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U.S. DEPARTMENT OF COMMERCE
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Thomas H. Williams[1], Brian C. Spence[1], Walt Duffy[2],
Dave Hillemeier[3], George Kautsky[4], Tom E. Lisle[5], Mike McCain[6],
Thomas E. Nickelson[7], Ethan Mora[1], Tom Pearson[1]

[1] Fisheries Ecology Division, NMFS/NOAA, Southwest Fisheries Science Center
110 Shaffer Road, Santa Cruz, California, USA 95060

[2] USGS, California Cooperative Fish Research Unit, Humboldt State University

[3] Yurok Tribe, Klamath, California

[4] Hoopa Valley Tribe, Hoopa, California

[5] USDA Forest Service, Pacific Southwest Research Station, Arcata, California

[6] USDA Forest Service, Smith River National Recreation Area, Gasquet, California

[7] Oregon Department of Fish and Wildlife (retired), Paso Robles, California

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U.S. DEPARTMENT OF COMMERCE
Carlos M. Gutierrez, Secretary
National Oceanic and Atmospheric Administration
William J. Brennan, Acting Undersecretary for Oceans and Atmosphere
National Marine Fisheries Service
James W. Balsiger, Acting Assistant Administrator for Fisheries

Preface

This report describes a framework for assessing coho salmon population viability that includes developing objective, measurable criteria that when met, would define when the Southern Oregon/Northern California Coast Coho Salmon Evolutionarily Significant Unit (ESU) is naturally self-sustaining with a low risk of extinction. Technical recovery planning for Pacific salmon and steelhead is intended to produce biologically based viability criteria for listed ESUs that will be considered in setting recovery goals.

The listing unit for Pacific salmon is the ESU. ESUs are defined as a population or group of populations that are substantially reproductively isolated from other conspecific population units and that represent an important part of the evolutionary legacy of the species. The Southern Oregon/Northern California Coast (SONCC) Coho Salmon ESU includes coho salmon populations from Elk River (Oregon) in the north to Mattole River (California) in the south. This report provides a framework to assess the viability of individual populations within this region and describes the spatial configuration of viable independent populations and dependent populations that would lead to a high likelihood of long-term ESU persistence.

This report constitutes a technical recommendation by the TRT intended to assist recovery planners in developing recovery strategies and prioritizing recovery actions. It does not constitute official agency policy.

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Executive Summary

The main purpose of technical recovery planning for Pacific salmon and steelhead is to produce biologically based viability criteria for listed Evolutionarily Significant Units (ESUs) that will be considered in setting recovery goals. These viability criteria, and the analyses from which they stem, must refer to specific populations and population groups (i.e., populations or groups of populations within a ESU). In this report we develop a framework for evaluating the viability of coho salmon populations and the greater Southern Oregon/Northern California Coast (SONCC) Coho Salmon Evolutionarily Significant Unit (ESU). Presented in this report are the recommendations of the Technical Recovery Team (TRT) for SONCC ESU as to the minimum population and ESU characteristics indicative of this ESU having a high probability of long-term (>100 years) persistence. The approach employs criteria representing three levels of biological organization: populations, diversity strata, and the ESU as a whole. Populations include both independent and dependent populations defined in Williams et al. (2006). Diversity strata are groups of geographically proximate populations that reflect the diversity of selective environments, phenotypes, and genetic variation across an ESU. A viable ESU comprises sets of viable (and sometimes non-viable) populations that, by virtue of their size and spatial arrangement, result in a high probability of persistence over the long term. In Chapter 1 of this report we provide background critical to understanding the context for viability criteria development. Chapters 2 and 3 define viability criteria at the population and ESU levels, respectively. In Chapter 4, we summarize our findings and provide recommendations.

We emphasize that the focus of this document is looking forward to viability and recovery goals. There are insufficient data to assess the risk to coho populations within the SONCC ESU, and therefore, we cannot assess the viability of the ESU using the quantitative approach developed in this report. Although the appropriate data are not available to assess population viability using the framework developed in this report, data sets that are available and were used in the most recent status review (Good et al. 2005) indicated that coho salmon populations in the SONCC ESU continue to be depressed relative to their historical numbers. As recovery planning

proceeds, ambitious research and monitoring programs will need to be initiated, as will long-term restoration solutions.

Population Viability Criteria

The report builds on the population structure report (Williams et al. 2006), describing a framework for assessing population and ESU viability for coho salmon in the SONCC ESU. Salmonid species are structured hierarchically based on the potential of exchange of individuals between similar components. This can range from sub-populations (e.g., breeding groups), to dependent populations, to independent populations, to population groups, and finally the ESU. The likelihood of exchange between components decreases as one builds from sub-populations, to populations, to population groups, to ESUs. Because an ESU is composed of a number of populations with varying features (e.g., habitat size, within population spatial distribution, etc.) and dynamics, an understanding of the biological organization of populations within an ESU and the temporal and spatial scales relevant to this organization is critical to developing meaningful biological viability criteria. A description of biological organization or biological structure of the SONCC ESU is presented in the TRT's report *Historical population structure of coho salmon in the Southern Oregon / Northern California Coasts Evolutionarily Significant Unit* (Williams et al. 2006).

The extinction risk of an ESU depends upon the extinction risk of its constituent populations. Therefore, development of objective and measurable viability criteria for an ESU must start with assessing the viability of its populations. In the Viable Salmonid Populations (VSP) document, McElhany et al. (2000) described four characteristics of populations that should be considered when assessing viability: abundance, productivity, diversity, and spatial structure.

Our approach to population viability extends the "viable salmonid population" concept of McElhany et al. (2000) to classify populations into various extinction risk categories based on a set of quantitative and qualitative criteria related to these parameters. Both the approach and the specific criteria have their roots in the IUCN (1994) Red List criteria (developed in part by Mace and Lande (1991)) and subsequent modifications made by Allendorf et al. (1997) to address populations of Pacific salmon. We have extended the Allendorf et al. (1997) criteria, adding criteria related to spawner density and to the potential effects of hatchery activities on wild populations.

Evaluation of extinction risk is based on the use of five surrogate criteria related to effective population size per generation, population decline, catastrophic decline, spawner density, and influence of hatchery fish (Table ES1). In addition, a rigorous, model-based population viability analysis (PVA) can be used to provide additional insight into extinction risk if appropriate data are available. The effective population size criteria address the loss of genetic diversity that can occur in populations with small numbers of individuals and can be estimated directly from demographic or genetic data, or in the absence of such data, by assuming a specific ratio of effective population size to total population size (census size). The population decline criteria address increased demographic risks associated with rapid or prolonged declines in abundance to small population sizes. The catastrophic decline criteria seek to capture effects of large environmental disturbances that produce rapid declines in abundance. These events are distinct from environmental stochasticity that arises from small or moderate disturbances that affect population growth rate. The density criteria are intended to capture several distinct processes not explicitly addressed in the Allendrof et al. (1997) criteria. The high-risk thresholds identify densities at which populations are at heightened risk of a reduction in per capita growth rate (i.e., depensation). Populations exceeding the low-risk density thresholds are expected to inhabit a substantial portion of their historical range, which serves as a proxy indicator that resultant spatial structure and diversity will reasonably represent historical conditions. The hatchery criterion of a less than 5% contribution of hatchery fish spawning in the wild is intended primarily to address potential genetic risks, recognizing that demographic and ecological risks that occur when hatchery fish interact also need to be also considered.

ESU-Level Criteria

In our proposed scenario for a viable ESU, we do not list specific sets of populations that must be viable to have a viable ESU. Instead, we provide a set of rules that will result in certain

Table ES1. Viability criteria for assessing extinction risk for populations of coho salmon in the Southern Oregon/Northern California Coast ESU. For a given population, the highest risk score for any category determines the populations overall extinction risk. Modified from Allendorf et al. (1997) and Lindley et al. (2007). See table footnotes for definitions of N_e, N_g, and N_a.

	Extinction risk		
Criterion	High	Moderate	Low
	- any One of -	- any One of -	- all of -
Effective population size[a]	$N_e \leq 50$	$50 < N_e < 500$	$N_e \geq 500$
- *or* -	- *or* -	- *or* -	- *or* -
Population size per generation	$N_g \leq 250$	$250 < N_g < 2500$	$N_g \geq 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[d]	Not apparent
Spawner density (adults/IP km)	$N_a/IP\ km \leq 1$	$1 < N_a/IP\ km <$ MRSD[e]	$N_a/IP\ km \geq$ MRSD[e]
Hatchery influence			Hatchery fraction <5%
			- *in addition to above* -
Extinction risk from PVA	≥20% within 20 yrs	≥5% within 100 yrs but <20% within 20 yrs	< 5% within 100 yrs[f]

[a] The effective population size (N_e) is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931); total number spawners per generation (N_g), for SONCC coho salmon the generation time is approximately three years therefore $N_g = 3\ N_a$.

[b] Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size of $N_a \leq 500$ spawners (historically small but stable populations not included) **or** $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.

[c] Annual spawner abundance N_a has declined to ≤500 spawners, but now stable **or** number of adult spawners (N_a) > 500 but continued downward trend is evident.

[d] Annual spawner abundance decline in one generation < 90% but biologically significant (e.g., loss of year class).

[e] MRSD = minimum required spawner density is dependent on the amount of potential habitat available. Figure 5 summarizes the relationship between spawner density and IP km.

[f] For population to be considered at low-risk of extinction, all criteria must be satisfied (i.e., not just a PVA). A population viability analysis (PVA) can be also included for consideration, but must estimate an extinction risk <5% within 100 years *and* all other criteria must be met. If discrepancies exist between PVA results and other criteria, results need to be thoroughly examined and potential limitations of either approach are carefully identified and examined.

configurations of populations that we believe will result in a viable ESU. The rules we propose are intended to capture our objectives of maintaining diversity throughout the ESU, providing connectivity among populations to maintain long-term demographic and genetic processes, and providing a buffer against potential catastrophic risks. Our overarching goal in developing these rules is that we desire an appropriate number and arrangement of populations that allows for the populations to track changes in environmental conditions and therefore be viable.

Representation Criteria

1. *All diversity strata should be represented by viable populations for the ESU to be considered viable..*

By requiring all diversity strata to be represented for the ESU to be viable, the range of environmental conditions historically available has a greater chance of being included and thereby a substantial portion of the historical diversity of the ESU. In addition, given the geographic make-up of the diversity strata, by requiring all strata be viable helps ensure that the ESU persists throughout a significant portion of its historical range. And finally, because of the arrangement of populations along portions of the SONCC ESU, representing each stratum also ensures that connectivity across the entire ESU is maintained.

Redundancy and Connectivity Criteria

Three additional viability criteria are proposed to provide a hedge against catastrophic risk by ensuring redundancy of viable populations and to promote connectivity within diversity strata and throughout the ESU.

2.a. *At least fifty percent of historically independent populations (functionally independent and potentially independent populations) in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria developed in this report. For strata with three or fewer independent populations, at least two populations must be viable.*

- and –

2.b. *Within each diversity stratum, the total aggregate abundance of independent populations selected to satisfy this criterion should meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all functionally independent and potentially independent populations.*

Recovery planners should seek configurations of populations within the ESU that emphasize historically independent populations that, by virtue of their size and distribution, formed the foundation of the ESU.

3. *Remaining populations, including historical dependent populations and historical independent populations (functionally and potentially independent populations) that are not expected to meet the low-risk threshold, must exhibit occupancy patterns that indicate sufficient immigration is occurring from the "core populations".*

This criterion acknowledges that while certain populations may no longer fulfill their historical role in ESU viability, the remaining portions of these populations can contribute substantially to connectivity among populations within the ESU, as well as represent important parts of the ESU's evolutionary legacy.

4. *The distribution of extant populations, both dependent and independent, needs to maintain connectivity across the stratum as well as with adjacent strata.*

This criterion stresses the importance of ensuring connectivity within and among diversity strata to maintain long-term evolutionary and demographic processes that result from natural dispersal.

The framework proposed in this report parallels efforts for other ESUs in California and is consistent in its underlying conceptual approach (e.g., VSP). The framework we propose consists of criteria and rules that are based in part on expert opinion and judgment, although we have attempted to provide support for our efforts from the general literature, specific references for Pacific salmonids, and when available, references specifically for coho salmon. Although the approaches used by this and other TRTs are to some degree based on expert judgment and subject to considerable uncertainty, the conclusions are not particularly sensitive to the exact threshold values of the criteria. Nor should uncertainty surrounding the proposed criteria deter recovery planners from proceeding with action to recover these populations. In most cases, populations appear to be well below the proposed viability thresholds, and the steps needed to move them toward viability will be similar, regardless of the specific recovery targets, which can be refined as more information becomes available.

As previously discussed, there are insufficient data to assess the risk of coho populations within the SONCC ESU, and therefore we cannot assess the viability of the ESU using the quantitative approach developed in this report. As recovery planning proceeds, ambitious research and monitoring programs will need to be initiated, as will long-term restoration solutions. Some very important actions that should be done as soon as possible for SONCC coho salmon (listed in no particular order):

- Secure all extant populations. Although the SONCC ESU is far short of being viable, extant populations, even if not currently viable, may be needed for recovery.

- Begin collecting distribution and abundance data throughout the SONCC ESU. These data are fundamental for developing effective recovery actions and future status assessment.

- Minimize straying from hatcheries to natural spawning areas. The goal of maximizing diversity within populations and the ESU is eroded with even low levels of straying from hatchery populations to wild populations.

- Begin conducting critical research on climate change and its potential impact to SONCC coho salmon. Resource managers should develop strategies to cope with climate change, perhaps though some sort of an adaptive management approach, given the uncertainty in the magnitude and timing of climate change.

Acknowledgments

This effort to develop a framework for assessing the viability of threatened coho salmon in the Southern Oregon / Northern California Evolutionarily Significant Unit was conducted by the SONCC workgroup of the Oregon and Northern California Coast Technical Recovery Team. The Workgroup consisted of scientists from NOAA National Marine Fisheries Service-Southwest Fisheries Science Center, USGS California Cooperative Fish Research Unit, the Yurok Tribe, the Hoopa Valley Tribe, USDA Forest Service-Pacific Southwest Research Station, USDA Forest Service-Smith River National Recreation Area, and the California Department of Fish and Game. Gayle Garman, California Department of Fish and Game, participated as a SONCC workgroup member during the later portion of the development of the viability criteria and contributed to discussions and reviews of material for this document. Throughout the preparation of this report, the Workgroup and the ONCC TRT were provided with information from numerous scientists and fishery managers. Specifically we would like to acknowledge the thoughtful and insightful reviews by the Oregon Department of Fish and Wildlife and the California Department of Fish and Game. Others we would like to acknowledge for meeting with the Workgroup or providing information include Richard Piaskowski (GeoEngineers, Inc.), Tom Satterthwaite (ODFW), and Russ Stauff (ODFW). The TRT would also like to thank Drs. Pete Adams, David Boughton, and Steve Lindley for many discussions about the viability framework. Thanks also to Heather Stout (NOAA NMFS Northwest Fisheries Science Center) and the support staff from the NOAA NMFS Southwest Fisheries Science Center that included Heidi Fish, Kerrie Pipal, Mark Jessop, and Alison Collins.

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1. Introduction

The Endangered Species Act (ESA) requires that the National Marine Fisheries Service (NMFS) develop recovery plans for anadromous species listed as threatened or endangered and that these recovery plans contain objective, measurable criteria that when met, would define when a species would be removed from the list. For Pacific salmon, the listing unit is the Evolutionarily Significant Unit (ESU), which typically comprises multiple individual populations within a particular geographic region[1]. The Southern Oregon / Northern California Coast (SONCC) Coho Salmon ESU, which comprises coho salmon populations found in coastal watersheds from Elk River (Oregon) in the north to Mattole River (California) in the south, was listed as threatened under the ESA in 1995 (70 FR 37160).

As part of the recovery planning process, the NMFS assembled a group of scientists to serve as a Technical Recovery Team (TRT), whose purpose was to provide a scientific context for identifying necessary actions to help the ESU recover. Among the TRT's responsibilities was the development of biological viability criteria for populations and the ESU that, if met, would indicate when the ESU is recovered and hence form the biological basis for formal delisting criteria[2].

For the TRT, development of biological viability was a two-step process. The first step was to define the historical population structure within the SONCC ESU. Salmonid species are structured hierarchically based on the potential of exchange of individuals between similar components. This can range from sub-populations (e.g., breeding groups) to dependent populations, to independent populations, to population groups, and finally the ESU (Bjorkstedt et

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[1] The ESA allows listing not only of species, but also "distinct population segments" of any species of vertebrate fish or wildlife which interbreeds when mature. The NMFS has developed policies that define distinct population segments as populations or groups of populations that are reproductively isolated from other conspecific population units and that are an important component in the evolutionary legacy of the species. For salmon, NMFS has termed these distinct population segments "Evolutionarily Significant Units" or ESUs (Waples 1991).

[2] Delisting criteria must consider not only the biological status of the listed species, but also the factors that led to the listing of the species in the first place (National Marine Fisheries Service 2006). The biological viability criteria proposed in this document represent what the TRT believes to be the minimum population and ESU characteristics required for the ESU to have a high probability of persisting into the future and at low risk of extinction.

al. 2005; Lawson et al. 2007). The likelihood of exchange between components decreases as one builds from sub-populations, to populations, to population groups, to ESUs. Because an ESU is composed of a number of populations with varying features (e.g., habitat size, within-population spatial distribution, etc.) and dynamics, an understanding of the biological organization of populations within an ESU and the temporal and spatial scales relevant to this organization is critical to developing meaningful biological viability criteria. A description of biological organization or biological structure of the SONCC ESU is presented in the TRT's report *Historical population structure of coho salmon in the Southern Oregon / Northern California Coasts Evolutionarily Significant Unit* (Williams et al. 2006).

The population structure report (Williams et al. 2006) adopts as its underlying foundation the concept of "independent populations," which McElhany et al. (2000) defined as "any collection of one or more local breeding units whose population dynamics or extinction risk over a 100-year time period are not substantially altered by exchanges of individuals with other populations." This concept is important, as it seeks to delineate population units that should form the appropriate fundamental units of conservation, since a strategy that focuses on units that are too small or those whose persistence depends on immigrants from neighboring populations could lead to a high probability of extinction for the ESU. Williams et al. (2006) describe the historical population structure of coho salmon in the SONCC ESU based on the location and amount of potential coho salmon habitat, with an assumption that the relative abundance of different populations mirrored the amount of intrinsic habitat potential in each watershed. In general, the SONCC ESU was characterized by 1) small-to-moderate-sized coastal basins in which habitat (as predicted by the intrinsic potential model used in Williams et al. (2006)) was concentrated in the lower portions of the basins, and 2) by three large basins in which some habitat was located in the lower portions of the basins, relatively little habitat was available in the middle portions of the basins, and the greatest amount of habitat was located in the upper sub-basins. This general description of historical coho salmon habitat was then interpreted to define the following historical population structure and distribution:

1. Nineteen *functionally independent populations*, defined as those sufficiently large to be historically viable-in-isolation (Bjorkstedt et al. 2005; Williams et al. 2006) and whose demographics and extinction risk were minimally influenced by immigrants from adjacent populations (Tables 1 and 2, Figure 1).

2. Twelve *potentially independent populations*, defined as those that were potentially viable-in-isolation but that were demographically influenced by immigrants from adjacent populations.

3. Seventeen small *dependent populations* of coho salmon, which are believed to have had a low likelihood of sustaining themselves over a 100-year time period in isolation and that received sufficient immigration to alter their dynamics and extinction risk.

4. Two *ephemeral populations*, defined as populations that were both small enough and isolated enough that they were only intermittently present.

In anticipation of developing viability criteria at the population scale and integrating population information into viability criteria at the ESU scale, Williams et al. (2006) also identified the diversity and distribution of coho salmon populations that historically existed within the ESU (Figures 1 and 2). The TRT organized the independent and dependent populations of coho salmon in the SONCC ESU into diversity strata largely based on the geographical arrangement of these populations and basin-scale environmental and ecological characteristics.

The current report constitutes the second step in the development of biological viability criteria. The report builds on the population structure report (Williams et al. 2006), describing a framework for assessing population and ESU viability for coho salmon in the SONCC ESU. The extinction risk of an ESU depends upon the extinction risk of its constituent populations. Therefore, development of objective and measurable viability criteria for an ESU must start with

Table 1. Independent and dependent populations of coho salmon in the Southern Oregon/Northern California Coast Coho Salmon ESU, including integrated intrinsic potential (IP) values with 21.5 °C air temperature mask (locations where IP was reduced by temperature mask have pre-mask values in parentheses) from Williams et al. (2006). Population types include functionally independent (FI), potentially independent (PI), dependent (D), and ephemeral (E) populations. Basins with integrated IP < 1.2 km with temperature mask were excluded from analyses. Basins in italics contained an integrated IP km value < 5 km and are not included by name or number in subsequent analyses. ID numbers are for reference to other tables and figures in this document and Williams et al. (2006).

			Population type		
Basin	ID	IP (km)	FI	PI	D
Elk River	1	62.64	X		
Mill Creek	2	7.25			X
Hubbard Creek	3	17.94			E[a]
Brush Creek	4	5.68			X
Mussel Creek	5	6.06			X
Euchre Creek	6	32.31			E[a]
Greggs Creek		3.40			X
Rogue River		2344.58 (2547.01)			
Lower Rogue River	7a	80.88		X	
Illinois River	7b	589.69	X		
Middle Rogue and Applegate rivers	7c	758.58 (760.67)	X		
Upper Rogue River	7d	915.43 (1115.77)	X		
Hunter Creek	8	14.63			X
Myers Creek		3.45			X
Pistol River	9	30.23			X
Sand Creek		1.62			X
Thomas Creek		1.36			X
Chetco River	10	135.19	X		
Winchuck River	11	56.50		X	
Gilbert Creek		1.80			X
Smith River	12	385.71	X		
Elk Creek	13	17.38			X
Wilson Creek	14	18.80			X
False Klamath Cove		2.17			X
Klamath-Trinity		2247.74 (3048.37)			
Lower Klamath River	15a	204.69	X		
Middle Klamath River	15b	113.49 (178.59)		X	
Upper Klamath River	15c	424.71	X		
Salmon River	15d	114.80 (145.90)		X	
Scott River	15e	440.87	X		

Table 1. continued.

