NOAA Technical Memorandum NMFS-NWFSC-118

Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon (*Oncorhynchus kisutch*)

June 2012

U.S. DEPARTMENT OF COMMERCE
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Executive Summary

Beginning in the 1990s, the National Marine Fisheries Service (NMFS) conducted a series of reviews of the status of West Coast populations of Pacific salmon and steelhead (*Oncorhynchus* spp.) with respect to the U.S. Endangered Species Act (ESA). This report summarizes the scientific conclusions of the most recent status review of the Oregon Coast Coho Salmon (*Oncorhynchus kisutch*) (OCCS) Evolutionarily Significant Unit (ESU).

On 10 August 1998, NMFS first listed the OCCS ESU as threatened under the ESA (NMFS 1998). From 2000 until 2008, considerable litigation surrounded the listing status of this species. (Additional information is online at http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Alsea-Response/Alsea-OCC.cfm.) The OCCS's listing status changed between "not warranted for listing" and "threatened" several times during this period. The most recent determination listed the ESU as threatened (NMFS 2008). As part of a legal settlement, NMFS agreed to initiate a new status review of OCCS on 29 April 2009 (NMFS 2009a). To conduct the status review, NMFS formed a biological review team (BRT) to evaluate the risk of extinction of the OCCS ESU based on the best available information. NMFS asked the BRT to judge whether the ESU was at low, moderate, or high risk of extinction based on current biological status and existing and projected threats, and to give particular attention to the status and trend of freshwater habitat conditions and marine survival conditions.

The BRT used a variety of information sources for this review, including the scientific literature; data and reports from federal, state, local, and tribal government sources; and information submitted by nongovernmental organizations. The BRT hosted a symposium in Corvallis, Oregon, on 14 September 2009, where the State of Oregon, comanagers, and other interested parties were invited to make presentations of scientific information related to OCCS status. The BRT met 15 September 2009 and 8−10 December 2009, and released its preliminary report 25 May 2010, concluding that the OCCS ESU was at moderate risk of extinction. Concurrent with the release of the preliminary status review report, NMFS proposed to retain the threatened listing of the OCCS ESU and invited public comment on the proposal (NMFS 2010, FR 75:29489–29506). In addition, the BRT solicited technical review of the draft status report from nine independent scientists selected from the academic and scientific community. Each of these reviewers is an expert in salmon biology, risk assessment methodology, ocean/salmon ecology, climate trend assessment, or landscape-scale habitat assessment. Eight of the reviewers responded.

This BRT report is a revision of the 25 May 2010 preliminary report. In revising the report, the BRT considered the comments from the expert reviewers, the scientific or technical comments submitted during the public comment period, updated spawner counts, other biological information that became available between May and December 2010, and the results of a joint Oregon Department of Fish and Wildlife (ODFW)/NMFS working group that reanalyzed freshwater habitat trend data. The BRT met on 18 January 2011 to discuss the updated information and analysis and reach a conclusion on the extinction risk of the OCCS.

In the past, BRTs have used a variety of methods to evaluate different categories of risk contributing to overall risk to an ESU. After 2000 the method was standardized to use a risk matrix method based on the four major criteria identified in the NMFS viable salmonid populations document (McElhany et al. 2000): abundance, growth rate/productivity, spatial structure, and diversity. For this analysis, the BRT followed that approach, but also included the work of the Oregon and Northern California Coast Technical Recovery Team on historical population structure (Lawson et al. 2007) and biological recovery criteria (Wainwright et al. 2008) as additional sources of information on OCCS status.

After considering the new and updated information, the BRT was uncertain about the status of the ESU, with opinion about evenly split between “moderate risk” and “not at risk,” and a small minority indicating “high risk.” Overall, a slight majority of BRT opinion considered the ESU to be at moderate or high risk of extinction. The uncertainty in risk status was largely due to the difficulty in balancing the clear improvements in some aspects of the ESU’s status over the last 15 years against persistent threats driving the longer term status of the ESU, which probably have not changed over the same time frame and are predicted to degrade in the future.

The BRT concluded that some aspects of the ESU’s status have clearly improved since the initial status review in the mid-1990s (Weitkamp et al. 1995). In particular, spawning escapements were higher in some recent years than they had been since 1970. Recent total returns (preharvest recruits) were also substantially higher than the low extremes of the 1990s, but still mostly below levels of the 1960s and 1970s. The BRT attributed these increases largely to a combination of lower harvest rates, reduced hatchery production, and improved ocean conditions. The BRT also noted that the ESU contained relatively abundant wild populations throughout its range, and that additional improvements to status from ongoing and past reductions in hatchery production could be expected in the future.

Despite these positive factors, the BRT had considerable concerns about the long-term viability of the ESU. Even with the recent increases, spawning abundance remains at approximately 10% of estimated historical spawning abundance. Despite some improvements in productivity in the early 2000s, the BRT was concerned that the overall productivity of the ESU remains low compared to what was observed as recently as the 1960s and 1970s. The BRT was also concerned that most of the improvement in productivity seen in the early 2000s was likely due to improved ocean conditions, rather than (presumably more lasting) improvements in freshwater conditions. The BRT noted that the legacy of past forest management practices combined with lowland agriculture and urban development have resulted in a situation in which the areas of highest potential habitat capacity are now severely degraded. The combined ODFW/NMFS analysis of freshwater habitat trends for the Oregon coast found little evidence for an overall improving trend in freshwater habitat conditions since the mid-1990s and evidence of negative trends in some areas, a result which concerned the BRT. The BRT was also concerned that recent changes in the protection status of beaver (*Castor canadensis*), an animal which creates coho salmon habitat, could result in further negative trends in habitat quality.

The BRT was particularly concerned that the long-term loss of high value rearing habitat has increased the vulnerability of the ESU to near-term and long-term climate effects. In the short term, the ESU could rapidly decline to the low abundance seen in the mid-1990s when ocean conditions cycled back to a period of poor survival for coho salmon. The BRT was also concerned that global climate change will lead to a long-term downward trend in freshwater and marine coho salmon habitat compared to current conditions. There was considerable uncertainty about the magnitude that most of the specific effects of climate change will have on salmon habitat, but the BRT was concerned that most changes associated with climate change are expected to result in poorer and more variable habitat conditions for OCCS in freshwater and marine environments.

Acknowledgments

We are grateful to the large number of people who participated in collecting and analyzing the information used in this status review, including Kelly Christiansen and Kelly Burnett, U.S. Forest Service; and Carol Volk, Jonathon Malstedt, Chris Moyer, Katie Barnas, Monica Diaz, and David Hamm, Northwest Fisheries Science Center. We also appreciate the assistance of Joy Vaughn, Oregon Department of State Lands; Andrew Herstrom, Oregon Department of Forestry; and Oregon Department of Fish and Wildlife (ODFW) Habitat Trends Working Group members Kim Jones, Kara Anlauf, and Jeff Rodgers.

We are also grateful for the contributions of participants in the September 2009 Oregon Coast Coho Salmon Symposium: Kelly Moore and Kim Jones, ODFW; Suzanne Knapp, Oregon State Governor's Office; Jim Paul, Oregon Department of Forestry; Robert Kennedy, Oregon State University; Joe Ebersole, U.S. Environmental Protection Agency; Paul Engelmeyer, Native Fish Society; Chris Frissell, Pacific Rivers Council; David Loomis, Douglas County Commissioners; Stan van de Wetering, Confederated Tribe of Siletz Indians; and Joe Moreau, U.S. Bureau of Land Management.

Introduction

Coho salmon (*Oncorhynchus kisutch*) is a widespread species of Pacific salmon, spawning and rearing in rivers and streams around the Pacific Rim from Monterey Bay in California north to Point Hope, Alaska, through the Aleutians, and from the Anadyr River in Russia south to Korea and northern Hokkaido, Japan (Godfrey et al. 1975, Laufle et al. 1986). From central British Columbia south, the vast majority of coho salmon adults return to spawn as 3-year-olds, having spent approximately 18 months in freshwater and 18 months in saltwater (Gilbert 1912, Pritchard 1940, Sandercock 1991). The primary exceptions to this pattern are jacks, sexually mature males that return to freshwater to spawn after only 5 to 7 months in the ocean. West Coast coho salmon smolts typically leave freshwater in the spring (April to June) and when sexually mature reenter freshwater from September to November and spawn from November to December and occasionally into January (Sandercock 1991). Coho salmon spawning habitat consists of small streams with stable gravels. Summer and winter freshwater habitats most preferred by young salmon consist of quiet areas with low flow, such as backwater pools, beaver (*Castor canadensis*) ponds, dam pools, and side channels (Reeves et al. 1989).

For purposes of U.S. Endangered Species Act (ESA) listings, coho salmon status has been reviewed repeatedly beginning in 1990. The first two reviews were in response to petitions to list coho salmon in the lower Columbia River and Scott and Waddell creeks in central California. Based on these reviews, the National Marine Fisheries Service (NMFS) concluded that no populations warranted protection under the ESA in the lower Columbia River (Johnson et al. 1991, NMFS 1991a), and that the Scott Creek and Waddell Creek populations were part of a larger, undescribed evolutionarily significant unit (ESU) (Bryant 1994, NMFS 1994).

Oregon Coast coho salmon (OCCS) were first petitioned for listing in 1993 (NMFS 1993). For a chronology of the legal history of this species, see Table 1. This and other petitions led NMFS to initiate a review of West Coast (Washington, Oregon, and California) coho salmon populations. This 1995 coast-wide review identified six coho salmon ESUs (Figure 1): the three southernmost ESUs (central California, northern California/southern Oregon and Oregon coast) were proposed for listing, two ESUs (Puget Sound/Strait of Georgia and lower Columbia River/southwest Washington) were identified as candidates for future consideration for listing, and one ESU (Olympic Peninsula) was deemed “not warranted” for listing (NMFS 1995, Weitkamp et al. 1995). In 1996 a biological review team (BRT) updated the status review for proposed and candidate coho salmon ESUs (NMFS 1996a, 1996b, 1996c). However, because of the scale of the review, requests from comanagers for additional time to comment on the preliminary conclusions, and the legal obligations of NMFS, the status review was finalized for proposed coho salmon ESUs in 1997 (NMFS 1997d) but not for candidate ESUs. In May 1997 NMFS listed the Southern Oregon/Northern California Coast (SONCC) Coho Salmon ESU as threatened (NMFS 1997b), while it announced that listing of the OCCS ESU was not warranted due to conservation measures in the Oregon Coastal Salmon Restoration Initiative (OCSRI) plan

Table 1. ESA chronology for the OCCS ESU.

Date	**Action**
May 2010	NMFS proposes to retain ESA threatened status of OCCS.
April 2009	NMFS initiates ESA status review of OCCS.
February 2008	In accordance with district court opinion, NMFS lists OCCS as threatened under the ESA.
October 2007	U.S. District Court in Oregon invalidates January 2006 decision not to list OCCS.
July 2007	U.S. District Court in Oregon rules that NMFS' decision not to list OCCS is "arbitrary, capricious, contrary to the best available evidence, and a violation of the ESA."
June 2006	Trout Unlimited challenges NMFS' decision not to list OCCS.
January 2006	NMFS concludes that OCCS are "not likely to become endangered" in foreseeable future and therefore listing them under ESA is not warranted; agency withdraws ESA listing proposal.
June 2005	NMFS releases final ESA hatchery listing policy and announces 6-month extension on listing determination for OCCS.
May 2005	Oregon releases final report of its Coastal Coho Assessment, concluding OCCS are viable and likely to persist into foreseeable future.
February 2005	NMFS requests public review and comment on Oregon's draft Coho Project Report.
June 2004	NMFS formally proposes to list OCCS as "threatened" under ESA and issues draft hatchery policy.
October 2003	Oregon begins Coastal Coho Project to evaluate effectiveness of Oregon Plan at recovering OCCS; state and NMFS work jointly on project.
November 2002	NMFS convenes OCCS Technical Review Team, charged with establishing biologically based delisting criteria and ESA recovery goals, and serving as science advisor to recovery planning.
July 2002	NMFS responds to ESA petition to redefine OCCS population.
February 2002	NMFS initiates ESA status review of West Coast salmon, including OCCS.
November 2001	NMFS begins developing new hatchery policy to address issues raised in Hogan decision and says it will apply new policy to all West Coast ESA-listed salmon and steelhead.
September 2001	Alsea Decision, U.S. District Court Judge Michael Hogan in Oregon finds that the ESA does not allow NMFS to split a salmon ESU into two components—hatchery and wild—then list only one of those components; functional effect of ruling is to delist OCCS.
August 1998	NMFS lists OCCS as threatened under ESA.
June 1998	Federal District Court for Oregon rules that "not warranted" determination for OCCS is arbitrary and capricious, saying the ESA doesn't allow NMFS to consider biological effects of future or voluntary conservation measures.
May 1997	NMFS determines OCCS is "not warranted" for listing under the ESA, based in part on Oregon's conservation measures contained in the plan.
March 1997	Oregon completes its Salmon Initiative Plan and submits it to NMFS.
October 1995	Oregon embarks on its Coastal Salmon Restoration Initiative to conserve and restore coastal salmon and steelhead.
July 1995	NMFS proposes to list OCCS as threatened under the ESA.
October 1993	NMFS receives petition from Pacific Rivers Council and 22 others requesting the agency list OCCS salmon under the ESA.

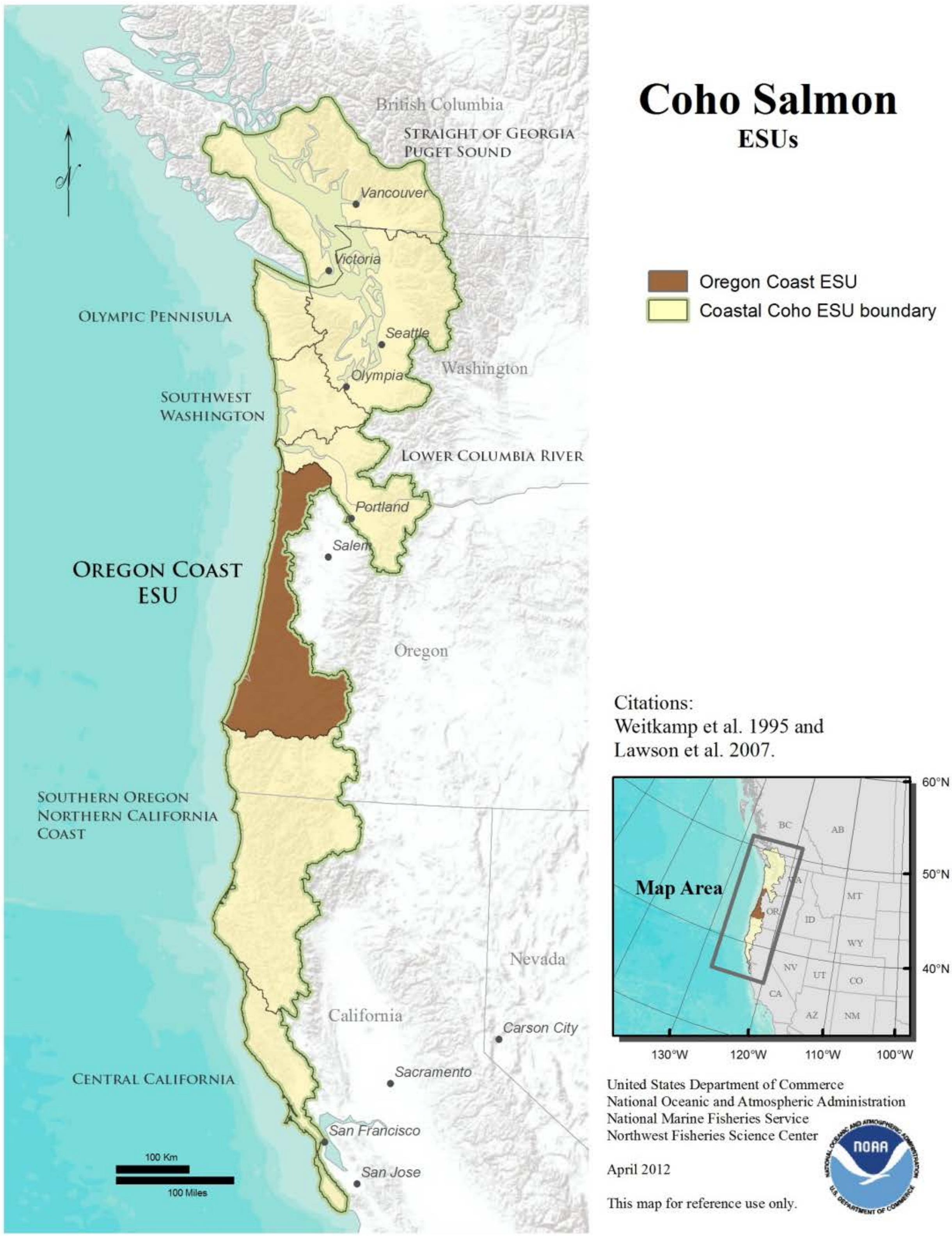

Figure 1. West Coast coho salmon ESUs.

