APPENDIX E.
ASSESSMENTS OF CURRENT STATUS

This chapter was drafted in 2004 and has not been revised for 2010. Revised analysis is reported in Volume I.

APPENDIX E. ASSESSMENTS OF CURRENT STATUS

Tables

Figures

E.1. Assessments of Current Status and Limiting Factors

Other sections of this Technical Foundation summarize the available information on fish status, limiting factors, and recovery standards. This chapter includes assessments of current population status relative to potential recovery benchmarks for each focal fish species. This chapter also describes analyses, based on a synthesis of the best available scientific information, of the relative significance of six actors for decline: fishing, hatcheries, stream habitat, mainstem and estuary habitat, dams, and predation. (Only factors within the realm of human management were included.) These evaluations provide a road map of possible avenues for recovery and a basis for more detailed assessments of recovery scenarios and strategies in the next phase of the recovery planning process. The assessment approach is an adaptation of alternatives previously identified in the Lower Columbia Fish Recovery Analytical Framework (TWC and SPCA 2003).

For effective interpretation by both highly technical scientific professionals and an informed lay audience, descriptions of current status and factors for decline must be technically defensible, based on the best available data, as well as intuitively easy to interpret. A sound technical approach was needed to provide effective guidance and to withstand intense scientific scrutiny. “Best available data” is the standard for evaluating Endangered Species assessments. In many cases, the “best available” may be less than ideal but scientific information can support informed decisions, provide direction, reduce uncertainty, and generate testable hypotheses even where the data is not definitive. Finally, descriptions need to be intuitively easy to understand by a mix of technical and non-technical people who will be called upon to make scientific and policy decisions based on this data.

Specific assessments for each species include: 1) estimates of current viability for each population, 2) comparisons of current fish numbers with recovery planning ranges, 3) descriptions of the biological significance of each population, 4) indices of the relative effects of each limiting factor for each fish population, and 5) subjective summaries of the recovery prospects for each focal fish species. Estimates of current viability provide a systematic representation of current status. Planning ranges will help identify biological objectives for recovery planning relative to the healthy and harvestable goal identified by the LCFRB. Biological significance will provide a useful index for sorting populations in future considerations of alternative recovery scenarios. Indices of limiting factor effects will help inventory threats to viability and potential avenues for recovery. Summaries will help highlight potentially effective recovery strategies.

E.2. Current Viability

The first step towards recovery is understanding current population viability, the long-term prospects for preservation of a naturally self-sustaining population. A population is viable where persistence probabilities are high. High persistence probabilities correspond to low extinction risks and constitute recovery for key species units (Evolutionarily Significant Units) under the Federal Endangered Species Act. Minimum component population levels required to ensure that ESUs do not go extinct constitute the low end of recovery planning targets identified by the Lower Columbia River Fish Recovery Board.

We evaluated viability based on standards developed by the Willamette/Lower Columbia Technical Recovery Team (TRT), consisting of a committee of scientists convened by NMFS to provide technical guidance in fish recovery. As detailed in the previous chapter on Recovery Standards, TRT viability guidelines are based on scores assigned to attributes related to the viability of each individual fish population within an ESU. Attributes include spawner abundance, productivity, juvenile outmigrant numbers, diversity, spatial structure, and habitat conditions (McElhany et al. 2003). Each population is rated for each attribute on a 0-4 scale based on the available information. Individual attribute ratings are averaged for each population. The rating scale corresponds to 100-year persistence probabilities: 0=0-40%, 1 = 40-75%, 2 = 75-95%, 3 = 95-99%, 4 > 99%. Population scores can then be counted and averaged across a geographic strata for each species for comparison with recovery benchmarks established by the TRT. The lower Columbia region includes Coast, Cascade, and Gorge strata identified by the TRT to capture within-ESU differences in population characteristics related to differences in geographical and environmental conditions in different ecological zones. These benchmarks include a strata average persistence probability greater than 2.25 with at least two populations at high persistence probabilities (≥3.0). Because this viability approach is a building block for population significance, it is described in more detail below.

Population status was scored independently by the TRT and by Washington or Oregon fish biologists with specific knowledge and expertise on lower Columbia River salmon populations. TRT and State scores were averaged for the purposes of this evaluation. Independent estimates in Washington were completed by LCFRB scientific consultants (Ray Beamesderfer and Guy Norman) and WDFW staff (Dan Rawding). Table 10-1 includes more detailed explanations of criteria applied to Washington scores. Population-specific rationales for LCFRB Washington scores may be found in technical appendices of Volume VI of this Technical Foundation. Oregon estimates were completed by Oregon Department of Fish and Wildlife Staff. Most of the Technical Foundation has been focused on lower Columbia River salmon populations in the Washington jurisdiction of this recovery planning effort. However, assessments of ESU viability also require information on Oregon populations. Recovery criteria address ESU-wide status and prospects for recovery. We therefore included summary information on Oregon stock status in this assessment to provide a context for Washington planning considerations.

Population trends and extinction risks are also reported based on analyses of population time series data by NMFS. TRT scores and time series analyses are alternative but related approaches to assessing population viability that can be used for cross-corroboration. In the NMFS time series analyses, abundance trends were described with median annual growth rates (λ) based on slopes fit to 4-year running sums of abundance (Holmes 2000). Values less than and greater than 1.0 indicate decreasing and increasing trends, respectively, over the period of record. Extinction risks were based on two different models that make slightly different assumptions about future patterns from recent abundance time series data. The first model estimates the probability of extinction using the Dennis-Holmes method based on the risk that a population starting with the most recent four year sum will decline to less than 50 spawners given the population growth rate (λ) and observed variation in abundance. The second model uses population growth rate and variance derived from time series data with different statistical assumptions and also incorporates a nonlinear stock-recruitment population function (McElhany et al. 2003).

Current population sizes were also compared with historical "template" numbers to provide a perspective on differences that have contributed to current viability. Historical numbers were available from EDT analyses based on assumed habitat conditions. For comparison, historical numbers were also independently estimated by NMFS based on a simple "back-of-envelope" (BOE) calculation – these estimates were only presented in our tables for comparison and were not used in the final summaries of this Technical Foundation. The BOE calculations extrapolated an assumed historical abundance of each ESU from literature sources and partitioned the total into populations based on respective fractions of accessible stream miles. The BOE was likely confounded by an assumption that all accessible streams supported similar densities of fish and relied on an assumed historical Columbia River run size. On the other hand, EDT estimated different stream-specific densities based on assumed differences in habitat conditions and relationships between habitat conditions and fish numbers.

Table E10-1. Population persistence categories used to score fish status relative to recovery criteria guidelines (Descriptions from McElhany et al. 2003, applications identified by WDFW & LCFRB staff).

Category	Description	Application[1]
	Population Persistence	
0	Either extinct or very high risk of extinction	Very low (0-40%) probability of persistence for 100 years
1	Relatively high risk of extinction	Low (40-75%) probability of persistence for 100 years
2	Moderate risk of extinction	Medium (75-95%) probability of persistence for 100 years
3	Low (negligible) risk of extinction	High (95-99%) probability of persistence for 100 years
4	Very low risk of extinction	Very High (>99%) probability of persistence for 100 years
	Adult Abundance and Productivity	
0	Numbers and productivity consistent with either functional extinction or very high risk of extinction	Extinction risk analysis estimates 0-40% persistence probability.
1	Numbers and productivity consistent with relatively high risk of extinction	Extinction risk analysis estimates 40-75% persistence probability.
2	Numbers and productivity consistent with moderate risk of extinction	Extinction risk analysis estimates 75-95% persistence probability.
3	Numbers and productivity consistent with low (negligible) risk of extinction	Extinction risk analysis estimates 95-99% persistence probability.
4	Numbers and productivity consistent with very low risk of extinction	Extinction risk analysis estimates >99% persistence probability.
	Juvenile Out-Emigrants	Evaluated based on the *occurrence* of natural production, whether natural production was *self sustaining* or supplemented by hatchery fish*, trends* in numbers, and *variability* in numbers.
0	Consistent with either functional extinction or very high risk of extinction[3]	No significant juvenile production either because no natural spawning occurs or because natural spawning by wild or hatchery fish occurs but is unproductive.
1	Consistent with relatively high risk of extinction[3]	Long term trend in wild natural production is strongly negative. Also includes the case where significant natural production occurs in many years but originates primarily from hatchery fish.
2	Consistent with moderate risk of extinction[3]	Sample data indicates that significant natural production occurs in most years and originates primarily from naturally-produced fish. No trend in numbers may be apparent but numbers are highly variable with only a small portion of the variability related to spawning escapement.
3	Consistent with low risk of extinction[3]	Sample data indicates significant natural production by wild fish occurs in all years. No long term decreasing trend in numbers is apparent. Juvenile numbers may be variable but at least some of this variability is related to fluctuations in spawning escapement.

