamath River to the Russian River are ESA listed (Good et al. 2005). Coho salmon from the Oregon border to Aptos Creek, the end of their range, are both CESA and ESA listed. Steelhead are listed throughout the entire coastal area except the Klamath River Basin and north to the Oregon border. Because species-specific needs require some prioritization, sampling will have to be geographically weighted to focus on species with greater risk of extinction. Very sophisticated sample draws will be necessary to obtain the most appropriate distribution of monitoring effort. In contrast, in the Southern Area, only steelhead are present, greatly simplifying the sampling design and the elements of drawing an appropriate sample.

In the Northern Area, the available information suggests that standard adult surveys can be successful (Ricker 2005, Gallagher and Gallagher 2005, Gallagher et al. 2010a). However due to very low steelhead abundance in the Southern Area, monitoring there will need to be very different. Low and patchy steelhead abundance adds considerable difficulty and therefore expense to monitoring in the south. The two Southern Area steelhead DPSs, South-Central California Coastal and Southern California, have severely reduced abundance, although how severely reduced is unknown due to lack of data for almost all populations (Good et al. 2005). In addition to the need for monitoring low or even rare abundances, there are concerns about the timing of the adult runs. Steelhead in the Southern Area are thought to migrate into rivers associated with one or a few large hydrologic or storm events, but this is unproven. The question of whether steelhead migrate in large groups or spread throughout the season raises concerns on how they should be monitored. In addition, there are also concerns about spatial distribution once steelhead are in the watershed. If grouped in only a few locations, how would those locations be targeted for sampling? Due to these concerns about monitoring Southern Area steelhead, counting the entire population (i.e., a census), usually at a counting station at a passage point in the lower portion of these watersheds should be used rather than dispersed and randomized adult surveys (Boydstun and McDonald 2005).

Finally, the division between the Northern and Southern areas occurs at the major change in hydrological and ecologic conditions, corresponding to the transition from the Coast Range Ecoregion to the Southern and Central California Chaparral and Oak Woodlands Ecoregion (EPA 2008). Ecoregions denote areas of general similarity in ecosystems and in the type, quality, and quantity of environmental resources, and are designed to serve as a spatial framework for the research, assessment, management, and monitoring of ecosystems and ecosystem components. The Coast Range Ecoregion extends from the U.S-Canadian border to just south of Aptos Creek and is characterized by low mountains covered by highly productive, rain-drenched coniferous forests. Coastal redwood (*Sequoia sempervirens*) forests originally dominated the fog-shrouded coast, while a mosaic of western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and Douglas-fir (*Pseudotsuga menziesii*) blanketed inland areas. The primary distinguishing characteristic of the Southern and Central California Chaparral and Oak Woodlands Ecoregion is the Mediterranean climate of hot dry summers and cool moist winters, and associated vegetative cover comprised mainly of chaparral, oak woodlands, and annual grasslands. Salmonid distribution mirrors these changes in habitat and hydrology, creating a logical boundary for the two monitoring areas.

MONITORING PLAN ORGANIZATION

The CMP (Figure 2) is designed to provide information to assess viability of CESA- and ESA-listed salmonids relevant to the four VSP parameters: abundance, productivity, spatial structure, and diversity, and to monitor trends in freshwater and ocean survival. Management decisions are routinely made by both State and Federal agencies based on understanding of these concepts and there is pressing need for improved information on salmonid abundance and distribution to better inform decision-makers. The CMP will also allow recovery partners to monitor salmonid populations in a consistent manner and to provide the essential data for other management purposes. Sampling will occur in a spatially explicit and balanced way, with flexibility in the analyses of larger or smaller spatial groupings of the data, a critical first step for this type of monitoring. The biological information from the CMP will be regularly organized by Northern and Southern areas, ESUs, and individual populations; although analyses can also be conducted at other scales. The CMP also provides organizational structure so data flows efficiently, effectively, and in a timely manner from the collection phase to central databases for editing and analysis.

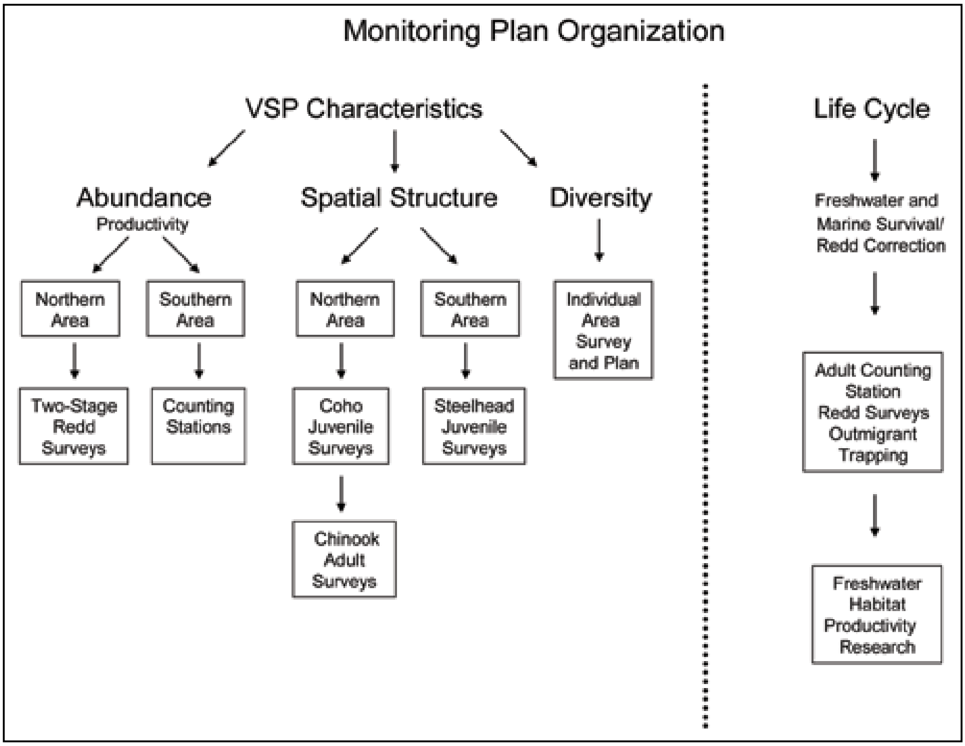

Figure 2. Overall California Coastal Salmonid Monitoring Plan organization based on VSP parameters and Life-Cycle Monitoring.

Adult abundance monitoring will be approached differently in the Northern and Southern areas due to differences in species composition, abundances, and habitat conditions. In the Northern Area, abundance monitoring is needed for Chinook salmon, coho salmon, and steelhead, and will occur through live fish, carcass, and redd surveys for Chinook salmon, and redd surveys for coho salmon and steelhead. These estimates will be expanded to compare to the various species and area goals in the State and Federal recovery plans. Random sampling of stream sample units will be spatially balanced (i.e., evenly distributed) because samples next to each other tend to be similar. Generalized Random Tessellation Stratified (GRTS) sampling is a commonly used method of selecting these types of spatially balanced random samples and is the best compromise between the need for randomization and the need for spatial balance. The proportion of hatchery fish will be estimated over various spatial scales, but these estimates will be contingent on the establishment of a consistent coast wide hatchery marking protocol and programs to conduct the marking, retrieval, and data analysis. In the Southern Area, steelhead is the only salmonid present, and monitoring is complicated because southern steelhead may spawn over an extended period, from January to June (Busby et al. 1996), but may enter rivers and streams in discrete pulses associated with freshets. This protracted spawning period with sporadic entry, coupled with the low abundance and highly patchy distribution once steelhead enter a watershed, pose extreme problems for accurately estimating population size. Therefore, population censuses taken at fish counting facilities are more likely to produce reliable estimates of abundance than adult surveys (Boydstun and McDonald 2005).

Spatial structure monitoring provides data to assess the extent to which populations can maintain connectivity to each other and whether the species distribution is expanding or contracting. Populations will be monitored by juvenile snorkel surveys that are spatially explicit and balanced in the same way as the Northern Area abundance surveys. Snorkel surveys will be adjusted to account for observer efficiency using a revisit technique, allowing the assessment of the data at flexible spatial scales from ESU to population. This approach of targeting juvenile fish in the summer or early fall is operationally simpler and less expensive than adult sampling, allowing for more sample reaches to be surveyed at a lower cost. In the Northern Area, juvenile surveys will be conducted only in sample units identified with coho salmon. Steelhead are widespread in the Northern Area. Chinook salmon spawn in only a few well-defined areas and outmigrate in the spring before the juvenile surveys take place. Some information on their spatial structure will come from adult surveys. In the Southern Area, juvenile surveys will be conducted for steelhead (and resident rainbow trout), since spatial structure is a particularly important characteristic for small populations.

Diversity monitoring is unique to each ESU, population, or individual area, since diversity characteristics are a response, in part, to the habitats where they occur and are strongly adaptive to those conditions. Primary diversity strata (e.g., ESUs) are defined genetically and so extensive genetic baseline surveys are necessary to determine whether subunit genetic diversity strata exist. In addition, each geographical unit requires a survey of phenotypic and other diversity characteristics and a plan for monitoring these characteristics.

Freshwater and ocean survival will be monitored using LCM stations, to assess whether population trends reflect changes in freshwater productivity or a response to changing ocean conditions. Each would have three essential components: an upstream adult counting station, adult surveys above the station, and outmigrant smolt trapping. LCM stations will provide an absolute measure of adult abundance, a survey estimate of adult abundance, and an estimate of outmigrating smolts. The adult counting station and outmigrant smolt counts will provide measures of "fish in and fish out" that will be used to evaluate freshwater and marine survival (Prager et al. 1999). The data from counting stations and adult survey estimates will be used to develop a correction factor between redd counts and adult numbers to calibrate adult surveys conducted in other watersheds. It is expected the LCM stations will be magnets for other kinds of recovery-oriented research, particularly fish habitat-productivity relationships and habitat restoration effectiveness.

SAMPLE DESIGN

The spatial extent of salmonid populations and certainly ESUs are too large to be measured completely. Therefore the biological and physical attributes of a population have to be inferred from a sample of measurements from the population of interest, the basic tenet of statistical inference. The plan develops a coast-wide sampling design based on the random selection of sample stream segments from a sample frame consisting of all possible reaches within a population of interest. This design allows for statistical inference to be made about the entire area within the sample frame, and the uncertainty of these inferences to be evaluated on the basis of probability theory. The unit utilized in the design is the collection of stream segments, and the attributes of this population are species abundance and distribution. This design is the structure within which species abundance and distribution monitoring occurs. Due to the biogeographic differences between the two areas explained earlier in this document (See Geographic Areas section), abundance of adult steelhead in key populations within the Southern Area will be censused, without the use of design-based abundance estimation, but design-based sampling will be used to estimate juvenile steelhead abundance.

Sample Frame Development

Field data cannot possibly be collected from all portions of all streams in either of the monitoring areas; consequently, a properly constructed and ordered sample frame is essential to the overall success of the CMP. As used here, the term sample frame refers to a list of all possible sample units that could potentially be selected as data collection sites in an area of interest. Sample units comprise the sample frame and are the basic stream entities over which sample measurements are made (e.g., approximately 1.6 – 3.2 km stream segment). The area of interest covered by the sample frame is dictated by overall project goals. Sample frames will be constructed with the goal of ensuring all units listed potentially contain one or more fish species of interest. Sample frame construction will target inclusion of units below impassable barriers that have been identified as potential habitat (Agrawal et al. 2005) and that do not have an obvious reason to exclude them. As of 2010, only work on the Northern Area sample frame has been started, but methods for the Southern Area sample frame are identical.

Despite careful thought during construction, some units in the sample frame may be excluded following the sample draw. Post-draw exclusion may be due to 1) failure to meet target population definitions after on-the-ground examination (e.g., inaccurate barrier or gradient measurements that render actual reaches unsuitable for salmonids) or 2) logistical inability to access the site (e.g., inability to secure landowner permission). These are distinct in that the former do not affect inference but the latter do, representing a non-response error. Removal of units completely at random from the post-sample draw reduces analysis efficiency slightly, but otherwise does not cause any ill effects on inferences or the ability of the study to meet monitoring objectives. However, removal of units due to logistical inability to gain landowner permission may introduce bias into the estimates. The segments that are excluded need to be examined carefully at the end each sampling year to determine their effects on data analyses.

Sample units proposed here are lengths of stream segments that will be sampled using the appropriate protocol. For both adult abundance and juvenile distribution sampling, the sample units will be stream segments of 1.6 - 3.2 km (1 - 2 mi) in length, chosen because it is the average distance that adult abundance monitoring crews can survey in one day. Crews measuring juvenile distribution will only be able to sample every other pool. Sample units are allowed to vary in length so that unit boundaries can be defined by easily observed landmarks in the watershed (e.g., bridges, confluences, cliffs, etc.). With boundaries at easily discerned landmarks, field crews readily know where to start and stop for the day's sampling.

Construction of the sample frame starts with all possible sample units; that is all possible 1.6 – 3.2 km stream segments. This will be done for both the Northern and Southern areas. The exact process has not been finalized, but will generally follow these methods. The sampling frame will be drawn from an Arcview GIS layer of the 1:24,000 routed hydrography containing the latitude-longitude (LLID) identifiers, stream and tributary names, and stream lengths. These stream lengths will then be reduced to include only those reaches below the upstream limits of anadromous habitat using historical occurrence and Intrinsic Potential salmonid habitat modeling (Agrawal et al. 2005; McCanne and Brown 2005). Both the Southern Oregon-Northern California Coast (SONCC) and North-Central California Coast (NCCC) TRTs used Intrinsic Potential habitat modeling for coho salmon and steelhead and current maps exist for all areas to be monitored. The sample frame is then reduced further by removing stream segments above barriers and areas known not to be used by fish (e.g., downstream extent of spawning) or areas that are unavailable for sampling for other reasons (e.g., access). Next, expert opinion will be used to define the up- and downstream limits of salmon spawning and rearing habitat including all tributaries in each stream. Consensus (Delphi-like) techniques will be used to rectify situations with differing expert opinions. Once established, each reach will be associated with the species and life stage assumed to be present. The reasons for all exclusions, especially those due to non-biological factors, will be documented.