(NMFS 1997b). This finding for OCCS was overturned by the Federal District Court for Oregon in August 1998 and the ESU was listed as threatened (NMFS 1998).

On 10 September 2001, Judge Michael R. Hogan, ruling in Alsea Valley Alliance v. Evans for the U.S. District Court for the District of Oregon, found that for the OCCS ESU, "NMFS's listing decision is arbitrary and capricious, because the Oregon Coast ESU includes hatchery spawned and naturally spawned coho salmon, but the agency's listing decision arbitrarily excludes hatchery spawned coho. Consequently, the listing is unlawful" (161 F. Supp. 2d 1154, D. Oreg. 2001). The lawsuit was brought by the Alsea Valley Alliance, partly in response to an action by the Oregon Department of Fish and Wildlife (ODFW) to terminate a domesticated coho salmon broodstock at the Fall River Hatchery on the Alsea River.

The effect of the ruling was to delist the OCCS ESU. The ruling was appealed by the appellant interveners to the U.S. Court of Appeals for the Ninth Circuit. On 14 December 2001 the court stayed the District Court ruling pending final disposition of the appeal (Alsea Valley Alliance v. Evans, Ninth Circuit appeal, No. 01-36071). This returned the OCCS ESU to threatened status under the ESA. In response NMFS initiated development of a new hatchery policy to address issues raised in the ruling.

In November 2002 NMFS convened the OCCS Workgroup (hereafter, workgroup), a subcommittee of the Oregon/Northern California Coast (ONCC) Technical Recovery Team (TRT). This workgroup was charged with establishing biologically based recovery criteria and ESA recovery goals as well as providing scientific advice to recovery planners. Results of the workgroup deliberations are published in Lawson et al. (2007) and Wainwright et al. (2008). In October 2003 Oregon began its Coastal Coho Project to evaluate the effectiveness of the Oregon Plan at recovering OCCS.[1]

The next coho salmon BRT met in January, March, and April 2003 as part of a coast-wide review of listed species to determine what portions of the artificially propagated salmon in each ESU should be listed with natively spawned fish and to discuss new data and determine whether conclusions of the original BRTs should be modified as the result of new information. In June 2004 NMFS published the proposal to list the OCCS ESU as threatened under the federal ESA (NMFS 2004a) and issued its draft hatchery policy (NMFS 2004b). The hatchery policy was finalized in 2005 (NMFS 2005a).

In May 2005 Oregon released the final Coast Coho Assessment (Nicholas et al. 2005), concluding that the OCCS ESU was viable and likely to persist into the foreseeable future. Subsequently, in January 2006 NMFS concluded that OCCS are "not likely to become endangered" in the foreseeable future, therefore that listing them under the ESA was not warranted, and withdrew its listing proposal (NMFS 2006).

In June 2006 Trout Unlimited challenged NMFS's decision not to list the OCCS ESU. In July 2007 a U.S. District Court in Oregon invalidated the January 2006 decision not to list the OCCS ESU. In February 2008, in accordance with the court's decision, NMFS listed the ESU as threatened under the ESA (NMFS 2008) and declared critical habitat (Figure 2).

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[1] Comments from the workgroup on the Oregon Coastal Coho Conservation Plan are online at http://www.oregon .gov/OPSW/cohoproject/PDFs/NOAA_Conservation_Plan_comments.pdf.

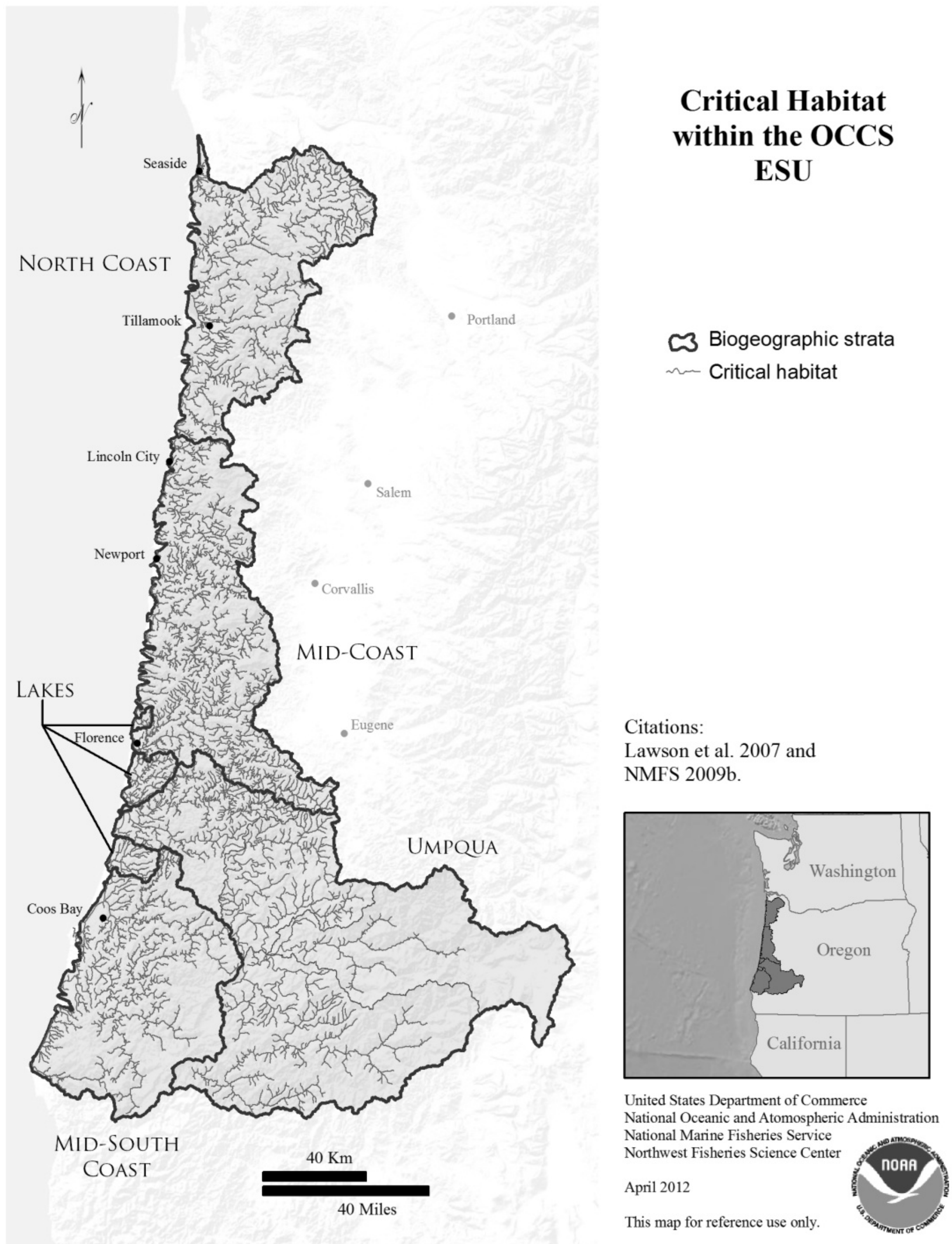

Figure 2. Critical habitat designation for the OCCS ESU, February 2008.

In 2008 NMFS, its Northwest Regional Office (NWR), and the Northwest Fisheries Science Center (NWFSC) formed the Recovery Implementation Science Team, a regional science team that provides scientific advice related to recovery plan implementation. Several TRTs, including the ONCC TRT, continued to provide local science support as subteams of the Recovery Implementation Science Team. In April 2009 NMFS announced a new status review for the OCCS ESU (NMFS 2009a) and formed a BRT.[2] The BRT met in September and December 2009.

This technical memorandum summarizes new information and the preliminary BRT conclusions on the OCCS ESU. It is intended as a summary of the information considered by the BRT in making its conclusion. It does not include specific recommendations for management; that is the role of the recovery plan. Many large sets of past analyses and conservation documents are included in this information and it is the goal of this presentation to include their information by reference to keep this document to a reasonable size. Details of previously published analytical methods are referred to in citations; details of those analyses can be found in the previously published documents. However, the BRT has utilized some analyses not previously published in this particular format, so a more detailed description of those is included in the appendices.

[2] The BRT consisted of Peter W. Lawson (chair), John Williams, Laurie Weitkamp, Robin Waples, Thomas Wainwright, Mark Scheuerell, George Pess, Robert Kope, Mike Ford, Chris Jordan, Tom Cooney, and Dan Bottom, NWFSC; Lance Kruzic, NMFS Northwest Regional Office; Tommy Williams, NMFS Southwest Fisheries Science Center; and Gordon Reeves, U.S. Forest Service. Heather Stout, NWFSC, acted as administrative staff for the BRT.

Summary of Previous BRT Conclusions

The OCCS ESU has been the subject of detailed assessments in three previous status reviews: one in 1994 (Weitkamp et al. 1995), another in 1996 (NMFS 1997d), and a third in 2003 (Good et al. 2005).

ESU Determination

As amended in 1978, the ESA allows listing of "distinct population segments" of vertebrates as well as named species and subspecies. However, the ESA provides no specific guidance for determining what constitutes a distinct population. To clarify the issue for Pacific salmon, NMFS published a policy describing how the agency will apply the definition of species in the ESA to anadromous salmonid species, including sea-run cutthroat trout (*Oncorhynchus clarkii clarkii*) and steelhead (*O. mykiss*) (NMFS 1991b). A more detailed discussion of this topic appeared in the NMFS definition of species paper (Waples 1991).

The NMFS policy stipulates that a salmon population (or group of populations) will be considered distinct for purposes of the ESA if it represents an ESU of the biological species. An ESU is defined as a population that 1) is substantially reproductively isolated from conspecific populations and 2) represents an important component of the evolutionary legacy of the species. The term evolutionary legacy is used in the sense of inheritance—that is, something received from the past and carried forward into the future. Specifically, the evolutionary legacy of a species is the genetic variability that is a product of past evolutionary events and that represents the reservoir on which future evolutionary potential depends. Conservation of these genetic resources should help ensure that the dynamic process of evolution will not be unduly constrained in the future.

The NMFS policy identifies a number of types of evidence that should be considered in the species determination. For each of the criteria, the NMFS policy advocates a holistic approach that considers all types of available information as well as their strengths and limitations. Isolation does not have to be absolute, but it must be strong enough to permit evolutionarily important differences to accrue in different population units. Important types of information to consider include natural rates of straying and recolonization, evaluations of the efficacy of natural barriers, and measurements of genetic differences between populations. Data from protein electrophoresis or DNA analyses can be particularly useful for this criterion, because they reflect levels of gene flow that have occurred over evolutionary time scales.

The key question with respect to the second criterion is: If the population became extinct, would this represent a significant loss to the ecological/genetic diversity of the species? Again, a variety of types of information should be considered. Phenotypic and life history traits such as size, fecundity, migration patterns, and age and time of spawning may reflect local adaptations of evolutionary importance, but interpretation of these traits is complicated by their sensitivity to environmental conditions. Data from protein electrophoresis or DNA analyses provide valuable insight into the process of genetic differentiation among populations, but little direct information regarding the extent of adaptive genetic differences. Habitat differences suggest the possibility for local adaptations, but do not prove that such adaptations exist.

The OCCS ESU was identified as one of six West Coast coho salmon ESUs in a coast-wide coho status review NMFS published in 1995 (Weitkamp et al. 1995). The six ESUs identified in that status review were: Puget Sound/Strait of Georgia, Olympic Peninsula, Columbia River/southwest Washington coast, Oregon coast, northern California/southern Oregon coast, and central California coast (Weitkamp et al. 1995). Subsequently, the Columbia River/Southwest Washington Coast ESU was divided into two ESUs (Columbia River and Southwest Washington Coast [NMFS 2001]), resulting in seven coho salmon ESUs.

Weitkamp et al. (1995) considered a variety of factors in delineating ESU boundaries, including environmental and biogeographic features of the freshwater and marine habitats occupied by coho salmon, patterns of life history variation and patterns of genetic variation, and differences in marine distribution among populations based on tag recoveries. Regarding the OCCS ESU, Weitkamp et al. (1995) concluded that Cape Blanco to the south and the Columbia River to the north constituted significant biogeographic and environmental transition zones that likely contributed to reproductive isolation and evolutionary distinctiveness for coho salmon inhabiting opposite sides of these features. These findings were reinforced by discontinuities in the ocean tag recoveries at these same locations. Finally, the available genetic data also indicated that OCCS north of Cape Blanco formed a discrete, although quite variable, group compared to samples from south of Cape Blanco or the Columbia River and northward.

Based on these sources of information, Weitkamp et al. (1995) described the OCCS ESU as follows:

> This ESU covers much of the Oregon coast, from Cape Blanco to the mouth of the Columbia River, an area with considerable physical diversity ranging from extensive sand dunes to rocky outcrops. With the exception of the Umpqua River, which extends through the Coast Range to drain the Cascade Mountains, rivers in this ESU have their headwaters in the Coast Range. These rivers have a single peak of flow in December or January and relatively low flow in late summer. Upwelling north of Cape Blanco is much less consistent and weaker than in areas south of Cape Blanco. Sitka spruce is the dominant coastal vegetation and extends to Alaska. Precipitation in coastal Oregon is higher than in southern Oregon/northern California but lower than on the Olympic Peninsula. Oregon coast coho salmon are caught primarily in Oregon marine waters and have a slightly earlier adult run timing than populations farther south.
>
> Genetic data indicate that Oregon coast coho salmon north of Cape Blanco form a discrete group, although there is evidence of differentiation within this area. However, because there is no clear geographic pattern to the differentiation, the area is considered to be a single ESU with relatively high heterogeneity.

Status review updates in 2001 (NMFS 2001) and 2003 (Good et al. 2005) did not reconsider ESU boundaries, with the exception of the Columbia River/southwest Washington Coast.