Category	Description	Application[1]
4	Consistent with very low risk of extinction[3]	Sample data indicates significant natural production by wild fish occurs in all years. Trend is stable or increasing over extended time period. Variability in juvenile production is low or a large share of the observed variability is correlated with spawning escapement.
	Within-Population Spatial Structure	
0	Spatial structure is inadequate in quantity, quality[2], and connectivity to support a population at all.	*Quantity* was based on whether all areas that were historically used remain accessible. *Connectivity* based on whether all accessible areas of historical use remain in use. *Catastrophic risk* based on whether key use areas are dispersed among multiple reaches or tributaries. Spatial scores of 0 were typically assigned to populations that were functionally extirpated by passage blockages.
1	Spatial structure is adequate in quantity, quality[2], and connectivity to support a population far below viable size	The majority of the historical range is no longer accessible and fish are currently concentrated in a small portion of the accessible area.
2	Spatial structure is adequate in quantity, quality[2], and connectivity to support a population of moderate but less than viable size.	The majority of the historical range is accessible but fish are currently concentrated in a small portion of the accessible area.
3	Spatial structure is adequate in quantity, quality[2], and connectivity to support population of viable size, but subcriteria for dynamics and/or catastrophic risk are not met	Areas may have been blocked or are no long used but fish continue to be broadly distributed among multiple reaches and tributaries. Also includes populations where all historical areas remain accessible and are used but key use areas are not broadly distributed.
4	Spatial structure is adequate to quantity, quality, connectivity, dynamics, and catastrophic risk to support viable population.	All areas that were historically used remain accessible, all accessible areas remain in use, and key use areas are broadly distributed among multiple reaches or tributaries.
	Within-Population Diversity	
0	All four diversity elements (life history diversity, gene flow and genetic diversity, utilization of diverse habitats2, and resilience and adaptation to environmental fluctuations) are well below predicted historical levels, extirpated populations, or remnant populations of unknown lineage	*Life history diversity* was based on comparison of adult and juvenile migration timing and age composition. *Genetic diversity* was based on the occurrence of small population bottlenecks in historical spawning escapement and degree of hatchery influence especially by non local stocks. *Resiliency* was based on observed rebounds from periodic small escapement. Diversity scores of 0 were typically assigned to populations that were functionally extirpated or consisted primarily of stray hatchery fish.

Category	Description	Application[1]
1	At least two diversity elements are well below historical levels. Population may not have adequate diversity to buffer the population against relatively minor environmental changes or utilize diverse habitats. Loss of major presumed life history phenotypes is evident; genetic estimates indicate major loss in genetic variation and/or small effective population size. Factors that severely limit the potential for local adaptation are present.	Natural spawning populations have been affected by large fractions of non-local hatchery stocks, substantial shifts in life history have been documented, and wild populations have experienced very low escapements over multiple years.
2	At least one diversity element is well below predicted historical levels; population diversity may not be adequate to buffer strong environmental variation and/or utilize available diverse habitats. Loss of life history phenotypes, especially among important life history traits, and/or reduction in genetic variation is evident. Factors that limit the potential for local adaptation are present.	Hatchery influence has been significant and potentially detrimental or populations have experienced periods of critical low escapement.
3	Diversity elements are not at predicted historical levels, but are at levels able to maintain a population. Minor shifts in proportions of historical life-history variants, and/or genetic estimates, indicate some loss in variation (e.g. number of alleles and heterozygosity), and conditions for local adaptation processes are present.	Wild stock is subject to limited hatchery influence but life history patterns are stable. Extended intervals of critical low escapements have not occurred and population rapidly rebounded from periodic declines in numbers.
4	All four diversity elements are similar to predicted historical levels. A suite of life-history variants, appropriate levels of genetic variation, and conditions for local adaptation processes are present.	Stable life history patterns, minimal hatchery influence, no extended interval of critical low escapements, and rapid rebounds from periodic declines in numbers.
	Habitat	
0	Habitat is incapable of supporting fish or is likely to be incapable of supporting fish in the foreseeable future	*Unsuitable habitat.* Quality is not suitable for salmon production. Includes only areas that are currently accessible. Inaccessible portions of the historical range are addressed by spatial structure criteria[2].
1	Habitat exhibits a combination of impairment and likely future conditions such that population is at high risk of extinction	*Highly impaired habitat.* Quality is substantially less than needed to sustain a viable population size (e.g. low bound in target planning range). Significant natural production may occur in only in favorable years.
2	Habitat exhibits a combination of current impairment and likely future condition such that the population is at moderate risk of extinction	*Moderately impaired habitat.* Significant degradation in habitat quality associated with reduced population productivity.
3	Habitat in unimpaired and likely future conditions will support a viable salmon population	*Intact habitat.* Some degradation in habitat quality has occurred but habitat is sufficient to produce significant numbers of fish. (Equivalent to low bound in abundance target planning range.)
4	Habitat conditions and likely future conditions support a population with an extinction risk lower than that defined by a viable salmon population. Habitat conditions consistent with this category are likely comparable to those that historically existed.	*Favorable habitat.* Quality is near or at optimums for salmon. Includes properly functioning through pristine historical conditions.

[1] Rules applied for each TRT criteria and category to develop integrated status assessments for example purposes of this technical foundation. Application rules were derived by project staff working in close association with WDFW staff. Application rules do not represent assessment by the Technical Recovery Team.
[2] Because recovery criteria are closely related, draft category descriptions developed by the Technical Recovery Team often incorporate similar metrics among multiple criteria. For instance, habitat-based factors have been defined for diversity, spatial structure, and habitat standards. To avoid double counting the same information, streamline the scoring process, and provide for a systematic and repeatable scoring system this application of the criteria used specific metrics only in the criteria where most applicable. This footnote denotes these items.
[3] This is a modification of the interim JOM criteria identified by the TRT. JOM scores consistent with persistence probabilities for other criteria. Consistent with an attempt to avoid double counting similar information in different criteria, data quality considerations were not included in the revised JOM criteria descriptions because they are scored separately for all criteria. This modification removes confounding effects of cases where no JOM data is available and provides

E.3. Recovery Planning Ranges

E.3.1. Definition

Recovery planning ranges provide approximate benchmarks for describing the biological objectives of recovery. Planning ranges are fish numbers for each population at: 1) minimum averages needed to ensure population viability (i.e. avoid extinction) and 2) realistic maximums that might be achieved by widespread restoration of favorable habitat conditions for salmon. The low bound of the planning range thus represents potential delisting goals for ESA populations. The high bound represents a limit to potential expectations rather than a goal.

Planning ranges were described both in terms of spawner numbers and population productivity. Greater fish numbers generally correspond to greater population productivity and increased population viability. Each alternative for describing status lends itself to different applications and analyses. Fish numbers can be measured directly and provide an intuitively easy-to-understand description of how well a population is doing. Productivity (replacement rate) provides a more direct description of the dynamics that determine status and viability. Viability level reflects persistence probabilities and extinction risks that are a particular concern for conservation and preservation of sensitive populations including those listed under the ESA.