Following initial sample frame construction, sampling units will be ordered using a one-dimensional method that starts by ordering units based on the geographic location of the watershed and the unit's location within that watershed. All watersheds in the CMP area will first be ordered from north to south along the coast. Sample units within each watershed will then be ordered starting at the lowest sampling unit in the drainage and moving upstream. All units in the main-stem of the system will appear first in the list, followed by units in tributaries. Tributaries will be ordered based on the stream distance of their confluence with the mainstem. That is, units in lower (farthest downstream) tributaries will appear before units in upper tributaries. In this way, ordering of the frame will continue from main-stem to tributaries until all units are placed in the frame. The location of a unit in the ordered frame defines its "spatial" location, where "space" here is a one-dimensional measure that generally represents river kilometer from the ocean, except that larger (main-stem) streams are inherently closer to the ocean in this "space" than tributaries. An example of a sample frame from the Mendocino Coast is shown in Figure 3.

The proposed frame ordering and its induced measure of "space" in the river system more closely mimics what salmonids experience than two-dimensional Euclidian space wherein distances are measured as straight lines between sample units. The difference between "spatial" locations of two units in the sample frame reflects both differences in distance from the ocean and stream order. This ordering, coupled with the sample drawing mechanism (McDonald 2003), ensures that sampled units will be spread out in this one-dimensional "space", and will represent all areas of the plan in proportion to size (number of river kilometers).

Completion of the sample frame for coastal California is currently underway. The sampling frame starts with stream segments predicted to be historical habitat by modeling (Agrawal et al. 2005). Stream segments are equivalent to sampling units as defined above. The historical predicted habitat will be further delimited using existing habitat data sets and local scientific expert opinion. The Eel River Basin, portions of the Eureka Plain in Humboldt County, Mendocino County, San Mateo County, and Santa Cruz County

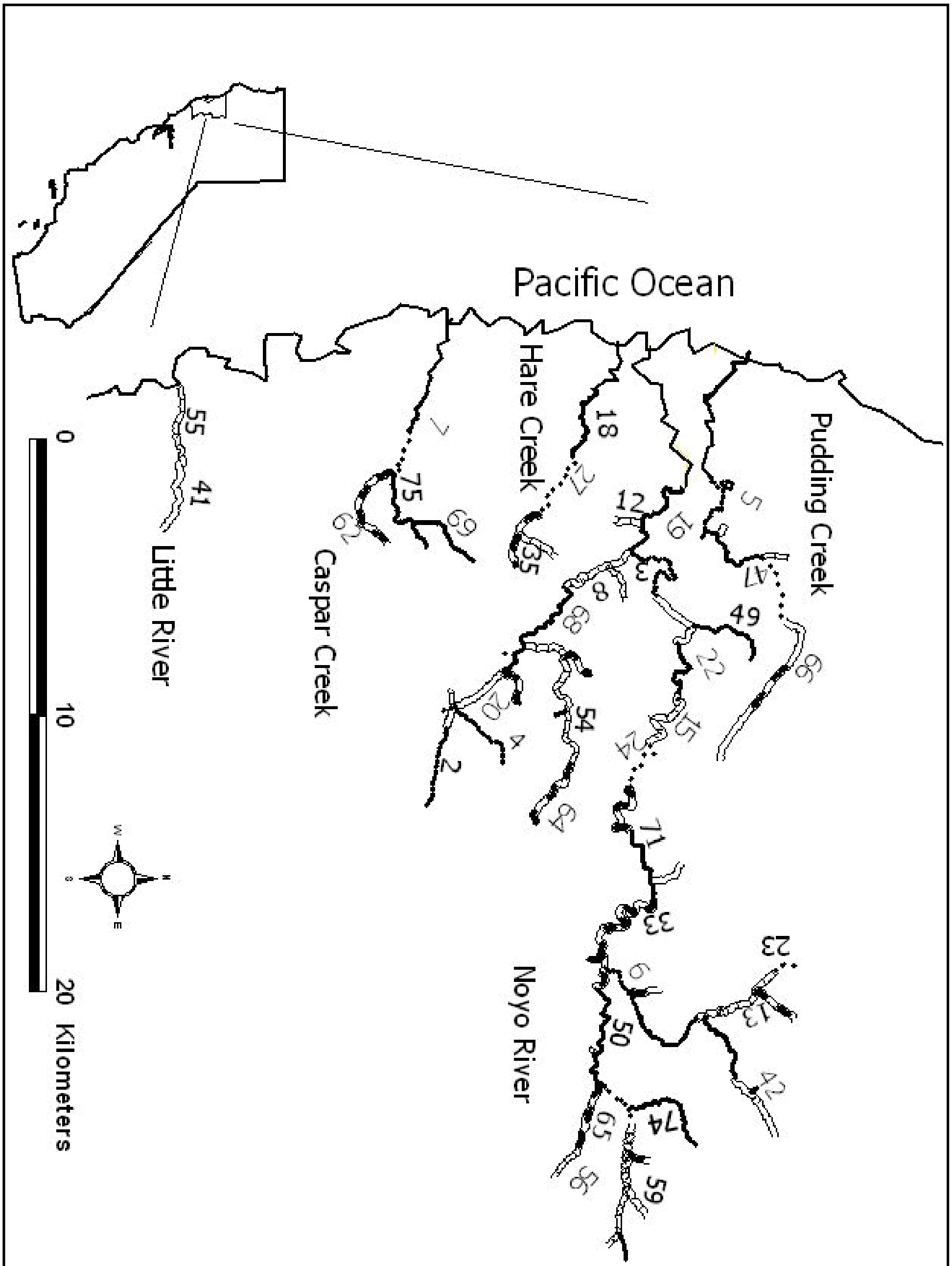

Figure 3. Mendocino Coast sampling frame example (McCanne, D. and A. Brown. 2005).

sampling frames have been completed. Sampling frames for other areas are in progress or have not been started (ibid, Figure 4).

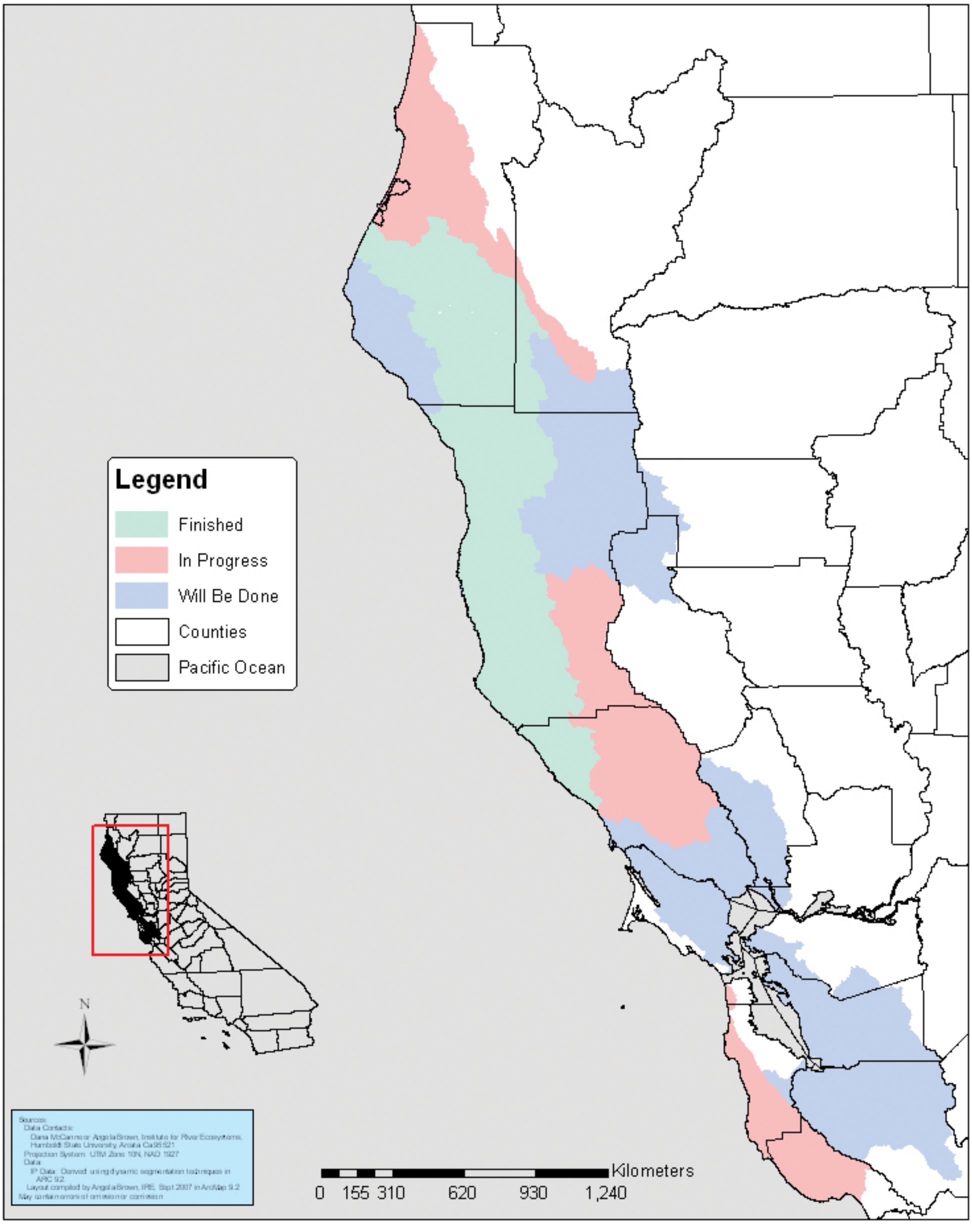

Figure 4. Status of small scale (1:24,000 routed hydrograph) sample frame development in the Northern Area.

An important refinement of the sampling frame will be dealing with species-specific estimates (Boydstun and McDonald 2005). Habitat modeling (Agrawal et al. 2005) suggests that adult steelhead occur in most of the stream segments in the Northern Area, but adult coho salmon only occur in about one half of the stream segments; half of the segments were estimated to contain steelhead only and half the segments were estimated to contain steelhead and coho salmon. The number of segments estimated to contain Chinook salmon is much smaller. If the Chinook salmon and coho salmon species estimates are derived from a sample selected without regard to hypothesized species composition, the Chinook and coho salmon estimates will have higher variances because they do not occur in many segments, increasing the number of zero counts.

One method to reduce variance in the adult estimates would be to attribute each segment in the frame with the species hypothesized to be present (i.e., Chinook salmon, coho salmon, steelhead) and draw a so-called "soft-stratified" sample (Larson et al. 2008). Assuming *n*-coho salmon segments are required from waters likely to contain, or likely historically did contain coho salmon, "soft-stratification" skips segments in the sample without the "coho" attribute until *n*-coho salmon segments with the "coho" attribute are obtained. Separate samples for each species result in separate samples from de facto populations of units containing each species. Alternatively, three separate samples could be drawn, one for each species. The primary advantage of "soft-stratification" over drawing three separate samples from three separate species-specific frames is that "soft-stratification" assures as much co-location of field sampling efforts as possible. Separate samples from separate species-specific soft stratification would reduce variance due to the elimination of zero counts from segments that never contained those species. However, costs, staffing, field time, and complexity of sampling would be greater using this strategy and would need to be organized and implemented carefully.

Juvenile habitat for coho salmon and steelhead is larger than adult spawning habitat due to dispersal of juveniles seeking food and space during their stream residency period (one year for coho salmon and, generally, two or three years for steelhead). Spatially balanced sampling is not proposed for Chinook salmon juveniles because of their short stream residency period and typical ocean-oriented downstream migration pattern. As described above, species distribution assessment for Chinook salmon will be based on adult spawning distribution sampling, although rotary-screw trapping could be used to assess timing and magnitude of Chinook salmon emigrations.

A similar "soft-stratification" scheme will be needed for the spatial structure juvenile surveys in the Northern Area. A "juveniles" attribute will be attached to all segments in the frame and will be invoked when drawing the juvenile survey samples. Using this attribute to delimit sampling will allow elimination of large areas uninhabited by juveniles, which should result in more accurate averages and reduced variances. High accuracy of the "soft-stratification" segment attributes assignments (i.e., Chinook salmon, coho salmon, and steelhead) will improve precision. In the Southern Area, there is no need for the species-based "soft-stratification" since only steelhead occur there.

Initially, a sampling frame workgroup will be formed to assign the segment attributes using available data and expert opinion. Since more information will arise as the sampling progresses, the workgroup will update these attributes annually or as better data become available. This is an important process and careful attention to detail and documentation is necessary to ensure that additions and removals of sample units are defensible. This process may influence estimates, but is unlikely to have a large effect as only a few smaller stream segments are expected to be changed out of a large area of known stream habitat. In general, it is expected that the sampling frame will be refined for several years after sampling has been initiated.

Utility of Generalized Random Tessellation Stratified (GRTS) Sample Selection

Generalized Random Tessellation Stratified (GRTS) is a compromise between systematic and simple random sampling that resolves problems with sampling patchy distributions and has several significant advantages useful for salmonid surveys. Because the GRTS procedure orders samples so that any consecutively numbered sample set is a randomly chosen, spatially-balanced sample, it has the ability to substitute consecutive samples when needed. This offers several major advantages. First, if any segment is unusable, the next segment in the GRTS draw can be substituted and the sample design will remain spatially balanced and randomized. Second, the GRTS sample can be decomposed to subregion sample sets that are still spatially balanced and randomized. Finally, if there is additional interest in a particular subregion, say a watershed, an additional number of successive samples from that subregion can be added to the survey and all of the samples can be used in the subregion estimates. These characteristics are of vast utility in regional-scale salmonid surveys.