Basin	ID	IP (km)	Population type FI	PI	D
Shasta River	15f	531.01 (606.86)	X		
South Fork Trinity River	15g	241.83 (342.47)	X		
Lower Trinity River	15h	112.01 (170.49)		X	
Upper Trinity River	15i	64.33 (533.79)	X		
Fern Canyon		3.66			X
Squashan Creek		2.66			X
Gold Bluff		2.88			X
Redwood Creek	16	151.02	X		
McDonald Creek	17	5.44			X
Maple Creek/Big Lagoon	18	41.30		X	
Little River	19	34.20		X	
Strawberry Creek	20	5.71			X
Norton/Widow White Creek	21	8.54			X
Mad River	22	152.87	X		
Humboldt Bay tributaries	23	190.91	X		
Eel River - Full		1459.81 (1773.37)			
Lower Eel and Van Duzen rivers	24a	393.52	X		
South Fork Eel River	24b	476.10	X		
Mainstem Eel River	24c	143.90 (156.73)		X	
North Fork Eel River	24d	53.97 (83.54)		X	
Middle Fork Eel River	24e	77.70 (252.77)		X	
Middle Mainstem Eel River	24f	255.50 (281.31)	X		
Upper Mainstem Eel River	24g	54.11 (124.39)		X	
Fleener Creek		3.87			X
Guthrie Creek	25	14.16			X
Oil Creek		3.09			X
Bear River	26	47.84		X	
Singley Creek		3.40			X
Davis Creek		1.71			X
Domingo Creek		1.36			X
McNutt Gulch	27	5.90			X
Mattole River	28	249.79	X		

[a] – Hubbard and Euchre creeks were designated as Ephemeral populations.

Table 2. Population unit boundaries of sub-basins in the Rogue, Klamath, and Eel rivers for SONCC Coho Salmon ESU.

Basin	ID	Population Unit	Boundaries
Rogue	7a	Lower Rogue River	Mouth of Rogue upstream to confluence of Illinois River.
	7b	Illinois River	
	7c	Middle Rogue River	Confluence of Illinois River upstream to confluence of Evans Creek (non-inclusive); includes Applegate River.
	7d	Upper Rogue River	Evans Creek (inclusive) upstream to IP limit.
Klamath	15a	Lower Klamath River	Mouth of Klamath upstream to confluence with Trinity River.
	15b	Middle Klamath River	Confluence of Trinity River upstream to Portuguese Creek (inclusive in Middle Klamath); Seiad and Grider creeks in Upper Klamath basin.
	15c	Upper Klamath River	Portuguese Creek (non-inclusive) upstream to Spencer Creek (inclusive).
	15d	Salmon River	
	15e	Scott River	
	15f	Shasta River	
	15g	S. Fk. Trinity River	Confluence of Trinity River is lower boundary.
	15h	Lower Trinity River	Confluence of Klamath River upstream to confluence with North Fork Trinity River (non-inclusive).
	15i	Upper Trinity River	Confluence of North Fork Trinity River (inclusive) upstream to Ramshorn Creek (inclusive).
Eel	24a	Lower Eel/Van Duzen River	Mouth of Eel River upstream to confluence with South Fork Eel River.
	24b	South Fork Eel River	
	24c	Mainstem Eel River	Confluence of South Fork Eel River upstream to confluence with Middle Fork Eel River.
	24d	North Fork Eel River	

Table 2. continued.

Basin	ID	Population Unit	Boundaries
	24e	Middle Fork Eel River	
	24f	Middle Mainstem Eel River	Confluence of Middle Fork Eel River upstream to Tomki Creek (inclusive), upstream in Outlet Creek and tributaries to IP limit.
	24g	Upper Mainstem Eel River	Eel River upstream of confluence of Tomki Creek (non-inclusive) to IP limit.

assessing the viability of its populations. In the Viable Salmonid Populations (VSP) document, McElhany et al. (2000) described four characteristics of populations that should be considered when assessing viability: abundance, productivity, diversity, and spatial structure. The viability of an ESU depends on the appropriate distribution and characteristics of its constituent populations to ensure that longer-term ecological and evolutionary processes are maintained. Consequently, ESU viability criteria should seek to maintain the diversity of an ESU across all or a significant portion of its historical range, to provide redundancy in order to reduce the effects of catastrophic events and to ensure connectivity among populations across the ESU (Shaffer and Stein 2000). This report proposes criteria for evaluating viability at both the population and ESU levels.

Before presenting a framework to assess the viability of the SONCC coho salmon ESU, we prepare the reader for the fact that almost no data sets of the appropriate time series or spatial scale are currently available to assess the viability of any population within the SONCC ESU using our criteria. The one exception is the Upper Rogue population unit, where counts from Gold Ray Dam provide an estimate of approximately 70% of the estimated historical IP km accessible. Therefore, assessing the current viability of the ESU using our approach is not currently possible given the lack of population specific data. Consequently, our effort has been directed at developing a framework that relates viability to extinction risk. The primary purpose

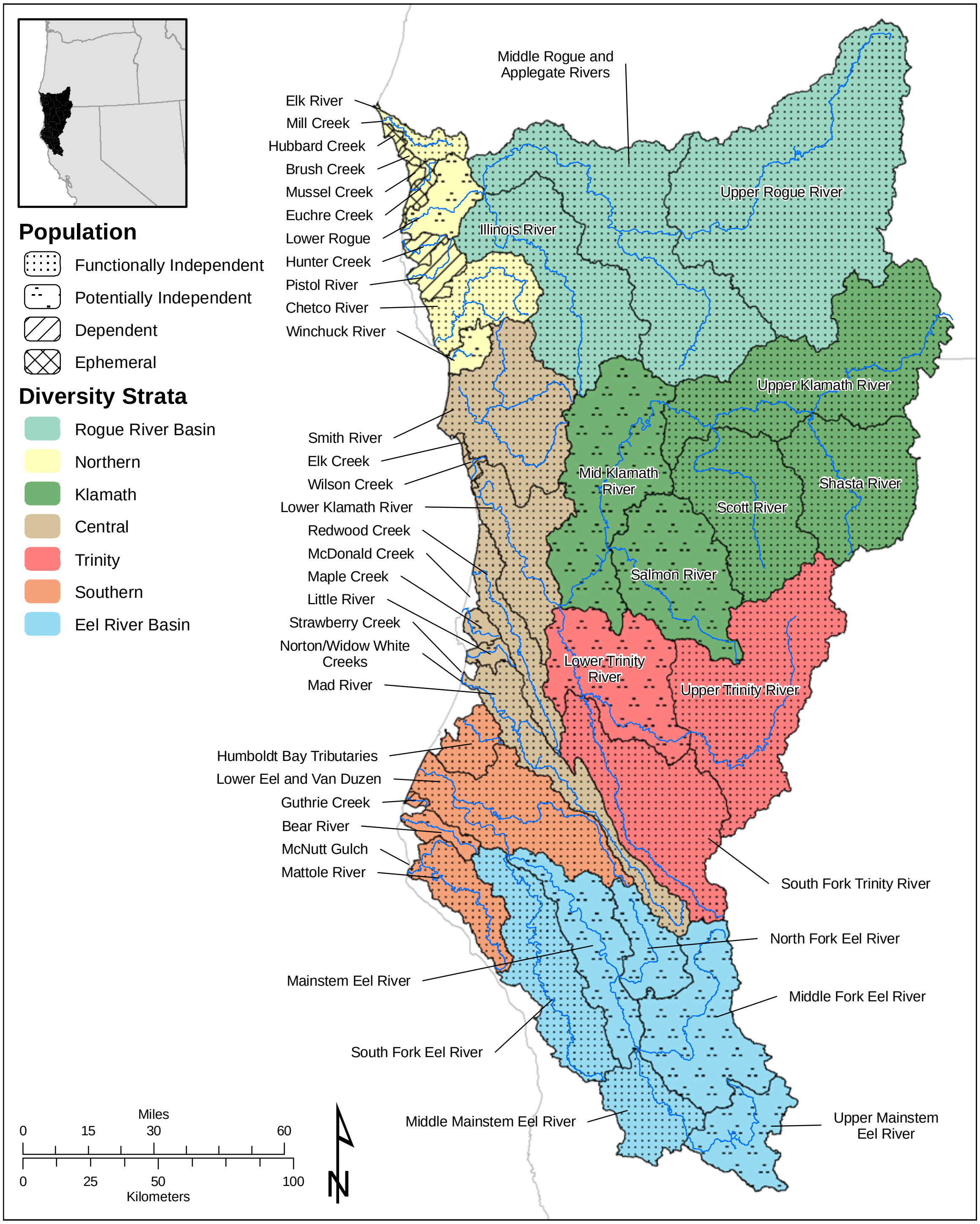

Figure 1. Southern Oregon/Northern California Coast Coho Salmon ESU population units and diversity strata proposed by the TRT (Williams et al. 2006).

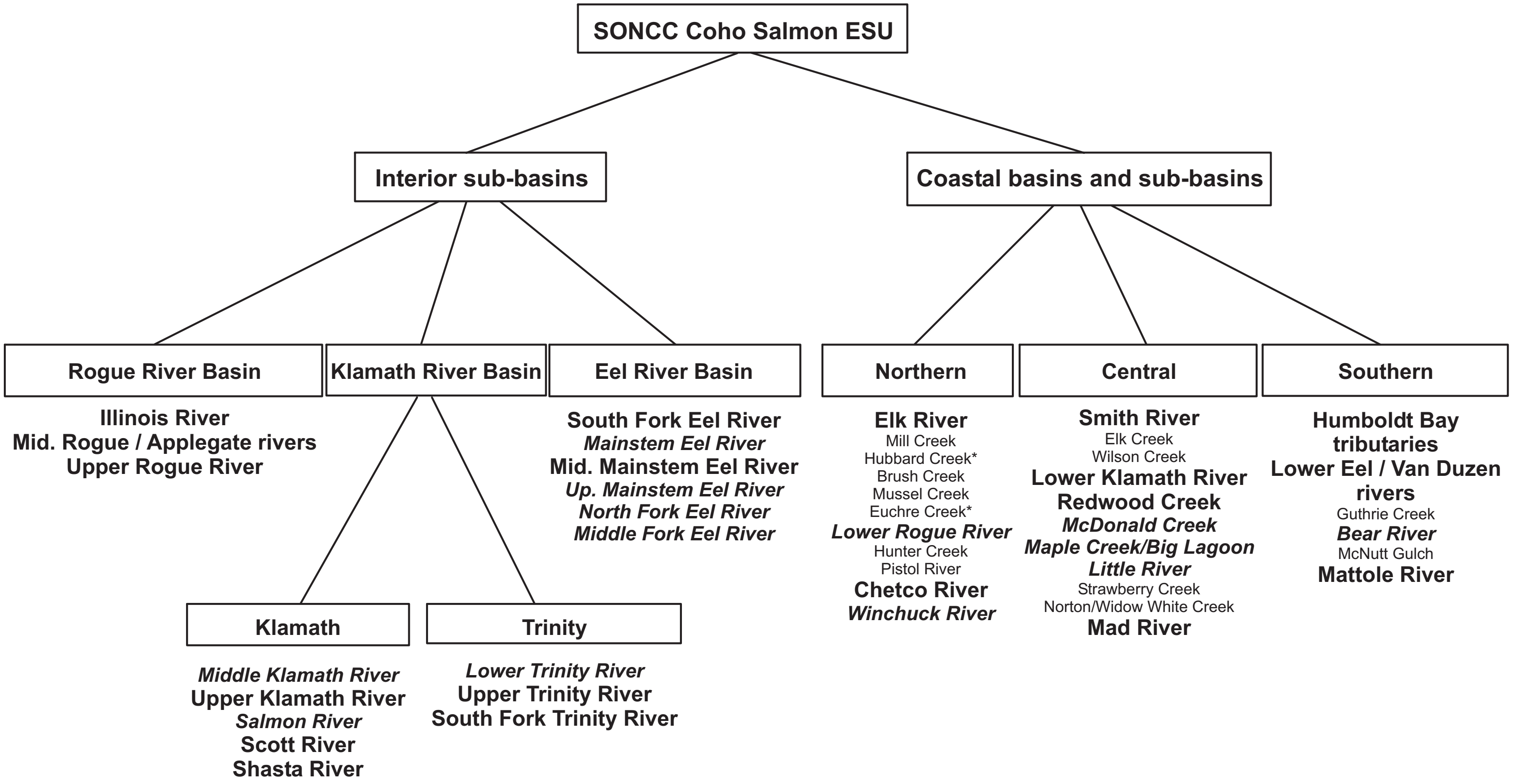

Figure 2. Arrangement of historical populations of the Southern Oregon/Northern California Coast Coho Salmon ESU into diversity strata. Functionally independent populations are listed in bold font, potentially independent populations are listed in bold italic font, other listed populations are dependent and ephemeral populations. Ephemeral populations are indicated with an astrick(*).

of this framework is twofold: 1) to provide a means by which population and ESU viability can be evaluated in the future, and 2) to propose preliminary biological targets that can assist recovery planners in prioritizing recovery actions.

Our approach is built upon a general approach used for assessing extinction risk developed by the International Union for the Conservation of Nature (IUCN) (Mace and Lande 1991; IUCN 1994) and later modified by Allendorf et al. (1997) specifically to prioritize conservation actions for Pacific salmonids. We took this path based on the widespread lack of data and recognition that appropriate data sets for performing more sophisticated analyses would not be available for many years. The general criteria thresholds and rules are such that, if met, we believe the ESU would have a high likelihood of being viable. Such an approach was advocated by Shaffer et al. (2002) and was advice given directly to the various TRTs by the Salmon Recovery Science Review Panel (RSRP, 27-29 August 2001 meeting)[3]. As ESU and population-specific research and monitoring occur, changes in our thresholds could be warranted. However, the framework proposed here will allow recovery planners to have initial population and ESU targets and thresholds of various performance measures that we believe address the VSP concepts of productivity, abundance, spatial structure, and diversity.

Although the appropriate data are not available to assess population viability using the framework developed in this report, data sets that are available and were used in the most recent status review (Good et al. 2005) indicated that coho salmon populations in the SONCC ESU continue to be depressed relative to their historical numbers. Good et al. (2005) reported strong indications that breeding groups had been lost from a significant portion of the historical range, and although the 2001 brood year appeared to be strong, it followed a number of relatively weak years. The one exception noted by Good et al. (2005) was the "Rogue River stock" where there were increasing numbers of spawners over the last several years, despite low numbers in 1998

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[3] The Recovery Science Review Panel was convened by the NMFS to help guide the scientific and technical aspects of recovery planning for listed salmon and steelhead species throughout the West Coast. More information about the RSRP and their reports are available at: http://www.nwfsc.noaa.gov/trt/rsrp.cfm. The RSRP made specific mention to the development of relatively simple objective, population-based criteria in their report based on the 27-29 August 2001 meeting.

and 1999. The Rogue River stock numbers reviewed by Good et al. (2005) are based on the Huntley Park seine data and represented the whole Rogue Basin, a composite of four independent populations: Lower Rogue River, Illinois River, Middle Rogue and Applegate rivers, and Upper Rogue River populations (Williams et al. 2006).

We stress that the viability criteria we propose should not be viewed as discrete knife-edge, pass/fail criteria for assessing population extinction risk and ESU viability. Although we have grounded the criteria as firmly as possible in the available science, the specific criteria thresholds are not without scientific uncertainty. Additionally, by their general nature, the criteria do not account for population-specific attributes that might make a population more or less vulnerable to extinction. Hence, there is some potential for the criteria to be overly conservative in some cases or not conservative enough in others. And finally, any estimation of population parameters will also be subject to some uncertainty. Accordingly, the criteria should be considered as general indicators of viability that are most appropriately viewed with these uncertainties in mind. Furthermore, in evaluating population risk, it is also important to consider each individual criterion in the context of other criteria. Populations that fail to satisfy several criteria are likely at greater risk than those that fail to satisfy a single criterion. Likewise, there will be higher uncertainty as to the status of populations that are borderline for several criteria than for those that are marginal for a single criterion.

Implementation of these viability criteria requires data, specifically time series data on the number of adult coho salmon in a population unit. The TRT recognizes that in some instances (e.g., Illinois River) obtaining these data is not a trivial endeavor (e.g., remote locations). In such situations, monitoring programs will need to be specifically developed to address the uncertainty around these abundance estimates. Moreover, recovery planners will need to accommodate an increased level of uncertainty into their assessments of viability and for recovery planning. There is a critical need for future research efforts to develop appropriate monitoring programs to deal with these remote or difficult geographic locations so that monitoring activities provide appropriate types of information and are able to provide some measure of uncertainty in the context of data from other more accessible locations.

This framework was developed in part to provide guidance on the types of population and ESU performance measures (i.e., VSP characteristics, McElhany et al. 2000) needed to assess the viability of the SONCC Coho Salmon ESU. We expect the implementation of rigorously designed and carried out research and monitoring efforts will lead to the identification of population-specific viability measures that form the foundation for future population- and SONCC ESU-specific criteria. We feel that this expectation is justified when examining situations outside of the SONCC ESU. For instance, the Oregon Coast Coho Salmon ESU has many populations where rigorously designed (e.g., statistically valid) monitoring has been in place for more than 10 years and is providing the TRT and recovery planners for that ESU more population- and ESU-specific data to develop viability criteria. Although differences exist in the final application of the viability criteria proposed for the Oregon Coast and SONCC ESU (i.e., decision support system for Oregon Coast, viability criteria table for SONCC), the approaches and threshold values are generally consistent based on the VSP foundations on which they were developed. In addition, the SONCC viability criteria were developed in concert with the TRT developing recovery criteria for the Central California Coast Coho Salmon ESU, making the viability analyses of these coho salmon ESUs extremely consistent.

The TRT discussed whether to use more population-specific measures to develop viability criteria. In the absence of population- and ESU-specific data, the general conclusion was that such specificity was inappropriate and would not approach viability assessment in a precautionary manner. In addition, with so few monitoring activities of the appropriate nature currently underway in the ESU, we believe our criteria will provide guidance for the types of data that should be collected and considered, including direction to the relevant spatial and temporal scales. Through the collection of these data and the use of population viability analyses, it may prove possible for recovery planners to more accurately assess population and ESU viability. In some instances, our criteria may need to be changed, but in the absence of data from well designed and carried out monitoring activities, the general criteria we propose will serve as a benchmark for recovery planning, as suggested by the RSRP (27-29 August 2001 meeting) and by Shaffer et al. (2002).

There was some discussion within the TRT for us to be more explicit in terms of the types of monitoring activities required for obtaining needed data. Given the current state of sampling throughout the ESU and the time required to have time series of the appropriate length, we opted for our more general approach. We believe that the initial efforts to 1) better understand the amount of habitat available to each population, and 2) obtain abundance estimates for each population are going to require a significant increase or redirection of current efforts as well as some trial and error. In terms of implementation (e.g., logistical considerations, methods standardization, site selection, etc.), we will be much better served by having these more general guidelines. Therefore, during the initial phase, more rigid, population-specific requirements may be impractical and overwhelm field efforts and perhaps discourage needed changes in current monitoring. However, in some portions of the SONCC ESU where the logistical and institutional infrastructure are in place, implementing more intensive monitoring programs may be possible in the near future.

2. Population Viability

2.1 Approach

McElhany et al. (2000) defined a viable salmonid population as “an independent population… that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame.” Population viability depends on various demographic properties of the population, such as population size, growth rate, variation in growth rate, and carrying capacity (Tuljapurkar and Orzack 1980). McElhany et al. (2000) proposed that four parameters are critical for evaluating salmon population viability: abundance, population growth rate, population spatial structure, and diversity.

Abundance is of obvious importance since, in general, small populations are at greater risk of extinction than large populations, primarily because many processes that affect population dynamics may operate differently in small populations than in large populations (Shaffer 1987; McElhany et al. 2000).

Population growth rate, the productivity over the entire life cycle, and factors that affect population growth rate provide information about how well a population is performing in the various habitats it occupies during the life cycle. Examining population growth rate allows one to assess if populations are able to replace themselves. Populations that consistently fail to replace themselves are at greater risk of extinction than populations that are consistently at or above replacement levels.

Spatial structure refers to the distribution of individuals within a population at a certain life stage throughout the available habitats, recognizing the abiotic and biotic processes that give rise to that structure. McElhany et al. (2000) gave two main reasons why spatial structure is important to consider when evaluating population viability: 1) overall extinction risk at longer time scales may be affected in ways not apparent from short-term observations of abundance and productivity, because there can be a time lag between changes in spatial structure and the resulting population-level effects, and 2) spatial population structure affects the ability of a population to respond to changing environmental conditions and therefore can influence evolutionary processes. Maintaining spatial structure within a population, and its associated benefits to viability, requires appropriate habitat conditions and suitable corridors linking the habitat and the marine environment to be consistently available.

Diversity relates to the variability of phenotypic characteristics such as life histories, individual size, fecundity, run timing, and other attributes exhibited by individuals and populations, as well as the genetic diversity that may underlie this variation. There are many reasons diversity is important in a spatially and temporally varying environment. Three key reasons are 1) diversity allows a species to use a wide array of environments, 2) diversity protects a species against short-term spatial and temporal changes in the environment, and 3) genetic diversity provides the raw material for surviving long-term environmental change (McElhany et al. 2000).

Developing objective, quantitative, and biologically meaningful viability criteria for Pacific salmonid populations is difficult, in part because of a paucity of data and a lack of understanding of some of the fundamental ecological and biological processes governing populations and ESUs (Wainwright and Waples 1998; Lindley et al. 2007). Although McElhany et al. (2000) provided a useful conceptual framework and guidance for developing viability assessments of Pacific salmonids, they did not propose specific quantitative criteria that would allow for assessing the viability of specific populations. Ideally, population-specific criteria could be developed that would take into account differences in the quantity or quality of freshwater habitat, variability in marine survival, or other conditions that influence viability. However, for many populations of Pacific salmonids, including SONCC coho salmon, very few population-specific data are available; therefore, viability criteria were developed based on biologically relevant criteria that are generic to *Oncorhynchus* species and in some cases tailored to other specific species (Lindley et al. 2007; Spence et al. 2008; Wainwright et al. 2008).

Because of the lack of information that would allow for development of population-specific viability criteria, we chose to adopt the general approach of the IUCN (1994) as modified for Pacific salmonids by Allendorf et al. (1997). Allendorf et al. (1997) classify populations into one of six categories: extinct, extinct in the wild, high risk, moderate risk, and low risk, or data deficient. They evaluate extinction risk by population viability analysis (PVA) or, if data are insufficient to construct a credible PVA, using four surrogate criteria related to population size and trend (see Table 1 in Allendorf et al. 1997).