Comparisons of current numbers and planning ranges provide an index of the difference between current, viable, and potential values (Figure E10-1). The low bound of the planning range is equivalent to a high level of viability as described by the Willamette/Lower Columbia Technical Recovery Team. Very high levels of viability are assumed to occur at population levels less than the potential reflected by the high bound on the planning range.

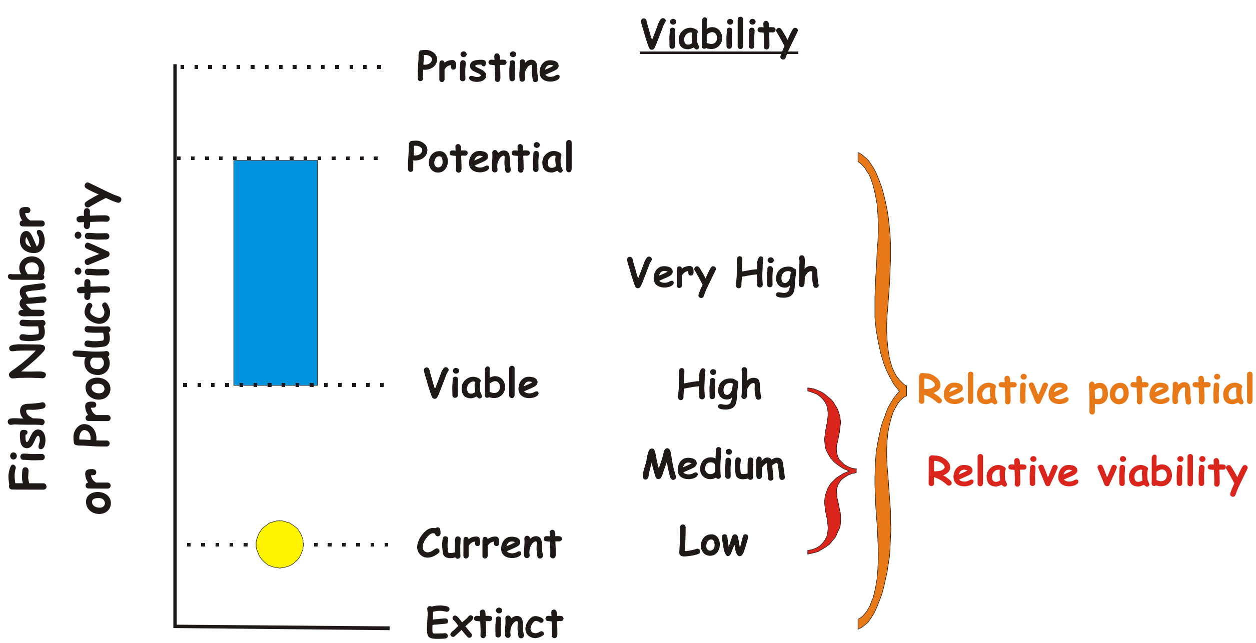

Figure E10-1. Depiction of generic recovery planning ranges relative to viability levels identified by the Willamette/Lower Columbia Technical Recovery Team.

E.3.2. Derivation

The low bound of the planning range was generally based on Population Change Criteria (PCC) developed by NMFS. PCC determines the population growth rate and average abundance after 20 years needed to minimize risks of falling below critical low population sizes over 100 years. Estimates were based on recent 4-year average spawning escapement of naturally produced fish for each population and annual variation in escapement of each species (McElhany et al. 2003).

Planning range abundance values at viability were expressed as 4-year average spawner numbers. Default PCC values of 600, 1,100, and 1,400 spawners were used for steelhead, chum, and Chinook, respectively, where either spawning escapement data were not available, numbers were thought to average less than 150 spawners per year, or estimated PCC values were less than default values. In populations where the available assessments indicate that extinction risks are not significant (i.e. less than 5% within 100 years), current abundance (recent 4-year natural spawning escapements) values were used as the low bound rather than the PCC values. (PCC derivation is based on assumption of an at-risk population. Where the population is not at risk, PCC numbers are undefined. Spawner numbers rather than EDT-derived population estimates (Neq) were used for comparability with PCC units.) Where PCC numbers exceed potential habitat capacity under properly functioning conditions estimated using EDT, the PFC+ EDT value was used as a minimum and no upper bound was specified. (This situation most commonly results from the apparent presence of large numbers of naturally-produced spawners from hatchery-origin spawners in preceding generations.)

Planning range productivity values at viability were expressed as median annual population growth rates (λ). Current estimates were derived by NMFS from escapement time series data analyses (Holmes 2000). Population productivity values needed to achieve PCC growth rates require proportionately larger increases where λ is less than 1.0 (McElhany, personal communication). Thus, viable median annual population growth rates were: $(1 + \Delta\lambda)$ where $\Delta\lambda$ = population change criteria for productivity derived by McElhany et al. (2003). Default PCC values ($\Delta\lambda$) of 9%, 14%, and 15% population growth per year were used for steelhead, chum, and Chinook, respectively, where population-specific PCC estimates were not available.

The upper end of the planning range represents the theoretical capacity if currently-accessible habitat was restored to good, albeit not pristine, conditions represented by the "properly functioning habitat conditions" identified by NMFS. Abundance and productivity at PFC was estimated using the Ecosystem Diagnosis and Treatment Model as describe in Volumes II and VI of this Technical Foundation. PFC describes stream conditions suitable for salmon throughout the accessible range. In this application, the upper end of the planning range also assumed no removal of existing dams, no fishing, and the estuary at historical productivity levels. PFC stream habitat conditions and historical estuary productivity levels are typically referenced as PFC+ to distinguish from PFC stream habitat conditions with current estuary productivity levels.

The upper bound of the abundance planning range was defined in terms of equilibrium spawner numbers. Equilibrium numbers are long term averages that can be expected based on average marine survival patterns. For planning purposes, we conservatively assumed an upper bound of two times the lower abundance bound where EDT was not available.

The upper bound of the productivity planning range was based on EDT values which are expressed as the asymptotic Beverton-Holt recruit per spawner parameter (the slope at origin or β-1: Ricker 1975). This parameter describes maximum adult spawner per spawner values which are realized at low spawner numbers. Spawner/spawner parameters were transformed into equivalent median annual population growth rates based on the following assumption:

$$\lambda_{pfc+} / \lambda_{current} = \text{Ln } \beta^{-1}{}_{pfc+} / \text{Ln } \beta^{-1}{}_{current}$$

Available estimates $\lambda_{current}$ (Holmes 2000), $\beta^{-1}{}_{pfc+}$ (EDT), and $\beta^{-1}{}_{current}$ (EDT) were used to solve for λ_{pfc+}.

E.3.3. Improvement Increments

Recovery scenarios based on TRT guidelines prescribe biological objectives that target different recovery levels for different populations. Some populations need to be restored to high levels of viability. Other populations need to be improved to contribute to ESU viability but need not reach high levels of viability. Yet other populations need to reach very high levels of viability to compensate for recovery uncertainties and to provide opportunities for other uses such as harvest. Comparisons of current status with recovery planning ranges provide a means of estimating improvement increments necessary to reach any given population level. Increments based on productivity differences also provide a means for relating necessary improvements to manageable impact factors.

Proportional improvements in population productivity were estimated for recovery of populations from current status to contributing, high, and very high levels of population viability consistent with recovery scenarios. Improvements to reach high levels of viability were based on the difference between current and viable median annual population growth rates. Thus, proportional productivity improvements to reach viability (θhigh) are:

$$\theta_{high} = [(1 - \lambda) + \Delta\lambda] / \lambda$$

Contributing populations were arbitrarily assumed to increase half the distance between current and viable productivities:

$$\theta_{contributing} = \theta_{high} / 2$$

Populations at very high levels of productivity were arbitrarily assumed to increase to half the distance between viable and potential (e.g. the mid-point of the recovery planning range):

$$\theta_{very\ high} = \theta_{high} + (\theta_{potential} - \theta_{high}) / 2$$

where

$$\theta_{potential} = (\text{Ln } \beta^{-1}{}_{pfc+} - \text{Ln } \beta^{-1}{}_{current}) / \text{Ln } \beta^{-1}{}_{current}$$

This alternative was chosen instead of using PFC+ for high viability under the presumption that persistence probability will approach 100% in many populations under conditions well below PFC+.