Perhaps the most valuable characteristic of the GRTS sample selection scheme is the ease with which unusable samples can be replaced and yet still maintain a randomized, spatially balanced sample design. The number of unusable samples is expected to be large, particularly in the early years of monitoring. There are several reasons why samples will be unusable, including landowners denying access to the site, the samples being above or below the limits of anadromy, dewatered reaches, sampler safety or health issues, or difficult access. Using GRTS, the next sample in the sample draw can simply be added to replace the unusable sample. GRTS sampling contrasts with systematic sampling, the other commonly used sampling scheme that assures spatial balance, in which it is difficult to replace unusable samples and still maintain spatial balance. For example, Oregon coho salmon surveys had 12% unusable samples between 1998-2006 (Jepsen and Leader 2007), even though coho salmon surveys have been conducted in these watersheds for over 50 years and many of these unusable samples had already been identified. A pilot effort in the Mendocino Coast had 22% unusable samples, primarily from denied access.

The second advantage of GRTS is that estimates can be made for subunits ("domain estimation," Lohr 1999) that remain spatially balanced and randomized. This means that population level estimates can be made from larger area estimates and additional samples can easily be added to the population estimate if greater precision is needed. Estimates of status and trend will frequently be needed for smaller parts of the study area for a variety of purposes, primarily recovery planning. Both CESA and ESA recovery plans are based on recovering a targeted set of designated populations within an ESU. These targeted populations are not currently identified, but their population abundances will be a critical part of recovery. Another need of domain estimation will be for Hatchery and Genetic Management Plans (HGMP). HGMPs will require monitoring survival and mingling of natural and hatchery fish at a number of spatial scales. Other needs for domain estimation include evaluating timber practices, habitat restoration activities, and evaluation of the effects of flow regimes.

A third advantage of using the GRTS sample selection scheme is flexibility in augmented sampling for domain or population estimates (Stevens 2002). Often, greater precision for particular domain estimates is necessary. To accommodate these additional needs, additional samples can be added as necessary to the existing sampling in the domain of interest. The need for domain estimates is expected to be common, since there will always be concerns about hatchery impacts, habitat restoration actions, and local watershed interests. The ability to include both the large scale samples and the additional domain samples in one estimate will greatly reduce the cost.

Stevens and Olson (2004) discuss the theory and details of spatially balanced sampling, and a detailed example of GRTS specific to CMP is described in

Boydstun and McDonald (2005, Appendix H). A GRTS sampling scheme is based on the concept of selecting a probability sample from a sampling frame arranged in a linear fashion. To do this, place all the stream segments in the sample frame on a linear line (see Sample Frame Development, above). Then create hierarchical addressing by splitting the sampling universe into quadrants and number the quadrants. Repeat this step by dividing the quadrants into subquadrants and number those until down to a single sample. This creates hierarchical addressing with the first digit being the first quadrant, the second digit being the first subquadrant and so on. Then randomize the hierarchical addresses and construct the sampling line. Select a systematic sample with a random start from sampling line and place the samples in reverse hierarchical address order. This procedure creates sampling schemes that emphasize spatial balance along with substantial flexibility to replace samples. Software is available to simplify this complex procedure (McDonald 2003, Kincaid and Olson 2009)

ADULT MONITORING

Northern Area

Goal and Methods

In the Northern Area, adult abundance monitoring will be used to measure progress toward adult abundance viability goals set in recovery plans (Spence et al. 2008, Williams et al. 2008). A time series of adult abundance estimates, adjusted for harvest mortality when appropriate, can then be used to estimate productivity. Abundance goals vary by species and area in the State and Federal recovery plans. The adult monitoring in the Northern Area will estimate coho and Chinook salmon and steelhead abundance from the Oregon-California border to Aptos Creek (Santa Cruz County) (See CDFG 2004 and Good et al. 2005). The Recovery Strategy for California Coho Salmon (CDFG 2004) has targets for downlisting Central California Coho Salmon from Endangered to Threatened status ranging from 1,350 spawning adults for the San Mateo County to 15,000 for the Mendocino Coast. State delisting targets will be determined in the future. For the Federal Southern Oregon-Northern California (SONC) and North-Central California Coast (NCCC) recovery domains, the low-risk coho salmon targets based on estimated habitat potential range from 1,400 for Little River (Humboldt County) to 18,000 for the Upper Rogue River (Williams et al. 2008). The Federal low-risk steelhead targets range from 600 for Casper Creek (Mendocino County) to 23,600 for the South Fork of the Eel River and low-risk Chinook salmon population targets range from 700 natural spawners for Little River (Humboldt County) to 11,900 for the Lower Eel River.

In the Northern Area, adult monitoring will consist of dispersed redd surveys (see Gallagher and Gallagher 2005, Gallagher et al. 2010b), augmented with adult to redd ratios estimated from LCM stations. The augmented redd surveys will be conducted over the appropriate areas in a probabilistic, spatially-dispersed fashion. The design will allow for increasing sample size in areas where there is interest in more precise estimates. Initially, live fish and carcass counts will also be recorded to insure that redd surveys are the most efficient estimation method since there is little cost associated with this extra data collection.

Adult Surveys

Differences in run timing among species and locations will require operational differences in the timing of the adult spawning surveys. Coho salmon and steelhead will be monitored throughout the entire Northern Area. Chinook salmon will be monitored in selected watersheds from Redwood Creek to the Russian River. The different species will require different beginning and ending survey dates with Chinook salmon being the earliest and steelhead the latest. Even within a species, there are major run-timing differences depending on latitude, distance from the ocean, and whether the river mouth bars over with sand. Northern California fall-run Chinook salmon enter larger rivers in August and September and spawn in late October and early November (Myers et al. 1998). Populations in smaller coastal watersheds may enter somewhat later. Chinook salmon surveys would need to begin in October but the ending date is less important since the surveys will need to extend beyond these dates for coho salmon and steelhead. Coho salmon run timing also varies by latitude. In California, coho salmon runs generally extend from September to February, with peak spawning in November to January (Weitkamp et al. 1995). Coho salmon surveys may need to start as early as October and run through February, with local adjustments. Steelhead run timing is even more variable and extended than Chinook or coho salmon. Steelhead in California typically spawn from December through April or even May, with peak spawning in January, February, and March (Busby et al. 1996). Steelhead surveys would need to be conducted the entire period from December through April, with some local adjustment.

Adult abundance estimates will be made from walking surveys that will record live fish, carcasses, and redds. Chinook salmon will be monitored using combinations of the three. Redd counts have been shown to be better estimates of coho salmon and steelhead in Mendocino watersheds (Gallagher et al. 2010a). Redd counts converted to adult numbers of fish using adult to redd ratios were similar to live fish capture-recapture estimates, but were operationally similar, cheaper, and less invasive to ESA listed fish (Gallagher et al. 2010a). Adult redd surveys can be conducted over widely distributed areas for ESU coverage and over smaller local areas (i.e. areas impacted by local watershed projects or hatcheries) with specialized needs for higher precision. Counting live fish and carcasses require no more effort and can be used as rough quality control measures for redd-based abundance estimates. Adult redd surveys will be conducted at sites from the GRTS sample draws made for multiple spatial scales. Again, the advantage of the GRTS sampling scheme is that it is flexible enough to draw samples for both of these purposes. Adult to redd conversions will be based on data obtained at the LCM stations (see LCM section below). Preliminary results from Mendocino County (Gallagher et al. 2010a) have found no differences in adult to redd conversions over a regional area.

At very high abundance, redds become difficult to count. Lestelle and Weller (2002) found that redd counts were better at low spawner abundance, but that area under the curve (AUC) escapement estimates were more reliable than redd count estimates at high spawner abundance. Our experience in Mendocino coast streams has found no superimposition, and hence, little difficulty in counting redds. This would be expected where fish are ESA listed for low abundance. Training and marking of redds can help reduce this potential difficulty. A study to measure observer error during coho salmon and steelhead redd surveys should be initiated (see Future Directions and Plan Refinement section). Previous studies (Durham et al. 2001, Muhlfled et al. 2006) found this to be insignificant when conducted on bull trout.

Abundance estimates can then be calculated for ESUs, for individual populations, and for even smaller units with management needs. As mentioned above, the advantage of the GRTS sampling scheme is that if higher precision is desired for a subunit (population or even smaller), then additional samples can be drawn for a subunit and all the samples can be used in the subunit estimators. Increasing sample size, and therefore precision, cannot be done efficiently using systematic sampling. For spawner surveys, sample sizes of 10% per year of the total sample universe for a given species are recommended based on precision levels estimated in Oregon for adult coho salmon spawners (Jacobs 2002). Recent work on the Mendocino Coast (Gallagher et al. 2010b) found that escapement estimated from sample sizes of 10% to 35% overlapped each other, and variation in the 95% confidence limits did not change after 15%. Censuses of every population within the ESU, or even intensive sampling, are not possible because of cost; thus a lower intensity probabilistic sampling with a precision of ± 30% may be acceptable.

Rotating Panels

Dividing sample units into rotating panels balances the dual goals of status estimation and trend detection. These goals conflict because sampling randomly drawn previously unvisited sites improves status estimates, while repeated sampling of the same sites improves trend detection. Rotating panel designs therefore provide the best compromise for achieving both goals. Sample units selected by the GRTS sampling scheme will be allocated to four panels that are in turn assigned different visitation schedules. The four different visitation schedules for panels are as follows: one panel that will be visited every year (Panel 1), three panels that will be visited once every three years (Panels 2 through 4), 12 panels that will be visited once every 12 years (Panels 5 through 16), and 30 panels that will be visited once every 30 years (Panels 17 through 46), the entire life of the project (Figure 5). Each panel will contain multiple sample units. The panel sampled every year is proposed to contain ~40% of the total number of reaches visited every year. The panels sampled every 3, 12, and 30 years are each proposed to contain ~20% of the total annual number of sampled reaches. In this way, one year of sampling will have both randomization for status estimation and retain consistency for trend detection. In the future, there will undoubtedly be some need for frame refreshment to account for sample change and attrition. This will be most serious if samples from every panel need to be replaced.

The one-year, three-year, twelve-year, and thirty-year rotational visitation scheme proposed here is slightly different than the rotation scheme used by the Oregon coho salmon monitoring plan. The Oregon Plan uses a 1, 3 and 9 year rotational visitation scheme with equal numbers of reaches in each panel. The Oregon Plan rotation scheme is based on the 3-year life history cycle of coho salmon. A series of visitation cycles (1, 3, 12, and 30-year) are proposed, based on the life histories of coho salmon, and also Chinook salmon and steelhead, both of which mature predominately at ages 3 or 4. We also propose to re-sample a higher proportion of sites every year given the importance of detecting population trends to the CMP. This scheme will need to be revisited iteratively to confirm the best allocation of sampling effort.

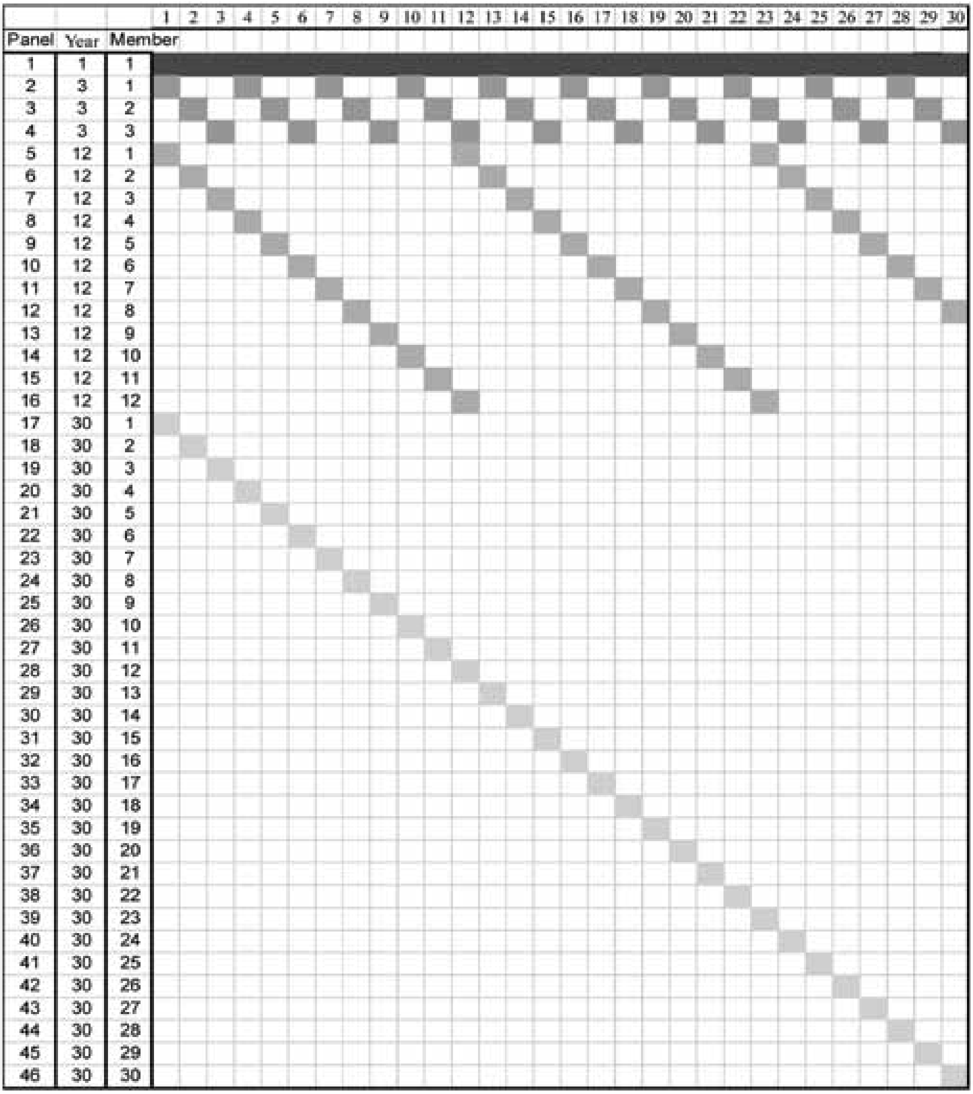

Figure 5. Rotating panel design for the California Salmonid Monitoring Plan by sampling year with 1, 3, 12, and 30 year panels with individual panel member rotation shown by the shading.