Status Evaluation in 1994

For the first review in 1994 (Weitkamp et al. 1995), extensive survey data were available for coho salmon in the Oregon coast region and information on trends and abundance was better for the OCCS ESU than for the more southerly ESUs. Overall, spawning escapements for OCCS had declined substantially during the twentieth century and natural production was at 5 to 10% of production in the early 1900s. Productivity and abundance showed clear long-term downward trends. Average spawner abundance had been relatively constant since the late 1970s, but preharvest abundance was declining. Average recruits per spawner were also declining and average spawner:spawner ratios were below replacement levels in the worst recent years. OCCS populations in most major rivers were found to be heavily influenced by hatchery stocks, although some tributaries may have maintained native stocks. The 1994 BRT noted widespread habitat degradation as a risk factor that, along with low abundance, posed a risk to the ESU due to increased variability. Because of these risks, the 1994 BRT concluded that the ESU was likely to become endangered in the foreseeable future if present trends continued.

Status Evaluation in 1996

Despite relatively good information on trends and abundance, the 1996 BRT (NMFS 1997d) faced some important uncertainties related to lack of information. Main uncertainties in the assessment included the extent of straying of hatchery fish, the influence of such straying on natural population trends and sustainability, the condition of freshwater habitat, and the influence of ocean conditions on population sustainability. For absolute abundance, the 1996 total average (5-year geometric mean) spawner abundance (44,500) and corresponding ocean run size (72,000) were less than one-tenth of ocean run sizes estimated in the late 1800s and early 1900s, and only about one-third of 1950s ocean run sizes (ODFW 1995). These abundances were well below estimated freshwater habitat production capacity for this ESU of run sizes of 141,000 under poor ocean conditions and 924,000 under good ocean conditions (OCSRI Science Team 1997). Abundance was unevenly distributed within the ESU through the early to mid-1990s.

Long-term trend estimates through 1996 showed that for escapement, run size, and recruits per spawner, trends were negative, while 6 years of stratified random survey population estimates showed an increase in escapement and decrease in recruitment. Furthermore, in the 1990s recruitment remained only a small fraction of average levels in the 1970s. Although spawner:spawner ratios had remained above replacement since the 1990 broodyear, recruit:spawner ratios for 1991–1993 broodyears were among the lowest on record. Recruits per spawner continued to decline after the OCCS ESU was reviewed in 1994. And the new data from 1994 to 1996 did not change the overall pattern of decline. This pattern was one of decline coupled with peaks in recruits per spawner every 4 to 5 years, with the height of the peaks declining over time. Risks that this decline in recruits per spawner posed to sustainability of natural populations, in combination with strong sensitivity to unpredictable ocean conditions, were the most serious concern the OCCS BRT identified in 1996, including whether recent ocean and freshwater conditions would continue into the future.

Widespread spawning by hatchery fish, as indicated by scale data, continued to be a major concern to the 1996 BRT, even though Oregon had recently made some significant changes in its hatchery practices including reduced production levels in some basins, switching to on-station smolt releases, and minimizing fry releases. Uncertainty regarding the true extent of hatchery influence on natural populations, however, was a strong concern. Another concern the BRT discussed in 1996 was asymmetry in the distribution of natural spawning in this ESU; a large fraction of the naturally spawned fish occurred in the southern portion. Northern populations were also relatively worse off by almost every other measure: steeper declines in abundance and recruits per spawner, higher proportion of naturally spawning hatchery fish, and more extensive habitat degradation.

With respect to habitat, the 1996 BRT had two primary concerns: 1) that habitat capacity for the OCCS ESU had significantly decreased from historical levels; and 2) that the Nickelson and Lawson (1998) model predicted that, during poor ocean survival, only high-quality habitat is capable of sustaining coho populations and subpopulations dependent on medium- and low-quality habitats would likely become extinct. Both of these concerns caused the 1996 BRT to consider risks from habitat loss and degradation to be relatively high for this ESU. The effects of the 1996 floods were not specifically discussed in this forum, but were included in factors for decline identified by NMFS NWR.

In addition to considering status based on recent conditions, the 1996 BRT was asked to consider ESU status if two sets of measures from the OCSRI were implemented: 1) harvest management reforms and 2) hatchery management reforms.

Some 1996 BRT members felt that the harvest measures were the most encouraging part of the OCSRI plan, representing a major change from previous management. However, there was concern among some of the 1996 BRT members that the harvest plan might be seriously weakened when it was reevaluated in 2000, while the ability to monitor nontarget harvest mortality and to control overall harvest impacts were also seen as a source of uncertainty. Of the proposed hatchery measures, the 1996 BRT thought substantial reductions in smolt releases would have the most predictable benefit for natural populations and marking all hatchery fish was anticipated to resolve uncertainties about the magnitude of those interactions.

In 1996 the BRT concluded that, assuming current conditions continued into the future (and that proposed harvest and hatchery reforms were not implemented), the OCCS ESU was not at significant short-term risk of extinction, but likely to become endangered in the foreseeable future. A minority disagreed, believing that the ESU was not likely to become endangered. The BRT generally agreed that implementation of the OCSRI's harvest and hatchery proposals would have a positive effect on the ESU's status, but was about evenly split as to whether the effects would be substantial enough to move the ESU out of the "likely to become endangered" category.

Status Evaluation in 2003

The OCCS ESU continued to present challenges to those assessing extinction risk in 2003. The 2003 BRT (Good et al. 2005) found several positive features compared to the previous assessment in 1996. For example, adult spawners for the ESU in 2001 and 2002 exceeded the number observed for any year in the past several decades and preharvest run size rivaled some of the high values seen in the 1970s (although well below historical levels), including increases in the formerly depressed northern part of the ESU. Hatchery reforms were increasingly being implemented and the fraction of natural spawners that were first generation hatchery fish were reduced in many areas, compared to highs in the early to mid-1990s.

On the other hand, the years of good returns just prior to 2003 were preceded by 3 years of low spawner escapements—the result of 3 consecutive years of recruitment failure, in which the natural spawners did not replace themselves, even in the absence of any directed harvest. These 3 years of recruitment failure were the only such instance observed in the entire time series. Whereas the increases in spawner escapement just prior to 2003 resulted in long-term trends in spawners that were generally positive, the long-term trends in productivity as of 2003 were still strongly negative.

For the OCCS ESU, the 2003 BRT received updated estimates of total natural spawner abundance based on stratified random survey techniques, broken down by ODFW's monitoring areas (Figure 29 in Lawson et al. 2007), for 10 major river basins and for the coastal lakes system.[3] In 2003 the total 3-year geometric mean spawner abundance was estimated at about 140,600 with spawners more evenly distributed than they had been previously.

The 2003 BRT used ODFW stratified random survey escapement data that indicated ESU-wide spawning escapement reached 30-year highs in 2001 and 2002. By contrast, in return years 1997–1999 (broodyears 1994–1996), and for the first time on record (since 1950), recruits failed to replace the parental spawners: a recruitment failure occurred in all three brood cycles, even before accounting for harvest-related mortalities. From 1999 until 2003, improving marine survival and higher rainfall were thought to be the factors contributing to an upswing in wild recruitment. However, it was far from certain that favorable marine conditions would continue and, with the freshwater habitat conditions, whether OCCS ESU could survive another prolonged period of poor marine survival remained in doubt.

In 2003 long-term (33-year) trends in spawner abundance for the lakes and rivers were slightly upward. Lakes increased about 2% per year and rivers increased about 1% per year. In lakes and rivers, long-term trends in recruits declined about 5% per year since 1970. For the ESU as a whole, spawners and recruits declined at a 5% rate from 1970 to 2003.

There had been notable changes in harvest management since the 1996 status review. The Pacific Fishery Management Council (PFMC) in 1988 (PFMC 1998) adopted Amendment 13 to its Salmon Fishery Management Plan, which was developed as part of the Oregon Plan for Salmon and Watersheds (Oregon Plan, formerly OCSRI). It specified an exploitation rate

[3] ODFW's monitoring areas are similar to but not identical to gene conservation groups (Figure 28 and Figure 29 in Lawson et al. 2007) that were the population units in the 1997 update.

harvest management regime with rates for naturally produced Oregon coast coho salmon (OCN)[4] dependent on marine survival and parental and grandparental spawning escapements. Allowable exploitation rates under the amendment can range 0−8% (poor marine survival) to a maximum of 45% (high survival and parent population).

Also, beginning in 1998 most adult hatchery-origin coho salmon in the Oregon Production Index (OPI) area were marked with an adipose fin clip, allowing the implementation of mark-selective fisheries. Recreational mark-selective fisheries were conducted on the Oregon coast in each year between 1998 and 2003, with quotas ranging from 13,000 to 24,000 marked fish. The 2003 BRT expressed concern that these incidental mortality rates estimated by PFMC were underestimates. Despite these uncertainties, there was no doubt that harvest-related mortalities were reduced substantially after 1994. This reduction was reflected in 2003 in positive short-term trends in spawner escapements despite continued downward trends in preharvest recruits. In summary, the higher returns in the early 2000s were tempered by the overall decline since 1970. When considered in the context of historical abundance and hatchery influence, this trend indicated a continuing decline in abundance across the ESU. Therefore, the BRT considered that future remedies outside of harvest management were required until the decline in productivity reversed.

As of 2003, the Oregon Plan (OCSRI Science Team 1997) was the most ambitious and far-reaching program to improve watersheds and recover salmon runs in the Pacific Northwest. The original OCSRI was written in 1997, so the plan had been in operation for several years by 2003.

Between 1991 and 2003, some Oregon coastal hatchery facilities were closed and the numbers of smolts released from the remaining facilities were reduced from 6.2 million in 1992 to 930,000 in 2001. Efforts to include more native broodstock were accomplished. The 2003 BRT considered that these changes would somewhat reduce risks to naturally spawning OCCS. As of 1999, most adult coho salmon of hatchery origin were marked with an adipose fin clip for fishery management; an additional benefit was better accounting of hatchery fish spawning in the wild.

The 2003 BRT conclusions for the ESU as a whole reflected ongoing concerns for the long-term health of this ESU: a majority of BRT opinion was in the “likely to become endangered” category, with a substantial minority falling in the “not likely to become endangered” category. Although they considered the significantly higher returns in 2001 and 2002 to be encouraging, most members thought that the factor responsible for the increases was more likely to be unusually favorable marine productivity conditions than improvement in freshwater productivity. The majority of BRT members thought that to have a high degree of confidence that the ESU was healthy, high spawner escapements should be maintained for a number of years and the freshwater habitat should demonstrate the capability of supporting high juvenile production from years of high spawner abundance.

The 2003 BRT considered the long-term decline in productivity to be the most serious concern for this ESU. With all directed harvest for these populations already eliminated, harvest

[4] Oregon Coast coho salmon naturally produced fish also includes SONCC populations in Oregon.

management (i.e., reducing harvest rates) could no longer compensate for declining productivity. The BRT was concerned that the long-term decline in productivity reflected deteriorating conditions in freshwater habitat and that the OCCS ESU would likely experience very serious risks of local extinctions during the next cycle of poor ocean conditions. With the cushion provided by strong returns in 2001−2003, the 2003 BRT had much less concern about short-term risks associated with abundance than did earlier BRTs.

New Contributions to Understanding and Assessing Status of OCCS

ESU Delineation

The 2009 BRT evaluated new information related to ESU boundaries. The biogeographical and environmental information summarized by Weitkamp et al. (1995) remains unchanged, and the 2009 BRT did not reevaluate this information. The data on tag recoveries that Weitkamp et al. (1995) evaluated were subsequently expanded, revised, and published (Weitkamp and Neely 2002). The revised analysis continues to show a distinct pattern of ocean tag recoveries for the OCCS ESU, consistent with the Weitkamp et al. (1995) conclusions.

Since the earlier status reviews, several new genetic studies of West Coast coho salmon that included samples from the Oregon coast have been published. Ford et al. (2004) analyzed data at 6 microsatellite loci from 22 populations of OCCS and several populations of Puget Sound coho salmon. Van Doornik et al. (2007) examined patterns of variation at 11 microsatellite loci from 84 coho salmon populations from northern California to southern British Columbia. Van Doornik et al. (2008) examined patterns of variation at eight microsatellite loci from coho salmon sampled from central California to Alaska. Johnson and Banks (2008) analyzed 23 populations of OCCS (including one population from the Rogue River in the SONCC Coho Salmon ESU) at 8 microsatellite loci.

The patterns of genetic variation in these newer studies are generally similar to those observed in the earlier studies summarized by Weitkamp et al. (1995). In particular, the new studies confirm that coho salmon are characterized by relatively low levels of population differentiation compared to other salmon species, particularly in the central part of their range. The new studies that include coast-wide samples (Van Doornik et al. 2007 and 2008) are also consistent with the data cited by Weitkamp et al. (1995), indicating genetic discontinuities at or around Cape Blanco and the Columbia River mouth. In particular, in a neighbor-joining tree cluster analysis, Van Doornik et al. (2007 and 2008) found 100% genetic bootstrap support for a cluster containing samples from the Rogue and Klamath rivers distinct from Oregon coast samples north of Cape Blanco. The same result has been confirmed with a more recent analysis of 18 microsatellite loci from approximately 6,000 coho salmon sampled coast wide.[5] These analyses indicate that the Oregon coast samples are distinct from the Columbia River and more northern populations, with moderate to high levels of genetic bootstrap support.

After considering the new information, the 2009 BRT concluded that a reconsideration of the ESU boundaries for the OCCS ESU is not necessary. The basis for this conclusion is that the environmental and biogeographical information considered by Weitkamp et al. (1995) remains

[5] Carlos Garza, SWFSC, 110 Shaffer Rd. Santa Cruz, CA. Pers. commun., November 2009.

unchanged, and new tagging and genetic analysis published subsequently to the original ESU boundary designation continue to support the current ESU boundaries.

Artificial Propagation—Membership in the ESU

At the time of the 2004 proposed rule and January 2006 final determination not to list the OCCS ESU, the Cow Creek (ODFW stock #37), North Umpqua River (ODFW stock #18), Coos Basin (ODFW stock #37), North Fork Nehalem stock, and Coquille River (ODFW stock #44) hatchery coho programs were considered part of the OCCS ESU. The Trask (Tillamook) and Salmon (Salmon, Siletz) stocks were excluded from the ESU due to observed or suspected divergence from natural populations (see Table 22.1 in NMFS 2004b). The North Umpqua, Coos, and Coquille programs have been discontinued since the 2006 final determination (NMFS 2008). The last year of returns for these programs was 2007. At the time of 2008 listing, only the Cow Creek stock was included in the ESU (the North Fork Nehalem was excluded from the ESU based on comments from ODFW, NMFS 2008). As of 2009, only three coho hatchery stocks are released within the freshwater boundaries of the OCCS ESU. These are the North Fork Nehalem, Trask (Tillamook Basin) and Cow Creek (South Umpqua) stocks.[6] The BRT found no new information to suggest that current ESU membership status of these stocks (Cow Creek in the ESU, others out of ESU) should be changed.

Population Delineation

Recently, the workgroup, a subcommittee of the ONCC TRT, published two documents, Identification of Historical Populations of Coho Salmon (*Oncorhynchus kisutch*) (Lawson et al. 2007) and Biological Recovery Criteria for Oregon Coast Coho Salmon (Wainwright et al. 2008). These defined historical population structure and biological recovery criteria and are discussed below. Because of these analyses, the discussion of risk to the species was focused at a finer scale than in previous status reviews.