Average species values were used for θ high where population-specific values were not available. We used whichever produced the greater increment: A) average of viable population productivities from populations with data or B) average of incremental improvements needed to move from current to viable in populations with direct estimates. Also note that in cases where λ was greater than 1.0, we assumed that it was 1.0. These assumptions were needed to reconcile differences between λ estimates and TRT status score assignments. For instance, some population productivities already exceed the viability average yet were scored as not viable under TRT criteria. Otherwise we would be saying no improvement is needed to get to viable for populations that were scored to be less than viable.

Estimated productivity increments highlight order-of-magnitude improvements in productivity needed to reach recovery. Population-specific estimates should be considered with caution because of large uncertainties in assessments. Species averages and ranges provide general guidelines. These estimates build upon results of existing analytical frameworks (EDT & PCC) to make a first approximation of the scale of needed improvements. Both EDT and PCC relied on simplifying and sometimes differing assumptions. Our extrapolation of results is also beyond the immediate intended application of each method. Given the ultimate uncertainty in the effects of recovery actions and the need to implement an adaptive recovery plan, this approximation should be adequate for developing order-of-magnitude estimates to which recovery actions can be scaled consistent with the current best available science and data. However, the adaptive research and evaluation component of the recovery plan should include data collection and further analysis based on an integrated life cycle framework that meshes an age-structured density-dependent population model like EDT with a stochastic empirical approach like PCC to directly relate persistence probabilities to population productivity.

E.4. Population Significance

To facilitate future development of recovery scenarios consistent with biological guidelines for recovery, we developed a simple index to systematically rate the biological significance of each population based on the available data. Biological significance is one of several elements including feasibility, equity, and efficiency that will considered in the development of recovery scenarios. Biological significance will inform but not necessarily drive the selection of recovery scenarios. For instance, less “significant” populations or subbasins might be targeted for more intensive recovery efforts where feasibility is greater.

The biological significance of each fish population can be described in terms of current viability, potential production, and genetic character:

Current viability: likelihood that a population will not go extinct within a given time frame. The healthiest, most robust current populations are the most viable.

Core potential: number of fish that could be produced in a given area if favorable historical conditions could be at least partially restored.

Genetic character: current resemblance to historical characteristics that were intended to be preserved.

Specific guidelines related to each of these attributes are the basis for population viability criteria identified by the Willamette/Lower Columbia Technical Recovery Team (McElhany et al. 2003). For instance, current viability was defined by the TRT in terms of population persistence probability. (Current viability was based on the scoring approach described in the previous section). Potential production is related to the TRT core population designation. Core populations “represented the substantial portion of the ESU’s abundance or contained life-history strategies that were specific to the ESU.” Thus, core populations were typically the largest historical populations. Finally, the TRT designated genetic legacy populations as having “minimal influence from non-endemic fish due to artificial propagation activities, or the population may exhibit important life-history characteristics that are no longer found throughout much of their historical range in the ESU.”

Biological significance ratings (B) were calculated for each population based on the following formula:

$$B = (V + C + G)/3$$

where

V = Current viability (Where are we now relative to the viability goal?)
C = Core potential (What is the potential of each population to produce fish?)
G = Genetic legacy (Which populations warrant extra consideration because they are most representative of the historical fish characteristics we are intent on preserving?)

The index is the simple arithmetic average of each of the three elements. Each factor was standardized to a scale of 0-1 so that each contributes equal weight in the calculation, unless there were compelling reasons for elevating any individual factor. Note that the TRT also identified criteria based on catastrophic risks that are not incorporated into this population index. Catastrophic risks are better considered later in the scenario development process where the net effect of population-specific risks on the strata risk can be controlled by the choice of specific combinations of populations (e.g. populations that are not next the same volcano.)

To facilitate qualitative consideration of biological significance in future development of recovery scenarios, populations were sorted in descending order and separated into up to 3 categories where values were similar. Categories were labeled A, B, and C. Splits were made based on incremental changes in the sequence within each strata. Categories represent rank relative to other populations within a species. Thus, each category may not be represented in every strata.

Current **population viability** (V) was calculated for each population based on the following formula:

$$V = P / 3.0$$

where

P = Population persistence category based on TRT criteria (see preceding section):

0 = very high risk of extinction (0-40% persistence probability in 100 years).
1 = high risk of extinction (40-75% persistence probability in 100 years).
2 = medium risk of extinction (75-95% persistence probability in 100 years).
3 = low risk of extinction (95-99% persistence probability in 100 years).
4 = very low risk of extinction (>99% persistence probability in 100 years).

Population persistence scores were based on fish population data for abundance, productivity, juvenile emigrant numbers, spatial structure, diversity, and habitat. According to TRT recovery guidelines, a population persistence score of 3 would correspond to a viable population (i.e. recovery under ESA). Thus, dividing the population persistence score by 3 normalized this element to a scale from 0 to 1.25 with a score of 1.0 denoting a viable population. A score of >1.0 would give extra credit for populations recovered to even greater levels although, because of the way TRT scores are defined, a score of greater than 3.0 is practically very difficult to achieve for any given population. Population persistence scores were standardized so that they would be equally weighted with potential production and genetic character scores that also contributed to the biological significance index.

Use of all TRT population persistence criteria (abundance, productivity, juvenile emigrant numbers spatial structure, diversity, and habitat) to index population viability will facilitate mapping population conditions back to specific TRT viability factors that can be addressed with specific recovery actions. Note that Population Change Criteria (PCC) thresholds identified by NMFS are not used directly in this approach but are implicit in the population persistence scores. We did not use the PCC viability thresholds because current population sizes used in the derivation of those thresholds are 4-year averages, are confounded in many populations by natural offspring of hatchery fish spawning in the wild, and may not be representative of long-term wild fish numbers.

Core potential (C) was calculated for each population based on the following formula:

$$C = NEQ_{PFC+} / mNEQ_{PFC+}$$

where

NEQ_{PFC+} = Potential population size if favorable habitat conditions are restored throughout the subbasin of origin [realized habitat capacity (equilibrium population size or Neq) inferred with EDT model from habitat data with universal restoration of Properly Functioning Conditions identified by NMFS plus estuary habitat improvements].

$mNEQ_{PFC+}$ = Maximum potential population size projected for any population of a given species and run type under favorable habitat conditions.

This approach addresses core population criteria of the TRT with EDT-based data. Core populations were designated by the TRT based on a qualitative review of the available information and expert opinion. However, EDT results represent the best available data on historical and potential size of each population. Core population designations by the TRT closely correspond with the core population potential estimates from EDT, but EDT estimates also provide for incremental scaling of core population potential rather than the all-or-nothing nature of the core designation. This data-driven approach thus provides for more fine-scale evaluations. Standardization of core population potential estimates versus the potential for the largest population in the species and run results in values being scaled from 0 to 1 where 1 is the largest potential population in each stratum. Values can then be compared among strata to flag the largest potential populations. Values are comparable among strata in an absolute scale.

Genetic Legacy (G) was scored directly from TRT designations:

$$G = \{1, 0\}$$

where

1 = Genetic legacy population according to TRT

0 = Not a genetic legacy population according to TRT

The all-or-nothing nature of the TRT designation flags key stocks but does not capture intermediate increments of genetic characteristics that might provide further guidance for scenario development. We examined data-driven approaches to quantifying the degree of genetic legacy for each population but suitable alternatives were limited by the available data. We often have good recent data on hatchery release numbers and broodstock origins as well as anecdotal historical information. For instance, the NMFS escapement dataset documents annual hatchery fraction in the natural escapement where data is available. Similarly, the recent NMFS status review classified the divergence from the wild for current hatchery stocks throughout the basin (1-4 scale where 4 is large divergence from wild). However, we lack similar information for the historical period when hatchery effects were substantially greater.