Adult spawning surveys have been the primary tool for assessing the status and trend of naturally reproducing salmonid populations since at least the 1930's (Ricker 1958), but have been conducted in different ways. This is particularly true for surveys targeting the three species of interest. Chinook salmon escapement is indexed using redd counts in Washington (Crawford and Volkhardt 2004) and peak counts of live fish and carcasses in Oregon (Jacobs et al. 2002). In the Klamath River in California, Chinook salmon escapement is indexed using a variety of methods including redd counts and peak counts (PMFC 2007).

Coho salmon survey escapement in Washington is indexed using redd counts, although live fish counts are also used in Puget Sound (Crawford and Volkhardt 2004). Oregon coho salmon escapement has a long history of using live fish counts (Jacobs et al. 2002). In California, preliminary results from coho salmon surveys (Gallagher and Gallagher 2005, Gallagher et al. 2010a) suggest that redd surveys provide estimates with higher precision than live fish counts and have greater consistency across streams at a lower cost.

Steelhead escapement is monitored using redd counts in Washington (Crawford and Volkhardt 2004) and Oregon (ODFW 2009). Gallagher et al. (2010a) found that steelhead redd counts were positively correlated with trap escapement counts and suggest that they should be considered as reliable indicators for steelhead in California.

Detailed redd survey methods are described in Gallagher and Gallagher (2005); Crawford et al. (2007); Gallagher et al. (2007); Gallagher et al. (2010a); and ODFW (2009). Selected sample units will be surveyed biweekly throughout the season. Two-person crews will walk or kayak the sample unit, searching for redds and noting live fish, carcasses, stream flow and visibility. All redds will be uniquely marked with flags to avoid double counting. Redds will be identified as to type and measured per Gallagher et al. (2007). Live fish and carcasses will be identified, tallied, sexed, and measured, and carcasses will be marked with tags. Obtaining this information from steelhead will be difficult due to unreliability of estimation on live fish and lack of carcasses, so information may be taken from LCM stations. Additional sampling may include otolith or scales for aging or microchemical analyses and tissue samples for genetic analyses.

Estimation of Hatchery and Natural Fish

In several coastal watersheds, hatcheries release salmonids in an attempt to supplement natural production. The returning adults typically return to the stream of origin, but some stray to neighboring streams. It is important to know how many returning fish are of hatchery origin and how many result from natural spawning. For recovery plans, the fraction of hatchery fish in a population needs to be below a specific proportion for the population to be considered viable (Spence et al. 2008, Williams et al. 2008). The proportion of hatchery fish must be less than 5% of the total population to avoid a significant negative effect (Good et. al. 2005).

For salmon, a fraction of each hatchery salmon brood-year is marked with a Coded Wire Tag (CWT) and the adipose fin is removed. The adipose clip is easily observed during adult surveys. When a salmon carcass with an adipose clip is observed, the head of the salmon will be removed for later CWT processing. Live salmon with adipose clips will also be noted. Unfortunately, several of the hatcheries have not yet begun implementing a constant fractional marking (CFM) program that would allow estimation of hatchery and natural proportions. Estimation of the fraction of hatchery fish will not be possible without coast-wide CFM programs. All hatchery steelhead receive an adipose clip, but do not receive a CWT. Therefore the hatchery fraction can be estimated directly from the combined live fish and carcass counts. Live or dead steelhead with an adipose clip observed by survey crews will also be noted.

Several procedures are available for estimating hatchery and natural proportions in watersheds where hatcheries are conducting CFM (Newman et al. 2004). The method proposed here assumes a constant fraction of hatchery fish have been marked for an extended period (e.g., three to four years) prior to the survey so that the same fraction of all age classes currently spawning are marked. This process essentially treats hatchery-marked fish as a separate species for estimation purposes, and applies the species-specific estimation methods outlined below to arrive at an estimate of total number of hatchery marked fish in a segment or system. This estimate of total number of hatchery marked fish will then be expanded by the constant proportion of hatchery fish that were marked prior to the survey. For example, redd counts will be converted into number of fish with hatchery marks in a particular segment using the methods outlined in the next Section. This estimate of fish with hatchery marks will then be divided by the proportion of marked fish. The estimated number of unmarked hatchery fish in a segment will have to be subtracted from the estimated number of non-hatchery fish. Total number of fish with hatchery marks in a larger system will be estimated using either the simple or regression estimators listed below, then divided by the proportion of marked fish. Because the proportion of marked fish is constant and assumed to be known, all variance estimates in the next section can simply be multiplied by the square of the proportion of marked fish.

Estimates of hatchery fish may be biased low due to the use of redd surveys as the principal sampling method because some marked fish will have been washed downstream after they completed redds, but before they have been counted. The question is whether marked or unmarked fish are washed down at different rates. Although this may not prove to be a concern, the question of bias associated with this problem can be evaluated if a LCM station is nearby. The number of marked fish found at the counting stations can be compared with the number of marked fish found in the surveys to assess whether bias associated with washed-down marked fish exists.

In watersheds where hatcheries are not yet conducting constant fractional marking, survey crews will count the hatchery marked fish encountered during sampling. Marked hatchery fish counts summarized over time and stream segments will provide a rough assessment of the proportion of hatchery and natural fish.

Estimation of Abundance

Redd survey estimates of the number of spawners in surveyed reaches will be expanded to regional or ESU scales. The use of reach specific redd surveys to estimate spawner abundance is conceptually simple and abundance estimates obtained are comparable to estimates obtained using other survey methods. These methods are flexible enough to allow abundance estimation over large regions or variously sized subunits within larger regions. Expansion of abundance estimates contains two steps: 1) estimation of the numbers of adults for each sample unit from redd surveys, and 2) scaling those reach-specific estimates to larger-scale abundance estimates. Redd-survey based estimation of number of spawners in a particular sample unit follows Gallagher and Gallagher (2005) and Gallagher et al. (2007, 2010b). Reducing over- and under-counting errors in redd counts (bias corrected) was accomplished using techniques outlined in Gallagher and Gallagher (2005). Methods to estimate larger-scale abundance are taken directly from Appendix H of Boydstun and McDonald (2005), and are included here for completeness. We assume that individual stream sample segments were selected in accordance with the proposed sampling design, and that unbiased estimates of the number of fish per sampled unit were obtained using calibrated redd counts. This method also assumes that the basins (or sub-basins) and time periods where expansion factors are established are representative of the sample frame in general. There is a significant relationship between escapements and redd counts in California coastal streams where these data are available (Gallagher et. al. 2010a). The large-scale estimators can be further applied to any unbiased estimate of a quantity associated with an individual segment, such as number of carcasses or live fish, fry to parr ratio, and habitat parameters like percent cover, temperature, large woody debris, etc.

Even though the redd count method had the best relationship to escapement of the survey methods considered in Gallagher et al. (2010a), they still have biases associated with their use to estimate abundance. In particular, redd detection probabilities may vary considerably depending on viewing conditions. Also, over the full temporal span of a spawning season, a "population" of redds must be treated as an "open" population with new recruits (i.e., new redds being constructed as fish enter the stream and spawn over the survey period) and mortalities (older redds becoming obscured (lost) due to gravel substrate migration during periodic high flow events) even with appropriate monitoring protocols.

Numbers of Adults per Sample Unit from Redds

Estimated number of adult fish in a particular sample unit will be computed by first classifying redds to species, then applying a species-specific fish to redd ratio. When the species that built a redd is unknown, estimated logistic regression equations computed from known Chinook salmon, coho salmon and steelhead redds will be used to attribute redds to species. This method follows Gallagher and Gallagher (2005) who developed a series of logistic equations for Mendocino County that were used to classify total redds into redds by species using day of year, redd area, and redd substrate data. The apparent error rate of redd misidentification of all species was 3.9% from a set of redd data where species was known, which compares favorably to field classification uncertainty ranging from 11% to 22%. The apparent error rate for discrimination of Chinook salmon and coho salmon was higher (5.9%), but this was probably due to a very low number of known Chinook salmon redds. The rate of redd misidentification was 6.8% when these equations were used to classify an independent set of steelhead redds. Gough (2010) developed logistic regressions to classify Chinook and coho salmon redds using just day number, and then day numbers, redd area, and redd substrate from Prairie Creek, Humboldt County. For unmeasured redds using only day number, Chinook salmon and coho salmon were classified correctly at a rate of 93.3% using a known redd data set. Adding measured redds and redd substrate to the regressions lead to a 97.7% correct identification rate. The two sets of regressions have different forms and the results were not completely compatible. The CMP will use the Gallagher and Gallagher (2005) regression to separate coho salmon and steelhead redds, and where necessary, use Gough (2010), to separate coho and Chinook salmon redds

Spawner to redd ratios are an active area of research. For Oregon steelhead, Susac (2005) suggest using a female to redd ratio of 1.04, but found ratios that ranged from 0.5 to 4.45. Washington assumes 2.5 fish for each redd for Chinook salmon, coho salmon and steelhead (Crawford et al. 2007). California does not have a standardized approach to spawner to redd ratios. Klamath-Trinity Basin Chinook salmon redd surveys use a ratio of two fish per redd (PFMC 2007). In some of the California Central Valley (CV) streams, Chinook salmon redd surveys are used as to assess escapement, as an index variable or to map distribution (Low 2007). On Mill Creek (CV) Chinook salmon redd surveys assume a female to redd ratio of 1:1 and a female to male ratio also of 1:1, for an overall redd to adult ratio of 2:1.

Another approach is to use a ratio of females per redd based on redd area (Gallagher and Gallagher 2005). This redd area method assumes that the number of females to redds is related to size of the redd. Coho salmon and steelhead redd sizes are scaled so that smaller redds represent fewer females. Female coho and female steelhead estimates from redd size measurements are then multiplied by the observed male-to-female ratios. Gough (2010) estimated population sizes for both Chinook and coho salmon using the one female per redd method and the redd area method. The estimates from the redd area method were consistently lower than estimates under the assumption of one female per redd.

Gallagher et al. (2010a) converted redds to fish using a three stream annual average of capture-recapture adult estimates divided by the number of redds. This method is conceptually simpler and estimates are similar to other methods. Redd to fish ratios will be calculated in different locales at LCM stations.

Finally, it should be noted that redd surveys in themselves without adjustment for adult to redd ratios would provide the same trend analysis as the adult numbers. However, much of the need for salmonid monitoring is to provide information for ESA and harvest management decisions. For these purposes, a measure of fish number carries more authority in these difficult decision-making processes.

Total Abundance Estimation over Large Geographic Regions (Status)

Status is estimated as total abundance (or escapement) over different geographic areas. Because the GRTS sample was selected with equal probabilities, estimation of current abundance in a study area is reasonably straight forward. Assume that $\tau_{p(i),t}$ is an unbiased estimate of the total number of fish present (abundance) in segment i of panel p when it was sampled during occasion t, and that $s_{p(i),t}$ is an estimate of the standard error of $\tau_{p(i),t}$. An estimate of total fish abundance in the entire study area during year t is,

$$T_t = N \sum_{p=1}^{P} \sum_{i=1}^{n_p} I_{p,t} \tau_{p(i),t} \Big/ \sum_{p=1}^{P} I_{p,t} n_p$$

where P is the total number of panels, n_p is the number of segments in panel p, $N = \sum_{p=1}^{P} n_p$ is total number of segments in the sample frame, and $I_{p,t}$ is an *indicator* function that takes on the value of 1 if panel p was sampled during occasion t and 0 otherwise (for the purposes of "soft stratification", Larsen et al. 2008). This "soft stratification" scheme will be expanded to include juvenile and habitat surveys. The scheme will result in vastly improved cost and efficiency from logistics such as landowner permissions, travel, and site set-up. Despite the complicated looking formula, this equation is simply the arithmetic average of fish abundance measured on all segments visited during occasion t [i.e., $\sum_{p=1}^{P} \sum_{i=1}^{n_p} I_{p,t} \tau_{p(i),t} \Big/ \sum_{p=1}^{P} I_{p,t} n_p$ = (sum of abundance on all segments sampled year t) / (number of segments sampled)] multiplied by frame size N.

Note that T_t does not contain terms that depend upon either GIS-estimated or actual segment length. This lack of dependence on unit length is intentional by design and avoids several potential pitfalls. First, segment lengths estimated from GIS data are notoriously inaccurate and errors in the GIS lengths compound if fish abundance is estimated as fish density times total stream length from the GIS. Second, field implementation of the CMP is unperturbed by stream channel mapping errors. For example, if a previously unmapped small stream, channel, or slough is discovered while field crews are collecting measurements on a particular segment, the additional habitat can be measured immediately and its data can be included in $\tau_{p(i),t}$ for that segment. This causes T_t to be "self-correcting" in the sense that it accurately estimates total abundance regardless of map inaccuracies in the GIS. Third, field workers do not absolutely need to measure real length of a segment, thus potentially simplifying field protocols. Contrary to intuition, empirical evidence suggests that the relationship between fish abundance and segment length is weak (unpublished data, North Cascade National Park). That said, stream length should be measured for use with the regression estimator with external variables described below. Finally, T_t remains unbiased for true total abundance regardless of true or measured variation in segment length.