The TRT's analysis of historical population structure of the ESU relies on a simple conceptual model of the spatially dependent demographics of the 56 populations that the workgroup considered likely to have been present historically within the ESU. This model classifies populations on the basis of two key characteristics: persistence (their relative abilities to persist in isolation from one another), and isolation (the relative degree to which they might have been influenced by adult fish from other populations straying into their spawning areas). The 56 populations are also used by ODFW and other resource agencies and have been incorporated into the State of Oregon's monitoring framework (ODFW 2007).

The TRT classified historical populations as dependent and functionally and potentially independent. For the purposes of this BRT, historical populations were reduced to two groups: independent and dependent (Table 2, Figure 3). Oregon coast drainage basins of intermediate to large size are thought to have each supported a coho salmon population capable of persisting in isolation. Some of them may have been demographically influenced by adult coho salmon straying into spawning areas from elsewhere in the ESU. Populations that appeared likely to

[6] Reported by BRT member L. Kruzic, December 2009.

Table 2. Independent and dependent historical populations of OCCS. (Adapted from Lawson et al. 2007.)

Population	Type	Population	Type
Necanicum	Independent	Alsea	Independent
Ecola	Dependent	Big (near Alsea)	Dependent
Arch Cape	Dependent	Vingie	Dependent
Short Sands	Dependent	Yachats	Dependent
Nehalem	Independent	Cummins	Dependent
Spring	Dependent	Bob	Dependent
Watseco	Dependent	Tenmile Creek	Dependent
Tillamook Bay	Independent	Rock	Dependent
Netarts	Dependent	Big	Dependent
Rover	Dependent	China	Dependent
Sand	Dependent	Cape	Dependent
Nestucca	Independent	Berry	Dependent
Neskowin	Dependent	Sutton (Mercer Lake)	Dependent
Salmon	Independent	Siuslaw	Independent
Devils Lake	Dependent	Siltcoos	Independent
Siletz	Independent	Tahkenitch	Independent
Schoolhouse	Dependent	Threemile	Dependent
Fogarty	Dependent	Lower Umpqua	Independent
Depoe Bay	Dependent	Middle Umpqua	Independent
Rocky	Dependent	North Umpqua	Independent
Spencer	Dependent	South Umpqua	Independent
Wade	Dependent	Tenmile	Independent
Coal	Dependent	Coos	Independent
Moolack	Dependent	Coquille	Independent
Big (near Yaquina)	Dependent	Johnson	Dependent
Yaquina	Independent	Twomile	Dependent
Theil	Dependent	Floras/New	Independent
Beaver	Independent	Sixes	Independent

have been capable of persisting in isolation were classified as independent (21 populations). Small coho salmon populations found in smaller coastal basins and that may not have been able to maintain themselves continuously for periods as long as hundreds of years without strays from adjacent populations were classified as dependent populations (Lawson et al. 2007).

The TRT concluded that dependent populations relied at times on the strength of adjacent larger populations for their continuous historical presence in the Oregon coast's smaller basins. As long as the larger persistent populations within the ESU remained strong, the smaller (dependent) populations would rarely if ever have disappeared from their basins. However, if some form of broadscale environmental change triggered a substantial decline in one or more of the larger populations, the reduction in migrants would have increased the possibility that the

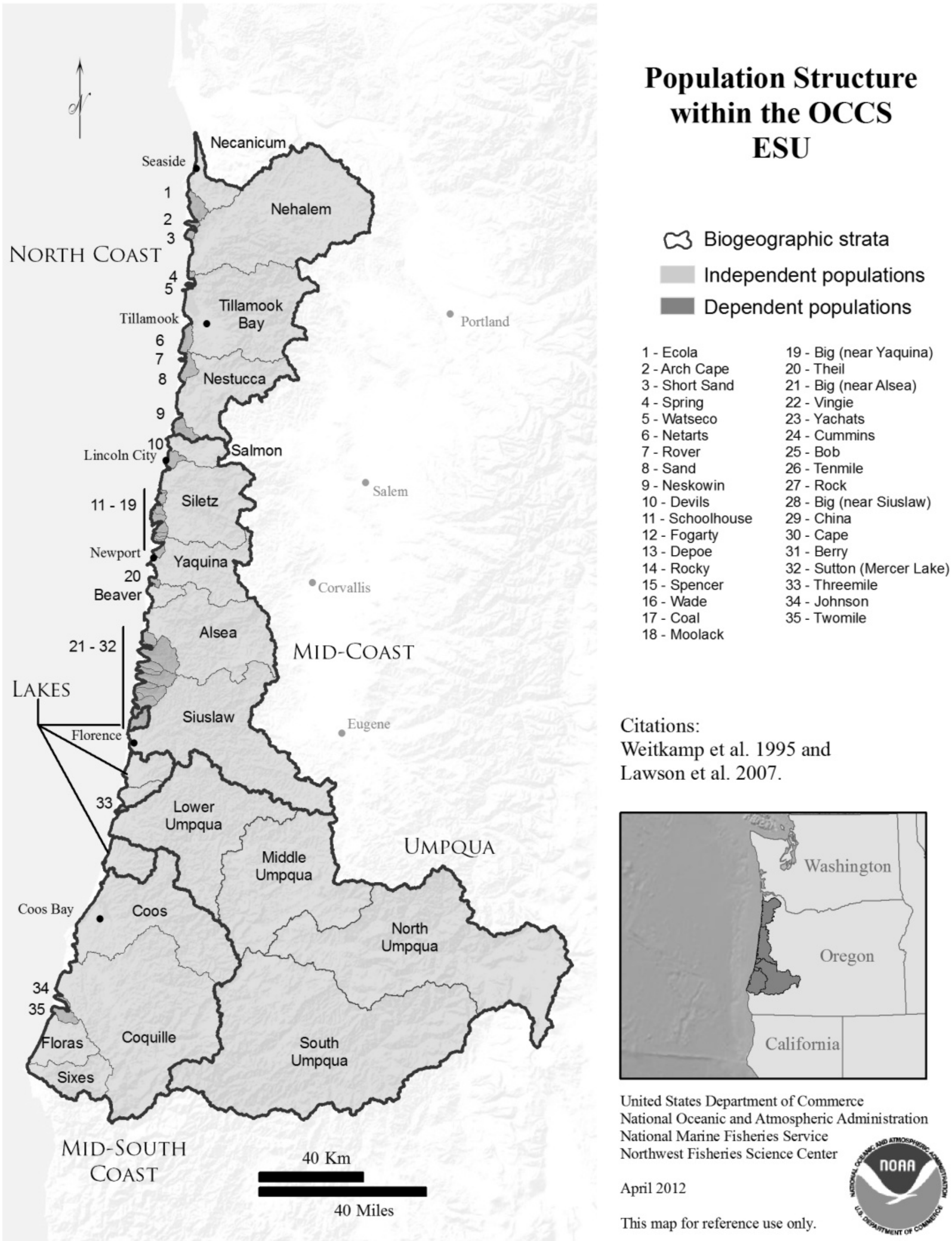

Figure 3. Historical populations and biogeographic strata for OCCS ESU.

same environmental change, perhaps coupled with local disturbances, would have resulted in the intermittent disappearances of the dependent populations found in some of the smaller basins. This may have occurred in the ESU in 1998 when no spawners were observed returning to Cummins Creek (Lawson et al. 2007.)

Definition of Biogeographic Strata

Within the OCCS ESU, there is substantial genetic and biogeographic structure, with populations clustering into a few larger geographic units that were identified by the workgroup. These biogeographic strata represent genetic and geographic similarities and assume that preserving sustainable populations in each of them will conserve major genetic diversity in the ESU, as well as spread risks to the maintenance of genetic and geographic diversity due to catastrophes. The workgroup considered that all strata must be secure for the entire ESU to be sustainable (Wainwright et al. 2008).

In defining the biogeographic strata, the workgroup considered that the four ODFW monitoring areas (Figure 29 in Lawson et al. 2007) in the ESU, for the most part, reflected the geography, ecology, and genetics of the landscape. However, the lakes are very different from the other portions of the Mid-South Coast Monitoring Area ecologically, geographically, and genetically. In order to reflect this diversity and reduce the risks to genetic and geographical diversity due to catastrophes, they accepted the Ford et al. (2004) Lakes Complex as a fifth biogeographical stratum for use in defining areas of diversity important in conservation (Figure 3).

Because these units represent biological diversity (genetic and ecological) and geographic variation, the workgroup considered that preserving all of them will accomplish two goals: 1) preserving major genetic and life history variation in the ESU and 2) spreading risks due to catastrophes. The 2009 BRT used these strata in considering risks to genetic and life history diversity.

Biological Recovery Criteria Used to Inform Risk Assessment

Wainwright et al. (2008) outlined biological recovery criteria (also called viability criteria) for the OCCS ESU, as identified in NMFS's status review for West Coast coho salmon (Weitkamp et al. 1995). The report was developed by the workgroup of the ONCC TRT. The BRT used the Wainwright et al. (2008) report as one important source to inform its risk assessment process. However, the BRT also considered other factors, such as environmental threats not included in the Wainwright et al. (2008) criteria, in making its overall risk determination.

A complete assessment of the biological condition of the ESU is necessarily multifaceted, including a variety of interrelated criteria, with varying data quality. The recovery criteria developed by the TRT relate to biological processes at a variety of time and space scales, with processes varying from individual stream reaches to the entire range of the ESU. To track this large suite of data and criteria in a transparent and logically consistent framework, Wainwright et al. (2008) constructed a knowledge-based decision support system (DSS).

The DSS uses a network framework to link criteria at a variety of scales and aggregate them from fine scale watershed-level criteria, through population-level criteria and biogeographic stratum-level criteria, to criteria for the entire ESU. The links take the form of logical operators that define specific relationships among the input values. In this knowledge-based system, a type of fuzzy logic extends the ability to work with imprecise knowledge of attributes of the OCCS ESU. The advantage of using this logic is that it allows evaluation and expression of certainty in an outcome, ranging from certainly false through uncertain to certainly true. The ability to work with a gradation of levels of certainty and uncertainty assisted the 2009 BRT in evaluating the degree of risk and uncertainty in its assessments. This analysis is further described in the Current Biological Status subsection below.

New Comments

Below are brief summaries of comments NMFS received in response to the April 2009 Federal Register notice asking for new scientific information for consideration by the BRT. Full texts of the comments are available from NMFS NWR.[7] The BRT considered these comments in its deliberations.

Trout Unlimited commented that it favors maintaining the current listing status of threatened. It asked the BRT to take a close look at hatchery practices, harvest, bycatch in estuary fisheries, and climate change during this status review. No specific information regarding these issues was provided (Trout Unlimited 2009).

U.S. Environmental Protection Agency (EPA) comments focused on the inadequacy of state programs to protect water quality and other OCCS habitat requirements. EPA presented comments previously included in other OCCS reviews that the Oregon Forest Practices Rules and Best Management Practices will not consistently meet water quality standards or protect riparian function. An EPA letter sent to the State of Oregon in 2005 regarding the inadequacy of the State's Oregon Coast Coho Conservation Plan was attached (EPA 2009).

The Pacific Rivers Council (PRC) and Center for Biological Diversity supported maintaining the current listing of threatened and asserted that they have no knowledge of credible information available to support changing this ESU's status. They included by reference that the State of Oregon was unable to secure an ESA Section 10 Habitat Conservation Plan for the Elliot State Forest Management Plan. They cited a letter from NMFS/NWR Habitat Conservation Division stating that NMFS has unresolved concerns regarding the ability of the plan to protect OCCS habitat. They also pointed out that other state forest plans throughout the range of OCCS are similar to the Elliot State Forest situation and may be equally inadequate. The PRC also suggested that watershed road density be used as a measure of risk for OCCS. It cited a new U.S. Bureau of Land Management (BLM) analysis indicating that road densities are relatively high throughout almost all subwatersheds occupied within OCCS distribution. The council's final point was about considering local extirpation and population homogenization due to the population dynamics driven by ocean conditions (PRC 2009).

The American Forest Resources Council commented on measurable habitat improvements since adoption of the Oregon Plan in 1997. It cited several publications and reports indicating that recent improvements in OCCS habitat demonstrate that the Oregon Plan is working. It also supported a population viability assessment that is tailored to this particular ESU and its naturally wide swings in abundance. The council cautioned against using the more simple population viability models relying primarily on abundance and productivity adopted by other West Coast salmon TRTs (e.g., Puget Sound TRT 2002) (American Forest Resources Council 2009).

[7] NMFS Northwest Regional Office, 1202 NE Lloyd Blvd., Suite 1100, Portland, OR 97232.

ODFW provided a significant amount of new information about the status of this ESU. Its comments highlight recent hatchery release reductions and changes to marine harvest. It pointed out that during times of low ocean survival, harvest will be managed under Amendment 13 of the Pacific Salmon Treaty, but stated that at times of higher abundance, coho salmon harvest may occur at levels that limit progress toward recovery but do not represent a threat to viability. ODFW also provided new population data and information on habitat conditions. It generally concluded that range-wide stream and riparian habitat conditions have remained relatively stable between 1998 and 2008. But ODFW also concluded that this habitat is in a condition suitable for producing enough smolts to maintain viability even during periods of low marine survival. ODFW reported that stream productivity seems to be improving slightly in all areas except the Umpqua Basin (Anlauf et al. 2009, ODFW 2009b).

The Douglas County Commissioners supplied a list of habitat improvement projects carried out in the Umpqua River basin. They stated that the number of projects occurring in this basin was evidence that the Oregon Plan works as intended. The commissioners also commented that significant harvest reform has been completed to ensure harvest no longer represents a threat to this ESU's viability. They asserted that recent high abundance realized by Umpqua populations and the cancellation of the North Umpqua hatchery program further demonstrate that this ESU does not need to be listed. Finally, they pointed out what they believe are some problems with the BRT population models. The commissioners were in favor of a not warranted finding for this ESU (Douglas County Commissioners 2009).

The State of Oregon Governor's Office commented that the State, "through its natural resource agencies, continues to put substantial effort on the ground and in policy to improve conditions for and status of coastal coho salmon." Its comments provided a summary of the Oregon Plan and the 2007 Oregon Coast Coho Conservation Plan. It also listed the state agencies responsible for implementation of these plans and some of their recent accomplishments (Oregon 2009a).

Coquille Indian Tribe comments mostly focused on coho salmon in the Coquille River basin. It listed several habitat limiting factors and provided an update on recent restoration projects carried out by the tribe and its partners. It also provided some information on recent abundance estimates and cited a study (Jacobs 2002) that concluded that recent surveys may have underestimated the abundance of returning coho spawners. The tribe stated that predators (marine mammals, birds, and fish) may be having significant effects on OCCS. The tribe concluded its comments by suggesting that some basins like the Coquille, Coos, and Umpqua be examined differently in the status review because listing is not warranted (Coquille Indian Tribe 2009).

Thad Springer (a private citizen) was not in favor of a listing and commented that the presence of a listed species is a disincentive to private landowners to carrying out restoration projects on their land. Mr. Springer also provided a list of information sources the BRT should consider, mostly related to state programs (Springer 2009).