E.5. Current Limiting Factors

E.5.1. Net Effect of Manageable Factors

We evaluated factors currently limiting Washington lower Columbia River salmon and steelhead populations based on a simple index of potentially manageable impacts. The index incorporated human-caused increases in fish mortality, changes in habitat capacity, and other natural factors of interest (e.g. predation) that might be managed to affect salmon productivity and numbers. We refer to this index approach as the AEIOU Index (Adult Equivalent Impacts Occurring Unconditionally).

To inform the development of recovery scenarios and strategies by technical and policy groups, we needed to inventory key factors and place them in perspective relative to each other. The AEIOU Index is a simple screening device to help educate a diverse audience and to provide general guidance for recovery decisions. The relative importance of each factor will guide both technical decisions on what combinations of recovery measures can prove effective and policy decisions on where to focus efforts and how to balance the responsibilities and costs of the effort. In popular parlance, the factors for salmon declines have come to be known as the 4-H's: hydropower, habitat, harvest, and hatcheries.

This approach represents the relative order of magnitude of key limiting factors. It does not constitute a fine-scaled mechanistic analysis of limiting factors and dynamics of every listed population. The question was not whether a factor might be responsible for a 50% or 55% impact with a confidence interval of 5% or 50%. Rather, we needed to know whether a factor represented a 5% or 50% or 90% impact.

Only the subset of factors we can potentially manage were included in the AEIOU Index – natural mortality factors beyond our control (e.g. naturally occurring ocean mortality) are excluded (Figure 10-2). For instance, tributary habitat changes, estuary habitat changes, fishing, hydro and hatchery effects are all obviously human impacts. Natural mortality in freshwater, the estuary, and the ocean that occurs independent of human effects was factored out. Predation by fish, birds, and marine mammals was included in the analyses, although it can only minimally be managed by humans, because of the widespread public interest in the magnitude of the predation effect relative to human factors.

The index was calculated as:

$$I_x = F_x / \sum F_x$$

where

I_x = relative impact of factor x.
F_x = proportional reduction in fish numbers as a result of factor x.
$\sum F_x$ = sum total of all proportional reductions selected for inclusion.

For instance, if we were concerned only with tributary habitat availability (e.g. 50% reduction due to development) and harvest (e.g. 25% average harvest rate), impacts would be calculated:

$I_{\text{tributary habitat}} = 0.50 / (0.50 + 0.25) = 0.67$ of the impacts of concern
$I_{\text{harvest}} = 0.25 / (0.50 + 0.25) = 0.33$ of the impacts of concern

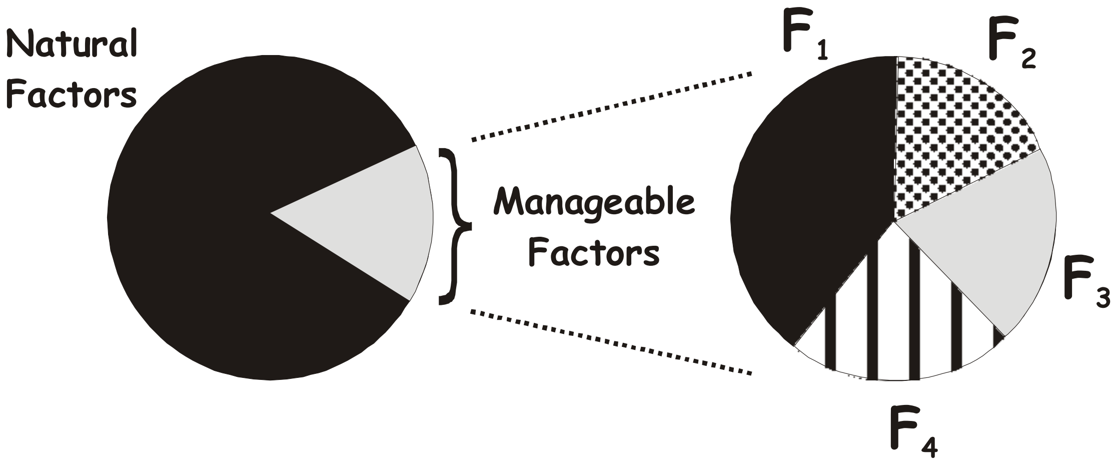

Figure E10-2. Manageable human factors affecting salmon mortality, productivity, and numbers represented as a portion of all factors and as their own pie.

With this index, the relative importance of any given factor decreases as additional factors are added. For instance, if we also included a 50% dam passage loss, the tributary habitat factor share of factors of concern is reduced from 0.67 to [0.50/ (0.50 + 0.25 + 0.50)] or 0.40. The factor effect is absolute (e.g. a 50% reduction) whereas the impact is relative to all factors of concern (0.67 becomes 0.40 when a new factor is added).

Factor level effects are most easily thought of as mortality rates. Our analyses include mortality associated with fishing, dam passage of juveniles and adult migrants, and predation by fish, birds, and marine mammals. Factor level effects also include other effects that reduce fish numbers and productivity including loss of tributary rearing capacity due to blockage and habitat degradation, reduced estuary survival due to habitat changes, and reduced natural population productivity due to interbreeding with less-fit hatchery fish.

The application of this index approach is limited to factors where we can reasonably quantify the effect. Other human-caused factors where data are sparse or effects are indirect may be overlooked or indistinguishable from natural productivity factors.

Factor level effects are described as unconditional adult equivalent effects that act independent of interactions with other factors. Unconditional factor effects are the proportional reduction in productivity or mortality of any given life stage. The reduction is relative to the potential number of that specific life stage rather than relative to numbers at an earlier or later life stage. Thus, the tributary habitat factor describes the reduction in smolt numbers relative to the number that would have been produced if habitat were unaffected, the harvest factor describes the reduction in adults relative to the number that would have survived in the absence of fishing, and so on.

Unconditional effects fairly represent factors that act on different parts of the life cycle. Each describes the proportional reduction associated with a given impact in the absence of the effects of other factors. Each factor level effect translates into an equivalent reduction in fish numbers or productivity (e.g. a 50% reduction in habitat quality reduces adult numbers by 50% just as a 25% harvest mortality reduces adult numbers by 25%). Because factor effects are unconditional, the sum of all factor effects can be greater than 1.0 where many factors are included. However, the general absence of significant density-dependent mortality factors after the freshwater rearing stage makes this approach relatively robust:

$$N = B\,(1\text{-}M)\,(1\text{-}F_1)\,(1\text{-}F_2)\,(1\text{-}F_3)\,\ldots\,(1\text{-}F_n)$$

where

N = fish numbers
B = density dependent births (e.g. eggs produced by all natural spawners on average)
M = natural fish mortality throughout the life stage
$F_1,\ldots F_n$ = proportional reduction in fish numbers as a result of factor x for n factors.

In our special case where factor level effects may be considered density-independent, the net impact (Z) of a series of unconditional effects can be estimated:

$$Z = 1 - [(1-F_1)\ (1-F_2)\ (1-F_3)\ \ldots\ (1-F_n)]$$

Thus, the net impact (Z) represents the net impact of all factors considered. We compared net impacts of potentially manageable factors including human impacts among population to identify the proportional reduction in productivity and numbers (1-E) from a historical baseline that included no human impacts. In our simple example with a 50% habitat quality reduction and a 25% harvest mortality, the net impact would be 1-[(1-0.5)(1-0.25)] or a 62.5% reduction due to habitat and harvest impacts (only 37.5% of the historical number remains).

In developing descriptions of the relative impact of various factors for decline, we considered a variety of model-based quantitative approaches currently in application by the scientific community. However, most available alternatives were based on nuances that were difficult to grasp except by the quantitative scientists who developed them or were considerably more complicated than was necessary for our purposes. Examples included elasticity (Heppell 2000) or sensitivity analyses (Zabel 2003) based on matrix population models (Casell 2001). We also examined simple run reconstruction analyses based on juvenile or adult equivalents (e.g. LCFRB 2003). Both our simple index and more complicated life cycle modeling approaches are based on similar fish demographic data which is referenced in this report.