We adopt the general framework of Allendorf et al. (1997) but propose several modifications and extensions that are important to our particular application. In making these modifications, we draw heavily from similar efforts for California salmonids (Lindley et al. 2007; Spence et al. 2008).

The most notable change the TRT made to the Allendorf et al. (1997) approach was de-emphasizing the use of PVA for assessing risk. Although PVA can be a valuable tool for assessing extinction risk, there is sufficient uncertainty associated with this use of PVA that the

TRT felt the general criteria (analogous to Allendorf et al.'s "surrogate" criteria) provide the most appropriate indicators of population viability, especially given the general lack of data appropriate for PVA. The use of PVA is encouraged for developing a better understanding of population-specific processes and may be used to identify risk not captured by the more general criteria. But unlike Allendorf et al. (1997), we recommend that PVA not be used to supersede the general criteria for low-risk designation.

Like Allendorf et al. (1997), we developed extinction risk categories from high to low defined by various quantitative criteria corresponding to specific extinction risks across specific time horizons (Table 3). We define criteria for a "low risk" category, which was implicit in Allendorf et al. (1997). In addition, we collapse their "very high" and "high" risk categories into a single "high" risk category to simplify analysis. Discriminating between "high risk" and "very high risk" was important to Allendorf et al. (1997) because their emphasis was on prioritizing stocks for conservation. The distinction between "high risk" and "very high risk" was less important for our effort since either categorization indicates that a population should not be considered viable over short-to-moderate time frames. In practice, the effects of combining these two categories are relatively minor, although it does result in a slightly different configuration and implementation of the viability criteria table than those of Allendorf et al. (1997). Specifically, we adopt a rule that the assignment of risk to a population is based on the highest risk category for any individual risk metric. For example, a population rated at "high risk" based on N_e, but moderate or low risk for the other metrics would receive the "high risk" rating. Allendorf et al. (1997) used a similar approach but added an additional rule where populations that rank at a certain risk level for more than one metric are moved to the next highest risk level when categorizing the population (e.g., a population rated at moderate risk for two metrics is considered at high risk overall). Because of this, the criteria listed in our "high risk" and "moderate risk" categories align themselves with the "very high risk" and "high risk" categories of Allendorf et al. (1997). In practice, a population that satisfies a single criterion (as opposed to

Table 3. Viability criteria for assessing extinction risk for populations of coho salmon in the Southern Oregon/Northern California Coast ESU. For a given population, the highest risk score for any category determines the populations overall extinction risk. Modified from Allendorf et al. (1997) and Lindley et al. (2007). See table footnotes for definitions of N_e, N_g, and N_a .

	Extinction risk		
Criterion	High	Moderate	Low
	- any One of -	- any One of -	- all of -
Effective population size[a]	$N_e \leq 50$	$50 < N_e < 500$	$N_e \geq 500$
- or -	*- or -*	*- or -*	*- or -*
Population size per generation	$N_g \leq 250$	$250 < N_g < 2500$	$N_g \geq 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[d]	Not apparent
Spawner density (adults/IP km)	$N_a/IP\ km \leq 1$	$1 < N_a/IP\ km <$ MRSD[e]	$N_a/IP\ km \geq$ MRSD[e]
Hatchery influence			Hatchery fraction <5%
			- in addition to above -
Extinction risk from PVA	≥20% within 20 yrs	≥5% within 100 yrs but <20% within 20 yrs	< 5% within 100 yrs[f]

[a] The effective population size (N_e) is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931); total number spawners per generation (N_g), for SONCC coho salmon the generation time is approximately three years therefore $N_g = 3\ N_a$.

[b] Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size of $N_a \leq 500$ spawners (historically small but stable populations not included) **or** $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.

[c] Annual spawner abundance N_a has declined to ≤500 spawners, but now stable **or** number of adult spawners (N_a) > 500 but continued downward trend is evident.

[d] Annual spawner abundance decline in one generation < 90% but biologically significant (e.g., loss of year class).

[e] MRSD = minimum required spawner density is dependent on the amount of potential habitat available. Figure 5 summarizes the relationship between spawner density and IP km.

[f] For population to be considered at low-risk of extinction, all criteria must be satisfied (i.e., not just a PVA). A population viability analysis (PVA) can be also included for consideration, but must estimate an extinction risk <5% within 100 years *and* all other criteria must be met. If discrepancies exist between PVA results and other criteria, results need to be thoroughly examined and potential limitations of either approach are carefully identified and examined.

two or more) receives the same ranking using either the Allendorf et al. (1997) or our approach. One additional risk category we define is “data deficient” for populations that are believed to still persist but where data for evaluating risk are partially or entirely lacking. In general, we viewed our configuration of the criteria table to be somewhat simpler to apply to extinction risk and viability.

In addition to these modifications in extinction risk categories, we also add two criteria not found in the Allendorf et al. (1997) framework. First, we add a criterion related to the percentage of fish spawning in the wild that are of hatchery origin. This criterion is intended to address potential genetic consequences of hatchery fish on their wild counterparts. Additionally, we add criteria related to spawner density, to address both potential demographic risks associated with very low population densities (i.e., depensation) and concerns about loss of spatial structure and diversity.

In this report, we adopt terms used by the North-Central California Coast Technical Recovery Team for the Central California Coast Coho Salmon ESU (Spence et al. 2008). We use three different terms to describe population size: the number of annual spawners (N_a), the number of spawners per generation (N_g), and the effective population size (N_e)[4]. The reason we express population size estimates as functions of both annual run size and the numbers of spawners per generation reflects the different time scales over which various processes considered by specific criteria occur. For example, many demographic processes operate at an annual time scale whereas it is usually more appropriate to consider genetic processes at generational time scales.

2.2 Population Viability Analysis

Population viability analysis estimates the probability of population extinction or collapse within a given time period (Beissinger and McCullough 2002). It focuses on how habitat loss,

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[4] The effective population size is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931).

environmental uncertainty, demographic stochasticity, and genetic factors interact to determine extinction risk (National Research Council 1995).

Numerous models for PVA have been proposed (Dennis et al. 1991; Ratner et al. 1997; Lindley 2003), although, the use of PVA has been viewed with skepticism by many scientists (Taylor 1995; Beissinger and Westphal 1998). The practical utility of any viability model depends on the validity of its underlying assumptions (National Research Council 1995). Ellner et al. (2002) discuss the need for some accountability for imprecision in parameter estimates and its consequences for risk assessments based on PVAs. Reed et al. (2002) reviewed various issues related to the use of PVA as a tool in endangered species management.

Data needed for PVA to estimate extinction risk include current population abundance, intrinsic population growth rate, habitat capacity, and variation in fecundity, growth, or survival (Belovsky 1987; Lande and Orzack 1988; Lande 1993); see also Beissinger and McCullough (2002) for a review of PVA. Either long-term time series data or intensive population or species-specific data sets (e.g., stage-specific survival rates, etc.) are required for a PVA. In general, short-term studies underestimate extinction risk since they fail to capture the greater temporal variation in population size and demographic parameters used in these models (Reed et al. 2002). We are not aware of data appropriate for PVA for any of the coho salmon populations in the SONCC ESU.

As previously mentioned, there has been concern expressed in the literature over how PVA models are used for making conservation decisions (see Beissinger (2002) for review). The dominant causes of uncertainty in the outcomes predicted by PVA model results include: 1) poor data, 2) difficulties in parameter estimation, 3) weak ability to validate models, and 4) effects of alternative model structures (Beissinger 2002).

There was much discussion within the TRT concerning how best to incorporate PVA into our low-risk criteria in Table 3. One underlying concern was that, given the range of possible approaches to PVA, an analysis might result in an extinction risk <5% within 100 years but may not provide a sufficiently accurate estimate of true extinction risk of a population, especially with the paucity of appropriate data for coho populations in the SONCC ESU. On the other hand, inclusion of the option to use a PVA in addition to the more generic criteria we propose is intended to encourage a more rigorous population-specific analysis integrating site-specific data. As pointed out in various responses to the Allendorf et al. (1997) approach (Wainwright and Waples 1998; Currens et al. 1998) and discussed by others in the literature, PVAs should be just one tool recovery planners use to evaluate extinction risks.

Allendorf et al. (1997) proposed the use of PVA to estimate extinction risk over a specified time and the use of alternate criteria when an acceptable PVA was not available. For our purposes, we propose the use of our general criteria (analogous to Allendorf et al.'s alternate criteria) to determine if populations are at low risk of extinction (Table 3). We suggest that as data become available, a PVA can be used to gain better understanding of population-specific dynamics and provide insight into possible population-specific characteristics to consider in the context of the general criteria to assess whether a population is at low-risk of extinction. Like Allendorf et al. (1997), PVA can be used to designate a population as at moderate- or high-risk of extinction (Table 3).

Because of the uncertainty expressed by many authors and members of the TRT, we require that all of the general criteria should be met for a population to be declared at low risk of extinction (Table 3). If a credible PVA can be constructed, results should be compared to results of the general criteria we propose, and by comparison of the outcomes, potential limitations of either approach identified and examined. A PVA is not required to determine a low-risk designation, but a PVA alone does not supersede the general criteria. For high-risk and moderate-risk determination, a PVA result alone can be used to establish risk level, although we strongly recommend that the PVA results be compared to results of the general criteria we propose. We also caution against using PVA analysis alone to assess population viability.

Our approach of advocating the use of general criteria rules over a PVA for assessing low-risk status of a population follows closely the recommendations of Shaffer et al. (2002):

> “As appealing as PVA models are for doing rigorous risk assessments of species extinction, our collective experience as a community of scientists and managers seems to be that, in most cases, we are probably overdriving our headlights. … The situation is unlikely to improve quickly. In the meantime, we have only two options for influencing the debate over how much habitat is enough to accomplish conservation: evolution of the status quo, or development of rules of thumb.”

Shaffer et al. (2002) listed the kinds of steps required for the evolution of the status quo to include: (1) developing standards of data, data analyses, and modeling that constitute an acceptable PVA, (2) model validation through laboratory and field experiments, and (3) long-term research to better understand how populations fluctuate. As part of the recovery planning process, recommendations on research and monitoring needs will be developed by the TRT and will address many of the steps suggested by Shaffer et al. (2002).

The development of rules of thumb suggested by Shaffer et al. (2002) is consistent with the framework the TRT has proposed. To this point Shaffer et al. (2002) states the following:

> “But what should be done when there are not data, or when we must prepare for a time horizon that exceeds the ability of the data and models to produce credible forecasts? It seems inescapable that the conservation biology community must, in fact, suggest rules of thumb, or guidelines, that will lead managers to make defensible judgments about how much habitat or what population size is enough to consider a species conserved.”

Shaffer et al. (2002) presented three principles that should be used in developing rules of thumb to make viability judgments without adequate data or models. The three principles are representation, redundancy, and resiliency. Of these, representation and redundancy are most directly captured by our ESU viability criteria, whereas resiliency is most directly captured by our spatial structure and diversity criteria at the population level (discussed below). Regardless, the foundation of all of these principles and the fundamental starting point of our approach is the population unit.

In summary, the TRT strongly recommends that PVA results be compared to results of the other criteria we propose, and that potential limitations of either approach be carefully identified and examined. In addition, any PVA analysis should be subjected to rigorous scientific peer review.

2.3 Effective Population Size / Total Population Size

These first two extinction risk criteria are intended to address concerns related to loss of genetic diversity within a population. Genetic variability is the source of adaptive potential of a population. Loss of genetic variability may affect the ability of a population (or ESU or species) to adapt to environmental change and may reduce survivorship and fecundity (Burgman et al. 1993; Allendorf et al. 1997; Willi et al. 2006). Deleterious genetic effects of small population size are a function of the effective population size (N_e), rather than the total number of spawners per generation (N_g), or census size.

The effective population size is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931). The effective population size of a population is usually smaller than the census population size because of variation in individual reproductive success, unequal sex ratios, and temporal variation in population size (Gall 1987; Burgman et al. 1993; Ardren and Kapuscinski 2003).

Direct estimates of N_e would be the most relevant measure for evaluating genetic risk to populations, but direct estimates are difficult to obtain from natural populations (Nelson and Soulé 1987; Waples 2002; Heath et al. 2002). In the absence of a direct estimate of N_e, an estimate based on the census population size can be made assuming a ratio of effective population size to total population size per generation, N_e/N_g. Allendorf et al. (1997) assumed a N_e/N_g ratio of 0.20, citing personal communication with R. Waples (NMFS, Northwest Fisheries Science Center). Later studies with Chinook salmon (Waples 2004) and steelhead (Heath et al. 2002) have reported N_e/N_g ratios between 0.05 and 0.3. Higher N_e/N_g ratios have occasionally been reported for salmonids. For example, for a single population of steelhead in Washington, Ardren and Kapuscinski (2003) reported N_e/N_g ratios over a 18-year period of 0.73 or 0.53, depending on the method of calculation (temporal method versus comprehensive demographic estimate). Nevertheless, we concluded that the 0.20 value proposed by Allendorf et al. (1997) was a reasonably precautionary default value to use for relating total fish abundance to effective population size when no direct estimates are available. The total population size criterion provides an alternative to the effective population size when estimates of effective population size are not available, as will likely be the case with most SONCC coho salmon populations.

We propose three different categories of genetic risk (i.e., high, moderate, low) related to effective population size, which are defined by two thresholds (i.e., $N_e = 50$ or 500). Populations are rated at high risk of extinction at $N_e \leq 50$ (or $N_g \leq 250$); at this level, populations are believed to be at high risk from random genetic risk such as inbreeding and fixation of deleterious alleles (Nelson and Soulé 1987; Frankel and Soulé 1981). Populations with $50 < N_e < 500$ (or $250 < N_g < 2500$) are considered at moderate risk, and populations with $N_e \geq 500$ (or $N_g \geq 2500$) are at low risk of extinction from genetic effects (Table 3). It should be noted, that for SONCC coho salmon with a generation time of approximately three years, the $N_g = 250$ would represent an annual abundance (N_a) of 83 fish, although effective population size should be considered at the generational time scale.

There has been an on-going discussion in the literature concerning the use of $N_e = 500$ as a threshold between low and moderate risk (Allendorf and Ryman 2002). Allendorf et al. (1997) proposed that long-term adaptive potential begins to be compromised at $N_e = 500$, although they noted that if populations are reproductively isolated from other populations then the N_e required to prevent loss of genetic variation might be as much as an order of magnitude greater (i.e, $N_e =$ 5,000; Nelson and Soulé 1987). Lande (1995) has suggested that an N_e of 5,000 rather than 500 may be necessary to maintain normal levels of adaptive genetic variance in quantitative characters under a balance between mutation and genetic drift. This was based on the concern that the models used to derive the N_e 500 rule assumed all mutations were mildly deleterious, whereas recent work suggests that most mutations with large effects are strongly detrimental, with perhaps only 10% being mildly deleterious. In contrast, the models of Franklin (1980) and

Soulé (1980) assume that populations are closed to immigration. Even low levels of immigration (i.e., straying) of as few as one or two individuals per generation, may prevent the loss of alleles through genetic drift (Lacy 1987). For most salmon populations, stray rates among populations at these levels are not unusual, or at least were not under historical conditions. Because violations in the assumptions act in opposition to one another, we accepted the $N_e = 500$ recommendation of Allendorf et al. (1997) as a reasonable default criterion for defining the break between populations at low and moderate risk.

Recently, Ardren and Kapuscinski (2003) developed demographic and genetic estimates of N_e for a steelhead population in Washington and concluded that at N_e levels between 50 and 500, the population was not losing diversity at a rate fast enough to warrant immediate concerns about inbreeding or loss of heterozygosity. However, by not having an N_e above 500 the population was unlikely to undergo increases in population genetic parameters, such as additive genetic variation and heterozygosity, that have been positively associated with long-term evolutionary potential. Ardren and Kapuscinski's (2003) results and discussion provide additional support for the use of the 50 and 500 threshold as default values.

The estimate of total population size per generation (N_g) we use is based on a harmonic mean of the running sum of adult spawner abundance over the mean generation time. We assume a mean generation time of three years throughout the SONCC ESU. If future research and monitoring indicate population differences in mean generation time, population-specific generation time values should be used.

The total population size per generation is calculated by

$$\widetilde{N}_{g(harmonic)} = \frac{1}{\frac{1}{n}\sum_{t=1}^{n}\frac{1}{N_{g(t)}}}$$

where $N_{g(t)}$ is the running sum of adult abundance at time t for a period equal to the mean generation time (three years) and n is the number of years for which the running sum can be calculated. We recommend a minimum of four generations (i.e., 12 years) since the effects of these genetic bottlenecks can linger for many generations. We also suggest that, if longer time series of data are available, this criterion be examined to determine if a specific population was subject to low effective population size at some time in the recent past (e.g., > 12 years). We use the harmonic mean, the reciprocal of the mean of the reciprocals, since it gives greater weight to low values of N_g and therefore captures our concerns over the potential long-term consequences of a genetic bottleneck on population persistence.

This estimate should be based on counts of naturally spawning fish (including jacks), exclusive of hatchery-origin fish, over a period representing at least four generations (i.e., 12 years). Allendorf et al. (1997) noted that spawner data often exclude jacks; however, jacks may contribute to subsequent generations and therefore need to be accounted for in the total population size estimate, although some adjustment for the relative reproductive success of jacks versus adults may be needed. Recent work by Van Doornik et al. (2002) estimated an effective proportion of 2-year-olds to be 35% in two naturally spawning populations in Washington, suggesting coho salmon should be treated as a species with overlapping generations.

In applying the total population size criteria, there are conditions that may lead to violations in our assumption of $N_e/N_g = 0.2$ that should be considered. The spatial structure of a population can affect the relationship between census size and effective population size (Whitlock and Barton 1997), as can highly skewed sex ratios, sex-biased differences in dispersal, and substantial among-family variation in survival rates (Gall 1987). Moreover, populations that have undergone a recent bottleneck may have a N_e/N_g ratio substantially below 0.2, indicating that the population remains at genetic risk even if total population size is large enough to suggest otherwise. Therefore, a population that has experienced a recent bottleneck may require a longer period of time with relatively high abundance to no longer be considered at risk. Also, it should be noted that Ardren and Kapuscinski (2003) found that a constant N_e/N_g ratio could not be assumed for a wild steelhead population. In the population they examined, they found that N_e/N_g

ratios increased with low numbers of fish, suggesting that genetic compensation and increased population productivity during brood years with few spawners can act to reduce demographic and genetic risks of extinction.

2.4 Population Decline

To address the increased demographic risks resulting from rapid or prolonged declines in abundance, we have included population decline criteria. The rationale for these criteria is that a severe and prolonged population decline resulting in small numbers of individuals is strong evidence that a population is at risk of extinction. In a variable environment any decline in population capacity proportionally increases the chances of population extinction (Shaffer 1987). We adopt criteria consistent with Allendorf et al. (1997), which include both a downward trend in population size and a minimum adult run-size.

A population is considered at high risk if it meets any of the following conditions: 1) the population has undergone a decline within the last two generations (i.e., 6 years) to an annual run size (N_a) of fewer than 500 spawners, 2) the population has an average annual run size $N_a > 500$ spawners but is declining at a rate of $\geq 10\%$ per year over the last two-to-four generations, or 3) the population currently has an annual average run size of $N_a > 500$ but is declining at a rate that would cause N_a to fall below 500 spawners within two generations (i.e., 6 years). Within this high-risk category, the progeny/parent ratio is less than one, indicating that populations are failing to replace themselves.

We consider the population at moderate extinction risk if it has declined to an annual run size below 500 spawners, but the numbers remain stable (i.e., progeny/parent ratio ≥ 1) or if $N_a > 500$ but continues to decline (i.e., progeny/parent ratio < 1), though not at a rate that will cause N_a to fall below 500 spawners within two generations. Populations with annual run sizes $N_a > 500$ adults and no apparent decline over the last two generations are considered at low extinction risk for this criterion. We note that these run sizes are distinct from N_e or census population size based on a N_e/N_g ratio used in evaluating effective population size; effective population size estimates represent abundance over an entire generation. For these population decline criteria we are evaluating the annual abundance (N_a). For consistency with Allendorf et al. (1997), we chose to use the value of 500.

The abundance threshold we adopt from Allendorf et al. (1997) is in the absence of information on intrinsic growth rate (i.e., growth rate when populations are released from competition at low population density). Modeling efforts to estimate extinction probability are often very sensitive to assumptions about intrinsic growth rate and environmental stochasticity (Goodman 1987; Lande 1993). For instance, a population with a high intrinsic growth rate and a low variance in the growth rate might have a relatively low extinction risk at $N_a < 500$, but a high probability of extinction if intrinsic growth rates were low and exhibited large variation. Recovery planners should be cautious about relaxing the thresholds for this criterion, especially when $N_a < 500$. Rigorous monitoring and evaluation is needed to support changes to this threshold. Recently Lindley (2003) suggested that a minimum of 30 years of data are likely required to obtain unbiased estimates of variance in population growth rate within reasonable confidence limits.

The population decline criteria require the calculation of two parameters, the mean annual spawner abundance (N_a) and the population trend (T). We recommend using the geometric mean of the most recent four generations (i.e., 12 years) to estimate annual population abundance. The geometric mean is slightly more conservative than the arithmetic mean. This estimate should be based on naturally spawning fish (including jacks), exclusive of hatchery-origin fish. This estimator is consistent with those used in recently published status reviews (Good et al. 2005). This estimator requires fewer data than spawner:spawner ratios where either age structure or an assumption of age structure is required. Moreover, our metric effectively captures our principle concern, which is to provide information on the long-term fate of the population.

Population trend should be evaluated for all populations to ascertain whether mean abundance is declining at a rate > 10% year, or if continued decline at the current rate would result in the annual spawner abundance to drop below the 500 fish threshold. In addition, examination of population trend will assist in the evaluation of populations where $N_a < 500$ and the populations appear to be stable.

Trend will be calculated as the slope of the regression of the number of natural spawners (ln-transformed) over the time series (minimum of 12 years for SONCC coho). To mediate zero values, 1 is added to natural spawners before transforming data. A slope less than zero (0) reflects a declining population (i.e., the long-term fate of the population is not good), a slope greater than zero reflect an increasing trend. The regression is calculated as:

$$\ln (N + 1) = \beta_0 + \beta_1 X + \epsilon$$

where β_1 is the slope of the equation and the value of interest, β_0 is the intercept, and ϵ is the random error term (Good et al. 2005).