Assuming $n_{t\bullet} = \sum_{p=1}^{P} I_{p,t} n_p$ is the number of segments actually sampled in year t, the estimated standard error of T_t is,

$$se(T_t) = N\sqrt{\left(1-\frac{n_{t\bullet}}{N}\right)\frac{sd^2\left(\tau_{p(i),t}\right)}{n_{t\bullet}} + \frac{1}{Nn_{t\bullet}}\left(\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t} s^2_{p(i),t}\right)},$$

where

$$sd\left(\tau_{p(i),t}\right) = \sqrt{\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}\left(\tau_{p(i),t} - \bar{\tau}_t\right)^2 \Big/ \left(n_{t\bullet}-1\right)}$$
$$= \sqrt{\left(n_{t\bullet}-1\right)^{-1}\left[\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}\tau^2_{p(i),t} - n_{t\bullet}^{-1}\left(\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}\tau_{p(i),t}\right)^2\right]}$$

(Thompson 1992, p. 129). This formula is the variance estimator of a total under two-stage sampling, assuming equi-probable sampling at stage one (whole segments), and unbiased sampling at stage two (sampling within segments). The finite population correction factor, 1 - $n_{t\bullet}/N$, for stage-one segments has been included, but a similar correction for sample size at the second stage has not, due to the varied nature of field measurements called for under the CMP. Certain analyses may wish to include a finite population correction factor in the last term of $se(T_t)$ if, for example, a large fraction of all pools in the segment were measured to obtain $\tau_{p(i),t}$ for the segment. The segment (first stage) correction factor will usually be negligible, and it can generally be dropped.

The above standard error estimators, and all other standard error estimators in this section, ignore the fact that the original sample of segments was selected using the GRTS algorithm. Ignoring the fact that the original sample was a GRTS sample, effectively treats it as if it were a simple random sample and results in an overestimate of variance. That is, standard errors calculated using these formulas are larger than the true standard errors of the associated estimator. This is unfortunate because a spatially balanced sample should result in estimates with lower standard error than simple random samples, and this lower standard error will not be realized because these standard error estimators assume simple random sampling. In other words, we know that GRTS sampling improves accuracy, but we don't know how much is due to the use of the simple random estimators. We used the simple random sampling estimators here because they are easy to calculate, and because the improvement in precision estimates afforded by more complicated estimators is slight for parameters with high residual variation. Nonetheless, analyses of data collected under this CMP should consider both the simple random variance and the local neighborhood variance estimators. The local neighborhood variance estimator $se(T_t)$ was proposed by Stevens and Olsen (2003) and software is available. The local neighborhood variance estimator averages variances estimated on local neighborhoods (on nearby segments) surrounding each segment.

Provided $n_{t\bullet}$ is large enough (generally > 30) a 95% confidence interval for

the true average fish abundance is,

$$T_t \pm 1.96 se(T_t)$$

regardless of the distribution of fish density in an individual segment. If sample size at occasion t is small, a confidence interval for mean fish density should be constructed using a nonparametric bootstrap method (Manly 2007, Ch. 3).

However, combinations of stream systems and fish species may exist where density is relatively constant throughout the system and correlation between fish abundance and segment length is strong. In addition, there may exist exogenous covariates such as average gradient or latitude that could potentially explain a significant fraction of the variation in T_t. Because of these potential advantages, regression estimators for T_t should be considered. Besides total abundance, regression estimation should yield excellent estimates of the true total kilometers of stream in a system that will in turn be used to estimate average fish density.

Assume that an auxiliary variable, say $x_{p(i),t}$, is known for all segments in the population, both sampled and unsampled. In most cases, $x_{p(i),t}$ will be derived from the GIS system. Examples of potentially useful $x_{p(i),t}$ include segment length as measured in the GIS, gradient of the segment as derived from Digital Elevation Model's, latitude (or longitude) of the segment's midpoint, average flow as predicted by a flow model, etc. Provided the true correlation between $x_{p(i),t}$ and $\tau_{p(i),t}$ is strong, we can use variation in $x_{p(i),t}$ to explain variation in $\tau_{p(i),t}$ and thereby improve the precision of T_t. We assume only one auxiliary variable is involved in estimation, even though it is possible to use more than one in a multiple regression estimator. Extension of the simple linear regression estimator to a multiple regression estimation is straightforward and is given in Thompson (1992, p. 86). Non-linear or scatter-plot smoother regression estimators are also possible.

The simple linear regression estimate of total abundance at a particular occasion is,

$$T_{R,t} = T_t + \hat{\beta}_1\left(T_{x,t} - N\overline{x}_t\right)$$

where

$$T_{x,t} = \sum_{p=1}^{P}\sum_{i=1}^{n_p} x_{p(i),t},$$

$$\overline{x}_t = \sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}x_{p(i),t} \Big/ \sum_{p=1}^{P} I_{p,t}n_p$$

are the known total of x in the population and the mean of x in the sample at time t, respectively. $\hat{\beta}_0$is the slope of a least-squares-estimated line through the scatter plot of $\tau_{p(i),t}$ on $x_{p(i),t}$ (Thompson, 1992, p. 80). Some care will be needed to avoid using the same segments multiple times. The estimated slope, $\hat{\beta}_0$, should be as accurate as possible and can be based on multiple years of data. An estimate of the standard error of $T_{R,t}$ is,

$$se\left(T_{R,t}\right) = N\sqrt{\left(1-\frac{n_{t\bullet}}{N}\right)\frac{\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}\left(\tau_{p(i),t} - \left[\hat{\beta}_0 + \hat{\beta}_1 x_{p(i),t}\right]\right)^2}{n_{t\bullet}\left(n_{t\bullet}-2\right)} + \left(\frac{1}{Nn_{t\bullet}}\right)\sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t}s^2_{p(i),t}}, \quad (1)$$

where $\hat{\beta}_0$ is the estimated intercept of the least squares regression fit (Thompson, 1992, p. 80 and 131). Note the first term under the square root is a function of the mean squared residual from the regression of $\tau_{(i),t}$ on $x_{p(i),t}$, thus affording a reduction in variance if $x_{p(i),t}$ indeed explains a large proportion of the variation in $\tau_{p(i),t}$.

To estimate total stream length in the population, we rely on correlation between segment length in the GIS and actual segment length measured in the field. If maps in the GIS are useful for locating stream segments, the correlation between map and actual length should be high. Assuming $l_{p(i),t}$ is the actual measured length of segment *i* in panel *p* at time *t*, and that $\lambda_{p(i),t}$ is length of the same segment reported by the GIS, we can apply the regression estimator above to estimate total length as,

$$L_{R,t} = N\bar{l}_t + \hat{\beta}_1\left(T_{\lambda,t} - N\bar{\lambda}_t\right)$$

where,

$$\bar{l}_t = \sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t} l_{p(i),t} \Big/ \sum_{p=1}^{P} I_{p,t} n_p \,,$$

$$T_{\lambda,t} = \sum_{p=1}^{P}\sum_{i=1}^{n_p} \lambda_{p(i),t} \,,$$

$$\bar{\lambda}_t = \sum_{p=1}^{P}\sum_{i=1}^{n_p} I_{p,t} \lambda_{p(i),t} \Big/ \sum_{p=1}^{P} I_{p,t} n_p \,.$$

The standard error of $L_{R,t}$ can be estimated using Equation (1), substituting l for τ and λ for x.

Average fish density in the population of stream segments can now be estimated by dividing estimated total stream length into estimated fish abundance. This is an instance of a ratio-of-totals estimator, and should yield highly accurate estimates. Prior to estimation, the best estimate of abundance, either T_t or $T_{R,t}$, should be determined. If estimates of fish per kilometer (or mile) are desired, the regression estimator $L_{R,t}$ should be used. If estimates of fish per hectare (or square meter or acre) are desired, measured values of segment area should be substituted for l in the regression estimator equations to obtain a regression estimator for total hectares in the system. Unless area of a segment can be estimated from GIS data, segment length should remain as the explanatory variable for estimating total area.

The ratio of totals estimate of average fish density at time t is either,

$$\bar{d}_t = \frac{T_t}{L_{R,t}} \text{ or } \bar{d}_t = \frac{T_{R,t}}{\dot{L}_{R,t}}$$

Assuming T_t is used, the estimated standard error of $\bar{d}_t$ is,

$$se(\bar{d}_t) = \sqrt{\frac{se^2(T_t) + \bar{d}_t^2 se^2(L_{R,t}) - 2\bar{d}_t \operatorname{cov}(T_t, L_{R,t})}{L_{R,t}^2}}$$

where $\operatorname{cov}(T_t, L_{R,t})$ is the estimated covariance between fish abundance and total length (Särndal et al., 1992, p. 179, eqn. 5.6.10). If $T_{R,t}$ is used to estimate density, $T_{R,t}$ should be substituted for T_t in this equation. Estimation of the covariance can be difficult in some surveys, and it is standard practice to drop this term during estimation. If the covariance term is dropped and if it can be assumed > 0, the resulting standard error is conservative in the sense that it is too large. However, after multiple years of sampling, covariance between fish abundance and total stream length can be estimated directly from data collected by the CMP. Furthermore, strong positive covariance between total number of fish and total stream kilometers in an entire system is expected, and the estimated standard error of $\bar{d}_t$ could be substantially reduced in this case. After m years of sampling under the CMP, the covariance between fish abundance and total steam length can be estimated as,

$$\operatorname{cov}(T_t, L_{R,t}) = \operatorname{cov}(T, L_R) = \frac{1}{m-1}\sum_{t=1}^{m}(T_t - \bar{T})(L_{R,t} - \bar{L}_R)$$

where

$$\bar{T} = \frac{1}{m}\sum_{t=1}^{m} T_t,$$

$$\bar{L}_R = \frac{1}{m}\sum_{t=1}^{m} L_{R,t}.$$

Again, $T_{R,t}$ should be substituted in place of T_t if $T_{R,t}$ is used to compute density.

Other estimates of average fish abundance and density at a particular point in time are available. The so-called MVLUE estimator reviewed by Binder and Hidiroglou (1988) is an alternate estimator of status that makes use of temporal correlation in fish abundance on individual segments (i.e., between $\tau_{p(i),t}$ and $\tau_{p(i),t-1}$) to improve estimates. The strength of correlation in fish abundances through time determines the magnitude of precision improvement, with higher correlation yielding higher improvement in precision. The MVLUE estimator is complicated and the improvement in precision afforded by it is unknown at present. The MVLUE estimator will therefore not be given here, but it should not be disregarded.

As described earlier, species estimation in subregions will be an important need in the CMP. Estimation of status in a subregion of the study area such as a watershed is called domain estimation (Lohr 1999) and is relatively straightforward. Samples that fall within the specified subregion are treated as if they were the sampling universe. If additional precision within the subregion is desired, then the next consecutive samples on the GRTS list that fall within the subregion can be added to the sample universe. If sample size for the subregion is sufficiently large (>30), then the simple random sampling formulas from above can be used for inference about the subregion total and variance. For smaller sample sizes, see Lohr (1999).

Southern Area

Goals and Methods

In the Southern Area, steelhead are the only salmonid present and, since abundances are known to be extremely low, monitoring is critical to assess recovery goals. These low abundances are difficult to monitor due to patchy spawner distribution and large stretches of uninhabited water. In addition, the Southern Area is very different from the northern area in terms of species composition, abundance, distribution, and run-timing, and dictate different adult monitoring approaches in the two regions. The major distinctive features of the Southern Area are: 1) only steelhead occur there; 2) steelhead population sizes are small and occur at widely spaced locations within watersheds, 3) stream flows in this area are generally very low, but in the winter can be episodic and short-lived, leading to erratic run-timing of steelhead that live there; and 4) the Southern Area has experienced greater habitat degradation than the northern area, particularly in the form of dams. Thus, our ability to sample adult steelhead in the Southern Area is confounded by small numbers of spawners coupled with the unpredictable shifting of steelhead run-timing and spawning locations. These conditions, particularly the small numbers of patchily distributed spawners, mean that the random spatially balanced surveys that will be used in the Northern Area would result in sampling an unacceptably large number of units containing no fish. This would lead to very small abundance estimates with large variances and little statistical power to detect change. These features argue for a complete census of steelhead in the Southern Area that would be accomplished by a fish counting station at the lower end of a number of watersheds. As in the Northern Area, decisions concerning adult abundance monitoring locations will be undertaken in other venues such as recovery plans, or for other specific needs. However unlike the spatially-balanced, random adult surveys used in the Northern Area, there will be no provision for subregion estimation. Abundance estimates will only be applicable to the specific streams surveyed and will have no variance estimates. While these individual population censuses cannot be expanded, this condition is being accepted due the lack of preliminary knowledge and the expectation that steelhead populations will be extremely sparse and highly clumped. As the level of background information is expanded, the Southern area monitoring plan may require modification.

Adult Monitoring

For the Southern Area, adult abundance monitoring goals will be to obtain complete adult censuses at existing or proposed fishways where possible and to conduct evaluations of new technologies for obtaining adult counts. Portable weirs may have limited usefulness in the Southern Area censuses because the few steelhead that inhabit these streams are known to move upstream and spawn on high flows when portable weirs usually have to be removed or cannot operate. This behavioral feature of steelhead in the region requires fixed location total census monitoring in the Southern Area rather than the random spatially balanced surveys proposed for the Northern Area. Counts at fixed stations lack the statistical rigor to assess regional status and trends in the same way as the methods used in the Northern Area (i.e., estimates cannot be statistically inferred to apply to non-sampled streams); however, over time this approach will create time series that will allow trend estimation on the set of monitored watersheds. The counting stations monitoring scheme is not as geographically flexible as the GRTS and greater care in selection of watersheds and locations of counting stations will be needed to insure that the collection will supply the needed data. Provided that the set of monitored watersheds include the major steelhead-bearing watersheds, this approach will provide the information necessary to guide management. Finally, in the past many attempts to establish fish counting stations have failed due to operational conditions. Operating a station that will provide reliable counts is not a simple undertaking and will need to be well thought out.

The proposal for monitoring Southern Area steelhead focuses on conducting complete censuses of the major watersheds considered the keystones for viability in recent recovery planning efforts (Boughton et al. 2007). This approach will increase reliance on spatial structure sampling over the entire Southern Area to provide information on other watersheds. As more funding becomes available, adult census monitoring can be expanded to more watersheds. In the South-Central California Coast Steelhead DPS, only the Carmel River is currently being monitored. In the Southern California Steelhead DPS, there are monitoring sites on the Ventura and Santa Clara rivers.