Estimated or assumed values for impact factors represent a reasonable first approximation and may be refined by more detailed evaluations of each individual factor. In many or most cases, we lack basin-specific fish population data. In some cases, current data are available but baseline historical data is almost invariably lacking. As a result, this exercise necessarily relied on a combination of inferences from other populations or areas, indirect analyses (EDT analysis of habitat data for instance), interpretations of our current scientific understanding of fish biology and system dynamics, or working hypotheses that are testable as part of recovery plan implementation. The diverse sources and nature of the information incorporated into this exercise makes it difficult to quantify the uncertainty in specific estimates. Clearly the uncertainty in specific point estimates is significant and caveats for their application are in order. Despite these limitations, these results are accurate representations of the available scientific information for the purpose of inventorying and generally describing the order-of-magnitude significance of potentially manageable factors for decline in a simple and intuitively understandable fashion.

The impact factors described in this assessment are a beginning rather than an end of the recovery scenario and strategy development process. As soon as the relative significance of various factors for decline is understood, the obvious next questions are: how big a change is needed to achieve recovery, what combinations of factor changes will be effective, and how difficult or costly will it be to affect each individual limiting factor by any given amount. A general sense of effective changes in any given factor can be gained by comparing specific impacts with increases in population growth rate or productivity identified by NMFS. For instance, if population growth rates need to increase by 10% to reach desired population persistence probabilities, then we would need to decrease impact factors by an absolute value of 10% per year. More complex fish life cycle modeling approaches will be required to compound the effects of factors acting on different life stages, to estimate the net change in population productivity in response to combinations of recovery actions, and to relate changes to population viability. The basic mortality and productivity data incorporated into the simple AEIOU index provides some of the raw materials for these determinations.

E.5.2. Fisheries

Fishery assessments include estimates of total impacts on each population and the distribution of impacts among different fisheries. Impacts include direct harvest and catch-and-release mortality of all ocean and freshwater sport, commercial, and tribal fisheries. Extensive mortality data are available from Federal and State fishery regulatory agencies and Indian Tribes. These data are detailed for each species in earlier chapters of this Technical Foundation. Fisheries at the time of listing under the ESA are the basis for values used in the AEIOU Index of relative significance. Current rates are also reported where reductions have occurred. Population-specific estimates are typically inferred from species or stock-specific impacts rather than subbasin-specific estimates because pooled data provides more robust estimates. Historical trends in impacts are also summarized to illustrate past impacts that may have shaped current fish populations. Allocations of impacts among various ocean and freshwater fisheries will identify opportunities for considering the consequences of fishery-related recovery measures.

E.5.3. Hatcheries

To provide a conservative estimate of the potential for negative hatchery impacts on wild populations relative to other impact factors, this assessment evaluated: 1) intra-specific effects resulting from depression in wild population productivity that can result from interbreeding with less fit hatchery fish and 2) inter-specific effects resulting from predation of juvenile salmonids of other species. Fitness effects are among the most significant intra-specific hatchery risks and can also be realistically quantified based on hatchery fraction in the natural spawning population and assumed fitness of the hatchery fish relative to the native wild population. Predation is among the most significant inter-specific effects and can be estimated from hatchery release numbers by species. The index is:

$$F_{Hatchery} = F_{Intraspecific} + F_{Interspecific}$$

where

$F_{Intraspecific}$ = proportional reduction in natural productivity at equilibrium due to interbreeding of native and hatchery fish where hatchery fish are different.

$F_{Interspecific}$ = proportional reduction in natural productivity due to predation by larger hatchery smolts on smaller wild juveniles.

Intra-specific effects were estimated:

$$F_{Intraspecific} = p\,(1\text{-}f)$$

where

p = proportion of natural spawners that are of first generation hatchery origin.
f = relative productivity of native and hatchery fish (scale = 0-1).

This index assumed that equilibrium conditions have been reached for the hatchery fraction in the wild and for relative fitness of hatchery and wild fish. This simplifying assumption was necessary because more detailed information is lacking on how far the current situation is from equilibrium. In practice, actual differences in fitness of hatchery and natural fish at any given time depend on inherent differences in fitness and the degree and period of interaction (Lynch and O'Hely 2001). The index may thus over or underestimate the true current impact of hatchery spawners on wild fitness depending on past history. Current numbers of hatchery releases in each basin are also summarized to place associated risks in perspective.

The hatchery fitness index increases with the proportion of hatchery fish and decreases as hatchery fish are less productive than the wild fish (Figure 10-3). For instance, where hatchery fish comprise 50% of the natural spawners and fitness is 0, the hatchery impact index would be 0.50 (i.e. 50% reduction in productivity). Thus, in the case of random interbreeding of hatchery and wild fish, spawners would average 25% W:W, 50% W:H, and 25% H:H. The index results assumes 100% productivity of the W:W pairs, 50% productivity of the W:H pairs (average of 100% wild fitness and 0% hatchery fitness), and 0% productivity in the H:H pairs. In the alternative case where hatchery fish are equally fit with wild fish (f = 1.0), no hatchery fraction reduces wild productivity. Finally, where 50% of spawners are hatchery fish and hatchery fish fitness is only 50% of the wild fish, the hatchery impact index would be 0.25.

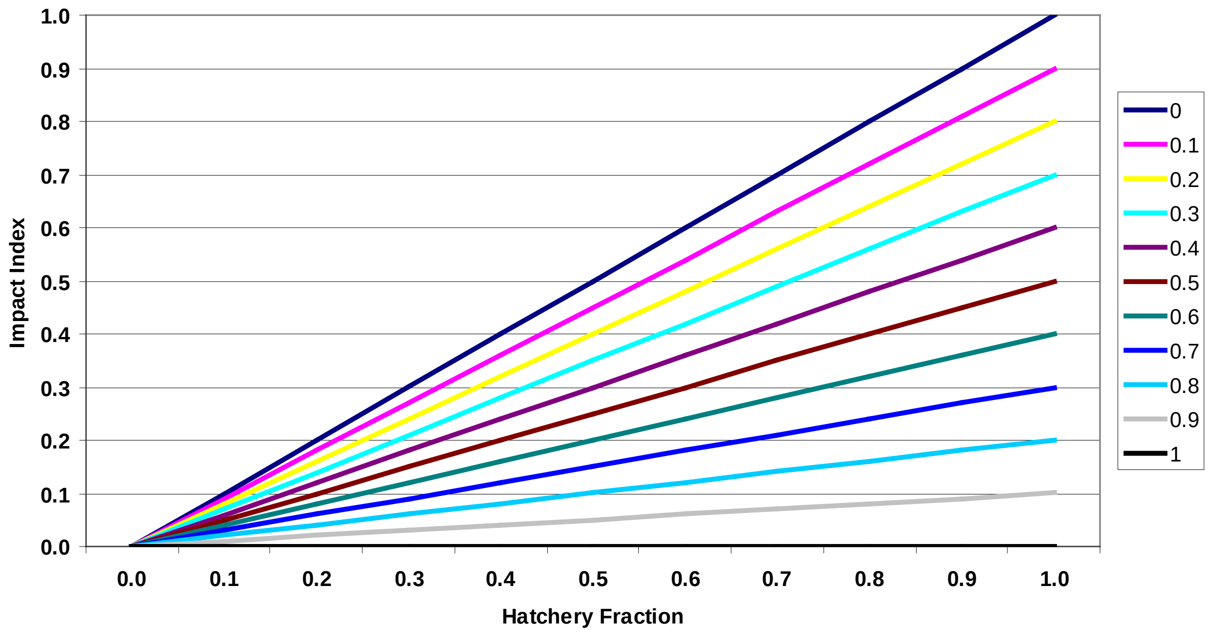

Figure E10- 3. Hypothetical effects of spawning by hatchery fish on wild population productivity relative to hatchery fraction and fitness of hatchery fish. Each line represents a different reduction in fitness (1-f) as depicted in the legend at right.