The population decline criteria are intended to capture recent, relatively rapid declines in abundance. Over longer periods of time, populations that decline at less than 10% annually may still be at high risk of extinction. In the SONCC ESU, there are few existing time series of population abundance spanning longer than 10 years. In these cases, long-term trends should be evaluated independently of the proposed population decline criteria.

Examination of the trend will provide context when considering population declines, especially for populations that are below an N_a of 500 but appear to be stable in their abundance. Interpretations of population trends can be greatly influenced by the tendency of salmon populations to naturally fluctuate at time scales ranging from annual to decadal or longer, which can lead to highly variable estimates of trend. Since few time series of the appropriate length currently exist for coho populations in the SONCC ESU, estimates of trend will likely be based on relatively short time series of N_a. Interpretation of trend must to be made in the context of marine and freshwater survival during the time period being examined. It is not unreasonable to contend that populations at high abundance (e.g., $N_a > 10{,}000$ adults) might experience declines on the order of 10% or more per year for two generations without appreciably increasing their extinction risk. However, currently within the SONCC Coho Salmon ESU there is little evidence to suggest that any population is approaching these abundances. In the event that such circumstances arise in the future, it would be appropriate to reevaluate this element of the population decline criteria, particularly if information on potential sources of variation in population abundance is available.

2.5 Catastrophic Decline

The Catastrophic decline criteria were used by Allendorf et al. (1997) and included in our approach to capture situations where there is a rapid decline in abundance, often resulting from a catastrophic event. Catastrophes are large environmental disturbances that produce rapid and dramatic declines in population abundance (Shaffer 1987; Lande 1993). These types of disturbances are different than smaller stochastic environmental events that result from the continuous disturbances that affect population growth rate (e.g., ocean conditions, interannual climate variability). Catastrophes can occur across a range of spatial scales, from localized disturbances affecting a few miles of stream and therefore only a portion of a population, to those that may encompass the habitats of several populations (e.g., large wildfires). For the purpose of developing population viability criteria, we focus on catastrophes that affect all or a substantial portion of a population, producing a rapid and dramatic decline in population abundance. (The issue of larger-scale catastrophes is treated in ESU viability criteria.) In general, populations are at increased risk of extinction following a major reduction in abundance.

The criteria of Allendorf et al. (1997) defined a very high-risk situation as a 90% decline in population abundance within one generation and a high-risk situation as “any lesser but significant reduction in abundance resulting from a single event or disturbance.” Their purpose was to capture situations where a population had experienced a drastic shift from a low risk to a higher risk level. Although Allendorf et al. (1997) built upon the IUCN criteria (Mace and Lande 1991), their criteria differed from those of the IUCN, which proposed that average population declines over two to four generations of 50%, 20%, and 10% corresponded to critical, endangered, and vulnerable status, respectively. Allendorf et al. (1997) did not elaborate on why they departed from the IUCN criteria, but they did acknowledge that Pacific salmon and trout often exhibit substantial natural variation in abundance. Based on Allendorf et al.'s (1997) general statements concerning variability in abundance, we concluded that they considered declines of the magnitude specified in the IUCN criteria to be within the range of natural variation observed in salmonids and therefore adopted their more strict criteria.

We adopt the criteria as presented by Allendorf et al. (1997). Populations that have experienced a 90% decline in abundance within one generation are considered to be at "high risk" of extinction. Populations that have experienced a lesser but significant decline are considered at moderate risk. Although Allendorf et al. (1997) did not specifically define what constituted a "lesser but significant decline," we consider events such as a loss or near loss of a year class resulting from a catastrophic disturbance to be an example of a lesser but significant decline that would warrant classifying a population as at moderate risk of extinction. Such losses may be particularly important to coho salmon, which have a relatively fixed three-year life cycle with minimal overlap in generations. Events such as landslides, fire, severe flood or drought, chemical spills, or some other catastrophic event affect populations differently, and the risk associated with these different events can vary substantially depending on the specific circumstances such as year class strength. Because of this, we do not propose specific numeric thresholds for moderate risk and instead recommend that risk be evaluated on a case-by-case basis.

There may be instances where a population exhibits a clear precipitous decline in abundance or suffers a major loss or alteration of habitat (e.g., chemical spill affecting an entire year class, migration blockage, or some other catastrophic event) that should result in an immediate elevated risk designation, even in the absence of a longer time series of adult spawner abundance data. In addition, there may be situations where a longer time series indicates that a population has experienced a catastrophic decline in abundance at some time in the past. In such cases, consideration should be given to the response of the population following the catastrophic decline (see Figure 3). How a population responds to a catastrophic decline in abundance can provide a measure of its viability. For example, a relatively stable abundance following a catastrophic decline or an apparent upward trend toward recovery following a catastrophic

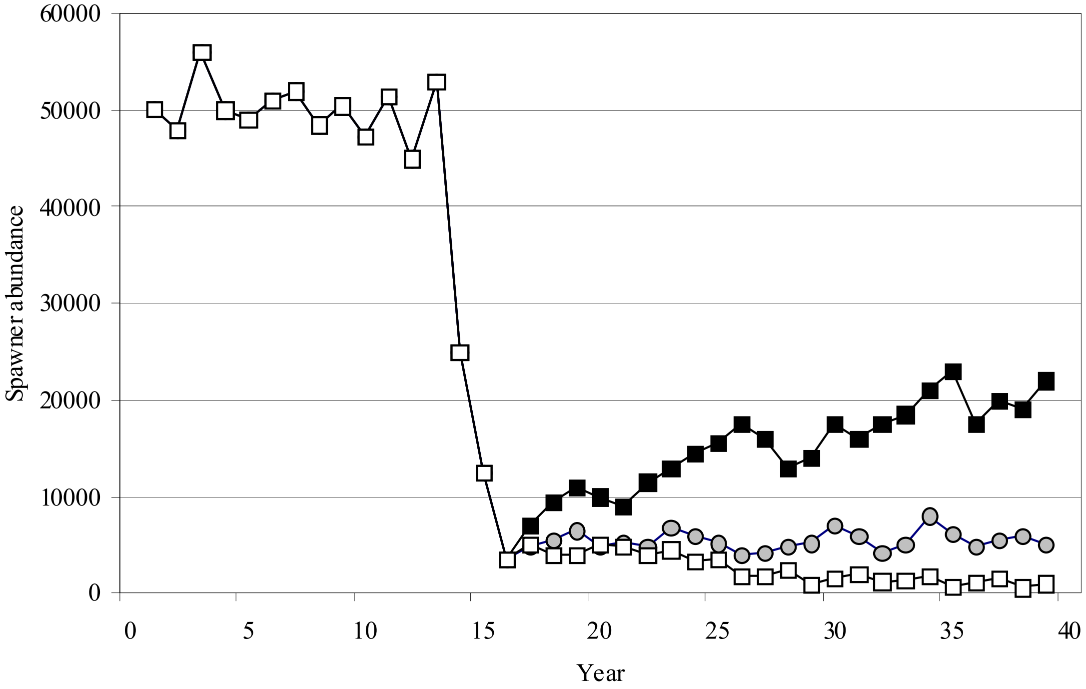

Figure 3. Hypothetical example of catastrophic decline in abundance showing three possible trajectories: a continued downward trend in abundance (open squares), a relatively stable abundance following decline (gray circles), and an apparent upward trend toward recovery following the decline (solid squares). Figure based on Spence et al. (2008).

decline might suggest a population tending towards viability (Figure 3). Although a stable abundance at a much-reduced level could also signal a fundamental reduction in carrying capacity (e.g., habitat degradation following a landslide). In contrast, a population that continues to decline following the initial catastrophic decline in abundance would indicate a population that may not be viable (Figure 3).

Certain types of catastrophic disturbances (e.g., wildfires, landslides that block access to habitats) may have long-term effects on carrying capacity, which may preclude a population from recovering rapidly. Other catastrophic disturbances (e.g., chemical spills), while they may strongly influence a year class or two, may not involve fundamental long-term shifts in habitat capacity, allowing for more rapid recovery as conditions improve.

The estimator we propose for catastrophic decline (*C)*, is the maximum proportional change in abundance from one generation to the next:

$$\hat{C} = maximum\left(1 - \frac{N_{g(t)}}{N_{g(t-2h)}}\right)$$

where $N_{g(t)}$ is the running generational sum of adult spawners in year t (i.e., $N_{a(t-2)} + N_{a(t-1)} + N_{a(t)}$), and $N_{g(t-2h)}$ is the running sum at time $t-2h$, where h is the mean generation time (i.e., three years for SONCC coho salmon).

The estimation of this criterion is based on the maximum proportional change in abundance from one generation to the next, although this is not simply based on the previous generation. Instead, we propose that the proportional change be based on the generation sum from two generations previously and therefore require a time series of adult spawner abundance (naturally spawning fish, exclusive of hatchery fish) of at least three generations (i.e., 9 years). For example, since coho salmon have a mean generation time of three years, the value calculated would be the sum of adult abundance for years 7, 8, and 9 (the most recent) divided by the sum of abundance for years 1, 2, and 3. We base these criteria on a time series over three generations because the value is highly influenced by the pattern of abundance during the transition period of high abundance to a period of low abundance as a result of it being based on a running sum of abundance. For example, consider the two time series of abundance presented in Figure 4. The “open-square line” depicts a situation where a population averaging around 50,000 spawners in years 1 though 13, drops in a single year to an average of 5,000 spawners in years 14 through 30. The “solid-square line” illustrates a similar scenario, but the decline occurs over a generation (3 years), rater than in a single year. If the running sum in the most recent generation were used in the denominator of the estimator, the value of C would exceed the 90% decline threshold only for the scenario where the decline occurred over one year (open-square line). In the second scenario, the intermediate abundances in years 14 and 15 moderate the value of C, such that the 90% criterion is never exceeded, despite the order of magnitude drop in abundance that occurred over a single generation (3 years). Our estimator would capture both scenarios as a catastrophic decline, regardless of whether the decline occurred over a single year or a full generation.

2.6 Spawner Density Criteria

As noted earlier, the spatial structure and diversity of populations can contribute to population persistence. Allendorf et al. (1997) included criteria for effective population size that address to some degree potential loss of diversity associated with small population size. However, no criteria are provided by Allendorf et al. (1997) that deal with potential loss of spatial structure or the loss of diversity that may result when populations no longer inhabit the range of environments that were historically occupied. Consequently, the TRT proposes as an addition to the Allendorf et al. framework criteria for spawner density that are intended to address these two population attributes (i.e., spatial structure and diversity).

Spatial structure and diversity influence population viability by spreading risk, both spatially and temporally, in addition to contributing to the resiliency of populations to various disturbances. The spatial arrangement of suitable spawning and rearing habitat within a watershed can be dynamic through time as a result of periodic disturbances that create a mosaic of varying habitat conditions (Reeves et al. 1995). Coho salmon distributed throughout a

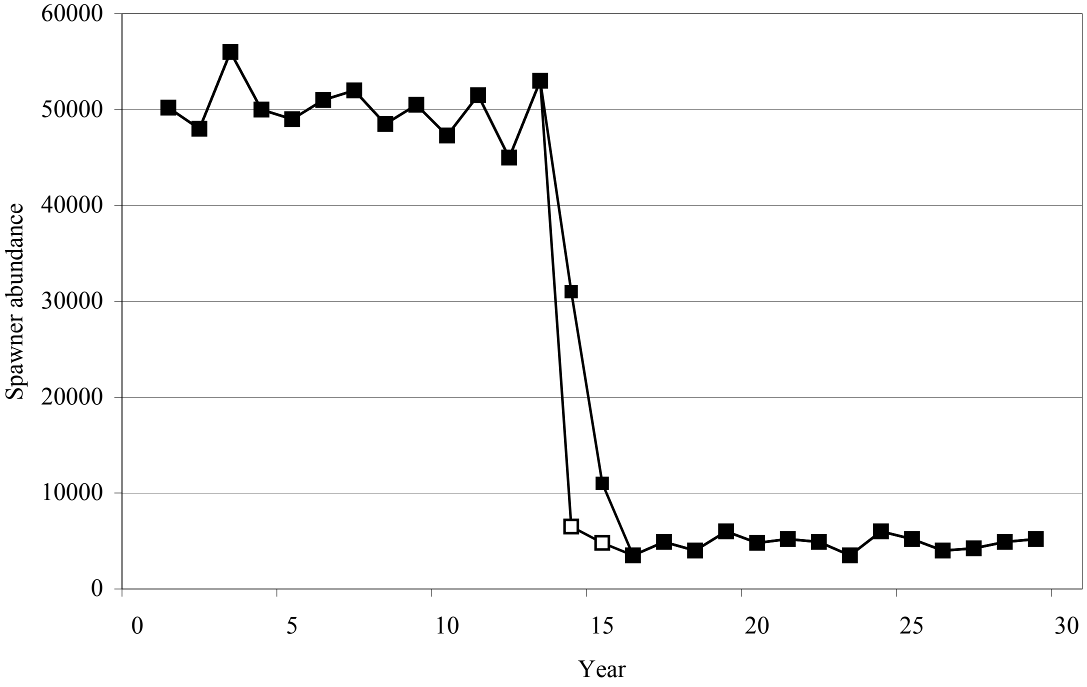

Figure 4. A hypothetical example where an order of magnitude decline in annual spawner abundance occurs over a single year (open squares) verses three years (solid squares). Figure based on Spence et al. (2008).

diversity of habitat conditions within a basin may be able to persist through periods of localized disturbance, with fish from areas not impacted by the disturbance available for recolonization (Parvinen et al. 2003; Kun and Scheuring 2006). A diversity of habitat conditions throughout a basin also results in exposure to a range of environmental conditions, which can lead to expression of greater phenotypic and genotypic diversity (Williams and Reeves 2003). Over the short term, greater phenotypic diversity (e.g., variation in adult or juvenile migration timing, length of freshwater residence, etc.) can help spread ecological risk (den Boer 1968). Over longer temporal scales, genetic diversity provides the material that a population draws upon to adapt to changes to the environment. As a population departs from its historical patterns of distribution and abundance as a result of habitat loss or degradation, the probability of persistence likely decreases, though numerous factors will determine how far a population can depart from historical conditions and still remain viable (Willi et al. 2006).

In contrast, populations that have been severely reduced in number may be subject to demographic processes that result in increased extinction risk. At very low densities, populations can experience a reduction in per capita growth rate with declining abundance, a phenomenon referred to as *depensation*. Depensation occurs when populations are reduced to very low densities and per capita growth rates decrease as a result of a variety of mechanisms (e.g., failure to find mates and therefore reduced probability of fertilization, failure to saturate predator populations, etc.; (Liermann and Hilborn 2001)). Depensation results in a negative feedback that accelerates a decline toward extinction.

The spawner density criterion varies with the size of the watershed the population occupies. Other viability metrics we have proposed are fixed values (i.e., effective population size and population decline criteria). The spawner density criterion accounts for historical differences in the total habitat available, the relative capacity of the habitat, the role of spatial structure and diversity in population persistence, the role of nutrient subsidies and physical actions of spawning in maintenance of ecosystem productivity, or the possibility of depensation when few individuals within a population are sparsely distributed across the available habitat. For example, an effective population size of 500 (or N_g of 2500) may be appropriate for a small basin and also represent an appropriate density of spawners, but the same number of fish spread throughout a large basin may be at a high risk of extinction. The spawner density criteria would require the larger basin to have more spawners than the smaller basin, although both basins would still have the 500-fish threshold for effective population size.

Within the framework of viable salmon populations, we considered population viability from two distinct but equally important perspectives (Spence et al. 2008). The first perspective involves defining the minimum viable population size (MVP) for which a population can be expected with some specified probability to persist over a specified period of time (Soulé 1987; Ralls et al. 2002). From this perspective, the minimum viable population size can be thought of as the lower bounds for a population where risks associated with demographic stochasticity, environmental stochasticity, severe inbreeding, and long-term genetic losses are negligible (Soulé 1987). This view of viability asks where a population is likely going in the future, but not necessarily where it has been in the past. With respect to genetic diversity, criteria related to a fixed MVP threshold size are intended to guard against further erosion of genetic diversity but do not necessarily consider diversity that may have been lost (Spence et al. 2008).

The second perspective in which viability can be considered is in respect to how a population is currently functioning in relation to its historical viability (Spence et al. 2008). From this perspective, the historical patterns of abundance, productivity, spatial structure, and diversity form the reference conditions about which there is high confidence that the population had a low extinction risk. This perspective takes a longer term and more broad view, and it asks how a population functioned in its historical context (e.g., what roles did spatial structure and diversity play in population persistence?) and what role the population played in relation to other populations within an ESU (Spence et al. 2008). As a population departs from historical conditions, its extinction risk likely increases and its functional role with respect to ESU viability diminishes.

Both perspectives are captured by our criteria. Our use of spawner density addresses the longer-term risks associated with loss of spatial structure and diversity that are important both for population resilience and therefore persistence and the ability of populations to fulfill their roles within the ESU and contribute to ESU viability.

In developing spawner density criteria to capture issues related to spatial structure and diversity, we made the following four assumptions. First, the historical distribution and abundance of spawners for an independent population represent reference conditions where extinction risk was likely low. Populations most likely tended towards a general carrying capacity, and the spatial structure, diversity, and productivity of the ecosystem resulted in low extinction risk in the absence of large-scale catastrophes. Second, the farther a population diverges from historical conditions, the greater the extinction risk and the greater the uncertainty of the population's viability. The more restricted and fragmented the distribution of individuals within a population, the higher the extinction risk. Some departure from historical conditions (e.g., diminished habitat conditions, reduced spatial distribution, loss of access to portions of habitat) may have little influence on population persistence, but the more these conditions diverge from historical conditions the greater the uncertainty of the population's viability. Third, the size of the population and its historical distribution largely determines how far it can deviate from historical conditions and remain viable. The thresholds we propose, based on a minimum amount of potential habitat capacity (IP km) required for viability-in-isolation, are based on the assumption that under historical conditions, populations were at or near carrying capacity. For example, a comparable percentage reduction of habitat would be less likely to increase extinction risk in a large watershed than it would in a small watershed where the attendant reduction in abundance and distribution would be more likely to move the population below levels required for viability. This is especially true for small populations that are near the IP km threshold for independence (Williams et al. 2006). Fourth, at extremely low densities populations are at a greater risk of extinction resulting from depensation.

The first three assumptions listed above relate directly to how far a population can diverge from historical conditions and remain viable, and therefore, provide our low risk-threshold. In practice, determining the low-risk threshold is also a difficult task since the scientific basis for quantitatively relating spatial structure, diversity, and ecosystem productivity to extinction risk is currently limited. The last assumption directly relates to the establishment of the high-risk threshold where we are concerned with the density at which depensation is likely to occur in coho salmon populations. Detecting depensation in salmonid populations is likewise difficult (Liermann and Hilborn 1997; Liermann and Hilborn 2001). Despite these uncertainties, we believe reasonable criteria can be developed from published literature and our general principles.

This potential habitat capacity, intrinsic potential (IP), is based on results from a Geographic Information Systems (GIS) model to predict the IP of coho salmon habitat (Burnett et al. 2003; Burnett et al. 2007). Its application to SONCC coho salmon was described by Williams et al. (2006). In brief, the model predicts the potential for a stream reach to exhibit habitat characteristics as a function of the underlying geomorphic and hydrologic characteristics of the landscape. A stream reach is a section of stream or river approximately 50 to 200 m in length and is generally defined with respect to geomorphologic and hydrologic features. Mean gradient, mean annual discharge, and valley constraint of stream reaches are used in the analysis. These characteristics are selected on the basis of being effectively constant features of the landscape that directly control the processes that create, alter, and maintain essential features of salmon habitat.

Specifically, IP is calculated as the geometric mean of suitability scores, which range from 0-1 and describe the potential that a stream reach with a specific value for a given characteristic will exhibit suitable habitat. These scores are generated by mapping the values for each of the three habitat characteristics (i.e., mean gradient, mean annual discharge, and valley constraint) onto suitability curves. The IP model itself has the structure of a limiting factors analysis, in that a low suitability score for a single habitat characteristic can greatly reduce (or eliminate) the potential for suitable habitat. We used this approach to generate predictions of IP for habitat of coho salmon using approaches developed by Burnett et al. (2003; 2007). The estimates of stream km based on IP model represent the total length of contiguous stream reaches with an IP score > 0. The IP score for each reach is weighted by the reach length (IP score X reach length), and the values for all reaches accessible by a given population are summed to obtain an estimate of IP km for that population. (see pages 12-14 in Williams et al. (2006) for more details of the IP model and how the historical distribution was derived).

It is important to note that the approach used by the TRT to set critical thresholds for the amount of habitat through the use of the IP model relies on the specific IP model we used and the associated suitability curves. The TRT's effort to propose the historical population structure of SONCC coho salmon also relied on this specific IP model to perform our viability-in-isolation analysis (Williams et al. 2006) and formed the foundation of our subsequent development of biological viability criteria. The use of a different model or different measure of habitat capacity (e.g., stream kilometers, basin size) could result in criteria thresholds different than those we propose and would likely require a rescaling of habitat capacity. For example, the 34 IP km required for a population to be viable in isolation would need to be re-examined, as would the threshold values for spawner density we propose for our spawner density. Most importantly, a consistent approach is needed to assess intrinsic potential (i.e., habitat capacity) across the ESU, and the IP model provides this consistency.

As discussed above, the use of spawner density defines two thresholds. The first addresses concerns resulting from depensation at very low densities and distinguishes between populations at high versus moderate risk. The second addresses concerns with spatial structure, diversity, and productivity and distinguishes between populations at moderate versus low risk. As previously discussed, defining a density at which depensation is likely to occur is extremely difficult since it can be highly variable and because of the limited number of spawner-recruit datasets where observations have been made at low abundances (Liermann and Hilborn 1997). Despite these difficulties, there have been numerous efforts to define thresholds at which depensation appears to occur in salmonids, and several of these efforts have focused on coho salmon (Chilcote 1999; Barrowman et al. 2003). In general, these and other studies have found little evidence of depensation in coho salmon unless densities were less than 1 female/km. Assuming a 50:50 sex ratio, this equates to 2 adult/km. In Chilcote (1999), it is suggested that coho salmon populations in the lower Columbia River were unlikely to recover if densities fell below 2.4 adults/km. The

Oregon Coast Workgroup of the ONCC (Wainwright et al. 2008) concluded that at spawner densities of 0.61 spawners/km (1 spawner/mile) demographic risks were certain to be significant.