The Native Fish Society (Paul Engelmeyer) commented that OCCS should remain listed as threatened. He asserted that the Oregon Forest Practices Act and state programs for agricultural lands are insufficient to protect water quality in the ESU. He also pointed out several other threats including pesticide runoff into coho salmon streams, designation of beaver as a nuisance species by the State of Oregon, and ongoing floodplain development. He commented against terminal recreational harvest of OCCS in TRT-identified independent populations (Umpqua, Coos, Yaquina, etc.) and was critical of Amendment 13 of the Pacific Salmon Treaty. Mr. Engelmeyer included numerous reports with his comments that have been made available to the BRT (Native Fish Society 2009).

Oregon Coast Coho Salmon Symposium

In order to provide an opportunity for the State of Oregon and ODFW to present their information to the BRT, a one-day symposium was organized. In addition to the State of Oregon, comanagers and interested parties were invited to make presentations of new scientific information and information on restoration activities. This section provides a short summary of the main topics presented at the symposium. The BRT considered the information presented in its deliberations.

OCCS ESU: Population status and conservation measures update from Oregon Department of Fish and Wildlife. Kelly Moore, ODFW, presented an overview of fish metrics for the ESU. These were wild or natural spawner and hatchery spawner abundance time series from 1950 to 2008, spawner abundance by population, spawner distribution, spawner surveys from 2005−2008 occupancy, modeled parr capacity, juvenile occupancy rates, hatchery release history, trends in hatchery influence, life cycle monitoring egg-to-smolt and smolt-to-adult survival, habitat limiting factors model (HLFM) smolt capacity, population abundance patterns and spatial distribution variability, and 2008−2009 returns. Oregon Plan and Oregon Coast Coho Conservation Plan information also were presented. This included additional factors for decline and issues of concern, the Oregon Plan habitat strategy, recognition of contributions to habitat by beavers, and what ODFW is doing to encourage conservation of beavers and the contributions of Oregon watershed councils.

The status and trends of physical habitat and rearing potential in coho bearing streams in the OCCS ESU. Kim Jones, ODFW, presented information on the Habitat Survey Program and included a discussion of factors for decline, Oregon Plan integrated monitoring, survey design, distribution of sites, and balancing status and trends sampling requirements. He discussed the four monitoring strata and the status of stream habitat in the OCCS ESU. The monitored aspects of wadeable streams are pools, large wood volume, fine sediments, and winter habitat. The HLFM (Anlauf et al. 2009) was discussed with a presentation of the capacity of differing kinds of pools, winter rearing, and spawner abundance. Monitoring trends analysis done by Kara Anlauf concluded that the ESU's streams are generally pool rich but structurally simple, mean values of the monitored attributes are all low, there are few off-channel habitats or beaver pools, and most streams have low volumes of wood and high fine sediment.

Oregon's plan for protecting salmon and watersheds. Suzanne Knapp, Governor's Natural Resources Office, discussed the Oregon Plan framework and which agencies are addressing limiting factors such as water removal, water quality, and stream complexity.

BLM and U.S. Forest Service (USFS) land management in OCCS ESU. Joe Moreau, BLM State Office, presented information on the types of restoration activities and the costs that BLM and USFS have engaged in to help restore OCCS habitat.

Satellite-based summaries of yearly timber harvest rates on all lands within the OCCS ESU from 1985 to 2008. Dr. Robert Kennedy, Department of Forest Ecosystems and Society, Oregon State University, presented his work on a new application of his land remote sensing satellite (Landsat) analytical method. He presented information that attempts to address the question: Does terrestrial habitat condition matter for coho salmon with respect to temperature, sediment type, and delivery? His conclusions were that the yearly disturbance information is useful for interpretation of impacts of policy and economics, that disturbance magnitude shows variability across ownerships and time, and that private lands dominate the land base and disturbance impacts. He also suggested that variation in disturbance rates and timing across basins may provide leverage for useful inferences about land management actions.

Maintaining Oregon's forest land base: The Forest Practices Act role in the conservation of forest values on nonfederal forest lands. Jim Paul, Oregon Department of Forestry (ODF), presented the Oregon Plan accomplishments of private timber landowners and discussed the threat of forest conversion to other land uses.

Road density, watershed condition, and implications for salmonid conservation in the range of the Oregon coastal coho salmon. Dr. Chris Frissell, PRC, presented a summary of his and others' work on applicable science from studies in the Columbia Basin on bull trout (*Salvelinus confluentus*). These centered on habitat response: more fine sediment, fewer pools, less wood, water quality decline (temperature and nutrients/toxics), watershed degradation, and salmon population response such as status, abundance, and species diversity.

Observations on water quality improvement under SB1010 and other lowland issues. Paul Engelmeyer, Native Fish Society, presented an overview of four issues that need significant change to improve the chances of coho population recovery at a landscape scale: agricultural water quality management, State of Oregon beaver policy, policies to protect floodplain function, and improvement in forest practices.

Recent observations of Oregon coast coho salmon in Smith River. Dr. Joe Ebersole, EPA, presented some of his work that attempts to help identify where habitat restoration activities should take place. He concluded that OCCS utilize intermittent stream habitat for a significant amount of winter rearing. In addition, what should be very good habitat (high intrinsic potential) is presently poor habitat in the study area due to legacy stream effects such as splash damming. He suggested that habitat that should be improved and conserved is the existing habitat currently at the center of coho salmon production. In other words, "fix the best first."

Hinkle Creek paired watershed study. Daniel Newton, working with the Watershed Research Cooperative, presented preliminary results of the Hinkle Creek paired watershed study. Their preliminary findings were that initial temperature response was small compared to the original Alsea Watershed Study, downstream recovery of nutrient increases following timber harvest is typical of other studies, sediment increased following timber harvest but was attenuated downstream, and that fish (coastal cutthroat trout and steelhead) survival and distributions were similar to preharvest patterns.

Comments of Douglas County, Oregon, on OCCS ESU. David Loomis, Douglas County Board of Commissioners and Public Works Department, presented information on the effects of unreported habitat restoration projects and the unreported decline of “likely to adversely affect” activities in freshwater habitat on the evaluation of habitat conditions. He also discussed habitat restoration projects reflecting specific needs of OCCS populations (instantaneous and long-term), spawner to spawner ratios, and generational health of individual populations. A discussion of recent ocean and in-river harvest history and the hatchery program level for North Umpqua and for Umpqua Basin spawning hatchery strays was included. He requested that the BRT use the North Umpqua Population Case Study as “truth value” of model sensitivity and risk of extinction status (Douglas County Commissioners 2009).

New Data and Updated Analyses

Current Biological Status

This section addresses new data and updated analyses for the viable salmonid population (VSP) parameters of abundance, growth rate (productivity), spatial structure, and diversity (McElhany et al. 2000). In addition, Harvest Impacts and Artificial Propagation subsections are included here for consistency with previous BRT analyses. Finally, a new analysis utilizing the ONCC TRT's biological recovery criteria is included.

Population Size

In past status reviews (Weitkamp et al. 1995, NMFS 1997d, Good et al. 2005), the BRTs considered two measures of recent population abundance—spawner abundance and preharvest recruits—and also considered recent estimates in the context of published estimates of abundance in the late 1800s and early 1900s (Mullen 1981a, Lichatowich 1989, ODFW 1995).

The 2009 BRT received updated estimates of total natural spawner abundance (and corresponding recruits) based on stratified random survey techniques broken down by historical populations for 10 major river basins and the coastal lakes system.[8] These data are shown in Table 3, Figure 4, and Figure 5. Since the previous status review, natural spawner abundance was generally up relative to the late 1990s and in all but 1 year (2007) has been well above the recent mean.

In the 1994 status review, Weitkamp et al. (1995, p. 113) considered historical estimates of abundance for this ESU and concluded that "these numbers suggest current abundance … may be less than 5% of that in the early part of the century." The current BRT reexamined that information and reanalyzed some of the historic abundance information. The earlier review based this conclusion largely on estimates of spawning escapement published by ODFW (Mullen 1981a). A reexamination of Mullen's tables found that he made a miscalculation such that the spawning escapement estimates included in-river gill net harvest and were thus inflated.

We made an independent estimate of spawners and recruits for the ESU for the period 1892–1956 using in-river gill net harvest estimates from other ODFW reports (Cleaver 1951, Mullen 1981b). Spawner escapement was estimated by expanding estimated gill net harvest, assuming a 40% harvest rate for 1892−1925 (Mullen 1981a, Lichatowich 1989), which was reduced as rivers were closed to fishing (Table 7 in Mullen 1981b, 12 rivers open in early years, 7 rivers remained open until 1956). Recruits were then calculated by adding in-river harvest and ocean troll catch. Ocean troll catch estimates are available for 1925−1927 and we assumed a 10% ocean harvest rate for 1912−1924 (Mullen 1981b). Results are shown in Figure 6,

[8] River basins from Pacific Coast Salmon Plan September 2003. Data from K. Moore, Research and Monitoring Supervisor, ODFW Corvallis Research Lab, Corvallis, OR. Pers. commun., September 2009.

Table 3. OCN coho salmon spawners and recruits (rivers, lakes, total) from 1969 to 2009, with approximate ocean exploitation rates. Data are from PFMC 2009 spreadsheet T6, version 1 (PFMC 2010).

	Spawners (thousands)			Exploitation	Recruits (thousands)		
Year	**Rivers**	**Lakes**	**Total**	**rate**	**Rivers**	**Lakes**	**Total**
1969	129.2	10.0	139.2	0.67	391.5	30.2	421.7
1970	51.2	21.5	72.7	0.65	183.1	61.7	244.8
1971	65.6	30.0	95.6	0.83	416.3	171.1	587.4
1972	24.1	10.6	34.7	0.84	185.5	67.5	253.0
1973	37.8	18.1	55.9	0.82	235.0	100.4	335.4
1974	28.1	6.8	34.9	0.84	196.4	41.0	237.4
1975	34.8	6.3	41.1	0.81	208.4	33.8	242.2
1976	39.2	1.7	40.9	0.90	451.7	17.1	468.8
1977	13.7	6.0	19.7	0.89	161.2	53.9	215.1
1978	18.2	1.8	20.0	0.83	111.6	10.3	121.9
1979	38.4	6.6	45.0	0.79	188.8	32.1	220.9
1980	25.6	5.0	30.6	0.73	108.3	18.6	126.9
1981	30.1	3.2	33.3	0.81	174.5	16.9	191.4
1982	68.3	8.3	76.6	0.62	188.4	22.0	210.4
1983	19.4	3.8	23.2	0.79	104.8	19.5	124.3
1984	59.7	16.3	76.0	0.32	95.3	24.1	119.4
1985	66.3	7.9	74.2	0.43	126.2	14.1	140.3
1986	58.2	12.3	70.5	0.34	98.9	18.7	117.6
1987	25.9	4.3	30.2	0.60	71.1	10.9	82.0
1988	51.0	6.2	57.2	0.56	127.3	14.6	141.9
1989	41.6	5.4	47.0	0.55	107.9	12.5	120.4
1990	16.5	4.7	21.2	0.69	60.6	15.3	75.9
1991	29.1	7.7	36.8	0.44	69.4	14.0	83.4
1992	38.6	2.1	40.7	0.51	87.7	4.4	92.1
1993	44.3	10.2	54.5	0.42	81.3	17.7	99.0
1994	37.5	5.9	43.4	0.07	40.3	6.0	46.3
1995	41.3	11.3	52.6	0.12	47.2	14.7	61.9
1996	59.5	13.6	73.1	0.08	64.9	15.9	80.8
1997	14.1	8.7	22.8	0.13	16.1	9.9	26.0
1998	19.8	11.2	31.0	0.08	21.5	12.0	33.5
1999	34.6	12.8	47.4	0.08	37.5	13.9	51.4
2000	54.1	12.8	66.9	0.07	58.4	13.8	72.2
2001	148.0	19.9	167.9	0.07	160.0	21.5	181.5
2002	231.4	22.3	253.7	0.12	264.2	25.2	289.4
2003	206.3	16.3	222.6	0.14	241.3	18.8	260.1
2004	149.2	19.3	168.5	0.15	175.2	21.8	197.0
2005	119.3	14.3	133.6	0.11	134.4	15.7	150.1
2006	87.2	22.7	109.9	0.06	92.8	23.6	116.4
2007	42.3	9.4	51.7	0.11	47.8	10.1	57.9
2008	142.1	23.6	165.7	0.04	146.5	24.6	171.1
2009	215.5	17.4	232.9	0.11	241.6	19.6	261.7

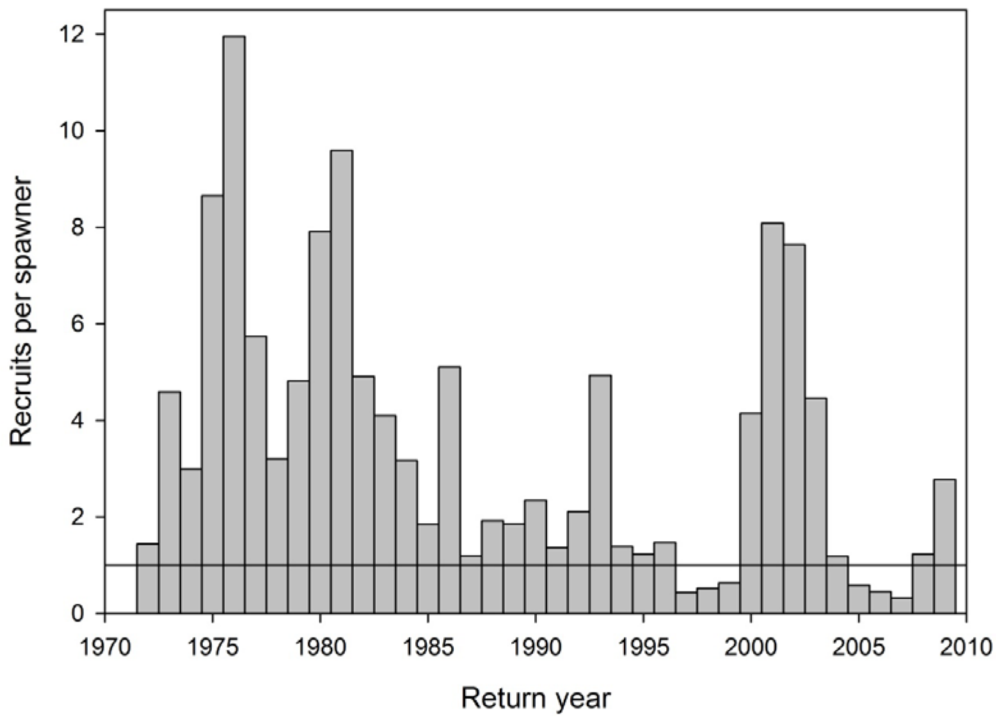

Figure 4. OCCS naturally produced recruits per spawner (recruits are from spawners 3 years earlier), 1972–2009. Horizontal unity line is replacement value. Data are from Table 3, this document.