Estimates of hatchery fraction were based on spawning ground survey data (typically CWT recoveries) where available. Where specific data were not available, approximate values are inferred from adjacent systems or available anecdotal information. Hatchery fractions are based on total hatchery and wild spawners that spawn within the same period. For instance, timing differences between hatchery and wild steelhead stocks often result in much less interbreeding than might be expected based on relative numbers of spawners (LCSCI 1998). These corrections were applied to steelhead populations where substantial differences in spawn timing occur but not Chinook or chum where hatchery and wild spawn timing is similar.

Because population-specific fitness estimates are not available for most lower Columbia River populations, we applied hypothetical rates comparable to those reported in the literature and the nature of local hatchery program practices. Published information on relative fitness of hatchery and wild fish is limited (Berejikian and Ford 2003, TOAST 2004). Reisenbichler & McIntyre (1977) reported relative survival rates of Deschutes wild and Round Butte hatchery steelhead from egg to migration of 78% for H:H pairs, 80% for H:W pairs and 86% for W:W pairs. These differences are analogous to a 91%

relative fitness of Round Butte hatchery fish which were only a few generations removed from the wild at the time of the study. In the Kalama River, Chilcote et al. (1986) reported a 28% relative fitness of Kalama wild summer and Skamania hatchery summer steelhead based on smolt production. This large reduction in fitness is likely driven by the high degree of domestication in the Skamania hatchery steelhead stock. Even larger differences become apparent where the hatchery stock is substantially different than the wild stock. For instance, a relative fitness of 0% was reported by Kostow et al. (2003) for a Skamania summer steelhead in Clackamas River relative to the native winter run. Finally, Oosterhout & Huntington (2003) assumed a 70% relative fitness for coastal Oregon hatchery and wild coho based on a recommended range of 0.5 to 0.9 by a technical scientific panel.

Increasing levels of domestication and interbasin transfers were assumed to reduce fitness consistent with hatchery categories identified by the salmon Biological Review Team based on historical data (Table 10-2). We generally assumed that hatchery fish are never as fit as the wild population even under the most enlightened hatchery practices. We described relative fitness values for each BRT category based on the literature review information above.

Interspecific hatchery effects were estimated:

$$F_{Interspecific} = (N_h)\ (r)$$

where

N_h = annual hatchery releases of salmon smolts with the potential to prey on the species of interest.

r = predation impact per hatchery fish released

For instance, intra-specific effects of 1 million potentially-predacious hatchery smolts would be 5% at an impact rate of 0.5% per 100,000 smolts.

Table E10-2. Fitness values assumed to correspond to hatchery categories reported by WCSBRT (2003).

Category	Description	Fitness
1	Hatchery population derived from native, local population; is released within range of the natural population from which it was derived; and has experienced only relatively minor changes from causes such as founder effects, domestication or non-local introgression.	0.9
2	Hatchery population was derived from local natural population, and is released within the range of the natural population from which it was derived, but is known or suspected to have experienced a moderate level of genetic change from causes such as founder effects, domestication or non-native introgression	0.7
3	The hatchery population was derived predominantly from other populations that are in the same ESU, but is substantially diverged from the local, natural populations(s) in the watershed in which it is released.	0.5
4	The hatchery population was predominantly derived from populations that are not part of the ESU in question; or there is substantial uncertainty about the origin and history of the hatchery population	0.3

Inter-specifies predation rates were assumed to be species-specific because of size and distribution differences. Natural fall Chinook which rear in the lower portions of most subbasins are subject to predation by hatchery coho, winter steelhead, summer steelhead, and spring Chinook that are typically reduced in lower to middle reaches (G. Norman, personal communication). Natural coho which rear in the lower and middle portions of most subbasins are also subject to predation by hatchery coho, winter steelhead, summer steelhead, and spring Chinook. Chum salmon rear in lower subbasins and are subject to predation by winter steelhead, summer steelhead, and spring Chinook which are released in March and April before juvenile chum have emigrated. Chum salmon are assumed not to be subject to significant predation by coho because coho are released in May after chum emigration. Inter-specific hatchery predation impacts on steelhead are not an issue because wild rearing areas of small juvenile steelhead are primarily in areas upstream of hatchery release sites. Impact rates were assumed to be 0.5% per 100,000 predators for fall Chinook and chum, and 0.125% per 100,000 predators for coho (G. Norman, personal communication). Coho predation rates are less than those on the smaller fall Chinook and chum. These rates provide reasonable magnitudes of predation impacts even in subbasins with large hatchery releases.

Fitness and predation effects of hatchery fish are two of a variety of potential positive and negative effects of hatchery and wild interactions. Because this exercise is primarily concerned with risks, the index did not consider the positive demographic benefits to natural spawner numbers from the additional hatchery fish and their progeny. Consideration of the numerical benefits of hatchery spawners to natural population numbers would substantially change the calculation, especially where wild and hatchery fitness are not substantially different. Nor does the index consider ecological interactions between hatchery and wild fish other than predation (e.g. competition, nutrient augmentation, or disease transfer). The net effect of direct and indirect ecological interactions may be either positive or negative and the occurrence and significance of each interaction is practically impossible to quantify.

E.5.4. Mainstem and Estuary Habitat

The effects of human-caused changes in mainstem and estuary habitat conditions on fish numbers are particularly difficult to quantify because of their complex and poorly understood nature. Salmon are affected during crucial smolt and adult migration stages. Mainstem and estuary areas also provide critical rearing habitats, particularly for spring Chinook, fall Chinook, and chum salmon which migrate to mainstem and estuary areas at pre-smolt life stages. Estimates of the impacts of human-caused changes in mainstem and estuary habitat conditions were generally based on changes in river flow, temperature, and predation as represented by EDT analyses for the NPCC Multispecies Framework Approach (Marcot et al. 2002).

In EDT analyses, estimates of the effects of human impacts on estuary habitat (Festuary habitat) were represented as the difference in fish numbers between EDT results for Properly Functioning Conditions (NEQPFC) and Properly Functioning Conditions plus estuary restoration (NEQPFC+):

$$F_{\text{estuary habitat}} = (NEQ_{PFC+} - NEQ_{PFC}) / NEQ_{PFC+}$$

The hypothesized change in fish survival corresponding to estuary habitat changes was an explicit input of the EDT model calculations. EDT model results translate those changes into fish equivalents. This calculation is a reasonable approximation of the actual effect of estuary changes that could be more directly calculated with focused EDT analyses (L. Mobrand, personal communication 11/7/03).

Note that this definition potentially incorporates some indirect effects of dam construction and operation on fish habitat. Dam effects on fish productivity are evaluated separately where they can be distinguished from other factors.

E.5.5. Stream Habitat

Stream habitat assessments evaluate the effects of changes in subbasin watersheds and stream conditions on fish habitat quantity and quality. Analyses are based on analysis of stream habitat data using the Ecosystem Diagnosis and Treatment Model (EDT). EDT provides a systematic basis for inferring fish numbers from habitat conditions. Conditions for fish are described based on 46 habitat attributes. Habitat conditions are described for each homogenous stream reach used by the population of interest. The EDT model translates the 46 specific attributes into 17 "habitat survival factors" that represent hydrologic, stream corridor, water quality, and biological community characteristics related to habitat suitability and favorability for fish. Among other things, EDT then estimates average or equilibrium fish population sizes (Neq) based on quantitative relationships between fish and limiting habitat factors distilled from an extensive literature review of salmon limiting factors.