Based on these studies, we set the threshold for high risk of depensation as those populations with an average spawner density of less than 1 adult per IP km. We chose to use IP km in the denominator in order to account for potential differences in habitat quality among watersheds. This was based on an assumption that IP km provides a reasonable measure of the relative productive potential of a watershed. For basins with similar IP km but different total km, the average density based on adults/km might be expected to be lower in the less productive watershed, perhaps leading to a greater depensation risk. However, we assume that in most cases fish will be distributed somewhat according to habitat quality. In general, the ratio of IP km:total km averages about 0.60 for watersheds within the SONCC ESU. Consequently, the Oregon Coast Workgroup value of 0.6 spawner/km translates to approximately 1 spawner/IP km, the criterion we use.

The value for our low-risk threshold varies as a function of population-specific estimates of habitat capacity (Figure 5). For the smallest watersheds capable of supporting an independent population in the SONCC coho salmon ESU, low-risk populations are those exceeding 40 spawners/IP km. For larger watersheds, the density requirement decreases with increasing capacity (i.e., IP km) to a minimum of 20 spawners/IP km based on our assumption that larger populations can diverge farther from historical conditions before extinction risk is substantially increased.

The low-risk criteria are based on the assumption that populations historically occurred, on average, at something close to the natural carrying capacity of the habitat (e.g., watershed) they occupied. Based on their viability-in-isolation analysis, Williams et al. (2006) defined the minimum threshold of potential habitat (expressed as IP km) required for a population to be considered viable-in-isolation to be 34 IP km for SONCC coho salmon. Therefore, for populations in the smallest watersheds (in terms of IP km) capable of supporting viable independent populations to remain viable, the population must function at something close to its

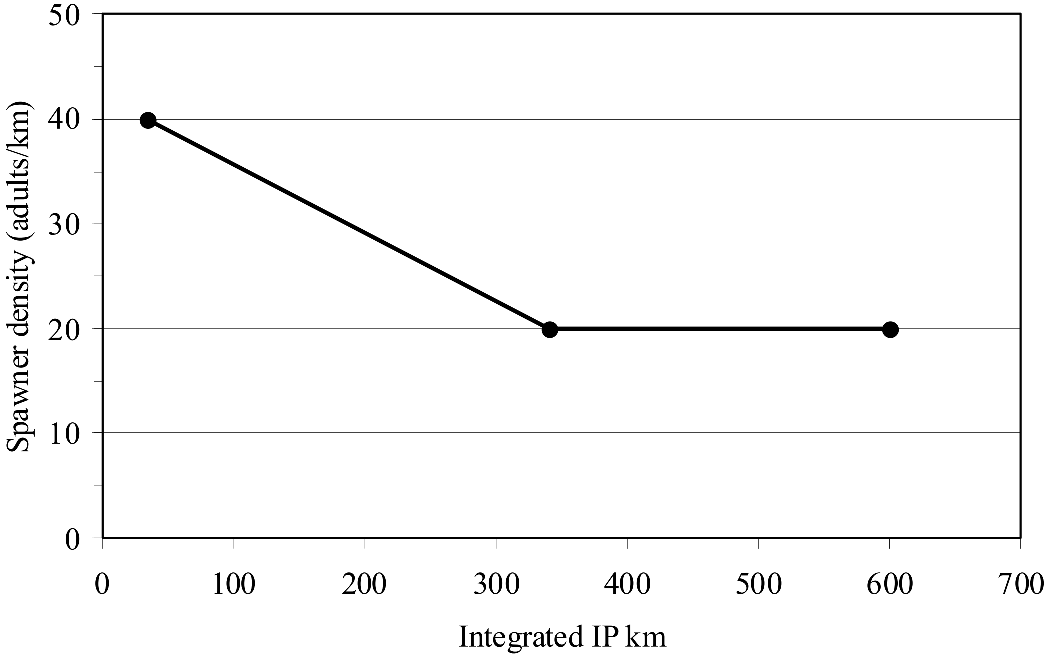

Figure 5. Minimum required spawning density (MRSD) based on amount of integrated IP km for coho salmon.

historical carrying capacity. Any reduction in capacity (either loss of access to portions of the historical habitat, or reduction in the productive capacity of the historical habitat) would drop the population below the threshold for viability (i.e., independence). Based on this reasoning, the average spawner density at historical carrying capacity serves as a reasonable basis for establishing the low-risk threshold for the smallest watersheds.

The TRT relied heavily on the work of Bradford et al. (2000) to address the difficult task of estimating the threshold for the low-risk criteria based on spawner density. Bradford et al. (2000) fit a model to 14 historical data sets of coho salmon from the Pacific Northwest and found that, on average, a density of 19 females/km is required to fully seed freshwater habitats with juveniles. Assuming a sex ratio that is slightly biased for males, we rounded the number to approximately 40 spawners/km for watersheds with a minimum IP required to be considered independent. It should be noted that Bradford et al. (2000) included watersheds in various states of disturbance and therefore does not necessarily indicate historical carrying capacity.

The establishment of the low-risk threshold of 40 spawners/IP km for the smallest populations was largely dictated by the threshold for viability-in-isolation proposed by Williams et al. (2006) and supported by empirical data and various modeling efforts reported in the literature. To accommodate our assumption that for larger populations a comparable percentage reduction in habitat is less likely to result in a substantial increase in extinction risk as it would in smaller populations, we assume that a population with ten-fold additional habitat potential than the smallest population requires an average spawner density of half that of the smallest population. This captures our general conclusion that the larger the historical population, the more it can depart from historical conditions and remain viable. The function we propose to capture this is a linear decline in required density between 40 spawners/IP km in the smallest populations to 20 spawners/IP km in the watersheds with greater than 10-fold the habitat potential of the minimum watershed (i.e., IP km > 340, Figure 5).

The development of this latter reference point was by the NCCC TRT (Spence et al. 2008) after much review and discussion, and although it is based largely on expert opinion, it provides results that are qualitatively consistent with the general hypotheses relating watershed size and density to spatial structure, diversity, and other factors that influence population persistence. The benefits of our approach for these criteria are that it establishes a population-specific abundance that is scaled to the amount of potential habitat and avoids the use of fixed abundance criteria. In addition, this approach captures the elements of spatial structure and diversity that contribute to viability without rigidly defining what the spatial structure must look like. For instance, in a large watershed the density criteria could be satisfied either by having fish distributed throughout the watershed at moderate densities or by having high densities in portions of the available habitat. Each of these scenarios has advantages and disadvantages from a population persistence perspective. For example, moderate densities spread throughout a watershed may be more resilient to localized disturbances than populations with more localized groups of fish at densities near carrying capacity densities. Conversely, localized areas of high productivity may be critical for population persistence during periods of unfavorable environmental conditions (Nickelson and Lawson 1998). The amount and distribution of productive habitat available to a population is dynamic and may change over time, especially given the dynamic nature of the geographic area of the SONCC ESU. Currently, we lack the appropriate data to make more spatially explicit criteria on spatial structure, but believe our approach captures the essence of the spatial structure and diversity elements outline by McElhany et al. (2000) for viable salmon populations. Future research and monitoring may allow for the development of explicit population-specific distribution criteria.

For the high-risk threshold that captures our concerns related to depensation, we propose an estimate of average spawner density (spawners/IP km) in the three consecutive years of lowest abundance (i.e., a moving three year average) within the last four generations (i.e., 12 years). Mathematically, we express this as follows:

$$\hat{D}_{dep} = \frac{\min\left(\frac{N_{g(t)}}{3}\right)}{X}$$

where N_g is the running generational sum of annual spawner abundance at time t and X is the estimate of potential habitat capacity in IP km for the watershed containing the population. We propose averaging the spawner density over three years within the last four generations versus a single year or over all years in order to have an indicator that is sensitive to a population that is at risk of depensation, without being too sensitive to natural fluctuations in abundance. For example, a population that experiences a single year of low abundance may be at minimal risk of falling into an accelerating pattern of depensation. Averaging over all years might lead to a few relatively good years masking a general pattern of very low spawner abundance. The use of the lowest three consecutive years looks for recurring evidence of population numbers sufficiently low that there is heightened potential for depensation. A concern we have that is not captured in this metric is the possibility of having two relatively healthy brood cycles of a coho salmon population masking the third brood cycle that is facing a high depensation risk. Taking the running sum may mask such a scenario, although such a situation may be captured by our population decline criteria or the "lesser but significant decline" element of the catastrophe criteria. For now, we recommend that recovery planners examine such situations on a case-by-case basis.

For the low-risk threshold, we propose the arithmetic mean of adult spawner density for all years over the last four generations, expressed as adult spawners/IP km:

$$\hat{D}_{ssd} = \frac{1}{4h} \sum_{t=1}^{4h} \frac{N_a}{X}$$

where N_a is annual spawner abundance and X is the estimate of potential habitat capacity in IP km for the watershed containing the population, and h is the mean generation time for the population. For coho salmon, we assume a mean generation time of three years throughout the SONCC ESU, in the absence of population-specific estimates of generation time. The estimated density is then evaluated against thresholds that are a function of population-specific estimates of potential habitat capacity or IP km (Table 4, Figure 5).

Estimates of density can be obtained from two different sampling approaches. First, it can be estimated from a direct weir count, or a count at some other fish passage facility, divided by the number of stream IP km accessible in the watershed. The count can be either a total fish count if all upstream migrating fish are counted or a total population estimate, if only a portion of upstream migrating fish are captured and the proportion can be accurately estimated. Both of these types of counts estimate annual run size, although they must represent the whole population being considered. Second, randomized spawner surveys can provide a total population estimate, which can then be divided by the total accessible IP km to obtain an average density over the entire watershed.

When considering the number of spawners for specific populations that result from our approach using spawner density it is important to view these numbers in the context of historical conditions. Given the current numbers of adult coho salmon returning to SONCC ESU streams and the current conditions of habitat throughout the region, these target values may seem overly ambitious. Habitat conditions in most of the watersheds of the SONCC ESU had been subjected to impacts from land use that included mining, water diversion, and timber harvest by the early 1900s, if not decades earlier. For example, it is reported that by 1880 hydraulic mining in the Applegate, Illinois, and Rogue River watersheds was occurring and resulted in dams and diversion ditches that interfered with fish migration (Atwood and Gray 2002). In Grants Pass, the Sugar Pine Door and Lumber Company was exporting an average of “four carloads of lumber a week, each containing 6,000 board feet” by 1886 (Atwood and Gray 2002).

The TRT also examined our targets for number of spawners by examining populations along the Oregon Coast and the Rogue River where there have been efforts to estimate historical abundances based on cannery records and also to examine a historical weir count in California.

Counts of coho salmon were made at Benbow Dam on the South Fork Eel River from 1938 to 1975. The dam was located 133 km upstream of the ocean and approximately 67 km upstream of the confluence with the mainstem Eel River. Consequently, counts at the dam represent only a

Table 4. Specific viability criteria of coho salmon populations in the SONCC ESU. Percent lost IP km represents the amount of habitat currently located upstream of dams. Depensation threshold represents the minimum number of spawners required to avoid depensation risk; spawner threshold is the minimum number of spawners required for a population to be considered at low-risk for the spatial structure and diversity threshold (based on spawner density value, fish/IP km, multiplied by IP km).

Population unit	Historical IP km	Depensation threshold (fish)	Spawner density (fish/IP km)	Spawner threshold low risk
Elk River (1)	62.64	63	38	2400
Lower Rogue River (7a)	80.88	81	37	3000
Illinois River (7b)	589.69	590	20	11800
Mid. Rogue/Applegate rivers (7c)	758.58	759	20	15200
Upper Rogue River (7d)	915.43	915	20	18300
Chetco River (10)	135.19	135	33	4500
Winchuck River (11)	56.50	57	39	2200
Smith River (12)	385.71	386	20	7700
Lower Klamath River (15a)	204.69	205	29	5900
Middle Klamath River (15b)	113.49	113	34	3900
Upper Klamath River (15c)	424.71	425	20	8500
Salmon River (15d)	114.80	115	35	4000
Scott River (15e)	440.87	441	20	8800
Shasta River (15f)	531.01	531	20	10600
South Fork Trinity River (15g)	241.83	242	26	6400
Lower Trinity River (15h)	112.01	112	35	3900
Upper Trinity River (15i)	64.33	64	37	2400
Redwood Creek (16)	151.02	151	32	4900
Maple Creek/Big Lagoon (18)	41.30	41	39	1600
Little River (19)	34.20	34	41	1400

Table 4. continued.

Population unit	Historical IP km	Depensation threshold (fish)	Spawner density (fish/IP km)	Spawner threshold low risk
Mad River (22)	152.87	153	32	4900
Humboldt Bay tributaries (23)	190.91	191	30	5700
Low. Eel/Van Duzen rivers (24a)	393.52	394	20	7900
South Fork Eel River (24b)	476.10	476	20	9500
Mainstem Eel River (24c)	143.90	144	33	4700
North Fork Eel River (24d)	53.97	54	39	2100
Mid. Fork Eel River (24e)	77.70	78	37	2900
Mid. Mainstem Eel River (24f)	255.50	256	25	6500
Upper Mainstem Eel River (24g)	54.11	54	39	2100
Bear River (26)	47.84	48	40	1900
Mattole River (28)	249.79	250	26	6500

portion of the independent population of coho salmon delineated in the population structure report. To compare historical abundance estimates with our density-based projections for coho salmon, we estimated the fraction of total IP km upstream of the dam and then multiplied this fraction by the overall abundance targets to obtain estimates of the contribution of above-dam habitat to the total population target. We then compared this estimate to the counts obtained from 1938 to 1950. This time period was presumed to be when the influence of land use was lowest (for the period of record), based on the fact that counts during these periods were generally higher on average than in the decades that followed.

The density-based abundance estimate used by the TRT projects a coho salmon population of 6,836 fish above Benbow Dam. The historical counts for the period 1938 to 1950 averaged 13,514 and ranged from 7,370 to 25,289 coho salmon. Our low-risk threshold was 51% of the average dam count, providing strong evidence that our methods do not overestimate the historical carrying capacities of coho salmon in the South Fork Eel River upstream of Benbow Dam. Moreover, our conclusion is strengthened when several additional factors are considered. Not accounted for in the historical counts at Benbow Dam are in-river and ocean harvest that was occurring in the period from 1930 to 1950s, the amount of habitat degradation that had likely already occurred in the South Fork Eel River by the late 1930s, and issues with fish passage at the dam that almost certainly underestimated the total number of fish passing the dam (see Spence et al. (2008) for more details). Hatchery activity did occur in the basin. Coho salmon were released from 1935 to 1938, with an average annual total of approximately 693,000 throughout Humboldt County. There are no data concerning distribution locations of these fish; thus, it is unclear if any of these fish, and if so, how many, were released into the South Fork Eel River and so may have influenced counts at Benbow Dam. The counts of coho salmon from 1938 to 1940, the years that would have been directly affected by hatchery releases in the South Fork Eel River, are lower on average (9,400) than counts in the period from 1941 to 1950 (14,900) when no planting occurred.

Comparisons of historical abundance estimates and hypothetical density-based abundance targets for coastal watersheds in Oregon also suggest that our methods do not overestimate the historical carrying capacities of coho salmon populations. Historical abundance estimates for Oregon populations were based on cannery records from 1892 to 1915 (Meengs and Lackey 2005). Meengs and Lackey (2005) estimated historical run sizes from cannery pack records through a series of steps including 1) converting salmon pack data (in cases) into pounds of salmon caught (by assuming a certain constant “waste” in processing); 2) converting pounds of salmon captured into numbers of adult fish (by assuming an average weight for adult fish of 4.46 kg); 3) converting numbers of harvested salmon into an estimate of total population sizes (assuming a specific catch efficiency rate); and 4) using the five years of highest abundance in each watershed as indicative of run size. The abundance targets that would result from application of our density-based criteria are well below, by an order of magnitude, historical estimates of abundance (Table 5). In all cases, the target abundance expressed as a percent of the historical estimates of abundance range between 3% and 12% (Table 5).

Meengs and Lackey (2005) also estimated salmon run sizes for the Rogue River for the late 1800s based on extrapolations from cannery pack. The historical estimate of coho salmon for the Rogue River was 114,000 and for Chinook salmon it was 154,000 (Meengs and Lackey 2005). The TRT has delineated four independent populations in the Rogue River Basin. The Lower Rogue River population unit is part of the Northern Coastal Basin diversity stratum. The Illinois River population unit, the Middle Rogue/Applegate rivers population unit, and the Upper Rogue River population unit make up the Interior – Rogue River diversity stratum. The ESU viability criterion (detailed in Section 3.2) requires 50% of the stratum total for the spawner density criteria be met for a stratum to be viable, which equates to 22,650, or about 20% of the estimated historical abundance for the greater watershed.

In summary, where we do have estimates of historical abundances of coho salmon to compare with abundance targets based on spawner density, our methods do not appear to overestimate the historical carrying capacities of coho salmon populations.

2.7 Hatchery Influence

Hatchery programs and the presence of cultured fish can impose various biological problems on salmon populations that include genetic and evolutionary risks, demographic risks, ecological risks, and problems due to the behavior, health status, or physiology of hatchery fish (National Research Council 1996). Specific genetic risks can include four fundamentally different adverse consequences: extinction, loss of within-population variability, loss of among-populations variability, and domestication (Busack and Currens 1995). Demographic risks can include direct risk, such as when wild adults are captured for hatchery broodstock, or indirect, such as when releases of large numbers of hatchery fish lead to excessive harvest on wild fish in mixed-stock fisheries. Ecological risks can include competition between hatchery and wild fish, predation on

Table 5. Comparison of abundance estimates and hypothetical density-based abundance targets for coastal watersheds in Oregon. IP km are integrated IP km values as described by Williams et al. (2006).

Population	Historical estimates of abundance derived from cannery records (Meengs and Lackey 2005)	IP km	Estimated historical spawner density (spawners/IP km)	Projected abundance target based on MRSD (20 spawners/IP km)[a]	Projected abundance target as percent of historical estimate
Nehalam	236,000	1,116	211	22,300	9.3%
Tillamook	234,000	537	436	10,700	4.7%
Nestucca	107,000	299	358	6,800	6.4%
Siletz	122,000	310	394	6,800	5.6%
Siuslaw	547,000	902	607	18,000	3.3%
Yaquina	65,000	385	169	7,700	12.3%
Alsea	153,000	466	328	9,300	5.9%
Coquille	342,000	883	387	17,700	5.3%
Coos	161,000	552	292	11,000	6.8%

[a] – The Nestucca and Siletz populations have less than 340 IP km, therefore the MRSD values used for these calculations were 23 spawners/IP km for the Nestucca population and 22 spawners/IP km for the Siletz population.

wild fish by hatchery fish or by predators attracted to abundant hatchery fish, and transmission of diseases between hatchery and wild fish.

The evaluation of the various types of potential impacts of hatchery fish on wild fish is extremely difficult, as many of these impacts can be highly context-dependent. For example, the potential for negative competitive interactions depends on numerous factors, including the number and size of hatchery fish released, the size of the recipient population relative to carrying capacity, and the timing and location of release, among other factors. Likewise, genetic impacts of hatchery fish on wild fish depends on the origin of hatchery broodstock, broodstock collection practices, mating and rearing protocols, the duration of exposure to hatchery practice, the number of hatchery fish spawning in the wild, and a host of other factors. Consequently, an analysis of various impacts, in many instances, is best done on a case-by-case basis when specifics of past, present, and future hatchery operations can be considered. For the SONCC

Coho Salmon ESU, we are not aware of any rigorous studies currently available that demonstrate no or negligible ecological or genetic effects resulting from current or past hatchery operations.

The TRT felt that criteria addressing the potential genetic risks of hatchery fish were warranted, as there exists a substantial literature on these risks (Chilcote 2003; Einum and Fleming 2001; Goodman 2005; National Research Council 1996). In the absence of such studies, populations are at low risk if it can be demonstrated that there are no ecological or genetic effects resulting from current or past hatchery operations. For our purposes, we consider a population to be at least at moderate risk if the fraction of naturally spawning fish that are of hatchery origin, as determined by appropriately designed surveys, exceeds 5%.

The TRT recommends a general low-risk threshold of 5% with recognition that although the appropriate low-risk threshold value is difficult to determine, there is very strong support in the literature for a precautionary approach when considering impacts of hatchery fish on wild populations (National Research Council 1996). Several researchers have suggested that even minimal contribution of hatchery fish can pose a risk to wild populations. In their review of literature data, Einum and Fleming (2001) reported that numerous studies have found interactions between wild and released salmonids resulting from current hatchery practices may be detrimental to the recipient wild populations. Chilcote (2003) suggested that if his findings for steelhead held for other salmonids, an effective method to increase the productivity of natural populations and associated conservation benefits may be to minimize the frequency of hatchery fish in natural spawning populations. Goodman (2005) found through modeling of phenotypic evolution of integrated hatchery and wild spawning programs a potential for substantial erosion of natural spawning fitness. Moreover, he cautions that the modeling shows that the depression of natural spawning fitness increases with the magnitude of the hatchery contribution. In their "Native Fish Conservation Plan" the Oregon Department of Fish and Wildlife (Oregon Department of Fish and Wildlife 2003) proposes that at least 90% of the spawners within a population must be naturally produced and not hatchery produced fish (with some exceptions).

We do not propose specific metrics for assessing genetic risk beyond the general low-risk threshold of 5%, acknowledging that the uncertainty in quantitatively relating the risk of hatchery fish to extinction risk by a single ESU-wide threshold might not be appropriate. For instance, various best-management procedures may be in place that might reduce the risk and allow for a more liberal threshold level. Also, an appropriately planned and executed phase-out of a hatchery program might also allow for a more liberal threshold level if the targeted endpoint is a viable wild spawning population, recognizing that in some situations hatchery programs may play a role in population recovery. The 5% threshold we propose is a default value for recovery planners in the absence of any population-specific research and monitoring that would more directly measure risks, both genetic and the host of other previously mentioned potential impacts of hatchery fish on the wild population of interest. In addition, this 5% is selected in the absence of information concerning the hatchery practices in place. Our 5% threshold relates directly to genetic effects; other effects (e.g., ecological) may not be protected by the 5% threshold and therefore should still be considered. Factors that contribute to genetic risk, such as origin of broodstock (e.g., within basin, within ESU, etc.), broodstock collection procedures, and mating and rearing protocols, should be considered by recovery planners in assessing the impacts of hatchery fish. In addition, hatchery operations change over time. The legacy of past hatchery practices as well as current hatchery protocols should be considered on a case-by-case basis.