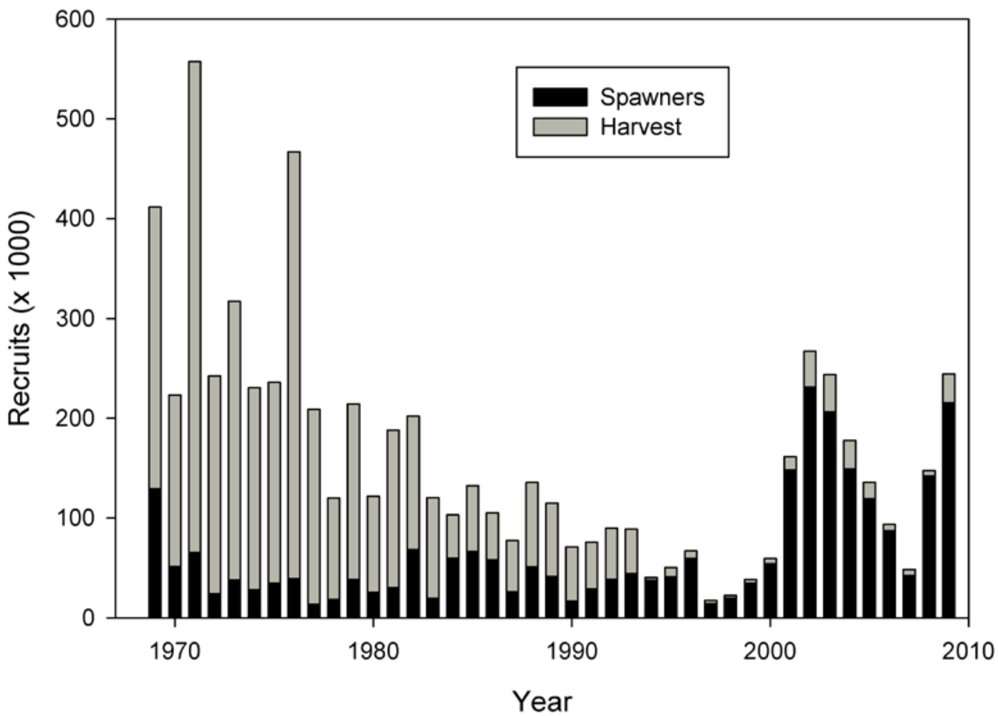

Figure 5. OCCS naturally produced recruits, 1969–2008. Each bar shows spawners and harvest. Harvest was gradually reduced starting in the late 1970s and sharply curtailed in 1994. Data are from Table 3, this document.

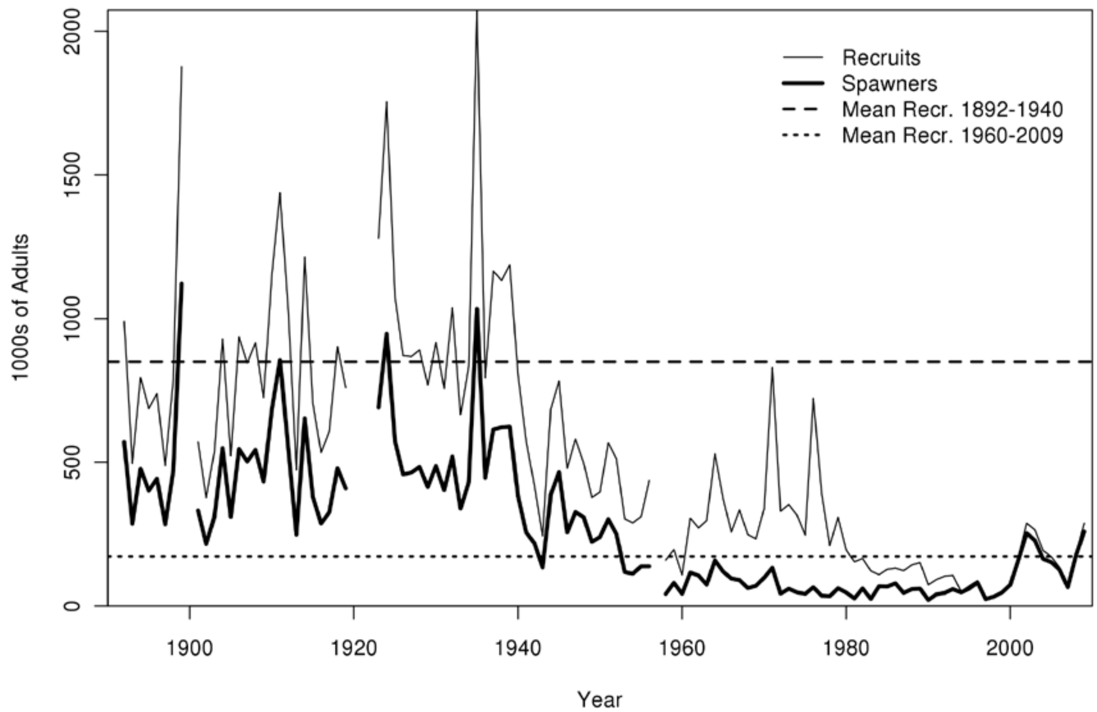

Figure 6. Comparison of historical (1892–1956) and recent (1958–2009) estimates of spawner abundance and preharvest recruits. Horizontal lines are the geometric mean recruits for 1892–1940 and 1960–2009. Analysis based on data from Cleaver 1951, Mullen 1981a, and Mullen 1981b; recent data from Wainwright et al. 2008 and ODFW 2009a.

compared with more recent estimates. From this historical perspective, recruits over the past few years have been close to the 1960−2009 average, but are only a fraction of the abundance before 1940.

While these historical abundance estimates are very rough and based on an assumed gill net harvest rate derived from expert opinion, they suggest that there has been a substantial decrease in ESU-wide abundance during the twentieth century. In fact, the decline was a concern to state biologists as early as the late 1940s (Cleaver 1951). Cleaver did not discuss causes of the decline other than to note that it was not caused by changes in harvest rates. However, Lichatowich (1989) related the overall decline to habitat loss, reporting a decline in production potential from about 1.4 million recruits ca 1900 to only 770,000 in the 1980s, likely resulting from habitat alterations related to timber harvest and agriculture, which both expanded on the coast between 1910 and 1950.

Population Growth Rate

Previous status reviews noted strong concerns regarding long-term and short-term trends in population productivity of the ESU. The BRT examined population growth rate (productivity) via two parameters: the ratio of recruits to spawners (R/S), and the natural return ratio (NRR). These measure different aspects of population dynamics. R/S indicates the basic productivity of populations in the absence of harvest, that is, the intrinsic ability of spawners in one generation to produce adults in the next generation. NRR is defined as the ratio of naturally produced spawners in one generation to the total (natural + hatchery-produced) spawners in the previous generation. NRR indicates the realized ability of populations to replace themselves, given intrinsic production and the effects of harvest and hatchery production.

Recruits from the return years 1997–1999 failed to replace parental spawners: a recruitment failure occurred in all three brood cycles even before accounting for harvest-related mortalities (Figure 4). This was the first time this had happened since data collection began in the 1950s. In most years since 2000, improved marine survival and higher rainfall are thought to be factors that have contributed to an upswing in recruits. However, in the return years 2005, 2006, and 2007, recruits also failed to replace parental spawners (Figure 4). There are several possible explanations for the more recent recruitment failure. It may reflect population dynamics that have not been allowed to occur since 1950; prior to 1994 harvest had consistently maintained spawner abundance near 50,000 fish (Figure 5). With harvest sharply curtailed in 1994, most recruits have been able to return to spawn. Ocean conditions improved for the 1998 broodyear, and recruits since 2001 have returned to spawn in numbers higher than previously observed. Harvest and hatchery reductions have changed the population dynamics of the ESU. Assuming these changes continue into the future, response of the system to fluctuations in environmental conditions and spawner escapements may show a different pattern than seen in the recent past. However, it is too soon to discuss with confidence the nature of these changes or the degree to which they may have improved the status of the ESU. In particular, it has not been demonstrated that productivity during periods of poor marine survival is now adequate to sustain the ESU.

Response of the system to higher escapements is essentially unknown; there are several possible interpretations of the recent patterns in productivity as measured by R/S and NRR. These productivity statistics are influenced by marine survival, freshwater survival, and freshwater carrying capacity (there is little evidence for an ocean carrying capacity). The current data set measures only the endpoints of spawners and ocean recruits, so we cannot easily separate the effects of freshwater and marine survival, although the ODFW life cycle monitoring sites help address this concern for recent years. Several lines of evidence based on patterns in the marine environment (CalCOFI 2010) and patterns of survival and abundance of a variety of salmon stocks in the Pacific Northwest (PFMC 2011) indicate that marine survival has been relatively good during the period from about 1998 to the present.

Despite apparently high marine survival, OCCS recruitment never exceeded 275,000, suggesting that current freshwater habitat capacity is substantially lower than it was up to the 1980s, when production potential was estimated at about 767,000 (Lichatowich 1989). The observation that recruits failed to replace spawners in 2005–2007 could be an indication that productive capacity had been reached or simply be a reflection of patterns in marine survival. Recent increases do not provide strong evidence that the century-long downward trend has changed. Abundance observations are consistent with the pattern of cyclical abundance overlying a downward trend as hypothesized by Lawson (1993).

While total spawners has been at its highest level since the 1950s, total recruits has not (Figure 5 through Figure 7). This suggests that the overall productivity (Figure 4 and Figure 8) and capacity (Figure 5 and Figure 6) of the system has, at best, been stable over the past half century.

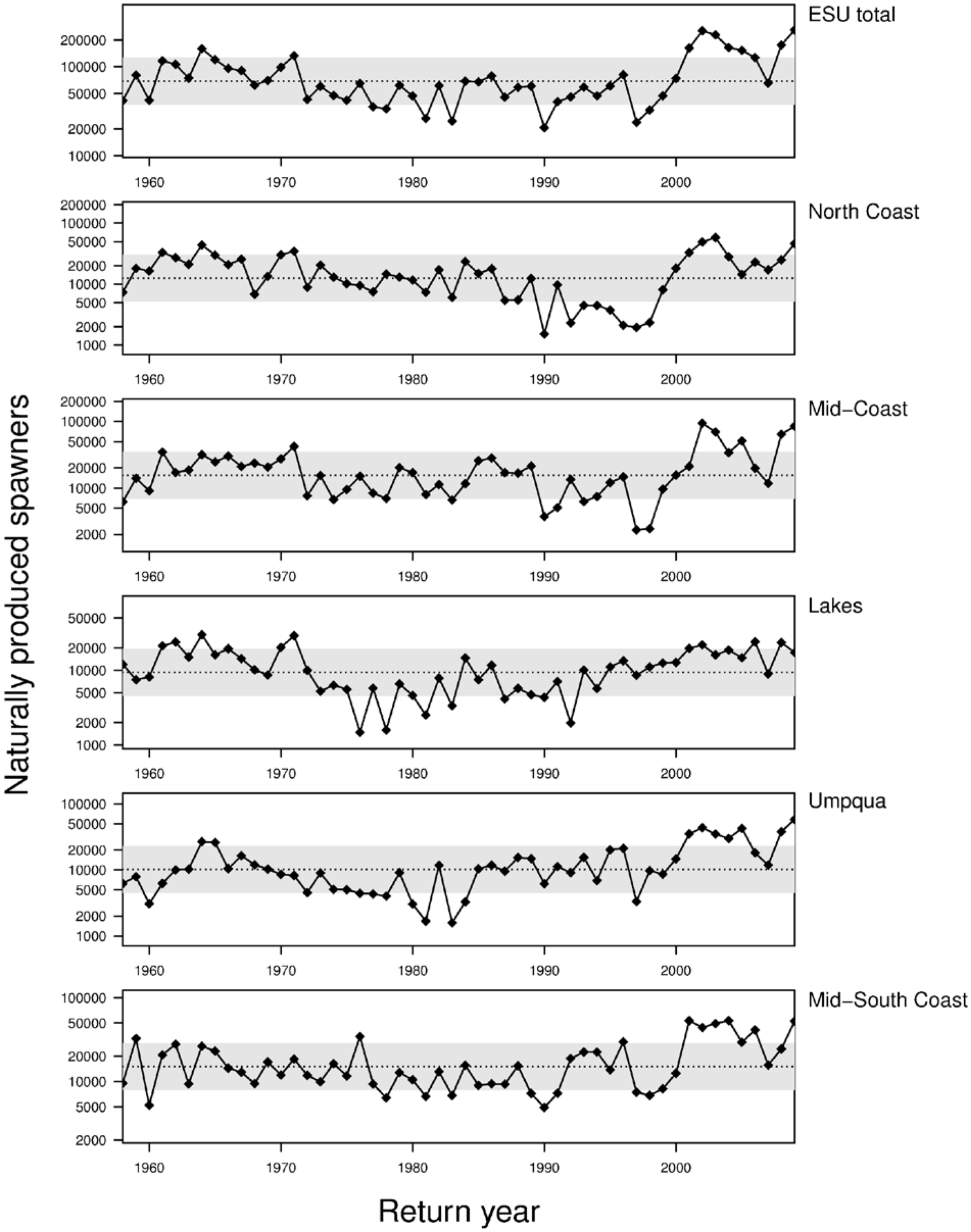

Figure 7. Trends in natural spawner abundance for the whole ESU (top panel) and the five biogeographic strata (lower panels). The dotted line marks the long-term mean and the gray background spans the mean ±1 SD. Note the logarithmic scale. Data from Wainwright et al. 2008 and ODFW 2009a.

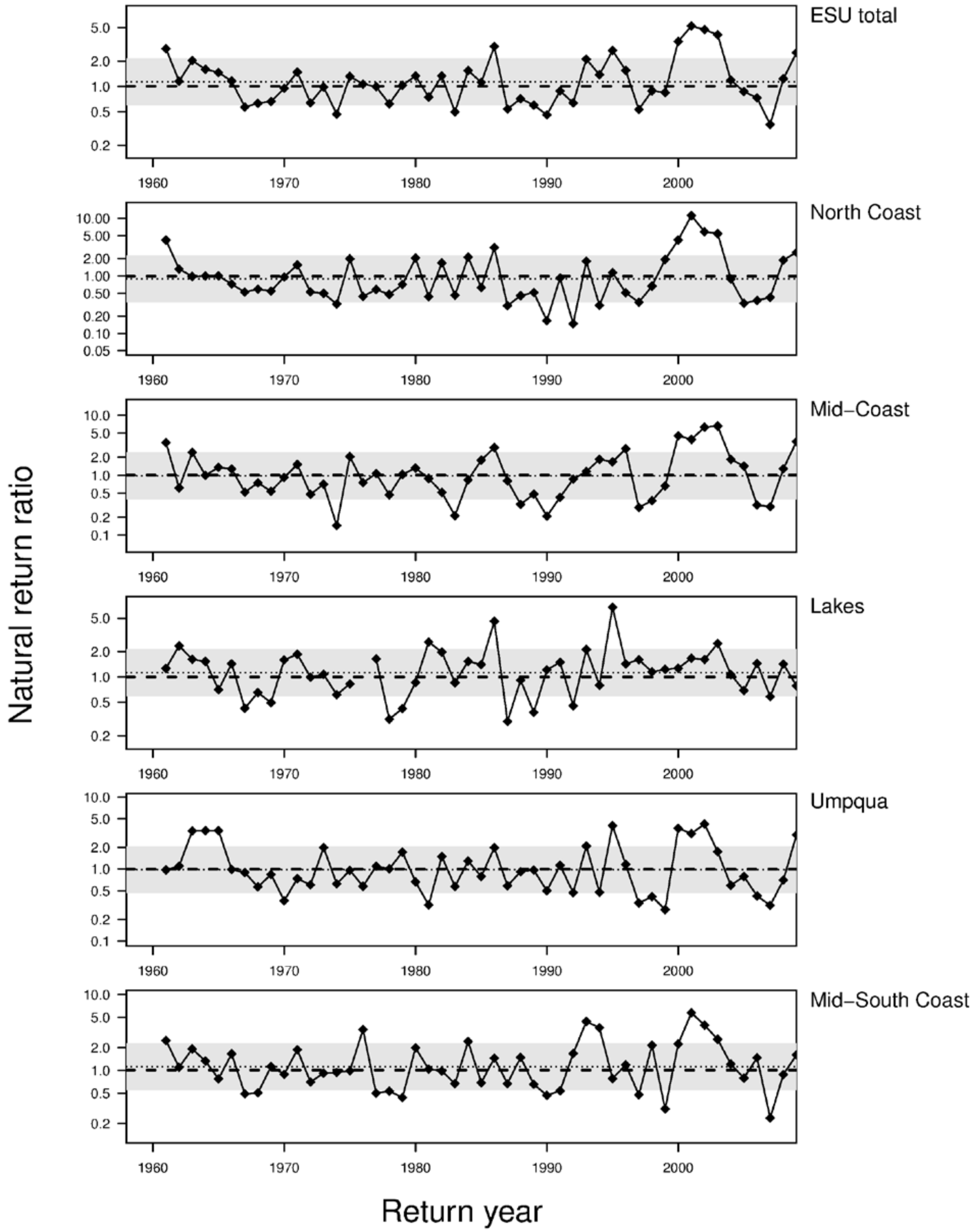

Figure 8. Trends in natural return ratio for the whole ESU (top panel) and the five biogeographic strata (lower panels). The dotted line marks the long-term mean and the gray background spans the mean ±1 SD. The dashed line is the replacement line. Note the logarithmic scale. Data from Wainwright et al. 2008 and ODFW 2009a.