EDT estimates are available in most subbasins for historical (template), current (patient), and "Properly Functioning" (PFC) habitat conditions. The historical/template condition is defined as pre-non-Native American/European influence and represents a hypothetical maximum. The current/patient condition represents the immediate past few years. PFC represents favorable habitat conditions for salmonids throughout the basin based on criteria identified in a general review of salmonid habitat requirements by NMFS (1996). The difference between historical and current conditions represents the degree of habitat degradation associated with subbasin development. The difference between current and PFC conditions represents the potential for improvement in fish numbers that might be achieved by restoring favorable habitat conditions throughout a given subbasin. PFC conditions are typically less than historical baseline. Current conditions are typically estimated from the available data including physical site surveys as well inferences from geospatial data, anecdotal evidence, and expert opinion. Detailed data on historical conditions are generally unavailable and so corresponding inputs are based on assumed conditions. The uncertainty of each EDT data input is also entered into the database that serves as an input for the model. Although data limitations frequently require significant assumptions in model inputs, our applications of results presumes that the model provides robust estimates of general habitat quantity and quality for fish, especially where results are used for relative comparisons of differences among areas or changes in conditions. More detailed descriptions and discussions of EDT methods, inputs, and results may be found in Technical Appendices (Volume VI) and Subbasin Chapters (Volume II).

Human impacts on stream habitat conditions were quantified based on differences in fish numbers between current and historical habitat conditions. The specific calculation also included corrections for estuary habitat effects that were contained in the historical EDT calculation:

$$F_{\text{tributary habitat}} = \{[NEQ_{\text{Historic}} * (1 - F_{\text{estuary habitat}})] - NEQ_{\text{Current}}\} / [NEQ_{\text{Historic}} * (1 - F_{\text{estuary habitat}})]$$

where

$F_{\text{tributary habitat}}$ = Proportional reduction in fish numbers as a result of human impacts on tributary habitat quantity and quality.

NEQ_{Historic} = Hypothetical average population size under pre-development habitat conditions in the subbasin and estuary.

NEQ_{Current} = Hypothetical average population size under current habitat conditions.

$F_{\text{estuary habitat}}$ = fish effects of human impacts on estuary habitat quality (see preceding section for definitions

The estuary correction was required because the difference between historical and current estimates produced by EDT is a function of both tributary and estuary habitat changes. However, we wanted to describe tributary and estuary changes separately because of the implications for recovery strategies and actions.

E.5.6. Dams

Dam impacts include access and passage effects. Access effects are the proportional reduction in available habitat where dams block passage. Access effects also include inundation of key spawning reaches in the lower portions of Bonneville Reservoir tributaries. Access impacts were based on historical EDT estimates of fish numbers produced from blocked areas versus the total produced in the subbasin. Access effects were included for the upper portions of the Lewis and Cowlitz basins. We also incorporated an assumed 20% reduction in productivity of chum salmon spawners in the mainstem below Bonneville Dam to account for flow effects during incubation. Loss of habitat availability because of dams was considered separate from other habitat impacts in tributaries.

Passage effects are loss rates of juveniles associated with attraction and collection efficiencies as well as direct mortality in all routes of passage. Passage effects were included for populations upstream from Bonneville Dam. Juvenile passage mortality rates at Bonneville Dam were assumed to average 10% for steelhead and Chinook based on a review of the historical data in the technical foundation. Recent PIT tag studies suggest that average passage mortality rates may be less than 10% in some years. However, we hypothesize that fish from Washington tributaries in Bonneville Reservoir are more likely to pass via powerhouse 2 where guidance efficiencies and survival are less than the basin-wide average. Data were not sufficient to develop species or subbasin-specific estimates for spring Chinook (yearling migrants) and fall Chinook (subyearling migrants). We did not incorporate other dam-passage sources of mortality such as gas bubble disease and delayed passage effects. In the absence of specific data, chum salmon juvenile passage mortality was assumed to be twice that of steelhead and Chinook because chum migrate at smaller, potentially more fragile sizes during early spring periods where spill measures to divert migrants from turbine passage are not in effect.

Adult passage mortality rates were assumed to be 5% for steelhead and 10% for spring and fall Chinook based on conversion rate analyses for lower Columbia mainstem dams using dam counts, tributary escapement, and estimated harvest (US v. Oregon Technical Advisory Committee, unpublished data). Data are not available for chum salmon conversion rates but anecdotal information suggests rates are poor (G. Norman, personal communication). Consistent with this observation, we hypothesized a 50% adult upstream passage rate for chum.

Assumed dam passage mortality rates for juveniles and adults in this analysis are similar to those identified in the NPCC Multispecies Framework Approach (Marcot et al. 2002).

E.5.7. Predation

Predation impacts were based on approximate total mortality rates by northern pikeminnow, birds, and marine mammals. Detailed data on predation rates are limited, especially for marine mammals. However, anecdotal information is sufficient to generate order-of-magnitude estimates that place this impact in perspective relative to other impact factors.

Estimates of pikeminnow predation on juvenile salmonids are available for the Columbia River mainstem based on a series of studies by the Oregon Department of Fish and Wildlife and the Biological Resources Division of U.S. Geological Survey (see Limiting Factors Chapter of this Technical Foundation). Pikeminnow are of particular concern because they are among the most common salmonid predators among fish. Pikeminnow were estimated to consume approximately 9.7 million salmonids per year in the mainstem between Bonneville Dam and the estuary (Beamesderfer et al. 1996). Assuming approximately 200 million juvenile salmon and steelhead are available in the lower river per year, pikeminnow predation translates into a rate of 4.85%. Of this, approximately half occurs in the Bonneville Dam tailrace (Ward et al. 1995). The remainder was apportioned throughout the mainstem based on distance between the tributary mouth and the ocean. Bonneville Reservoir pikeminnow predation rate was calculated in a similar fashion (0.5%) with half assumed to occur in the Bonneville Forebay. The forebay rate was included for salmon populations originating in Bonneville Reservoir tributaries. Data were inadequate to estimate species differences in pikeminnow predation rates. Predation by other fishes, including walleye and hatchery salmonids, is not considered separate, hence, gets subsumed into estimated natural mortality. Walleye are substantially less abundant than pikeminnow. Data on predation rates by hatchery salmonids are not available. A pikeminnow sport reward fishery program has been implemented with a goal of reducing predation mortality on salmonids by 50%.

Tern predation on juvenile salmonids was based on Rice Island estimates by Roby et al. (1998) with corrections for recent translocation of the breeding colony to East Sand Island where salmonids are a less important diet item. We used a Rice Island predation mortality rate of 20% based on Roby et al.'s (1998) reported range of 10-30%. We estimated the corresponding East Sand Island predation rate at 9% by applying the difference in salmonid share of the diet at Rice (85%) and East Sand (40%) Islands (20% * 0.40/0.85). We hypothesize that tern predation accounts for the majority of the potentially manageable avian predation. Predation by other bird predators or birds in other areas is not addressed because of lack of data.

Estimates of marine mammal predation on adult salmonids were based on reported population sizes, literature values for daily ration, and reported diet shares of salmonids. Spring and fall predation mortality rates were estimated at 12% and 3% based on the following method. NMFS (2000) reported population sizes of about 2,000 in spring (1,700 harbor seals, 100-200+ sea lions). Fall population sizes were substantially less (1,000 total). Espenson (2003) quoted a daily ration equivalent to 1.2 – 2.0 salmon per day. To generate conservative minimum estimates we applied diet shares of 20% salmonids in spring NMFS (2000) and 50% salmonids in fall to an assumed daily ration equivalent to 1 salmon per day. Fall diet shares were assumed to be greater than spring because of fewer alternative foods and switching to more abundant salmon. This resulted in per predator consumption rates of 0.2 salmon per day in spring and 0.5 salmon per day in fall. Spring mortality rates were based on 24,000 salmon eaten versus average spring run sizes of 200,000 adult salmonids. Fall mortality rates were based on 30,000 salmon eaten versus average fall run sizes of 1,000,000 adult salmonids.

Because of the assumptions required by these calculations, our predation rates should be considered with caution. However, site-specific predation rates suggest that a 3-12% annual loss rate to marine mammals is reasonable. NMFS (2000) reported 250 salmon per year eaten by 10 sea lions at Willamette Falls based on direct observation. This translates into a 0.5% annual mortality rate based on a minimum Willamette Falls fish run of 50,000. Similarly, Espensen (2003) reported a 1.5% mortality rate by 100 sea lions in the Bonneville Dam tailrace.