The types of analyses used to assess the impacts of hatchery contributions on a population should include a suite of considerations. Although not exhaustive, the list below provides an example of the types of questions that should be addressed before a credible determination of hatchery risk can be made:

- Is the productivity of the naturally spawning component of a population consistent with demographic viability?
- Are there changes in characteristics of the integrated population that indicate reduced ability of naturally spawning component to persist in the absence of hatchery production?

- Is the hatchery prone to outbreaks of disease that would place a wild or integrated hatchery-wild population at heightened demographic risk?
- If wild fish are being used for broodstock, is the remaining wild population at heightened risk?

We also suggest an approach that builds upon the efforts by the Interior Columbia Basin TRT (Interior Columbia Basin Technical Recovery Team 2005a; Interior Columbia Basin Technical Recovery Team 2005b), which recognized that the risk associated with hatcheries is heavily influenced by not only fraction of hatchery fish spawning in the wild, but also the degree of genetic similarity between hatchery and wild broodstock and the number of generations over which the impact has occurred.

2.8 Population Viability of SONCC Coho Salmon Populations

As discussed in the introduction of this document, there are currently almost no data at the appropriate spatial scale or temporal scale (i.e., enough years of data from present back 9 to 12 years) to assess the viability of coho salmon populations in the SONCC ESU. This is not to say there are not efforts currently underway within the SONCC ESU to collect various types of data on coho salmon. While these programs may be important for answering specific questions about the ecology of coho salmon populations in general, or local patterns of trend and abundance, they are generally not sufficient for assessing status and trends at the population level. A benefit of many of the projects currently underway is that they will provide recovery planners some insight into various biological, ecological, and logistical considerations needed to develop a more comprehensive monitoring program for coho populations in the SONCC ESU.

For recovery planners, our criteria (Table 3) provide guidance on the types of data of interest, the spatial scale needed (population unit), and the temporal length of time series required to assess viability, even with the very general approach we have proposed. The approach we have taken provides a framework that can assist recovery planners in developing recovery strategies and evaluating progress toward recovery, providing targets that can help prioritize recovery efforts within the SONCC ESU (Table 4). Clearly, considerable data are needed to perform a relatively simple and objective assessment of status. Recovery planners need to be extremely cautious concerning efforts to assemble various incomplete or incompatible data sets in hopes of obtaining the various population-specific values we have proposed to assess viability. Conclusions from such efforts can be based on incomplete data or sampling efforts that are not representative of the greater population of interest and could lead to erroneous conclusions about current viability.

Past status reviews of coho salmon in the SONCC ESU have necessarily relied on existing data that were not intended to provide population-level estimates of abundance. In the most recent federal status review, Good et al. (2005) found that coho salmon populations in the SONCC ESU continue to be depressed relative to their historical number and that there were strong indications that breeding groups had been lost from a significant portion of the historical range. Although the appropriate data are lacking to assess population viability using the framework we have proposed, data available at the present and used by Good et al. (2005) are in agreement with an earlier assessment (Weitkamp et al. 1995) that SONCC coho salmon are likely to become endangered in the foreseeable future. In these status reviews, uncertainty that arises from the rigor of sampling designs, representation of the sampling unit, and length of time series are recognized and factor into final conclusions about status. The framework we have proposed provides a consistent and robust means for evaluating status that reduces these uncertainties, in addition to providing the population and ESU targets needed for recovery planning.

Future research and monitoring may also lead to changes in estimates of historically assessable habitat that could lead to changes, either greater or lesser, in the spawner density threshold and the depensation threshold that are based on the amount of historical IP km. For example, recent temperature monitoring in the Kalmiopsis Wilderness Area of the Chetco River has found stream reaches within the basin with water temperatures that might limit use by juvenile coho salmon. Similar pre- and post-fire temperatures recently observed in this wilderness area and areas downstream have been suggested to indicate that portions of the stream with positive IP km values may not have been historically available to juvenile coho salmon and therefore the total IP km should be reduced, decreasing the target abundance based on the spawner density criteria. The unique environmental conditions of the Kalmiopsis Wilderness Area of the Chetco River could exhibit conditions that are not appropriately captured by the IP model and temperature mask used by Williams et al. (2006) to delineate historically assessable habitat.

The TRT considered this, and other similar situations and advises recovery planners to accept changes to the baseline IP km only after critical review (i.e., peer review) of each unique situation. Historical context is the basis, so pre-1900 conditions should be considered even if limited land use has occurred over the past 50 to 100 years. In addition, stream reaches and tributaries with appropriate temperatures for juvenile coho salmon may exist in tributaries both downstream and upstream of portions with less than suitable water temperatures (Welsh et al. 2001; Madej et al. 2006); specifically, the occurrence of unsuitable water temperatures in a reach of stream should not, by default, exclude all upstream portions of the basin to coho salmon use. Exclusion of stream reaches should be based on a thorough examination of water temperatures throughout the basin and within a historical context. In addition, movement among stream reaches and among tributaries should be considered. Recently Ebersole et al. (2006) describe within basin movement of juvenile coho salmon in a coastal Oregon basin (West Fork Smith River, Douglas County, Oregon). Although the focus of their work was on winter movement and overwinter use of tributaries, Ebersole et al. (2006) suggested that during wetter years, small tributaries could provide improved summer survival and subsequently higher densities of juvenile salmonids prior to the overwinter period.

Future research and monitoring should examine basin-wide conditions (e.g., inter-tributary variability in water temperatures and within-basin fish movement) to estimate historically available habitat in basins such as the Chetco River and Klamath Basin in order to better capture the historically assessable habitat and therefore the baseline for the density based viability criteria. Poole and Berman (2001) describe important processes and pathways that should be considered when considering the external drivers of stream temperature, the internal structures and processes that insulate and buffer stream temperatures, and the mechanisms of human influence on stream temperature. Two of the key points made by Poole and Berman (2001) are that 1) inaccurate or incomplete conceptual understanding of complex spatial and temporal stream temperature response patterns to anthropogenic influences can jeopardize stream temperature research and monitoring, and 2) analyses of land-use history and the historical versus contemporary structure of the stream channel, riparian zone, and alluvial aquifer are critical prerequisites for applying mechanistic temperature models.

2.8.1 Upper Rogue River Population Unit Example

The Upper Rogue River population unit is one of three independent populations in the Interior/Rogue River Basin stratum of the SONCC Coho Salmon ESU. Historically, we estimate that approximately 915 IP km were accessible to this population. However, Lost Creek Dam (Rkm 254), which began operations in February 1977, blocks access to an estimated 94 IP km that were historically accessible. Cole Rivers Hatchery, located at Rkm 252, began operation in late fall of 1972 and was built to mitigate for spawning and rearing areas blocked by the construction of Lost Creek Dam, as well as Applegate and Elk Creek dams, which are located downstream.

Fish counts have been made at Gold Ray Dam (Rkm 202) each year since 1942 (Figure 6, Table 6). These counts represent a partial count of the Upper Rogue coho population, with approximately 716 IP km of the historical habitat occurring upstream of Gold Ray Dam and about 200 IP km located downstream of the dam. An adjustment to the total IP km has been made for purposes of this example of application of the viability criteria with currently available population abundance data for the Upper Rogue population unit. Specifically, in applying density criteria, we use the historical value of 716 IP km to estimate spawner density (Figure 7).

Fish counted at Gold Ray Dam are distinguished as wild or hatchery based on fin clips. Except for possible strays from hatcheries outside of the basin, only wild adults returned until the mid-1970s (Oregon Department of Fish and Wildlife 1989). Releases of hatchery-reared coho salmon began in 1976. Available data includes counts at Gold Ray Dam from 1942 to 2007, including the critical last 12 years (1996 – 2007), which represent the last four generations for coho salmon most useful for application of the viability criteria.

Effective Population Size/Total Population Size–We know of no direct estimates of effective population size (N_e) for this population, so we used an estimate based on census population size assuming a ratio of effective population size to total population size per generation (N_g) of 0.20.

When using total population size, the critical thresholds for the various risk levels are:

High risk: $N_g \leq 250$

Moderate risk: $250 < N_g < 2500$

Low risk: $N_g \geq 2500$

Over the past 12 years (1996 – 2007) the harmonic mean of N_g of wild fish has not been below 2,500, placing this population unit at a ***low risk*** for this criterion (Table 6). However, when data from the entire time series were examined, we found harmonic mean N_g values dropped below the moderate risk threshold of $N_g = 2{,}500$ for 25 consecutive years from 1966 to 1990. This suggests the possibility that some loss of genetic diversity may have occurred during this period of low abundance.

Population Decline–A population is considered at high risk if it meets any of the following conditions: 1) the population had undergone a decline within the last two generations (i.e., 6 years) to an annual run size (N_a) of fewer than 500 spawners, 2) the population has an average annual run size $N_a > 500$ spawners but is declining at a rate of ≥ 10% per year over the last four generations, or 3) population currently has an annual average run size of $N_a > 500$ but is declining at a rate that would cause N_a to fall below 500 spawners within two generations (i.e., 6 years). We consider the population at moderate extinction risk if it has declined to an annual run size below 500 spawners, but the numbers remain stable (i.e., progeny/parent ratio > 1) or if $N_a >$

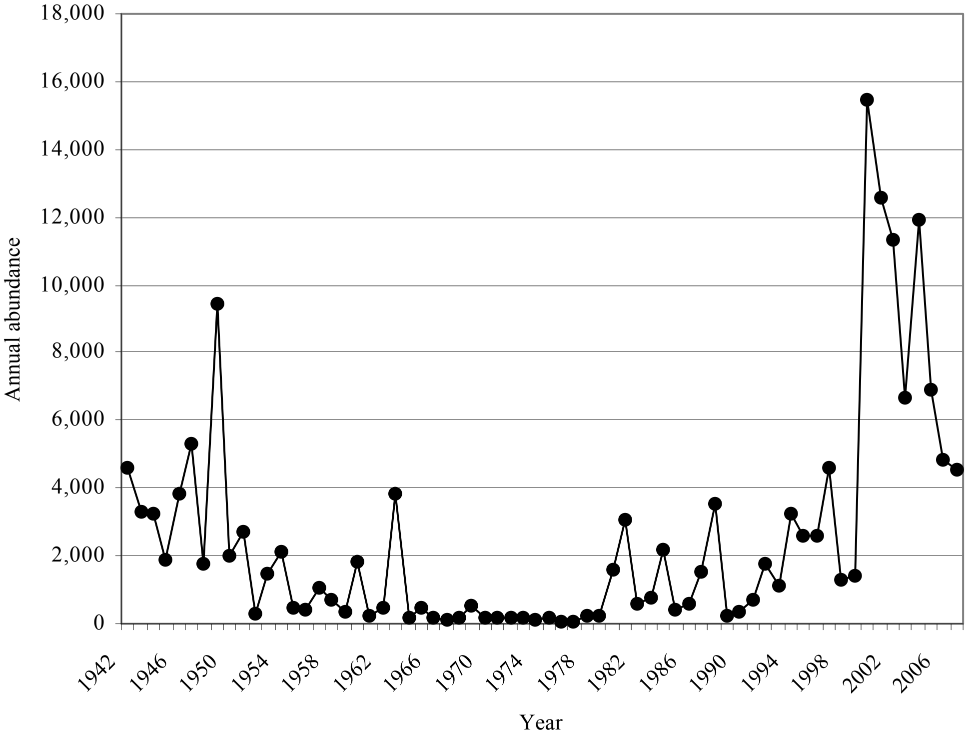

Figure 6. Annual estimates of wild adult and jack coho salmon at Gold Ray Dam (Rkm 202) on the Rogue River between 1942 and 2007.

Table 6. Gold Ray Dam coho salmon counts from 1942 to present.

	Adults ($\geq$ 20 inches) (Wild, Hatchery, Total)			Jacks (< 20 inches) (Wild, Hatchery, Total)			Total (adults+jacks)		Wild adults + jacks							
Year	W	H	T	W	H	T	Wild	Hatchery	N_g	12-yr Harmonic mean of N_g	12-yr geometric mean of N_a	ln (N_a)	Catatropic rate of decline ($\hat{C}$)	Hatchery fish (%)	Depensation 3-yr running mean of N_a	12-yr mean of N_a
1942	4,391	0	4,391	217	0	217	4,608	0								
1943	3,089	0	3,089	201	0	201	3,290	0								
1944	2,894	0	2,894	336	0	336	3,230	0	11,128						3,709	
1945	1,823	0	1,823	84	0	84	1,907	0	8,427						2,809	
1946	3,629	0	3,629	211	0	211	3,840	0	8,977						2,992	
1947	5,174	0	5,174	166	0	166	5,340	0	11,087						3,696	
1948	1,679	0	1,679	85	0	85	1,764	0	10,944						3,648	
1949	9,034	0	9,034	406	0	406	9,440	0	16,544						5,515	
1950	1,770	0	1,770	237	0	237	2,007	0	13,211						4,404	
1951	2,508	0	2,508	230	0	230	2,738	0	14,185						4,728	
1952	313	0	313	7	0	7	320	0	5,065						1,688	
1953	1,319	0	1,319	134	0	134	1,453	0	4,511						1,504	3,328
1954	1,907	0	1,907	231	0	231	2,138	0	3,911						1,304	3,122
1955	434	0	434	46	0	46	480	0	4,071	7,368	2,050	6.17	0.75		1,357	2,888
1956	398	0	398	23	0	23	421	0	3,039	6,425	1,730	6.04	0.77		1,013	2,654
1957	998	0	998	77	0	77	1,075	0	1,976	5,321	1,649	6.98	0.86		659	2,585
1958	648	0	648	84	0	84	732	0	2,228	4,629	1,437	6.60	0.56		743	2,326
1959	353	0	353	18	0	18	371	0	2,178	4,052	1,150	5.92	0.52		726	1,912
1960	1,757	0	1,757	94	0	94	1,851	0	2,954	3,740	1,155	7.52	0.24		985	1,919
1961	230	0	230	2	0	2	232	0	2,454	3,375	848	5.45	0.40		818	1,152
1962	457	0	457	0	0	0	457	0	2,540	3,098	750	6.12	0.16		847	1,022
1963	3,513	0	3,513	318	0	318	3,831	0	4,520	2,982	771	8.25	-1.29		1,507	1,113
1964	168	0	168	0	0	0	168	0	4,456	2,962	731	5.12	-1.00		1,485	1,101
1965	470	0	470	12	0	12	482	0	4,481	2,961	666	6.18	-1.06		1,494	1,020
1966	178	0	178	0	0	0	178	0	828	2,398	542	5.18	0.72		276	857
1967	89	0	89	0	0	0	89	0	749	1,969	471	4.49	0.69		250	824
1968	149	0	149	0	0	0	149	0	416	1,469	432	5.00	0.84		139	801

Table 6. continued.

	Adults (≥ 20 inches) (Wild, Hatchery, Total)			Jacks (< 20 inches) (Wild, Hatchery, Total)			Total (adults+jacks)		Wild adults + jacks							
Year	W	H	T	W	H	T	Wild	Hatchery	N_g	12-yr Harmonic mean of N_g	12-yr geometric mean of N_a	ln (N_a)	Catatropic rate of decline ($\hat{C}$)	Hatchery fish (%)	Depensation 3-yr running mean of N_a	12-yr mean of N_a
1969	530	0	530	0	0	0	530	0	768	1,338	407	6.27	0.83		256	756
1970	95	0	95	65	0	65	160	0	839	1,236	359	5.08	0.81		280	708
1971	181	0	181	0	0	0	181	0	871	1,154	338	5.20	0.81		290	692
1972	185	0	185	0	0	0	185	0	526	1,003	279	5.22	0.36		175	554
1973	193	0	193	0	0	0	193	0	559	899	275	5.26	0.25		186	550
1974	146	0	146	0	0	0	146	0	524	808	250	4.98	-0.26		175	524
1975	151	0	151	3	0	3	154	0	493	720	191	5.04	0.36		164	218
1976	27	0	27	17	0	17	44	0	344	620	171	3.78	0.59		115	208
1977	38	339	377	14	124	138	52	464	250	519	142	3.95	0.71	89.9%	83	172
1978	170	465	635	70	46	116	240	511	336	482	145	5.48	0.36	68.0%	112	177
1979	189	0	189	50	1,505	1,555	239	1,505	531	472	158	5.48	0.05	86.3%	177	189
1980	1,064	1,832	2,896	544	2,087	2,631	1,608	3,919	2,087	510	193	7.38	-2.98	70.9%	696	311
1981	2,771	3,377	6,148	285	292	577	3,055	3,670	4,902	535	223	8.02	-8.94	54.6%	1,634	521
1982	178	17	195	412	63	475	591	79	5,254	560	248	6.38	-14.27	11.8%	1,751	557
1983	459	286	745	337	411	748	796	697	4,442	586	281	6.68	-16.77	46.7%	1,481	609
1984	1,839	928	2,767	364	105	469	2,203	1,033	3,590	636	346	7.70	-9.68	31.9%	1,197	777
1985	320	502	822	91	257	348	411	759	3,410	691	368	6.02	-5.42	64.9%	1,137	795
1986	278	3,147	3,425	313	334	647	591	3,481	3,205	761	414	6.38	-0.54	85.5%	1,068	832
1987	1,456	2,979	4,435	80	880	960	1,537	3,858	2,539	848	501	7.34	0.48	71.5%	846	947
1988	3,240	2,999	6,239	305	338	643	3,545	3,337	5,673	1,052	722	8.17	-0.08	48.5%	1,891	1,239
1989	163	1,097	1,260	89	52	141	253	1,148	5,335	1,579	824	5.53	-0.20	81.9%	1,778	1,256
1990	306	329	635	25	37	62	331	366	4,129	2,466	846	5.80	-0.15	52.5%	1,376	1,263
1991	621	1,691	2,312	78	172	250	699	1,863	1,283	3,190	925	6.55	0.62	72.7%	428	1,302
1992	1,640	1,446	3,086	130	790	920	1,770	2,236	2,800	3,297	933	7.48	0.13	55.8%	933	1,315
1993	971	817	1,788	136	1,562	1,698	1,106	2,380	3,575	3,230	857	7.01	-0.41	68.3%	1,192	1,153
1994	2,742	6,432	9,174	503	1,022	1,525	3,244	7,455	6,120	3,253	988	8.08	-0.08	69.7%	2,040	1,374

Table 6. continued.

	Adults (≥ 20 inches) (Wild, Hatchery, Total)			Jacks (< 20 inches) (Wild, Hatchery, Total)			Total (adults+jacks)		Wild adults + jacks							
Year	W	H	T	W	H	T	Wild	Hatchery	N_g	12-yr Harmonic mean of N_g	12-yr geometric mean of N_a	ln (N_a)	Catatropic rate of decline ($\hat{C}$)	Hatchery fish (%)	Depensation 3-yr running mean of N_a	12-yr mean of N_a
1995	2,068	10,046	12,114	501	903	1,404	2,570	10,948	6,920	3,326	1,089	7.85	-0.30	81.0%	2,307	1,522
1996	1,745	9,799	11,544	827	1,228	2,055	2,572	11,027	8,386	3,480	1,103	7.85	-1.03	81.1%	2,795	1,552
1997	4,218	10,380	14,598	370	782	1,152	4,587	11,163	9,729	3,683	1,349	8.43	-6.58	70.9%	3,243	1,900
1998	1,074	3,686	4,760	251	1,031	1,282	1,325	4,717	8,484	3,917	1,443	7.19	-2.03	78.1%	2,828	1,962
1999	1,091	5,349	6,440	325	957	1,282	1,417	6,305	7,329	4,276	1,433	7.26	-1.05	81.6%	2,443	1,952
2000	12,971	9,488	22,459	2,489	3,843	6,332	15,460	13,331	18,202	4,469	1,620	9.65	-1.97	46.3%	6,067	2,945
2001	10,720	17,629	28,349	1,857	2,756	4,613	12,577	20,385	29,454	4,740	2,244	9.44	-3.26	61.8%	9,818	3,972
2002	8,751	17,878	26,629	2,584	4,941	7,525	11,335	22,819	39,372	5,184	3,012	9.34	-3.69	66.8%	13,124	4,889
2003	5,044	7,281	12,325	1,600	3,254	4,854	6,644	10,535	30,556	7,652	3,633	8.80	-2.14	61.3%	10,185	5,384
2004	10,792	8,221	19,013	1,125	1,564	2,689	11,918	9,784	29,897	9,643	4,259	9.39	-2.52	45.1%	9,966	6,230
2005	6,061	6,596	12,657	840	1,135	1,975	6,901	7,731	25,463	11,952	4,961	8.84	-2.47	52.8%	8,488	6,713
2006	4,414	5,895	10,309	452	607	1059	4,866	6,502	23,685	13,593	5,132	8.49	-0.30	57.2%	7,895	6,848
2007	4,229	4,077	8,306	295	134	429	4,524	4,211	16,291	15,006	5,379	8.42	0.45	48.2%	5,430	7,011

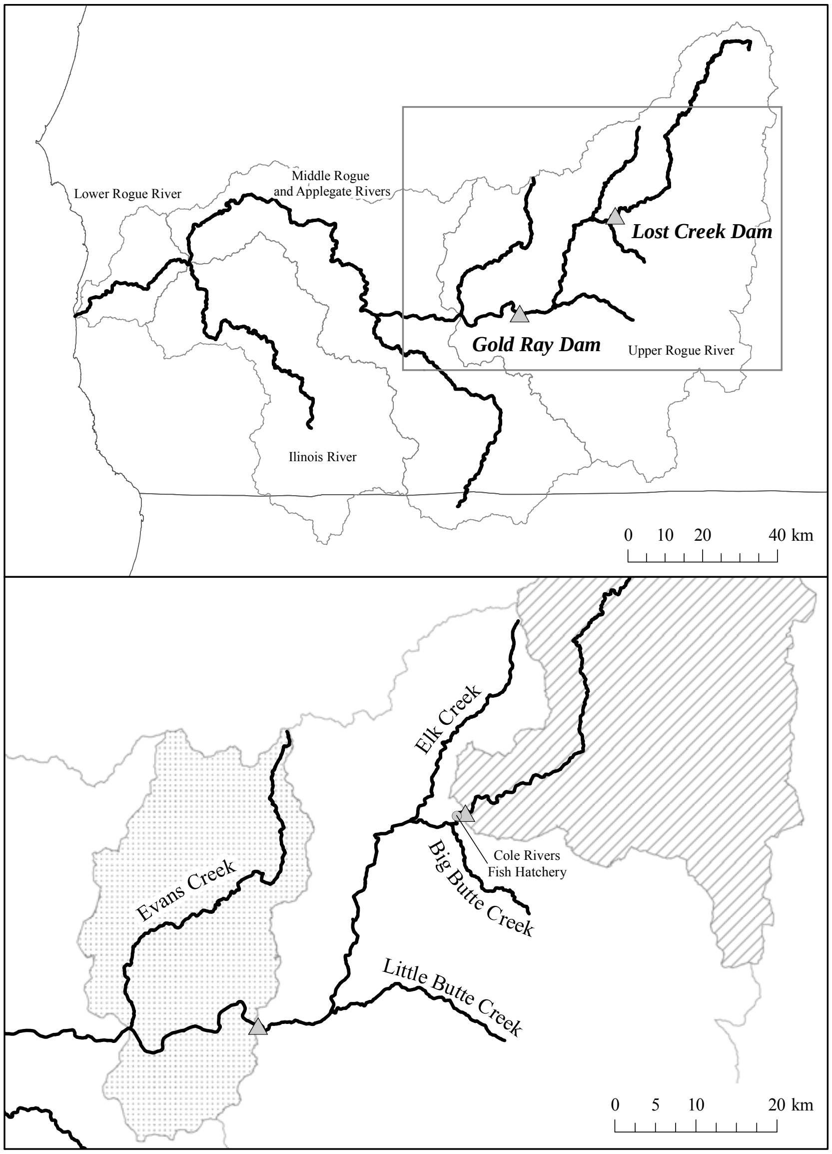

Figure 7. The Rogue River basin, the top figure shows all five population units in the Rogue Basin with the Upper Rogue population unit indicated, the lower figure shows the Upper Rogue population unit and the habitat area downstream of Gold Ray Dam not included in example and habitat upstream of Lost Creek Dam no longer accessible to coho salmon.