Given current habitat conditions, OCCS are thought to require an overall marine survival rate of 0.03 to achieve a spawner:recruit ratio of 1:1 in high quality habitat (Nickelson and Lawson 1998). Less productive habitats require higher marine survivals to sustain populations. Based on OPI hatchery survival rates (Table 4), marine survival (adults/smolt) exceeded 0.03 only in 2001 and 2003. Assuming natural spawners survive at twice the hatchery rate (Lawson et al. 2004, Wainwright et al. 2008), in 11 out of 18 years since 1990 marine survivals were high enough to sustain the ESU. Increases in recruits (Figure 6) reflect improved marine survival after 2000. Marine conditions will continue to cycle (Lawson 1993) and, with current freshwater habitat conditions, the ability of the OCCS ESU to survive another prolonged period of poor marine survival remains in question.

For the ESU as a whole, the 12-year NRR is higher than the long-term NRR mean, with an up-and-down trend over the recent 12 years (Figure 8). The pattern is similar for the North Coast, Mid-Coast, and Mid-South Coast strata. The Lakes and Umpqua River strata have recent mean NRRs close to the long-term mean. The trend for the Umpqua River stratum is similar to the other riverine strata, while the Lakes stratum has a flat trend in recent years.

Population Spatial Structure

The 2009 BRT utilized historical populations defined and classified in Lawson et al. (2007). The TRT identified 56 populations, 21 independent and 35 dependent. The dependent populations rely on strays from other populations to maintain them over long time periods. The TRT also identified five biogeographic strata. This is a change from the 1996 status review, which partitioned OCCS into ODFW's three gene conservation groups, and from the 2003 status review, which partitioned OCCS into ODFW's four monitoring areas (Figure 28 and Figure 29 in Lawson et al. 2007).

Spatial structure was identified as a problem in the 1980s and 1990s, when it was observed that river systems on the north coast had substantially lower spawner escapements than those on the south coast. This problem persisted into the late 1990s (see Table 5). Causes of these disproportionately lower escapements were never clearly identified, but contributing factors may have included more intense fisheries north of Cape Falcon near the mouth of the Columbia River and high percentages of hatchery fish on the spawning grounds (Table 6). Stray hatchery spawners originated primarily from large hatcheries in the Nehalem and Trask rivers, but also came from the Columbia River hatcheries. Harvest was generally reduced in 1994 (although not as severely north of Cape Falcon as south). Hatchery releases in the Nehalem and Trask rivers have been reduced or eliminated so that the percentage of hatchery fish on the spawning grounds has declined from a high of 67% in 1996 to less than 5% in most recent years (Table 6). Since about 1999, the north coast basins have had escapements more on a par with the rest of the ESU. Reduced harvest, reduced hatchery influence, and improved ocean conditions are all likely contributors.

Current concerns for spatial structure focus on the Umpqua River. Of the four populations in the Umpqua stratum, two—the North Umpqua and South Umpqua—were of particular concern. The North Umpqua is controlled by Winchester Dam and has historically been dominated by hatchery fish (Table 6). Hatchery influence has recently been reduced, but the natural productivity of this population remains to be demonstrated. The South Umpqua is a

Table 4. OPI hatchery marine survival calculated as adults per smolt from adult recruits and smolts in previous year, 1970–2008 (t-1 refers to year previous). Smolt data are from PFMC 2009 Table T3, adult data are from OPITT 2009 Table T6, version 1, and 2009 data are preliminary (PFMC 2010).

Year (t)	Smolts (t-1) (millions)	Adults (t) (thousands)	Adults/smolt
1970	28.8	2,765.1	0.096
1971	33.3	3,365.0	0.101
1972	35.3	1,924.8	0.055
1973	33.6	1,816.9	0.054
1974	32.6	3,071.1	0.094
1975	34.0	1,652.8	0.049
1976	34.2	3,885.2	0.114
1977	36.6	987.5	0.027
1978	37.4	1,824.2	0.049
1979	35.1	1,476.7	0.042
1980	36.4	1,223.9	0.034
1981	43.9	1,064.5	0.024
1982	35.9	1,250.8	0.035
1983	32.2	572.8	0.018
1984	35.9	679.2	0.019
1985	37.0	705.1	0.019
1986	42.6	2,383.0	0.056
1987	35.5	876.7	0.025
1988	37.1	1,634.6	0.044
1989	38.1	1,660.7	0.044
1990	40.0	717.2	0.018
1991	42.1	1,898.3	0.045
1992	39.7	538.5	0.014
1993	39.5	260.2	0.007
1994	32.3	201.2	0.006
1995	29.5	144.8	0.005
1996	31.6	185.5	0.006
1997	24.6	199.4	0.008
1998	29.1	211.6	0.007
1999	29.7	334.0	0.011
2000	32.1	668.8	0.021
2001	26.8	1,410.7	0.053
2002	25.2	641.9	0.025
2003	24.5	934.1	0.038
2004	23.4	614.4	0.026
2005	22.0	433.4	0.020
2006	21.8	454.0	0.021
2007	22.7	546.2	0.024
2008	22.8	565.3	0.025
2009	22.3	1,066.1	0.047

Table 5. OCCS ESU spawning escapement reported by ODFW, 1990–2009. Estimation methodologies changed in 2005 to enable ODFW to assess populations as defined by the OCCS TRT. Populations, reporting groups, and escapement numbers are different from those used by the TRT, and may be considerably different in some populations in some years. ODFW is currently working with the BRT to reconcile the differences. General patterns are consistent among all versions of this data set.

Monitoring area,	Spawner abundance by return year																			
	Basin/group based estimates															Population based estimates				
Basin/group	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
North Coast																				
Necanicum River and Elk Creek	191	1,135	185	941	408	211	768	253	946	728	474	5,247	2,896	3,068	3,142	1,218	750	431	1,105	3,827
Nehalem River	1,552	3,975	1,268	2,265	2,007	1,463	1,057	1,173	1,190	3,713	14,285	22,310	20,903	33,059	21,479	10,451	11,614	14,033	15,690	21,753
Tillamook Bay	265	3,000	261	860	652	289	661	388	271	2,175	1,983	1,883	15,715	14,584	2,290	1,995	8,774	2,295	4,897	16,251
Nestucca River	189	728	684	401	313	1,811	519	271	169	2,201	1,171	3,940	13,003	8,929	6,152	686	1,876	394	5,444	4,252
Sand Lake and Neskowin Creek	—	240	24	41	77	108	275	61	0	47	0	71	16	0	0	—	—	—	—	—
Miscellaneous	—	204	—	—	—	—	—	—	—	—	—	—	—	—	—	2,116	1,121	376	540	2,052
Total	2,197	9,282	2,422	4,508	3,457	3,882	3,280	2,146	2,576	8,864	17,913	33,451	52,533	59,640	33,063	16,466	24,135	17,529	27,676	48,135
Mid-Coast:																				
Salmon River	385	39	28	364	107	212	271	237	8	175	0	310	372	0	2,374	79	513	59	784	753
Siletz River	441	984	2,447	400	1,200	607	763	336	394	706	3,553	1,437	2,252	9,736	6,399	14,567	5,205	2,197	14,519	24,070
Yaquina River	381	380	633	549	2,448	5,668	5,127	384	365	2,588	647	3,039	23,981	13,254	4,989	3,441	4,247	3,158	8,710	11,182
Devil's Lake and Beaver Creek	23	—	756	500	1,259	—	1,340	425	1,041	3,366	738	5,274	8,754	5,812	7,179	2,264	1,950	611	1,182	3,575
Alsea River	1,189	1,561	7,029	1,071	1,279	681	1,637	680	213	2,050	2,465	3,339	6,170	8,957	6,005	13,907	1,972	2,146	11,431	14,638
Yachats River	280	28	337	287	67	117	176	99	102	150	79	52	1,245	1,635	641	—	—	—	—	—
Siuslaw River	2,685	3,740	3,440	4,428	3,205	6,089	7,625	668	1,089	2,724	6,767	11,024	57,129	29,257	8,443	16,907	5,869	3,552	17,042	30,607
Miscellaneous	207	—	700	180	250	231	1,188	13	71	0	12	764	4,063	217	4,364	242	1,468	547	4,204	1,610
Total	5,591	6,732	15,370	7,779	9,815	13,605	18,127	2,842	3,283	11,759	14,261	25,239	103,966	68,868	40,394	51,407	21,224	12,270	57,872	86,435
Umpqua																				
Lower Umpqua and Smith River	589	1,316	1,759	4,804	1,689	6,803	4,904	935	5,118	2,323	3,696	8,850	14,492	12,760	8,046	18,591	7,994	4,237	12,267	19,245
Mainstem Umpqua	455	—	192	1,431	1,240	352	339	397	444	1,289	2,774	8,177	9,349	5,770	5,309	7,608	4,852	1,587	4,594	15,075
Elk and Calapooya Creek	185	—	—	—	708	2,315	1,709	196	379	434	1,864	2,581	1,555	4,450	2,602	—	—	—	—	—
South Umpqua R.	2,508	2,284	201	2,415	579	755	1,685	512	678	1,219	479	6,482	1,670	2,345	9,333	14,364	2,246	4,549	12,007	15,944
Cow Creek	—	—	—	661	269	1,124	1,112	193	1,807	1,234	1,582	6,661	6,745	1,277	2,351	—	—	—	—	—
Winchester Dam (wild adult coho)	376	1,273	1,607	933	851	1,460	1,075	727	727	1,186	1,838	2,951	3,780	3,005	3,705	2,113	3,062	1,410	3,438	7,720
Total	4,113	4,873	3,759	10,244	5,336	12,809	10,824	2,960	9,153	7,685	12,233	35,702	37,591	29,607	31,346	42,676	18,154	11,783	32,306	57,984
Lakes																				
Siltcoos Lake	1,622	2,895	391	3,622	1,426	4,497	4,775	2,653	3,122	2,819	3,835	5,104	4,812	7,225	8,025	4,364	5,473	1,447	3,873	5,197
Tahkenitch Lake	1,085	1,215	318	954	1,062	1,627	1,627	1,858	2,817	3,769	634	3,526	3,489	3,203	3,496	1,897	3,718	3,551	2,604	2,977
Tenmile Lake	1,687	3,141	1,277	5,569	3,354	5,092	7,092	4,092	5,169	6,123	8,278	11,039	13,861	6,260	7,166	8,464	15,187	3,957	17,131	9,175
Total	4,393	7,251	1,986	10,145	5,841	11,216	13,493	8,603	11,107	12,710	12,747	19,669	22,162	16,688	18,687	14,724	24,378	8,956	23,608	17,349
Mid-Coast South																				
Coos Bay and Big Creek	2,273	3,813	16,545	15,284	14,685	10,351	12,128	1,127	3,167	4,945	5,386	43,301	35,688	29,559	24,116	17,048	11,266	1,329	13,312	26,979
Coquille	2,712	5,651	2,115	7,384	5,035	2,116	16,169	5,720	2,466	3,001	6,130	13,310	8,610	23,909	22,276	11,806	28,577	13,968	9,874	22,286
Floras & Sixes R.	—	—	—	—	—	—	—	—	252	164	1,440	1,945	20	310	5,498	NAS*	1,104	340	637	3,203
	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	NAS	NAS	NAS	77	176
	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	NAS	NAS	NAS	NAS	188
Total	4,985	9,464	18,660	22,668	19,720	12,467	28,297	6,847	5,885	8,110	12,956	58,556	44,318	53,778	51,890	28,854	40,947	15,637	23,900	52,832
Oregon Coast ESU	21,279	37,602	42,197	55,344	44,169	53,979	74,021	23,398	32,004	49,128	70,110	172,617	260,570	228,581	175,380	154,127	128,838	66,174	165,362	262,735

*NAS = not adequate surveys or samples for estimate.

Table 6. Hatchery influence at coho population, stratum, and ESU scales, 1994–2010. Hatchery influence expressed as percent of total spawning escapement. Table from ODFW 2009a.[a] EL = extremely low, L = low, and M = medium.