Traps and weirs associated with passage facilities can quantify the escapement of adult salmonids in streams and rivers. In addition to providing absolute counts of fishes migrating beyond a fixed point in the system, they can be used to determine species composition, determine sex ratio, place and recover tags, and collect tissue or scale samples. Passive integrated transponder (PIT) tagging can be used at the trap site to gather data on travel time, passage timing, and survival. The trap site and other appropriate locations should be equipped with PIT antenna arrays to detect tagged fish passage. Additional PIT monitoring stations can be added to collect data to answer specific questions at relatively low cost. As noted above, portable weirs may have limited usefulness in the Southern Area censuses because the few steelhead that inhabit these streams are known to move upstream and spawn on high flows when portable weirs usually have to be removed or cannot operate. Detailed procedures for these types of sampling operations are described in Zimmerman and Zabkar (2007).

Video systems have been used to count many salmonid species in a wide range of circumstances; however they are only likely to be successful when placed in a passage facility (O'Neal 2007a). For video systems to work, fish need to be crowded into a narrow area to be counted due to the limited imaging range of video recording systems. They provide a time-saving, cost effective method for obtaining weir counts and avoid actually handling the fish, which is an important consideration in dealing with a listed species. In addition, video systems provide the opportunity to record fish behavior. Video systems can provide all of the data from a passage facility including species composition, numbers, direction of passage, body size, and hatchery marking. However, video systems lose resolution with even limited turbidity.

A video counting system is currently being operated at San Clemente Dam on the Carmel River. The system is operated on a fish ladder where as fish jump up from one ladder step to another, they break an electronic beam that turns on the video camera, and the fish's image is recorded. Commercially available video systems have become very sophisticated and many specialized needs can be met, including automatic processing of the video and long distance real-time viewing. Detailed procedures and advice for constructing, installing, and operating video fish counting systems is given in O'Neal (2007a).

For smaller systems where larger, more permanent systems are not feasible, Dual-Frequency Identification Sonar (DIDSON) can be used to provide adult abundance numbers. DIDSON is an acoustic "camera" that has been recently adapted to fisheries monitoring by the Alaska Department of Fish and Game (Maxwell and Gove 2004). The device is a high-frequency sonar system with a lens capable of focusing sound waves onto a high-resolution sensor array. It is self-contained and operates much like a video camera except that it processes reflected sound rather than reflected light. The resulting acoustic image is grainy compared to light-based images, but is a considerable improvement over older-style sonar units. This unit is not much more difficult to operate than a video camera and requires little training to use. Its advantage over video systems derives from the fact that it is a sonar device, and is therefore not limited by turbidity, and does not require a fish crowding structure.

Like a video image, the DIDSON sonar image is detailed enough to identify, count, and measure the size of fish swimming through the beam, but unlike a video camera the unit can detect images when video cannot (e.g., in opaque water during high-flow events when steelhead are known to move in this region). The unit can view up to 20 m of stream width thereby making the installation of a weir unnecessary. Thus, DIDSON has the potential to provide complete steelhead counts even during peak Southern California flow events in small (< 20 m wide) streams. Steelhead are the only anadromous salmonid found in streams from the Pajaro River southward, so species identification is not an issue. Fish counts can be automated or the view sequence can be shortened to periods when the DIDSON software detects movement.

The Alaska Department of Fish and Game has conducted a series of pilot studies using DIDSON to assess its use in monitoring salmon runs (Maxwell and Gove 2004, D. Burwen, ADFG, pers. comm.). They found that DIDSON produces precise estimates of fish passage over a wide range of abundances, including at high passage rates (Figure 6).

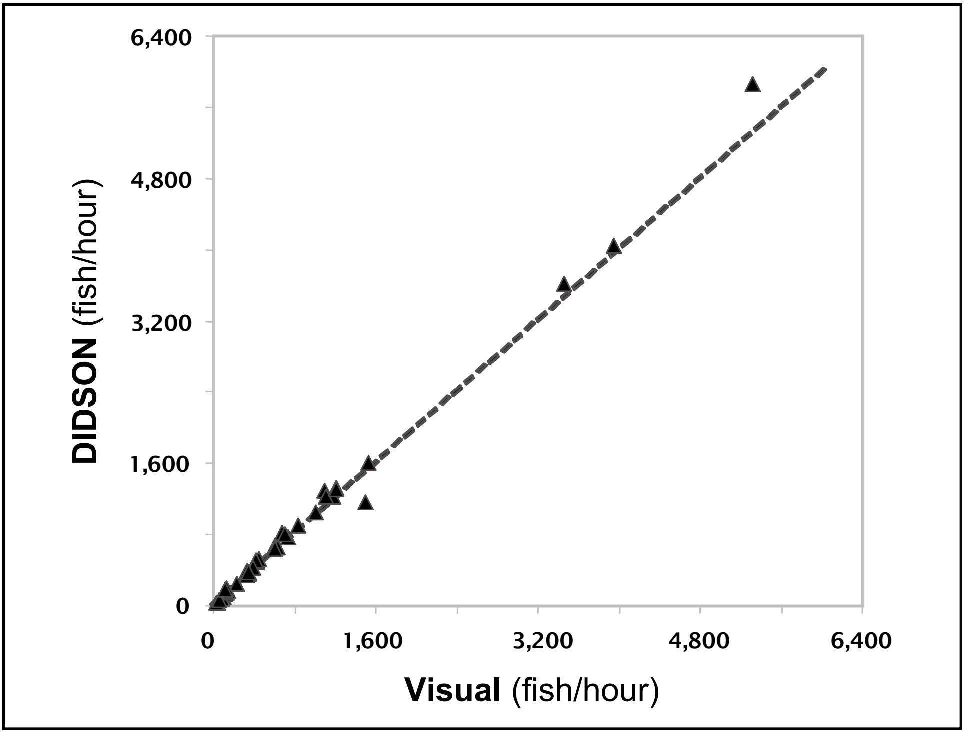

Figure 6. Comparison of salmon counts from the DIDSON camera vs. visual tower counts (made by an observer sitting in a streamside tower, currently considered the benchmark data type by Alaska Dept of Fish and Game).

To specifically test the DIDSON acoustic camera as a tool to monitor Southern California steelhead, a DIDSON unit was deployed in the San Lorenzo River, Santa Cruz County for limited periods during 2006 (Pipal et al. 2010). The DIDSON unit was installed approximately 200 m downstream of a diversion dam with a fish counting trap. The DIDSON unit and trap counts were compared over the same time period (March 17, 2006 to March 24, 2006) when both the DIDSON and the fish trap were operating.

Counts made with the DIDSON unit and the traps were very similar. The DIDSON unit counted 41 net upstream migrants (46 upstream migrants minus 5 downstream migrants Table 1). The trap collected 46 fish (Table 2). Evidence to date supports the potential for using DIDSON units for salmonid assessments in low abundance Southern California rivers and streams (Pipal et al. 2010).

DIDSON's greatest strengths—that it does not require handling of fish and it does not impede passage of low-abundance, high extinction risk salmonids—

are also it's greatest weakness in that certain types of biological information (e.g. sex ratio, scale samples) cannot be taken. DIDSON use for monitoring does have some of the same potential problems as video, such as providing power and security. The most difficult problem comes from the fact that its image spans the entire stream bed. The DIDSON records much more natural fish behavior than a situation where the fish has to move through a fish trap or narrow viewing channel. This behavior can, at times, be difficult to interpret as simply either upstream or downstream migration. Detailed operational guidance for using DIDSON as a salmonid counting method can be found in Pipal et al. (2010).

Table 1. Summary of DIDSON footage from the San Lorenzo River between March 17 and March 24, 2006.

DIDSON File Review				
DIDSON FILE DATE	**File Type**	**Fish Up**	**Fish Down**	**Net Fish Up**
3/17/2006	Sonar	1	0	1
3/21/2006	Sonar	4	0	4
3/22/2006	Sonar	15	4	11
3/23/2006	Sonar	22	0	22
3/24/2006	Sonar	4	1	3
Totals		**46**	**5**	**41**

Table 2. Totals from the fish trap on the San Lorenzo River approximately ~200m upstream from the DIDSON location.

Trap Totals				
Trap Operation	**File Type**	**Male STH**	**Female STH**	**Total Trapped**
3/17/2006		7	4	11
3/21/2006	Trap not operated	N/A	N/A	N/A
3/22/2006		5	5	10
3/23/2006		4	6	10
3/24/2006		9	6	15
Totals		**25**	**21**	**46**

SPATIAL STRUCTURE

Goals

Effective spatial population structure can provide protection from local catastrophic extinction risks that are separate from those due to abundance and productivity (McElhany et al. 2000). Salmonids have high fidelity to their spawning locations (Groot and Margolis 1991) and therefore have a naturally patchy distribution because of their spawning conditions and/or the nature of their habitat dynamics. At the same time, some individuals move from one natal spawning area to another (straying; Quinn 1997). Therefore, a population's spatial structure is the result of these population characteristics; fidelity to spawning location, straying, and the nature and dynamics of their habitats. Spatially structured populations are often generically referred to as "metapopulations" (Levins 1969). Though the term metapopulation has taken on a number of different meanings, the general meaning is a group of spatially separated populations of the same species that interact at some level through dispersion. Since the dynamics of a metapopulation can include individual population extinction and recolonization, understanding the population spatial structure can have important consequences to salmonid population viability (Hanski and Gilpin 1997). Metapopulation theory has shown that spatial structure can have both within-population and within-ESU aspects.

Population-level spatial structure is a function of the population's habitat dynamics and the rate at which individuals move from one location to another. Spatial structure is important to viability because extinction risk occurs at longer time scales and may not be apparent from short-term observations of abundance and productivity (McElhany et al. 2000). If habitat is being destroyed faster than it is being created, then population viability will decrease. Also, where straying among subpopulations decreases due to increasing distance among occupied habitat patches, population viability will again decrease. Often under anthropogenic stress, both mechanisms are occurring at the same time. In these situations of decreasing population viability, strong source subpopulations should be indentified and maintained as an essential element of recovery. As population decline becomes more pronounced, monitoring of these spatial structure characteristics is increasingly important since isolated groups of fish are much more vulnerable to rapid extinction.

Within-ESU spatial structure is important to salmonid viability due to risks of catastrophic events (Bisson et al. 1997). Catastrophic events affect entire populations and occur rarely over a 100-year time scale. They can be natural or anthropogenic events; and often natural catastrophes will increase in magnitude or frequency due to anthropogenic disturbances. Catastrophes can profoundly affect extinction risk. In fact, models predict that the rate and severity of catastrophes can be the most important factor in determining a population's extinction risk (Lande 1993, Mangel and Tier 1993). The Sacramento winter-run Chinook salmon ESU is an example of the risks associated with poor spatial structure characteristics—the entire run is composed of a single bottlenecked population that spawns in one location below Shasta Dam. In a nearby area above Shasta Dam, the Cantara herbicide spill caused a wide-spread fish kill. The spill wiped out the downstream fish and invertebrate populations, including native rainbow trout, but since the spill was confined above Shasta Dam, impacts to anadromous salmonids did not occur. Within-population spatial structure can also have serious extinction risks and these risks will often coincide with low abundance.

Due to the potential for catastrophic events, there should be multiple populations within ESUs that do not share common catastrophic risks. At the same time, some populations within ESUs should be geographically close to each other so that metapopulation interactions can occur. The TRTs built this concept into their approach with the use of diversity strata and requirements for strata viability throughout the ESU and requirements for population viability within a stratum (Spence et al. 2008, Williams et al. 2008). Anthropogenic impacts have often reduced habitat in ways that further concentrate salmonids into any remaining higher quality habitat. Therefore, assessing current spatial structure is an important measure of viability. Measured improvements in spatial structure are also a strong indication of progress toward recovery.

Spatial structure monitoring is important in both the Northern and Southern areas. In the Northern Area, some information for Chinook salmon, coho salmon, and steelhead will come from adult monitoring, although that information will not be comprehensive. These data will be used to assess the spatial patterns that indicate sufficient immigration is occurring to ensure connectivity and to assess whether distribution is expanding or contracting using simple binomial probability of segments occupied. Different species have different levels of extinction risk associated with loss of spatial structure, so that different sampling frames may be used in the sample draw, perhaps sampling one species more intensively than another. This assessment of occupancy patterns will be used to record that connectivity is maintained between populations and to monitor whether the species distribution is expanding or contracting. In the Northern Area, spatial structure monitoring will be conducted for juvenile coho salmon using snorkel sampling throughout the entire area. Spatial monitoring for steelhead is a lower priority because they are more widely distributed than coho and Chinook salmon. However, we realize that, even for this resilient species, spatial patterns may change rapidly. This is especially a concern in the face of climate change. The CMP will revisit prioritization of steelhead spatial structure surveys, incorporating them as soon as possible after project implementation begins. Juvenile Chinook salmon leave the watershed and enter the ocean too early in the year to be surveyed with snorkel methods. For Chinook, adult surveys will provide the primary information about spatial distribution of spawners, and outmigrant monitoring (e.g., using rotary screw traps) could be used to obtain opportunistic watershed-level information on spatial structure. If Chinook salmon spatial structure information is considered sufficiently important, adult surveys in Chinook salmon habitat could be expanded to provide that information. In the Southern Area, steelhead will be monitored for spatial structure. Due to the very small populations in Southern California, spatial structure monitoring may need to be more localized and focused than in the north. In the Southern Area, rainbow trout, the nonanadromous forms of *O. mykiss*, occurs more commonly than the anadromous form even in anadromous waters. Snorkel surveys for steelhead are difficult and their results are inconsistent because it is difficult to visually distinguish steelhead from rainbow trout while snorkeling. The only reliable way to distinguish between the two forms is an evaluation of otolith microchemistry—a lethal and time consuming procedure. Discrimination of steelhead from resident rainbow trout, and ways to evaluate the relationship between these two life history forms is an active area of research. However, presently there is no simple way to distinguish between them for routine population monitoring.