500 but continues to decline (i.e., progeny/parent ratio < 1), though not at a rate that will cause N_a to fall below 500 spawners within two generations. Populations with annual run sizes $N_a >$ 500 adults and no apparent decline over the last two generations are considered at low extinction risk for this criterion.

The geometric mean of annual spawner abundance over the past 12 years (4 generations) for wild fish including jacks was 5,379, and the annual abundance was never below 500 fish for this period based on Gold Ray Dam counts (Table 6).

The slope of the regression of the number of natural spawners (ln transformed) over the time series of a minimum of 12 years is used to describe the population trend, providing information on the long-term trajectory of the population (Figure 8). For the time period between 1996 and 2007, the slope of the regression was 0.09, suggesting an increasing trend.

When possible, examination of longer time series will provide recovery planners with a context within which to consider the most recent (i.e., 12 years) trend in abundance in comparison with that over a longer time period. For this example, the slope of the regression of the number of natural spawners (ln transformed) over the complete time series (1942 – 2007, 22 generations) was 0.02 (Figure 9), suggesting a slight increasing trend, although at a slower rate than over the most recent four generations. In summary, for this population unit, the annual abundance was > 500 fish over the past 12 years and the slope of the regression was positive. Therefore, the population is considered at ***low risk*** for this criterion.

Catastrophic population decline–This criterion is based on the geometric mean of the last 12 years of the generational sum of abundance. A three-year generation time is assumed for SONCC coho salmon; therefore 14 years of data are needed to obtain a mean of the last 12 years. In general, this criterion is used to identify populations that are at high risk given a recent catastrophic decline in numbers. Populations that have experienced a 90% decline in abundance within one generation are considered at high risk. Although the threshold for moderate risk is not explicitly defined, events such as a loss or near loss of a year class resulting from a catastrophic

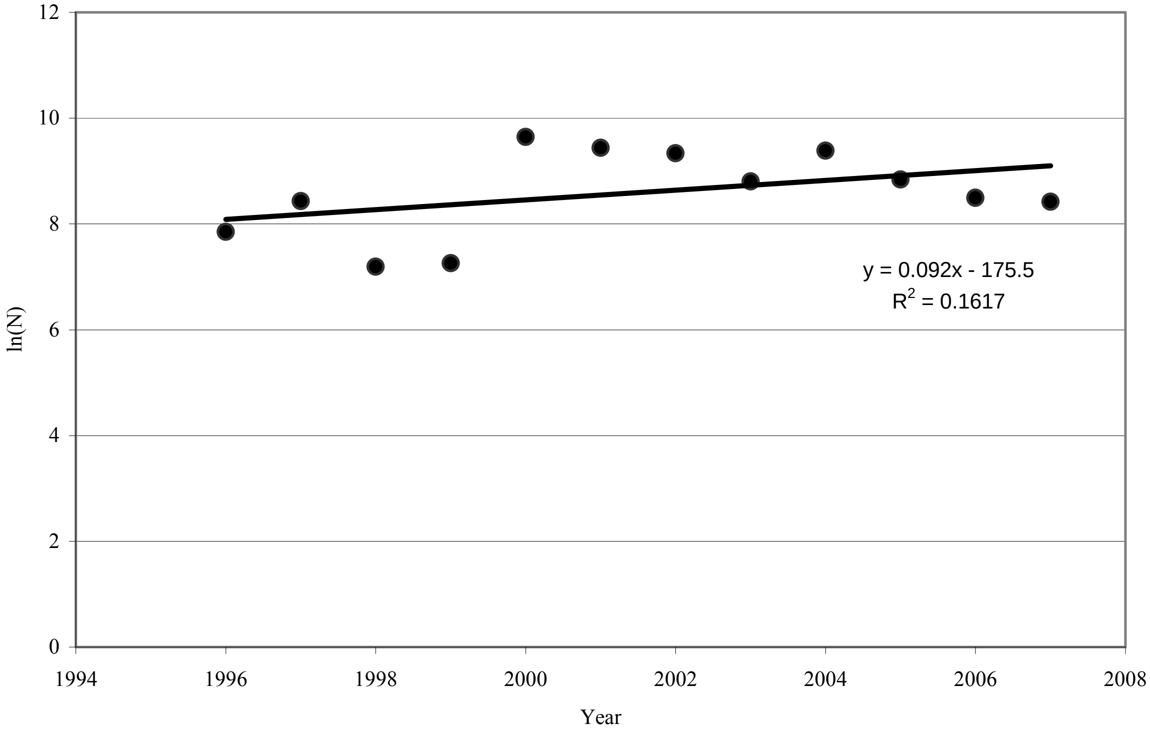

Figure 8. Trend in abundance (ln transformed annual abundance) over the past 4 generations (1996 to 2007) including regression line fitted to data indicating an increasing trend.

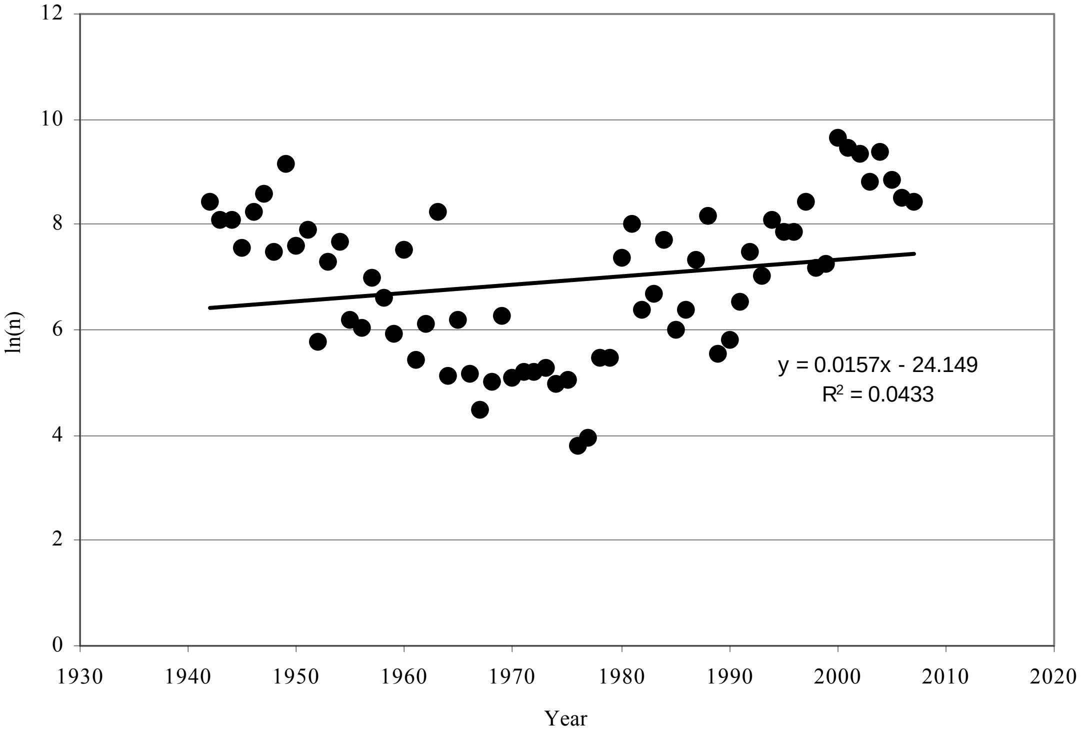

Figure 9. Trend in abundance (ln transformed annual abundance) over the complete record (1942 to 2007) including regression line fitted to data indicating an increasing trend.

disturbance would be an example of a lesser but significant decline that would warrant classifying a population as at moderate risk of extinction.

For the Upper Rogue population unit, the recent generational changes in abundance have been positive, indicating a ***low risk*** designation for this criterion (Table 6). However, it should be noted that during late 1960s and early 1970s there were several generational declines of approximately 80% (Table 6). The population appears to have rebounded from these events, perhaps aided by the initiation of the Cole Rivers Hatchery program.

Spawner Density Criteria–The amount of IP km in the Upper Rogue population unit is 915 IP km, resulting in a density-based spawner abundance target of 18,300. An IP km adjustment (for purposes of this example, 200 IP km) to compensate for the amount of IP km downstream of Gold Ray Dam results in an IP km total of 716 with a resulting density-based spawner target of 14,320.

Based on an IP km total estimate of 716, the depensation threshold for the adjusted IP km would be 716 spawners. The running 3-year average over the past 12 years (14 years of data) does not drop below 2,040 fish (Table 6); therefore, the Upper Rogue population unit is ***not at high risk*** for this criterion. Interestingly, in 1987 and in the majority of the years between 1957 and 1980 the number of spawners fell below the depensation threshold (Table 6). The fact that the population has rebounded from these low levels suggests several possible scenarios. One scenario is that the spawners were concentrated and did not descend into a depensation spiral. Another possibility is that contributions from the hatchery after 1977 contributed to the number of spawners and therefore prevented depensation effects.

For the spawner density threshold for low risk, the Upper Rogue population unit does not meet or exceed the "adjusted" spawner density (20 spaweners per IP km) or resulting target abundance of 14,320. The 12-year (four generations) mean abundance for the period from 1996 to 2007 is 7,011 with a spawner density = 9.8 spawners/IP km (Table 6). Based on this criterion, the Upper Rogue population unit is at ***moderate risk***.

Hatchery Influence–Under our criteria, hatchery risk is considered low when fewer than 5% of fish spawning in the wild are of hatchery origin. We are not aware of population-specific research and monitoring that would more directly measure genetic risks. Over the past 12 years, the percent hatchery fish counted at Gold Ray Dam has exceeded 60%, with three years greater than 80% (Table 6). These data provided by the Gold Ray Dam counts relate to number of fish counted and do not necessarily reflect ratio of hatchery to wild fish on the spawning grounds.

Jacobs et al. (2002) as well other surveys by the ODFW suggest that the fraction of hatchery fish on spawning grounds is very low, with the highest incidence of hatchery fish occurring in areas that are in close proximity to the hatchery. Other monitoring activities in place within the Rogue Basin that provide information concerning hatchery versus wild fish numbers include Elk Creek Dam, which provides a count of fish passing the site (the dam is currently being notched and therefore this count will no longer be available) and fish recovery at the Cole Rivers Hatchery trap. Sampling at Huntley Park on the mainstem Rogue River near its entry to the Pacific Ocean provides an estimate for the Rogue Basin, but is difficult to parse into the four population units upstream of the sampling location. The fish at Elk Creek Dam have been predominately wild fish whereas the majority of fish collected in the hatchery trap are of hatchery origin (i.e., fin clipped, marked). Indications are that a low percentage of the wild fish observed at Gold Ray Dam are later captured at the hatchery trap.

Although low incidence of hatchery fish spawning in the wild has been detected based on sites surveyed, there is still a high level of uncertainty concerning the fraction of hatchery fish spawning in the wild throughout the population unit. This uncertainty arises from the number of hatchery fish (i.e., marked fish) that are estimated at Gold Ray Dam but are not recovered at either Elk Creek Dam or at Cole Rivers Hatchery. For the years 1994 to 2007 where data are available (missing data for one of the three counts in 1997, 2003, and 2007) the percentage of hatchery fish unaccounted for (i.e., not recovered at either Elk Creek Dam or Cole Rivers Hatchery) vs. wild fish passing Gold Ray Dam ranges between 12 and 91% (Table 7). Conversations with area biologists suggest that the incidence of hatchery fish spawning in the wild is greatest in areas that are in close proximity to the hatchery and decreases rapidly with

Table 7. The percentage of marked fish that passed Gold Ray Dam but were not recovered at either Elk Creek Dam or Cole Rivers Hatchery versus the number of unmarked fish counted at Gold Ray Dam. Reference and data reports used are listed below table[a].

	Gold Ray Dam		Elk Creek Dam		Cole Rivers Hatchery		Unrecovered marked fish vs. unmarked fish counted at Gold Ray Dam
Year	Marked	Unmarked	Marked	Unmarked	Marked	Unmarked	
1994-95	7,454	3,245	52	232	6,967	499	13.4%
1995-96	10,949	2,569	70	349	9,262	209	62.9%
1996-97	11,027	2,572	123	319	8,564	199	91.0%
1997-98	11,162	4,588	71	982			-
1998-99	4,717	1,325	94	388	3,755	66	65.5%
1999-2000	6,306	1,416	64	298	5,369	67	61.7%
2000-01	13,331	15,460	62	710	11,359	436	12.4%
2001-02	20,385	12,577	102	1,446	13,003	903	57.9%
2002-03	22,819	11,335	68	1,382	12,609	1,006	89.5%
2003-04	10,535	6,644			7,303	824	-
2004-05	9,785	11,917	87	2,718	7,717	975	16.6%
2005-06	7,731	6,901	233	1,552	5,009	391	36.1%
2006-07	6,502	4,866	244	795	3,142	273	64.0%
2007-08	4,211	4,524			2,025	113	-

[a] - Satterthwaite et al. 1996a; Satterthwaite et al. 1996b; Satterthwaite and Leffler 1997; Satterthwaite 1998; Satterthwaite 1999; Satterthwaite 2000; Satterthwaite 2001; Satterthwaite 2002; Satterthwaite 2003; Evenson et al. 2005; Evenson et al. 2006; Evenson et al. 2007.

distance from the hatchery. Our understanding is that current survey sites are selected through the EMAP process (i.e., randomly selected from designated sampling universe) that could incorporate a greater density of sampling in close proximity to the hatchery to better ascertain the fraction of hatchery fish spawning in the wild. In addition, the universe from which spawning survey sites are randomly selected may not represent the full extent of the spawning distribution throughout the population unit. If areas in close proximity to the hatchery are not included in the sample universe but do indeed have spawning activity, even if only hatchery fish, a biased picture of hatchery spawners in the wild could result.

A positive aspect of this situation is that the ODFW has developed a sampling effort that can address these types of questions. We see this as a tractable issue. The sample selection process used by ODFW allows for increasing the density of survey sites in a specific area to address specific questions. In addition, it might be prudent for a review of the sampling universe so that all areas accessible to coho salmon spawners are included in the sampling frame.

As monitoring efforts continue and more extensive coverage of spawning survey sites is included throughout the Upper Rogue population habitat area, specifically in areas proximate to the hatchery, greater certainty about the occurrence and distribution of hatchery fish spawning in the wild will be achieved. For purposes of this example, the Upper Rogue population unit is determined to be at ***moderate risk*** based on this criterion. Given the number of hatchery fish observed at Gold Ray Dam but not captured at Elk Creek or Cole Rivers Hatchery, some portions of the population might have very high fractions of hatchery fish spawning in the wild. A directed effort to address the distribution of hatchery fish spawning in the wild in areas near Cole Rivers Hatchery would provide greater confidence and might result in a low-risk designation. A question that will need to be addressed in a case-specific manner for the Upper Rogue population unit is what the impacts are of having a small percentage of the available spawning habitat exceed the 5% threshold, possibly by a substantial amount. Most likely, this will need to be considered in the context of specific hatchery practices.

Summary–Based on the proposed viability criteria, we classify the Upper Rogue population unit as at moderate risk. The population did not meet or exceed the low-risk threshold for spawner density criteria and fraction of hatchery fish spawning in the wild. Currently available spawning surveys suggest low fractions of hatchery fish spawning in the wild, but specific issues related tothe low density of survey sites located near the hatchery and the adequacy of the currently used sampling frame when considered in the context of marked fish uncounted for at Elk Creek and Cole Rivers Hatchery result in a high degree of uncertainty. Also adding to the uncertainty is concern over the failure to meet the moderate risk threshold for the effective population size criteria for 25 consecutive years from 1966 to 1990. For the spawner density criteria, the population is currently significantly short of being considered at low risk. The abundance of coho salmon in the Upper Rogue population unit was 49% of the density based on the low-risk abundance target of 14,320 (for the adjusted amount of habitat). For purposes of this example and as a lead in for a later example of strata viability, if habitat isolated upstream of Lost Creek

Dam were excluded from consideration, the current density would increase to 11.3 spawners/IP km, and the population unit gets about 7% closer to the viability target, although the strata aggregate abundance target remains the same and the difference would need to be made up elsewhere (see strata viability example with Interior Rogue Basins stratum).

The issues related to the hatchery risk we see as very tractable given the sophistication of the ODFW monitoring plan, although development of an appropriate sampling frame will be required. With an understanding based on monitoring information, the risk associated with localized high hatchery spawning fractions can be considered with a resulting reduction in the uncertainty currently resulting from the hatchery fish detected passing Gold Ray Dam but not accounted for upstream at Elk Creek and Cole Rivers Hatchery.

3. ESU Viability

The viability of an ESU depends on several factors, including the number and status of populations, spatial distribution of populations, the characteristics of large-scale catastrophic risk, and the collective diversity of the populations and their habitat (Lindley et al. 2007). In anticipation of developing viability criteria at the population scale and integration of population information into viability criteria at the ESU scale, groups of populations spanning the diversity and distribution that historically existed within the ESU were identified by the TRT (Williams et al. 2006); Figures 1 and 2). These groups or “diversity strata” reflect diversity of (potential) selective environments. The TRT organized the independent and dependent populations of coho salmon in the SONCC ESU into diversity strata largely based on the geographical arrangement of the populations coupled with multivariate analysis of basin-scale environmental and ecological characteristics (Williams et al. 2006).

3.1 Characteristics of a Viable ESU

The foundation of ESU viability is built upon the ability of populations to function in an integrated manner and persist across the landscape. This integration includes dispersal among populations (i.e., connectivity) and a diversity and distribution of habitat types and conditions that allow for the expression of a range of life-history types (Williams and Reeves 2003). For an ESU to be viable the number and distribution of its constituent populations would exist in a balance between connectivity through dispersal and isolation from common catastrophic risks; viable populations need to be in close enough proximity to ensure connectivity, but not so close as to have a high likelihood of being affected by the same catastrophic event.

For an ESU to persist, populations within the ESU must be able to track changes in environmental conditions. When the location or distribution of a species' (or ESU's) habitat changes, a species can avoid extinction either by adapting genetically to the new environmental conditions or by spatially tracking the environmental conditions to which it is adapted (Pease et al. 1989). A species or ESU persists in places where it is able to track environmental changes, and becomes extinct if it fails to keep up with the shifting distribution of suitable habitat (Thomas 1994). These changes in environmental conditions can range spatially and temporally. Spatial scales can range from localized impacts (e.g., affecting portions of one or a few populations) to regional impacts from severe events such as droughts that affect all populations within a diversity stratum. Temporal scales can range from a site specific impact resulting from a short-term, albeit catastrophic, event (e.g., landslide temporarily blocking passage on a large mainstem river), to interannual variability of various environmental conditions (e.g., marine conditions, annual precipitation patterns), to long-term environmental changes such as climate change that have the potential to impact all populations within the ESU.

Interaction among populations of an ESU buffers against catastrophic loss of many populations, maintains long-term demographic and evolutionary processes through connectivity, and maintains sufficient diversity so that the ESU has the evolutionary potential to deal with changing environmental conditions. Some populations need to have sufficiently large numbers of individuals to disperse and provide the needed connectivity among populations, including strays to dependent populations, thereby increasing connectivity throughout the ESU. Thomas (1994) proposed that population declines in the larger habitats may be as worrying as local extinction in small patches. Recently, Isaak et al. (2007) found that for Chinook salmon in central Idaho, the size and connectivity of habitat were the strongest predictors of occupancy. In addition, these large populations increase the overall abundance of the ESU and also provide an additional buffer against catastrophic disturbance. For these purposes, functionally independent and potentially independent populations are essential to ensure connectivity based on their historical functional roles. Dependent populations that occupy smaller watersheds also contribute to connectivity and may provide an essential contribution to ESU viability.

A critical element for assessing ESU viability and conservation planning is representation (Groves et al. 2002; Carroll et al. 2006). By representation, we mean establishing populations across the full range of historical potential habitats and ecological settings within the ESU, to the greatest extent possible. By incorporating representation into recovery criteria, it is recognized that a single population may not represent species recovery, even if it is large enough to be significantly resilient to extinction (Carroll et al. 2006).

The TRT worked from the assumption that the historical conditions are the conditions where we are most certain that the ESU could have persisted for long periods of time (e.g., hundreds of years). Therefore, the historical population structure of coho salmon populations that make-up the SONCC ESU provides a template against which the outcome of our proposed ESU viability assessment could be evaluated. As with population viability, our certainty concerning ESU viability decreases as conditions depart from historical conditions. The historical population structure of SONCC coho salmon proposed by Williams et al. (2006) recognized different functional roles that populations played within the historical ESU (i.e., functionally independent, potentially independent, dependent, and ephemeral) and a general structure for ESU viability represented by the diversity strata (Figures 1 and 2). The TRT's approach to developing their picture of historical population structure was built upon the foundation that persistence of the ESU was contingent upon the ability of its populations to track changes in the environment. As discussed previously for population viability, there are certain attributes that populations must exhibit to also track and persist with changes in the environment, but these are mostly at smaller spatial scales and temporal scales than those considered at the ESU level.

3.2 ESU Criteria

In our proposed scenario for a viable ESU, we do not list specific sets of populations that must be viable to have a viable ESU. Instead, we provide a set of rules that will result in certain configurations of populations that we believe will result in a viable ESU (Table 8). The rules we propose are intended to capture our objectives of maintaining diversity throughout the ESU, providing connectivity among populations to maintain long-term demographic and genetic processes, and providing a buffer against potential catastrophic risks. Our overarching goal in developing these rules is that we desire an appropriate number and arrangement of populations that allows for the populations to track changes in environmental conditions and therefore be viable. Shaffer et al. (2002) and Shaffer and Stein (2000) discuss three principles that should be

Table 8. Summary of ESU viability criteria for SONCC coho salmon.

ESU viability characteristic	Criteria
Representation	1. All diversity strata should be represented by viable populations
Redundancy and Connectivity	2.a. At least fifty percent of historically independent populations in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria. For strata with three or fewer independent populations, at least two populations must be viable.
	AND
	2.b. Total aggregate abundance of the populations selected to satisfy 2a must meet or exceed 50% of the aggregate viable population abundance predicted for the stratum based on the spawner density
.	3. All dependent and independent populations not expected to meet low-risk threshold within a stratum should exhibit occupancy indicating sufficient immigration is occurring from the "core populations".
	4. The distribution of extant populations, both dependent and independent, needs to maintain connectivity across the stratum as well as with adjacent strata.

considered when making viability judgments: representation, redundancy, and resiliency. In simple terms, Shaffer and Stein (2000) defined representation as "saving some of everything … not simply as species and communities but as the complexes of populations, communities, and environmental settings", while the measure of resiliency can be thought of as the size of the sites and the number of the sites can be thought of as a measure of redundancy. Resiliency is something more than just the size of a site (or a population's habitat); it has been described as the capacity of a system to absorb disturbances, reorganize, and maintain adaptive capacity (Gunderson 2000; Bengtsson et al. 2003). We have incorporated these principles into our viability framework at the ESU level and believe they are equally pertinent even when adequate data or models are available. Though the availability of more complicated models might reduce our uncertainty at the population viability level, we believe capturing many of the issues related to representation, redundancy, and resiliency is best done at the ESU spatial and temporal scale.

3.2.1. Representation

1. All diversity strata should be represented by viable populations for the ESU to be considered viable.

As previously discussed, the diversity strata were largely based on the geographical arrangement of the populations and basin-scale environmental and ecological characteristics. A represented diversity stratum is one that is viable (as defined below). By requiring all diversity strata to be represented for the ESU to be viable, the range of environmental conditions historically available have a greater chance of being included and thereby a substantial portion of the historical diversity of the ESU. In addition, given the geographic make-up of the diversity strata, by requiring all strata be viable helps ensure that the ESU persists throughout a significant portion of its historical range. And finally, because of the arrangement of populations along portions of the SONCC ESU, representing each stratum also ensures that connectivity across the entire ESU is maintained.

3.2.2 Redundancy and Connectivity

Three additional viability criteria are proposed to provide a hedge against catastrophic risk by ensuring redundancy of viable populations, and promote connectivity within diversity strata and throughout the ESU.

2.a. At least fifty percent of historically independent populations (functionally independent and potentially independent populations) in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria developed in this report. For strata with three or fewer independent populations, at least two populations must be viable.

- and -

2.b. Within each diversity stratum, the total aggregate abundance of independent populations selected to satisfy this criterion must meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all functionally independent and potentially independent populations.

Recovery planners should seek configurations of populations within the ESU that emphasize historically independent populations that, by virtue of their size and distribution, formed the foundation of the ESU. Ideally, the second portion of this criterion will be met if the first portion is satisfied. However, in some cases it may prove infeasible to restore the larger independent populations in a stratum to levels that meet population viability criteria based on historical conditions (e.g., density-based abundance targets). In such cases, recovery planners may need to identify stratum-scale recovery strategies that include (1) restoring some (presumably historically large) independent populations so that they are demonstrably viable (i.e., have a negligible risk of extinction over a 100-year time frame (McElhany et al. 2000)) but occupy only a remnant of the historical population's range, and so cannot be considered as being entirely representative of the historical population, and (2) restoring additional (presumably smaller) independent populations to a sufficient degree for stratum abundance to satisfy the second part of this criterion.

For a diversity stratum to be viable, the above criteria require that at least two or 50% of the independent populations (functionally independent or potentially independent) within a diversity stratum, whichever number is greater, should be viable and that the abundance of these viable populations collectively must meet or exceed 50% of the abundance predicted for the stratum by the density-based spawner abundance criteria (i.e., the 50% of total abundance predicted for all independent populations within the stratum when at low risk). Any functionally independent or potentially independent populations that contribute to the aggregate stratum abundance should (1) meet the threshold for being viable-in-isolation of 1,360 (i.e., N_a >40 fish x 34 IP km from Williams et al. 2006) with the distribution of fish such that the density criterion is satisfied within the remaining accessible habitat, and (2) meet minimum thresholds for low genetic risk ($N_g \geq 2500$).

Dependent populations, as well as independent populations that fail to meet minimum standards for viability, by definition are not expected to persist over long time frames in the absence of subsidies from other neighboring populations. Consequently, only populations that are expected to persist and could do so in isolation are counted toward the aggregate population criterion.

In the case of populations affected by impassable dams or other human-caused barriers to fish passage, the remaining accessible habitat will consist of habitat downstream of the obstruction. In areas still accessible to anadromous fish but affected by severe and irreversible habitat modification, recovery planners will need to explicitly define those portions of a watershed expected to contribute to a viable population.

By requiring at least two populations or 50% of populations within a stratum to be viable, the criterion ensures that redundancy is provided to help guard against loss of significant diversity as a result of catastrophic events. Requiring selected populations to meet or exceed 50% of the abundance predicted for the stratum accomplishes two important objectives (Table 9). First, it ensures that sufficient numbers of fish are present throughout the stratum. Second, meeting the

Table 9. Diversity strata of the SONCC Coho Salmon ESU including population type (F: functionally independent, P: potentially independent, D: dependent, and E: ephemeral), population unit name (ID number), the spawner density threshold value of number of spawners for independent populations (both functionally and potentially independent populations), and the number of spawners needed to satisfy the 50% of the total number of spawners in a strata needed to meet stratum viability (see text for details).

Stratum	Pop. type	Population unit	Low risk N_a
Northern Coastal Basins	F	Elk River (1)	2,400
	P	Lower Rogue River (7a)	3,000
	F	Chetco River (10)	4,500
	P	Winchuck River (11)	2,200
	D	Mill Creek	
	E	Hubbard Creek	
	E	Euchre Creek	
	D	Hunter Creek	
	D	Pistol River	
		50% total stratum N_a	6,050
Central Coastal Basins	F	Smith River (12)	7,700
	F	Lower Klamath River (15a)	5,900
	F	Redwood Creek (16)	4,900
	P	Maple Creek/Big Lagoon (18)	1,600
	P	Little River (19)	1,400
	F	Mad River (22)	4,900
	D	Elk Creek	
	D	Wilson Creek	
	D	McDonald Creek	
	D	Strawberry Creek	
	D	Norton/Widow White Creek	
		50% total stratum N_a	13,200

Table 9. continued.

Stratum	Pop. type	Population unit	Low risk N_a
Southern Coastal Basins	F	Humboldt Bay tributaries (23)	5,700
	F	Low. Eel/Van Duzen rivers (24a)	7,900
	P	Bear River (26)	1,900
	F	Mattole River (28)	6,500
	D	Guthrie Creek	
	D	McNutt Gulch	
		50% total stratum N_a	11,000
Interior – Rogue River	F	Illinois River (7b)	11,800
	F	Mid. Rogue/Applegate rivers (7c)	15,200
	F	Upper Rogue River (7d)	18,300
		50% total stratum N_a	22,650
Interior - Klamath	P	Middle Klamath River (15b)	3,900
	F	Upper Klamath River (15c)	8,500
	P	Salmon River (15d)	4,000
	F	Scott River (15e)	8,800
	F	Shasta River (15f)	10,600
		50% total stratum N_a	17,900
Interior - Trinity	F	South Fork Trinity River (15g)	6,400
	P	Lower Trinity River (15h)	3,900
	F	Upper Trinity River (15i)	2,400
		50% total stratum N_a	6,350
Interior - Eel	F	South Fork Eel River (24b)	9,500
	P	Mainstem Eel River (24c)	4,700
	P	North Fork Eel River (24d)	2,100
	P	Mid. Fork Eel River (24e)	2,900
	F	Mid. Mainstem Eel River (24f)	6,500
	P	Upper Mainstem Eel River (24g)	2,100
		50% total stratum N_a	13,900

stratum abundance requirement means that proposed recovery scenarios should include historically independent populations that by virtue of their size and location were disproportionately important to ESU function and persistence. Historically large populations were critical not only because their large size and spatial distribution imparted greater resiliency in the face of longer-term environmental change, but also because they were major sources of dispersers, which likely affected the dynamics of adjacent populations. In practical application, recovery of the larger independent populations will satisfy both portions of this criterion. In some strata, particularly those with several historically independent populations that were smaller in size, the second portion of this criterion encourages recovery planners to focus efforts on restoring some presumably historically larger independent populations (even if only portions of the population's habitat are still accessible or suitable) or restore additional smaller independent populations to a sufficient degree for the within stratum abundance to satisfy the second part of this criterion. In other words, a stratum cannot be considered recovered by focusing exclusively on the smallest historical populations within it.

3. Remaining populations, including historical dependent populations and historical independent populations (functionally and potentially independent populations) that are not expected to meet the low-risk threshold, must exhibit occupancy patterns that indicate sufficient immigration is occurring from the "core populations".

This criterion addresses our concerns that connectivity be maintained or reestablished among populations. We propose that recovery planners place a high priority on populations that are remnants of historically independent populations with a minimum standard that most historically independent populations should be at no greater than moderate risk of extinction (i.e., not at high risk) when evaluated as independent populations. This recommendation would require a higher standard for occupancy than just presence of individuals. It should be recognized that these independent populations no longer fulfill their historical role within the ESU, but they can play a critical role in connectivity and have the potential for representing critical components of the evolutionary legacy of the ESU.

Dependent populations also fulfill a role in maintaining connectivity among populations, especially in situations where historically independent populations are at high risk or have been extirpated. In these cases, dependent populations contribute to maintaining genetic diversity within a stratum and provide strays (i.e., colonizers) that can reduce the genetic and demographic risks to adjacent independent populations and support restoration of these independent populations. After reestablishment of adjacent independent populations, dependent populations can increase resilience, buffering larger populations against future disturbances. Dependent populations can also play the role of "steppingstones," allowing individuals, and thereby a population, to track changes in environmental conditions.

4. The distribution of extant populations, both dependent and independent, needs to maintain connectivity across the stratum as well as with adjacent strata.

As with the previous criterion, this criterion also reflects our concern with maintaining or reestablishing connectivity among populations within and among diversity strata. In order to ensure that connectivity is maintained within and among diversity strata, there may be specific populations (and the watersheds that they occupy) that are essential for filling what might otherwise be substantial spatial gaps within a diversity stratum. These watersheds may contain populations considered to have been historically dependent on immigration; therefore, ensuring that such populations persist requires that their source populations also be at sufficient status to maintain connectivity.

This distribution criterion addresses the need for a rather continuous set of populations across the ESU to allow for connectivity and provide a buffer against catastrophes and smaller disturbance events. There are currently few data on stray rates and stray distances for Pacific salmonids along the southern Oregon and California coastal region to provide strict guidance on how close adjacent populations should be to maintain connectivity. However, it would be preferable to avoid having all the viable independent populations and occupied dependent populations isolated into a single geographic region of a stratum. Typically, the greater the distances among populations, the less likely it is that dispersal will occur. Genetic data supports this strong concordance of geographic and genetic distances, often referred to as "isolation by distance" (Bucklin et al. 2007). While data are too limited to provide a specific geographic distance at this time to help guide recovery planners, we propose this more general criterion of avoiding a clumped distribution of viable populations across the ESU. Studies of Chinook salmon (Hard and Heard 1999), pink salmon (Wertheimer et al. 2004), chum salmon (Tallman and Healey 1994), and Atlantic salmon (Jonsson et al. 2003) from other regions suggest that the majority of straying occurs within a few tens of kilometers from the natal stream (or stream of release). Assuming coho salmon in the SONCC ESU exhibit similar tendencies, unoccupied gaps along the coastline of more than 20 to 30 km may be sufficient to limit patterns of dispersal and connectivity.

Interior Rogue Basins Stratum Viability Example

How recovery planners should consider the "50% of independent populations should be viable" and "aggregate abundance" requirements for stratum viability can be illustrated with available data from the Interior Rogue Basin Stratum (IRBS). In addition, the IRBS provides an example of how recovery planners could consider situations where anthropogenic migration barriers (e.g., dams) now block access to historically accessible habitat.

The IRBS is made up of three independent populations, the Illinois River, the Middle Rogue and Applegate rivers, and the Upper Rogue River. The 50% rule states that at least fifty percent of historically independent populations (functionally independent and potentially independent populations) in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria developed in this report. For strata with three or fewer independent populations, at least two populations must be viable.

Therefore, for the IRBS, two of the three independent populations need to be viable for the stratum to be considered viable. As illustrated in a previous section of this report, the Upper Rogue population unit is currently assessed at being at moderate risk of extinction using the proposed population viability criteria. This moderate risk designation resulted from the density-based spawner abundance failing to meet the low-risk threshold and also uncertainty related to the percentage of hatchery fish spawning in the wild. It should be noted that there is uncertainty surrounding the distribution and number of hatchery fish spawning in the wild. Monitoring procedures and protocols currently available could reduce this uncertainty in the very near future and lead to a different assessment of the hatchery fish criterion.

For the aggregate abundance requirement for the stratum, the "50% of total" number of spawners required is 22,650 (Tables 9 and 10). The proposed viability requirements state that any functionally independent or potentially independent populations that contribute to the aggregate stratum abundance should (1) meet the threshold for being viable-in-isolation of 1,360 (i.e., N_a >40 fish x 34 IP km from Williams et al. 2006) with the distribution of fish such that the density criterion is satisfied within the remaining useable habitat, and (2) meet minimum thresholds for low genetic risk ($N_g \geq 2500$). For coho salmon with a 3-year generation time, an N_g of 2500 results in an annual abundance of 834.

For purposes of this example, we will assume that the only historically accessible habitat lost is that above Lost Creek Dam in the area supporting the Upper Rogue population unit. The Upper Rogue population unit had an estimated 915 IP km (Williams et al. 2006). Lost Creek Dam (Rkm 254) began operations in February 1977 and blocks access to an estimated 94 IP km that were historically accessible. Therefore, in this example the target density-based spawner abundance would be based on 821 IP km (915 IP km – 94 IP km) and result in a target abundance of 16,400 (rounded from 16,420) instead of the 18,300 spawners based on historically accessible habitat.

The "aggregate abundance" requirement is based on historically available habitat and does not change if population-specific adjustments are made for habitat no longer accessible. The rationale for this is that the historically derived abundances capture the function and role the populations and the stratum played in the greater ESU. Populations might be viable (i.e., persist for longer than 100 years and be demographically independent), but the population's function and role within the stratum and ESU may change, particularly if the number departs greatly from historical conditions. By requiring the stratum total to be based on the historically derived total

Table 10. A hypothetical example of stratum viability using the Interior Rogue Basin Stratum and an adjustment for population viability for the Upper Rogue population unit.

	Spawner abundance low-risk threshold		
Population unit	Historical	Adjusted	Stratum viability needs
Illinois River	11,800	- none -	11,800
Middle Rogue / Applegate rivers	15,200	- none -	
Upper Rogue	18,300	16,400	16,400
50% aggregate	22,650		28,200

abundance, the historical role and function of the suite of populations that make up a stratum are best captured. This approach is based on our assumption that there is a difference between a population's historical capacity and the level at which it could reasonably perform its historical role within the stratum. As discussed earlier in this document, our density-based abundance targets were largely developed from Bradford et al. (2000), which included watersheds in various states of disturbance and therefore does not indicate historical carrying capacity. Yet, we concluded that if the population met the density requirement, the spatial structure and diversity would represent historical diversity and spatial structure to a sufficient degree and therefore substantially increase the likelihood of persistence.

To illustrate how the criteria might be applied in this case, let us suppose that recovery planners have targeted the Illinois River population unit and the Upper Rogue population unit for meeting the "50%" rule. In addition, let us assume that habitat behind Lost Creek Dam will be permanently inaccessible. The aggregate abundance target for the stratum is 50% of the combined abundances deemed to be low risk based on the density criteria for the three populations within the stratum, or 22,650 fish (Table 10). The individual population abundance targets based on the historically accessible habitat are 11,800 and 18,300 for the Illinois and Upper rogue rivers, respectively. However, because some habitat is no longer accessible due to Lost Creek Dam, the adjusted abundance target for the Upper Rogue is 16,400 fish. If both the Illinois and Upper Rogue populations met the low-risk targets, the aggregate abundance would be 28,200, which exceeds the target abundance for the stratum. Consequently, the stratum would be considered viable despite the loss of habitat resulting from the dam.

The above example is hypothetical. There are other options that could include the Middle Rogue/Applegate rivers population unit and result in a viable stratum. We do not intend to imply that this is the only course recovery planners should consider.

In other strata of the SONCC coho salmon ESU, the requirement that any independent populations that contribute to the aggregate stratum abundance should (1) meet the threshold for being viable-in-isolation and (2) meet minimum thresholds for low genetic risk ($N_g \geq 2500$) can be achieved through various configurations and provides recovery planners multiple scenarios that would result in stratum viability and fulfill TRT requirements that historical roles and functions are captured. Because of the nature of the IRBS population configuration, mostly the relatively large size of the sub-basins, there are fewer options for satisfying the criteria, though any two of the three populations being viable would meet the aggregate abundance rule.

This hypothetical example emphasizes the more efficient approach of targeting stratum viability when developing recovery plans. By building recovery scenarios around viable strata, the daunting task of requiring all populations to meet low-risk viability criteria shifts to the task of recovering and conserving processes that operate at larger spatial and temporal scales and captures the ESU viability focus on representation, redundancy, and resiliency.

3.3 Viability of the SONCC Coho Salmon ESU

We were not able to assess the viability of the SONCC Coho Salmon ESU with the quantitative approach proposed in this report due to data limitations. There are almost no data with which to assess the status of any of the populations of coho salmon in the SONCC ESU described by Williams et al. (2006). Good et al. (2005) concluded that SONCC coho salmon were likely to become endangered in the foreseeable future, a conclusion consistent with an earlier assessment (Weitkamp et al. 1995). Although there were few data, the information that was available for SONCC coho salmon indicated the component populations were in decline and strongly suggested the ESU was at risk (Weitkamp et al. 1995; California Department of Fish and Game 2002; Good et al. 2005).

4. Summary and Recommendations

The purpose of this report was to develop a framework for evaluating the viability of coho populations and the greater SONCC Coho Salmon ESU. This framework is built upon a foundation proposed by Allendorf et al. (1997) of relatively simple criteria and rules that have modest data requirements. The framework proposed in this report parallels efforts for other ESUs in California (Lindley et al. 2007; Spence et al. 2008). In addition, the approach is consistent in its underlying conceptual approach (e.g., VSP) and the general level of criteria thresholds used for assessing viability of the Oregon Coast Coho Salmon ESU (Wainwright et al. 2008). The framework we propose consists of criteria and rules that are based in part on expert opinion and judgment, although we have attempted to provide support for our efforts from the general literature, specific references for Pacific salmonids, and when available, references specifically for coho salmon. Although the approaches used by this and other TRTs are to some degree based on expert judgment and subject to considerable uncertainty, the conclusions are not particularly sensitive to the exact threshold values of the criteria (Lindley et al. 2007). Nor should uncertainty surrounding the proposed criteria deter recovery planners from proceeding with action to recover these populations. In most cases, populations appear to be well below the proposed viability thresholds, and the steps needed to move them toward viability will be similar, regardless of the specific recovery targets, which can be refined as more information becomes available.

As previously discussed, there are insufficient data to assess the risk of coho populations within the SONCC ESU, and therefore, we cannot assess the viability of the ESU using the quantitative approach developed in this report. As recovery planning proceeds, ambitious research and monitoring programs will need to be initiated, as will long-term restoration solutions. The TRT is preparing a summary of research and monitoring needs for SONCC coho salmon. In the meantime, Lindley et al. (2007) identified some very important actions that should be done as soon as possible for Central Valley salmonids, several of which are pertinent to SONCC coho salmon (listed in no particular order):

- Secure all extant populations. Although the SONCC ESU is far short of being viable, extant populations, even if not currently viable, may be needed for recovery.

- Begin collecting distribution and abundance data throughout the SONCC ESU. These data are fundamental for developing effective recovery actions and future status assessment.

- Minimize straying from hatcheries to natural spawning areas. The goal of maximizing diversity within populations and the ESU is eroded with even low levels of straying from hatchery populations to wild populations.

- Begin conducting critical research on climate change and its potential impact to SONCC coho salmon. In their report on endangered and threatened fishes in the Klamath River basin, the National Research Council (2004) discussed the need for resource managers to develop strategies to cope with climate change in the Klamath Basin, suggesting an adaptive management approach given the uncertainty in the magnitude and timing of climate change. They go on to state that climate change could make temperature an even greater issue than it is currently for the future of salmonids in the Klamath basin and suggest that interior basins like the Shasta River with headwater and groundwater recharge areas at high elevation may be more resilient than most stream reaches in the event of increases in temperature and changes in precipitation patterns (National Research Council 2004). It is not unreasonable to suspect that similar scenarios could occur throughout the SONCC ESU, particularly in the Rogue and Eel river basins that penetrate far inland and to high elevations.

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