<u>Sample Design and Methods</u>

Although spatial structure could be monitored using adult spawner distributions, the approach of targeting the distribution of juvenile fish in summer or early fall is operationally simpler and less expensive and is more comprehensive as it takes into account species dispersal following hatching. Therefore, the CMP proposes using juvenile salmonid surveys as the most efficient means to monitor spatial structure. Sites will be selected using the protocol described in the Adult Sampling section. In the future, a modified sampling frame to represent summer rearing areas will be developed using a "soft-stratification." The procedure would include selection of a GRTS sample, and allocation of that sample into panels that receive rotating effort over the years. This will allow for estimating spatial structure of fish and habitat condition at both the population and ESU levels.

The sample draw process starts with establishing a desired sampling intensity. More intense sampling would be required as the species becomes rarer to maintain the same coefficients of variation. Sampling rare species usually leads to greater uncertainty in the estimate even with higher sampling intensity. As in the adult sampling, a random spatially balanced sample is desired due to the patchy distribution of the fish. A GRTS-based sample draw at the desired intensity will be selected. Additional samples will need to be drawn since some of the samples will not be useable due to inaccessibility, unsuitable habitat, poor water quality, or other reasons. Increased subsampling will be accomplished by drawing additional samples in the same manner as for the adult sampling.

Snorkel surveys for juveniles are effective, cost efficient, and cause the least impact on ESA/CESA-listed species. Juvenile surveys during the summer and fall will allow the widest measurement of species distribution for a fixed cost. Sampling and access is far easier and less expensive at that time of the year. Snorkel surveys are both cheaper and faster than the next most common juvenile sampling method: electrofishing. Snorkel surveys and electrofishing have different levels of precision depending on conditions like water clarity, habitat structure/complexity, and fish density. Also, snorkel surveys can give inaccurate counts if moving fish are recounted. However, more samples can be obtained for the same cost using snorkel surveys. Snorkel surveys can also be conducted in conditions where other survey techniques are not feasible, such as low water or when sampling sites are far from roads. Finally, snorkel surveys can provide qualitative information on fish behavior such as habitat associations, feeding, and resting activities that may help us to evaluate the reasons underlying observed distributions.

Comparative studies of snorkel and electrofishing for coho salmon and steelhead have shown that snorkel-survey abundance estimates are higher than electrofishing estimates (coho salmon 1.6 times and steelhead 2.0 times, Jepsen 2005). Other research evaluating population estimation methods for juvenile coho salmon in small Oregon streams found that mark-recapture, electrofishing, and snorkel techniques accounted for 85%, 67%, and 40%, respectively, of the known summer populations in pools (Rodgers et al. 1992). But this range of densities may not have been great enough to show an influence (J. Rogers pers. comm.). High fish density influences snorkel counts more than other methods, as accurate visual counting is more difficult with larger numbers of fish (Heggenes et al. 1990). However, Rodgers et al. (1992) found no effect of fish density on the accuracy of snorkel or removal techniques. So while snorkel surveys may be biased toward lower abundance estimates than two stage sample designs (Hankin and Mohr 2009), this is less important in evaluating spatial structure than the increase in the number of samples that can be obtained at fixed cost. Finally, snorkel surveys do not require handling fish and in general cause much less stress for the fish than methods that require handling. This is an important advantage of this method for sampling listed species.

The CMP proposes using standard snorkel survey procedures as described in detail in Peck et al. (2003), and O'Neal (2007b), which we will only briefly review here. Teams of two snorkelers will be trained prior to the sampling season. Inter-observer reliability will be assessed and calibrated during training. The snorkelers will alternate counting and recording on smaller sections within the reach. In those few instances where the tributary is too wide for one snorkeler to survey, teams will snorkel side-to-side. Sampling will count all individuals of whatever species encountered in the sample unit through the entire length. Data will be entered in handheld electronic recording devices to be downloaded after return from the field. The accuracy of the snorkel counts will be assessed by revisiting between 10-20% of sites that were occupied by the species of interest. This will allow evaluation of initial count precision and provide the basis for variance estimates using the methods of Stevens (2002). More detailed field protocols can be obtained from O'Neil (2007b).

Spatial structure from juvenile snorkel surveys will be measured as the proportion of sample units occupied by at least one fish, the average number of pools per sampling unit occupied by at least one fish, and the average number of fish per sampling unit. Spatial structure might be compromised even if a relatively high proportion of sites are occupied but they are geographically concentrated. The proportion of sampling units occupied by at least one fish will be reported as a simple percentage. Since the sampling units were drawn using the GRTS procedure, this may be considered a random sample and representative of the entire population of sample units. The average number of pools per sampling unit with at least one fish is a simple mean of a binomial distribution and means and confidence intervals can be calculated by the standard methods. Then again, the degree of occupancy of any sampling unit is the number of segments containing at least one fish and is also estimated as a simple mean of a binomial distribution and its variance. We expect that, because of their relative rarity, data for coho salmon will include a large number of sample units without any fish. Because of that, for coho salmon, the median may be a more useful measure of central tendency and should be considered along with the mean. The average number of fish per sample unit can then be estimated over larger sampling units of interest. A composite variance estimator using the revisit data is provided by Stevens (2002). ODFW uses the SVB metric (Stevens 2006) to assess whether the occupied sites are dispersed or clumped, and this can be considered in the future.

DIVERSITY MONITORING

Goals

Salmonids in coastal California possess and exhibit a wide range of physical and behavioral characteristics that affect population and ESU-level viability (McElhany et al. 2000; Table 3). Expression of diverse life history, behavioral, and physiological traits allow salmonid populations to tolerate irregular or cyclical environmental variation, and provide a buffer against habitat change, food web shifts, and varying predation pressure. Diversity is frequently assessed by analyzing allele frequencies of neutral genetic markers (e.g., microsatellite DNA). Although the genetic underpinnings of most life history traits cannot currently be directly assigned and quantified, the expressed traits themselves (e.g., run timing, outmigration timing, and age structure) can be assessed. Diversity traits are expressed on different spatial scales. Major diversity traits, which have the strongest genetic signal, are observable at the species- and ESU-level. Species and ESU-level diversity, and diversity patterns over large geographic areas (e.g., California coast-wide), are incorporated into listings and recovery plans. Population-level diversity traits, which are more difficult to track, will have to be identified on a case-by-case basis. Due to these ESU level and even local differences in diversity traits, it will be almost impossible to compare diversity over larger areas and diversity monitoring will be used largely for trend monitoring within an area of sampling interest.

The CMP goals for diversity monitoring are to: 1) establish and maintain genetic baselines for all salmonid runs and ESUs, and 2) identify important and variable life history characteristics of specific populations within each ESU that can be measured as part of existing field surveys, at LCM stations, and at hatcheries, or for other Diversity traits for which specific data collection still needs to be designed.

Methods

Relevant diversity characteristics vary with species, population, and ESU. This makes it very difficult to provide specific advice about sample design and methods for monitoring diversity across all of coastal California. Also, we expect that substantial research will be necessary to understand the way in which many of the most important diversity traits operate before we can understand how to monitor them.

The CMP proposes a stepwise process to identify relevant diversity characteristics and incorporate appropriate monitoring into adult and juvenile field surveys and LCM station data collections. In some cases, additional surveys may be required to address diversity monitoring.

Step 1: CDFG and NOAA Fisheries will jointly convene meetings of local expert teams that will indentify diversity characteristics relevant to each species at the population- and ESU-level. ESU-level diversity characters will include those identified in federal and State status reviews and recovery plans.

Step 2: Local expert teams will develop ESU-wide programs to monitor identified diversity traits. Whenever possible, diversity monitoring will be incorporated into established population monitoring protocols at hatcheries, established LCM stations, and regional spawning and juvenile sampling surveys. Otherwise, new surveys will be added to the CMP to collect and evaluate specific diversity information.

General Diversity Characteristics

Evaluation of genetic diversity at population and ESU scales is essential. Therefore, the CMP proposes collecting tissues for genetic analysis from all fish handled in surveys and at LCM stations. Tissue collection and archiving will follow protocols established by NOAA Fisheries SWFSC's Coastal Salmonid Tissue Archive in association with CDFG's Central Valley Anadromous Salmonid Tissue Archive. Genetic baselines will be periodically revisited.

Tissue samples will also be collected from hatchery broodstock and, when appropriate, juveniles. In places with significant hatchery influence, tissue samples will be sought from natural-origin and hatchery-origin fish that spawn naturally. Hatchery and spawning ground genetic data will be used to evaluate and improve hatchery operations and to assess interactions of hatchery fish with naturally spawning stocks.

A program will be developed to collect, read, and archive scales (and/or otoliths) from adult fish collected at LCM stations, hatcheries, and those encountered in limited carcass surveys. These data will be used to assess age structure. In some cases (e.g., steelhead, and to some extent perhaps coho salmon), carcasses may not be available or accessible in large enough numbers to allow age structure estimation. In these cases we will use either hatchery fish data as a surrogate, or rely solely on data gathered at LCM stations. Age structure estimation for the small numbers of steelhead in the southern area will likely rely on data collected at LCM stations.

The CMP will also collect seasonal abundance data at LCM stations to evaluate adult run timing and juvenile outmigration at these selected sites.

Table 3. Diversity Characteristics of Salmonids Ranging from Strongest level of Genetic Separation to Weakest.

I. Strongest levels of genetic separation
   A. Species

II. Significant levels of genetic separation
   A. Major geographic divisions: Distinct Populations Segments (DPSs) and Evolutionarily Significant Units ESUs)
   B. Within geographic division traits (Generally labeled as run timing, but includes a wide variety of genetically inherit traits that enable these reproductive strategies)
      1. Strong genetic signal – Separate Central Valley Chinook salmon ESUs, Columbia River Chinook salmon ESUs
      2. Weak genetic signal – Klamath Mountain Province summer steelhead, Klamath spring run Chinook salmon

III. Major life history traits (Small to no genetic signal)
   A. Anadromy/resident
   B. Sex ratio
   C. Fecundity (Includes egg size)
   D. Age and size structure
   E. Habitat utilization patterns (Freshwater and marine)
   F. Emigration age and timing
   G. Maturity patterns (Includes winters at sea)
   H. Adult spawning timing
   I. Physiological tolerances

LIFE CYCLE MONITORING STATIONS

Goals

Work within the past few years has demonstrated the effects of ocean conditions on salmonid abundance (Loggerwell et al. 2003, Botsford et al. 2005, Mueter et al. 2007). Salmonids experience wide variation in marine survival due to cyclic and non-cyclic changes in ocean conditions which can mask both species recovery and declines. Effective recovery monitoring should provide an independent measure of ocean survival so that recovery can be accurately assessed. The CMP proposes long-term, intensive monitoring at fixed Life Cycle Monitoring (LCM) stations to evaluate the effects of changing ocean conditions on salmonid populations by providing measures of freshwater and ocean survival. Salmonid population abundances are known to change dramatically from year to year (PMFC 2007) due to changes in ocean survival. This variation has long been considered in harvest management. For coho salmon, abundance has been shown to have decadal scale variability due to ocean survival (Botsford et al. 2005). For example, coho salmon experienced a decadal scale decline in ocean survival from near 10% in the early 1970's to values less than 1% in the 1990's. Similar patterns of variability in ocean survival are thought to occur with other salmonid species, but are not as well documented. Therefore, the measures of freshwater and ocean survival that will be obtained from LCM stations are essential for effective interpretation of observed variation in adult abundance. Also, secondary questions such as geographical patterns in survival rates can be used to help explain differences in effectiveness of recovery and restoration actions.

LCM stations will include an adult counting station, spawner surveys upstream from the counting station, and outmigrant juvenile trapping. The adult station is necessary to adjust the results of the larger-scale redd surveys for estimating adult abundance and to link variation in survival at different life cycle stages to adult abundance. Redd to adult bias corrections will be estimated from the LCM data (see Adult Escapement per Sample Unit from Redds Estimation Section, above). These corrections are essential components of the larger-scale adult abundance estimates and can best be gained from data collected at LCM stations. Currently, it is not clear how much geographical or annual variation these redd to adult bias corrections have, and results over a number of locations and years will be necessary to establish these relationships. This means that the LCM stations need to be established as soon as possible. The outmigrant juvenile trapping along with the adult counting station will provide estimates of freshwater and marine survival.

It is expected that the LCM stations will attract a wide range of salmonid research projects. The most obvious research focus will be salmonid habitat productivity studies. In particular, specific links between fish production and freshwater habitat condition are difficult to determine, and have not been well established (Smokorowski et al. 1998, Roni et al. 2002, Feist et al. 2003). Current thinking tends toward the view that population viability is more dependent on a complicated collection of spatial features and processes at the landscape level (Dunning et al. 1992, Bond and Lake 2003, Williams and Reeves 2003). It is hoped that CMP habitat assessments and population monitoring will further our understanding of these habitat-productivity relationships.

Locations

LCM stations will need to be distributed in a way that captures regional marine and freshwater dynamics and at least two LCM stations per recovery domain will be necessary. Oregon Department of Fish and Wildlife (ODFW 2008) and Washington Department of Fish and Wildlife (WMOC 2002) have prepared a list of considerations for location of the LCM stations and this information has been updated and modified specifically for California (Boydstun and McDonald 2005). Specific consideration criteria are presented in these documents and will not be repeated here except for the following general comments. The LCM stations will not be located randomly, due to accessibility requirements and the need to restrict locations to watersheds of manageable size. LCM stations will probably be placed on smaller systems that are in single ownership with good access or where there are existing counting weirs. This will probably lead to the stations being placed in systems that are smaller and perhaps in places with better habitat condition than average. Also, locations where there would be substantial or erratic mortality between the outmigrant trapping and ocean entry locations should be avoided, since this would bias the ocean survival estimates. However, LCM stations will still provide important information for understanding salmonid recovery, even with these unavoidable limitations. We will not provide specific location recommendations in this document. But, Boydstun and McDonald (2005) provide a list of existing counting weirs and some general guidance. ODFW (2008) suggests pairing geographically close stations, since one person can operate two locations in one day. Currently, Oregon operates eight LCM stations (ODFW 2008) while Washington has nineteen LCM stations (Crawford et al. 2002). The numbers of LCM stations that will be needed are unclear at the present, but should reflect major biotic areas along the California coast. At a minimum, there should be at least two LCM stations per recovery domain in the Northern Area. In the Southern Area, there are liable to be a large number of LCM stations due to the reliance on counting stations to estimate abundance.

Methods

The essential components of the LCM stations are an adult counting station (e.g., a weir), adult escapement surveys above the counting station, and outmigrant juvenile trapping. The standard adult surveys can be conducted following the procedures presented in the Adult Monitoring section. The adult counting station and trapping of outmigrant juveniles will be described briefly below.

Counting Stations

Zimmerman and Zabkar (2007) describe detailed sampling methods for operating fixed station and weir counting stations. A few major points are outline here. Gallagher et al. (2010a) operated both fixed counting stations and the more-common PVC resistance weirs in Mendocino County and found that the fixed counting station performed better. The fixed counting station performs at much higher flows than the resistance weir. However, new fixed counting stations are unlikely to be built except in association with new or renovated water storage or hydroelectric projects. Therefore, resistance weirs are much more likely to be used as adult counting stations. Procedures at weirs are straightforward where fish will be counted, measured and biological samples (scales, tissue samples, etc.) will be taken. Fish will be marked as they pass through the weir to estimate double counting and uncounted passage in the watershed.

In circumstances where high flow events allow significant numbers of salmonids to pass counting stations unmonitored, mark and recapture experiments will be needed. The fish will be marked at the counting station and recovered in the adult spawning ground surveys either as live re-sightings or carcass recoveries. The fish should be marked with tags that are individually numbered and/or have a color scheme that indicates the week in which they were marked. In addition, an operculum punching system that is stream-specific should be used to evaluate tag loss. Mark and recapture data can be analyzed for either live fish (re-sight) recoveries that assume replacement or closed population models using carcasses that are recaptured only once (see Seber 1982 and Gallagher et al. 2010a).

DIDSON methods for counting adult abundance (see Adult Monitoring, Southern Area) appear to provide reliable estimates where species identification is not an issue (see data presented here and Maxwell and Grove 2004). DIDSON equipment can be operated at higher flows than resistance weirs and provides salmonid estimates unconstrained by fixed counting stations. However, where two or more similar salmonid species inhabit a stream, reliable species identification can be problematic. Adult Chinook salmon can be separated from coho salmon and steelhead by size and date, but identifications of coho salmon and steelhead need to be validated before species specific estimates can be accepted.

Outmigrant Juvenile Trapping

The CMP proposes using outmigrant juvenile trapping to assess freshwater habitat quality both through estimators of freshwater survival and through outmigration characteristics (e.g., numbers of fish, fish size, and timing). Trap type and design (fyke net, inclined plane, or rotary screw traps) will be dictated by local conditions, species present, stream size, and flow conditions. Extensive advice is available about trap site selection, operations, and data management (Volkhardt et al. 2007, O'Neill 2007a, ODFW 2008) and will only be dealt with briefly here. Traps for coho salmon and steelhead are generally fished near the head of a pool, just below a section of fast flowing water. Stream flow should be moving in a straight line as it enters the trap. Juvenile Chinook salmon trapping will generally occur in the main stem using rotary screw traps, which require several feet of water to use. Trapping usually begins by the first week in March and continues until the catch decreases to low levels, usually ending by the first of June except for Chinook salmon which may continue migration through July. However, juveniles are out-migrating both earlier and later than these dates and there is some level of juvenile outmigration throughout the year (S. Harris, pers. comm.; S. Hayes, pers. comm.). Traps generally are operated 24 hours per day 7 days per week and are at a minimum monitored daily. All fish should be counted each day and a subset of at least twenty should be measured per species and size class. `Generally all fish handled for length measurements or marking for trap efficiency experiments should be anesthetized and care should be taken to reduce fish stress due to water temperature and fish density. Holding water should be cool and well-oxygenated.

Measures of trap efficiency, or the probability that an individual will be captured in a trap, are necessary to properly assess juvenile outmigrant data. Volkharht et al. (2007) gives examples of trap efficiencies ranging from 63% to 13% so that the estimates would be expanded by 1.6 to 7.7 times actual catch. Flow is often the dominant factor affecting trap efficiency since downstream migration is often prompted by high flow events. Turbidity, visibility, fish species and size, noise, and location are also factors that may affect efficiency. The varying influence of these factors indicates that trap efficiency measures are needed throughout the migration period and particularly to cover a wide range of flow events. Detailed instructions for measuring trap efficiency are given in Bjorkstedt (2005), Volkhardt et al. (2007) and ODFW (2008).

The basic trap efficiency procedure is to release marked juvenile outmigrants above the trap, estimate the proportion of released fish captured, and expand the total trap numbers by that proportion. Fish previously captured in the trap or in a secondary upstream trap are marked using fin clips, dying, pit tags, or panjet marking. Care should be taken in the selection of the marked fish so that they are representative of the entire population. In some circumstances, hatchery fish may be used as the marked component, but because hatchery fish can behave differently than natural fish, using them in this way introduces unknown biases to the efficiency estimates. The marked fish should be released far enough upstream that they redistribute in a natural pattern, but not so far that other factors such as predation become issues. Volkhardt et al. (2007) suggest a release point at least two pool/riffle sequences above the trap. Trapping efficiencies over discrete experiments and time periods described here are estimated using the Petersen method (See Seber 1982, Volkhardt et al. 2007). Confidence intervals are commonly estimated using bootstrap procedures (Manly 2007).

Mark and recapture experiments have a number of assumptions: populations are closed, marked and unmarked individuals are well-mixed, and marks are not lost (Ricker 1958). Also there is the assumption that these factors do not vary over time that is often unlikely given the influence of flow on smolt outmigration. Stratified mark and recapture estimates provide a means to incorporate variability in trapping operations and for improving the precision of the abundance estimate (See Schwarz and Dempson 1994 and Bjorkstedt 2005). Fish are marked in a different unique manner over discrete time periods, often one week. Mark and recapture experiments then can be decomposed into a series of time periods when recaptured individuals can be traced back to the period in which it was marked. Stratified mark and recapture experiments can use two traps (either partial or complete) or a single trap, where fish are captured, marked, moved upstream and released. Stratified mark and recapture estimates can be modeled against other variables, most often flow, to greatly improve total abundance estimates. The methods of calculation are well developed and will not be covered here, but are available from Seber (1982), Schwarz and Dempson (1994), Arnason et al. (1996), and Bjorkstedt (2005). Software for analyzing stratified mark and recapture experiments is available from Arnason et al. (1996) and Bjorkstedt (2005).

Survival Indices

Three survival indices will be calculated: total survival, freshwater survival, and marine survival. Together, these indices will be used to assess links between life-stage specific mortality and adult abundance. Total survival will be calculated as the number of adults returning in a year (recruits) divided by the number of spawning adults that contributed to the recruits. This is the same quantity as Productivity or the cohort replacement rate. Freshwater survival will be calculated as the number of smolts divided by the estimated number of eggs deposited by the spawning adults in that season. Whether an overall or an annual egg per female body weight estimator will be needed is yet to be determined. Marine survival is calculated as the returning adults (recruits) divided by the number of smolts from the corresponding brood year. While this is relatively straightforward for coho salmon due to their more rigid three-year cycle, both Chinook salmon and steelhead have less rigid life cycles and will require knowledge of their age structure. These marine survival indices should be calculated as geometric means to reduce the influence of extreme values and to be consistent with other ESA analyses (Good et al. 2005). Precision estimates for these survival indices will be computed by bootstrapping the underlying data (Manly 2007). Knowledge of life stage specific mortality features will allow us to assess which recovery actions or scenarios will be most likely to lead to improvements in adult numbers. Marine survival can also be estimated from CWT hatchery Chinook salmon data. These estimates are not routinely made, but have been calculated for specific situations (M. Mohr, pers. comm,) The nature of survival indices of hatchery salmonids may be very different than those of natural spawned fish (Lindley et al. 2009) and may vary among species.

DATA MANAGEMENT

Need for a Centralized Database

Data management is a very important part of the CMP. These data are required for analyses intended to inform decision makers, and if these data are not easily and clearly available, then they have no value. Data management is often neglected as an integral element of biological sampling programs and it is therefore essential that data management be addressed at the beginning of the monitoring program and become central to its development.

Planning for a central data management scheme needs to begin immediately. The data management system should be a distributed one, accessible through a web-based platform. Data will be entered directly from the field, range checked, and held for data review and editing. After the data review and editing, the data will be entered into the central database. These data will then be accessed through a web-based platform for analyses. There will also be a regularly updated web-based interface for the general access. The following tasks will need to be completed for the creation of the centralized database:

1. Development of a common set of spatial and user identification data fields to allow queries and relationships between data sets;
2. Software programming for a fully relational database and interface;
3. Metadata standards to facilitate use of a metadata catalog for keyword and thematic searches across data sets;
4. Guidelines for standardizing data structures and management; and
5. Identification of lookup tables and data definition standards.

More detailed descriptions of these tasks can be found in Toshach et al. (2007). In addition, there are a number of preexisting data sets that will need to be incorporated into the database and standards for their conversion will need to be developed.

Data Flow

Data flow for the monitoring would include: 1) the capture of data collected by field operations including geolocation, 2) transfer of raw data to the database, 3) data editing and range checking, and 4) making these data available to agencies and the public. Each of these steps will contribute to improved data collection, organization, management, estimation, interpretation and display to the user audiences.

Actual data collection activities will occur over a wide range of locations and conditions, including wet winter sampling. Data recording in the field can be accomplished by using direct data recorders. Field electronic devices programmed for data entry are currently widely used for this purpose. These data entry systems have the advantage that some data checking can occur at the time of data collection (Johnson et al. 2009). Most importantly, this will minimize the time required for data entry and the errors introduced during data entry. The direct data recorders can then be downloaded into laptops each day and backed-up onto external media or if possible to the central database via a web-based platform. These direct data recorders are not expected to be a major program expense and are preferred to the traditional method of having support personnel enter field data, usually at a remote location, without oversight. Their disadvantage is if they fail, the crew could potentially lose all of the raw data collected on any given day's field survey because there is no paper backup. Timely archiving of daily field data in both electronic and hard-copy formats will mitigate or eliminate this disadvantage of the direct data recorders. These data logging methods are now widely used and data losses have not been significant. Some of the authors have used these methods for over seven years and have not experienced this sort of data loss.

Data can be uploaded over the web to a central database upon the field crew's return to field offices. This will offer an additional chance for data editing and range checking and also allows a rapid transfer to the central data management center. The close to real time data transfer allows almost immediate data summary and examination and will result in much better operational control for monitoring sampling activities.

Raw data from field activities will be managed by a central data management system. This central data management system will be part of the operational control of the sampling as well as charged with basic data management of the raw field collection data. The data center will be responsible for regional data management, including final data editing and storage of data and development of data expansion factors and variance calculations. This central data management also will be responsible for annual data reports and summaries that will be necessary to insure that the year's sampling is being completed in a timely fashion. This type of in-season sampling information is invaluable for operational control of sampling activities.

FUTURE DIRECTIONS AND PLAN REFINEMENT

The strategy, design and methods outlined here, as well as data collection methods and analytical tools will be continuously reviewed in an effort to improve accuracy, implementability, usefulness, and cost effectiveness. CDFG and NOAA Fisheries have convened joint agency committees to oversee the initial development and implementation of a coastal monitoring program. Joint working groups will be established to address the technical field and scientific analysis responsibilities and data management (e.g., database structure, storage, retrieval, distribution). Once these groups are established and operating, a multi-agency advisory committee will be established, to provide a forum for State, federal, county, academic, and private partners to collaborate with CDFG and NOAA in implementing and maintaining a statewide monitoring program. The objectives of this organizational strategy are to ensure consistent permanent membership to oversee the program, analyze alternative methods, implement program improvements, apply consistent methodologies across the State, maintain a single, comprehensive data set, promote collaboration with other organizations and agencies, and advance widespread availability of monitoring data, analyses, and reports.

The following list contains important initial tasks and investigations that will be taken up by the joint CDFG-NOAA committees. Many of these issues were either identified by the authors in the process of writing this technical summary or brought to our attention by reviewers. However, in the interest of beginning program implementation as soon as possible, and because many monitoring elements are subject to multiagency approval, these issues could not be resolved in this paper. These issues, and others, will be taken up and resolved by the appropriate CMP committees or working groups as program implementation progresses.

1. Finalize the habitat monitoring plan to be integrated with CMP population monitoring;
2. Improve analytic techniques to provide answers from sampling data (Inference Design);
3. Add steelhead in the Northern area to spatial structure monitoring, with a minimum goal of establishing baseline distribution over their range in California;
4. Explore the potential value of incorporating additional or different analytical methods and tools (e.g., local neighborhood variance estimator);

5. Identify and implement research components that would improve CMP data collection, sampling strategy, and data analysis;
6. Further develop the relationship of redd counts to true abundances for all species and locations across the northern monitoring area. Explore the alternative use of local polynomial regression estimates (Breidt and Opsomer 2000) to evaluate these relationships;
7. Use initial years' data to evaluate and improve allocation of sample size to each panel type. Revisit sample size required to obtain sufficient power to detect trends and sufficient precision to accurately estimate status;
8. Investigate PIT tag-based methods for estimating separate winter and summer juvenile survival rates at LCM stations, specifically when methods would allow for evaluating habitat restoration activities;
9. Initiate studies to measure redd detection and inter-observer reliability of redd surveys in a variety of locations across the northern monitoring area. Studies have shown that redd detection was not a significant issue. These studies were conducted on bull trout, a non-anadromous salmonid with a much shorter spawning season than CMP's target species. Investigations based on coho salmon and steelhead in California would provide corroboration and support for this methodology under CMP;
10. Consider using the SVB metric to assess whether spatial structure occupied sites are dispersed or clumped;
11. Add coastal cutthroat trout as a target species;
12. Finalize the Shaffer document; and
13. Establish a website portal to provide organizations and the public with information and guidance on a) status and trend monitoring for anadromous salmonids and b) the State's coastal monitoring program.

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