ESU/strata Population	Return year / Marine survival	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
		EL	EL	EL	EL	EL	L	L	M	L	M	M	L	L	M	EL	M	L
Oregon	**Total**	**59,418**	**71,219**	**107,150**	**28,237**	**40,614**	**51,730**	**82,644**	**186,139**	**265,122**	**239,743**	**183,366**	**166,211**	**141,352**	**72,121**	**182,957**	**269,645**	**258,719**
Coast ESU	**Hatchery**	**12,394**	**10,448**	**26,128**	**4,576**	**8,139**	**4,688**	**8,953**	**24,321**	**12,028**	**12,186**	**10,633**	**11,616**	**12,784**	**5,850**	**3,271**	**6,910**	**3,953**
	% hatchery	**20.9%**	**14.7%**	**24.4%**	**16.2%**	**20.0%**	**9.1%**	**10.8%**	**13.1%**	**4.5%**	**5.1%**	**5.8%**	**7.0%**	**9.0%**	**8.1%**	**1.8%**	**2.6%**	**1.5%**
North Coast	**Total**	**8,239**	**7,026**	**6,484**	**3,451**	**2,837**	**8,860**	**18,704**	**33,944**	**50,465**	**58,768**	**29,953**	**16,509**	**25,524**	**18,126**	**25,777**	**50,505**	**53,655**
stratum	**Hatchery**	**3,755**	**3,267**	**4,368**	**1,500**	**496**	**766**	**486**	**1,076**	**1,222**	**672**	**1,131**	**43**	**1,389**	**597**	**206**	**2,370**	**495**
	% hatchery	**45.6%**	**46.5%**	**67.4%**	**43.5%**	**17.5%**	**8.6%**	**2.6%**	**3.2%**	**2.4%**	**1.1%**	**3.8%**	**0.3%**	**5.4%**	**3.3%**	**0.8%**	**4.7%**	**0.9%**
Necanicum	Total	448	301	693	161	958	370	378	5,112	2,143	2,535	2,339	1,252	843	464	1,183	3,869	3,183
River	Hatchery	179	120	277	64	383	19	19	280	96	158	141	34	93	33	128	42	0
	% hatchery	40.0%	39.9%	40.0%	39.8%	40.0%	5.1%	5.0%	5.5%	4.5%	6.2%	6.0%	2.7%	11.0%	7.1%	10.8%	1.1%	0.0%
Nehalem	Total	5,556	3,818	4,293	2,538	1,257	4,155	14,580	22,342	17,862	32,801	18,825	10,451	12,816	14,458	17,205	23,493	29,729
River	Hatchery	2,712	2,118	3,766	1,351	51	600	118	414	698	284	89	0	1,202	425	0	1,740	354
	% hatchery	48.8%	55.5%	87.7%	53.2%	4.1%	14.4%	0.8%	1.9%	3.9%	0.9%	0.5%	0.0%	9.4%	2.9%	0.0%	7.4%	1.2%
Tillamook	Total	1,922	1,096	979	481	384	1,978	2,477	2,119	13,707	13,129	3,360	1,995	8,774	2,429	4,906	16,811	14,346
Bay	Hatchery	817	755	246	44	26	147	299	175	373	121	828	0	0	134	78	560	103
	% hatchery	42.5%	68.9%	25.1%	9.1%	6.8%	7.4%	12.1%	8.3%	2.7%	0.9%	24.6%	0.0%	0.0%	5.5%	1.6%	3.3%	0.7%
Nestucca	Total	313	1,811	519	271	238	2,357	1,269	4,371	16,753	10,303	4,768	695	1,895	399	1,844	4,252	5,129
River	Hatchery	47	274	79	41	36	0	50	207	55	109	73	9	19	5	0	0	26
	% hatchery	15.0%	15.1%	15.2%	15.1%	15.1%	0.0%	3.9%	4.7%	0.3%	1.1%	1.5%	1.3%	1.0%	1.3%	0.0%	0.0%	0.5%
North Coast	Total	NA[b]	NA	NA	NA	NA	NA	NA	NA	NA	NA	661	2,116	1,196	376	639	2,080	1,268
Dependents	Hatchery	—	—	—	—	—	—	—	—	—	—	0	0	75	0	0	28	12
	% hatchery	—	—	—	—	—	—	—	—	—	—	0.0%	0.0%	6.3%	0.0%	0.0%	1.3%	0.9%
Mid-Coast	**Total**	**12,219**	**16,156**	**24,278**	**3,529**	**5,067**	**10,879**	**15,824**	**23,731**	**95,721**	**71,535**	**44,066**	**53,402**	**22,695**	**13,663**	**70,742**	**88,044**	**53,661**
stratum	**Hatchery**	**4,805**	**4,104**	**9,633**	**1,197**	**2,626**	**1,261**	**262**	**2,656**	**1,514**	**2,135**	**1,996**	**1,995**	**1,471**	**1,393**	**2,604**	**1,609**	**150**
	% hatchery	**39.3%**	**25.4%**	**39.7%**	**33.9%**	**51.8%**	**11.6%**	**1.7%**	**11.2%**	**1.6%**	**3.0%**	**4.5%**	**3.7%**	**6.5%**	**10.2%**	**3.7%**	**1.8%**	**0.3%**
Salmon	Total	1,554	1,325	2,703	417	432	173	394	877	1,108	1,738	3,525	817	1,160	993	2,664	753	1,438
River	Hatchery	1,463	1,220	2,621	401	346	159	215	652	565	1,696	1,883	738	647	934	2,012	0	92
	% hatchery	94.1%	92.1%	97.0%	96.2%	80.1%	91.9%	54.6%	74.3%	51.0%	97.6%	53.4%	90.3%	55.8%	94.1%	75.5%	0.0%	6.4%
Siletz River	Total	1,200	607	763	336	357	1,364	3,387	2,454	2,504	8,421	8,179	15,234	5,323	2,416	20,634	25,032	5,814
	Hatchery	579	293	368	38	41	155	0	859	375	383	0	667	118	219	0	962	0
	% hatchery	48.3%	48.3%	48.2%	11.3%	11.5%	11.4%	0.0%	35.0%	15.0%	4.5%	0.0%	4.4%	2.2%	9.1%	0.0%	3.8%	0.0%
Yaquina	Total	2,448	5,668	6,104	529	644	2,567	638	3,760	23,800	16,484	5,652	3,613	4,306	3,355	10,913	11,690	8,727
River	Hatchery	408	945	1,526	110	134	4	1	171	0	0	113	172	59	197	0	508	0
	% hatchery	16.7%	16.7%	25.0%	20.8%	20.8%	0.2%	0.2%	4.5%	0.0%	0.0%	2.0%	4.8%	1.4%	5.9%	0.0%	4.3%	0.0%
Beaver	Total	675	308	1,701	644	520	1,511	1,510	2,114	3,360	5,552	4,569	2,264	2,122	611	1,218	3,575	2,442
Creek	Hatchery	0	0	405	147	119	0	46	282	143	0	0	0	172	0	0	0	0
	% hatchery	0.0%	0.0%	23.8%	22.8%	22.9%	0.0%	3.0%	13.3%	4.3%	0.0%	0.0%	0.0%	8.1%	0.0%	0.0%	0.0%	0.0%
Alsea River	Total	1,279	681	1,637	928	1,732	2,071	3,363	3,920	9,254	10,281	5,233	13,907	1,972	2,146	13,442	14,777	8,218
	Hatchery	451	240	577	327	1,624	730	0	692	181	0	0	0	0	0	122	139	0
	% hatchery	35.3%	35.2%	35.2%	35.2%	93.8%	35.2%	0.0%	17.7%	2.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.9%	0.9%	0.0%
Siuslaw	Total	5,063	7,567	11,370	675	1,382	3,193	6,532	10,606	55,695	29,059	8,729	17,321	6,260	3,581	17,864	30,607	24,594
River	Hatchery	1,904	1,406	4,136	174	362	213	0	0	250	56	0	414	391	29	373	0	0
	% hatchery	37.6%	18.6%	36.4%	25.8%	26.2%	6.7%	0.0%	0.0%	0.4%	0.2%	0.0%	2.4%	6.2%	0.8%	2.1%	0.0%	0.0%
Mid-Coast	Total	NA[b]	NA	NA	NA	NA	NA	NA	NA	NA	NA	8,179	246	1,552	561	4,007	1,610	2,428
Dependents	Hatchery	—	—	—	—	—	—	—	—	—	—	0	4	84	14	97	0	58
	% hatchery	—	—	—	—	—	—	—	—	—	—	0.0%	1.6%	5.4%	2.5%	2.4%	0.0%	2.4%

Table 6 continued. Hatchery influence at coho population, stratum, and ESU scales, 1994–2010. Hatchery influence expressed as percent of total spawning escapement. Table from ODFW 2009a.[a] EL = extremely low, L = low, and M = medium.

ESU/strata Population	Return year	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
	Marine survival	**EL**	**EL**	**EL**	**EL**	**EL**	**L**	**L**	**M**	**L**	**M**	**M**	**L**	**L**	**M**	**EL**	**M**	**L**
Umpqua	**Total**	**9,639**	**22,423**	**32,758**	**5,126**	**14,639**	**11,004**	**22,787**	**52,842**	**52,036**	**43,702**	**37,160**	**51,845**	**27,647**	**15,630**	**38,302**	**60,428**	**62,735**
stratum	**Hatchery**	**2,735**	**2,311**	**11,578**	**1,792**	**4,888**	**2,428**	**8,193**	**17,758**	**8,532**	**8,919**	**7,240**	**9,313**	**9,555**	**3,847**	**434**	**2,444**	**3,114**
	% hatchery	**28.4%**	**10.3%**	**35.3%**	**35.0%**	**33.4%**	**22.1%**	**36.0%**	**33.6%**	**16.4%**	**20.4%**	**19.5%**	**18.0%**	**34.6%**	**24.6%**	**1.1%**	**4.0%**	**5.0%**
Lower	Total	2,918	10,854	8,435	1,445	4,552	2,708	5,896	12,872	19,787	16,529	9,053	19,014	9,478	4,661	9,332	20,026	14,819
Umpqua River	Hatchery	156	0	450	188	0	85	115	1,233	906	35	64	423	1,484	424	309	781	90
	% hatchery	5.3%	0.0%	5.3%	13.0%	0.0%	3.1%	2.0%	9.6%	4.6%	0.2%	0.7%	2.2%	15.7%	9.1%	3.3%	3.9%	0.6%
Middle	Total	2,309	3,250	5,431	601	1,336	1,914	4,719	9,817	11,669	11,090	6,433	8,203	6,111	1,763	4,472	15,075	11,649
Umpqua River	Hatchery	147	0	345	38	79	166	164	877	931	0	58	595	1,259	176	0	0	0
	% Hatchery	6.4%	0.0%	6.4%	6.3%	5.9%	8.7%	3.5%	8.9%	8.0%	0.0%	0.9%	7.3%	20.6%	10.0%	0.0%	0.0%	0.0%
North Umpqua	Total	1,889	3,049	4,812	1,956	4,144	3,173	9,262	16,728	10,063	11,746	10,265	10,264	9,662	3,975	3,563	8,185	10,124
River	Hatchery	990	1,756	3,743	1,379	3,379	1,979	7,585	14,094	6,695	8,884	6,706	8,295	6,662	2,565	125	465	662
	% hatchery	52.4%	57.6%	77.8%	70.5%	81.5%	62.4%	81.9%	84.3%	66.5%	75.6%	65.3%	80.8%	69.0%	64.5%	3.5%	5.7%	6.5%
South Umpqua	Total	2,523	5,270	14,080	1,124	4,607	3,209	2,910	13,425	10,517	4,337	11,409	14,364	2,396	5,231	20,935	17,142	26,143
River	Hatchery	1,442	555	7,040	187	1,430	198	329	1,554	0	0	412	0	150	682	0	1,198	2,362
	% hatchery	57.2%	10.5%	50.0%	16.6%	31.0%	6.2%	11.3%	11.6%	0.0%	0.0%	3.6%	0.0%	6.3%	13.0%	0.0%	7.0%	9.0%
Lakes	**Total**	**5,842**	**11,216**	**13,494**	**8,603**	**11,108**	**12,711**	**12,747**	**19,669**	**22,097**	**16,091**	**18,642**	**14,725**	**24,127**	**8,955**	**23,608**	**17,349**	**38,859**
stratum	**Hatchery**	**130**	**132**	**68**	**16**	**0**	**168**	**0**	**65**	**120**	**15**	**45**	**0**	**251**	**0**	**0**	**0**	**0**
	% hatchery	**2.2%**	**1.2%**	**0.5%**	**0.2%**	**0.0%**	**1.3%**	**0.0%**	**0.3%**	**0.5%**	**0.1%**	**0.2%**	**0.0%**	**1.0%**	**0.0%**	**0.0%**	**0.0%**	**0.0%**
Siltcoos	Total	1,426	4,497	4,775	2,653	3,122	2,819	3,835	5,104	4,749	6,628	7,998	4,364	5,452	1,447	3,873	5,197	7,843
	Hatchery	124	82	68	0	0	63	0	0	113	0	27	0	21	0	0	0	0
	% hatchery	8.7%	1.8%	1.4%	0.0%	0.0%	2.2%	0.0%	0.0%	2.4%	0.0%	0.3%	0.0%	0.4%	0.0%	0.0%	0.0%	0.0%
Tahkenitch	Total	1,062	1,627	1,627	1,858	2,817	3,769	634	3,526	3,487	3,203	3,496	1,897	3,611	3,551	2,604	2,977	10,631
	Hatchery	6	50	0	16	0	105	0	16	7	15	0	0	107	0	0	0	0
	% hatchery	0.6%	3.1%	0.0%	0.9%	0.0%	2.8%	0.0%	0.5%	0.2%	0.5%	0.0%	0.0%	3.0%	0.0%	0.0%	0.0%	0.0%
Tenmile	Total	3,354	5,092	7,092	4,092	5,169	6,123	8,278	11,039	13,861	6,260	7,148	8,464	15,064	3,957	17,131	9,175	20,385
	Hatchery	0	0	0	0	0	0	0	49	0	0	18	0	123	0	0	0	0
	% hatchery	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.4%	0.0%	0.0%	0.3%	0.0%	0.8%	0.0%	0.0%	0.0%	0.0%
Mid-South	**Total**	**23,479**	**14,398**	**30,136**	**7,528**	**6,963**	**8,276**	**12,582**	**55,953**	**44,803**	**49,647**	**53,545**	**29,730**	**41,359**	**15,747**	**24,528**	**53,319**	**49,809**
Coast stratum	**Hatchery**	**969**	**634**	**481**	**71**	**129**	**65**	**12**	**2,766**	**640**	**445**	**221**	**265**	**118**	**13**	**27**	**487**	**194**
	% hatchery	**4.1%**	**4.4%**	**1.6%**	**0.9%**	**1.9%**	**0.8%**	**0.1%**	**4.9%**	**1.4%**	**0.9%**	**0.4%**	**0.9%**	**0.3%**	**0.1%**	**0.1%**	**0.9%**	**0.4%**
Coos River	Total	15,207	10,447	12,128	1,127	2,985	4,818	4,704	34,259	33,265	25,950	23,450	17,305	11,266	1,342	14,881	27,216	27,228
	Hatchery	707	145	0	15	0	0	0	664	145	189	113	257	0	13	0	237	194
	% hatchery	4.6%	1.4%	0.0%	1.3%	0.0%	0.0%	0.0%	1.9%	0.4%	0.7%	0.5%	1.5%	0.0%	1.0%	0.0%	0.9%	0.7%
Coquille River	Total	5,119	2,116	16,169	5,720	2,412	2,667	6,253	15,665	7,866	22,565	22,182	11,806	28,577	13,968	8,791	22,513	16,374
	Hatchery	0	82	355	0	0	0	0	1,832	190	162	44	0	0	0	0	227	0
	% hatchery	0.0%	3.9%	2.2%	0.0%	0.0%	0.0%	0.0%	11.7%	2.4%	0.7%	0.2%	0.0%	0.0%	0.0%	0.0%	1.0%	0.0%
Floras Cr.	Total	2,893	1,751	1,628	525	958	730	1,477	5,752	3,568	1,038	7,446	506	1,214	340	803	3,203	5,629
	Hatchery	240	400	109	43	79	60	0	88	296	86	0	0	110	0	17	0	0
	% hatchery	8.3%	22.8%	6.7%	8.2%	8.2%	8.2%	0.0%	1.5%	8.3%	8.3%	0.0%	0.0%	9.1%	0.0%	2.1%	0.0%	0.0%
Sixes River	Total	260	84	211	156	608	61	148	277	104	94	467	113	302	97	53	190	104
	Hatchery	22	7	17	13	50	5	12	182	9	8	64	8	8	0	10	14	0
	% hatchery	8.5%	8.3%	8.1%	8.3%	8.2%	8.2%	8.1%	65.7%	8.7%	8.5%	13.7%	7.1%	2.6%	0.0%	18.9%	7.4%	0.0%
Mid-South	Total	NA[b]	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0	197	474
Dependent	Hatchery	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0	9	0
	% hatchery	—	—	—	—	—	—	—	—	—	—	—	—	—	—	NA	4.6%	0.0%

[a]Updated by K. Moore, ODFW, Corvallis, OR. Pers. commun., January 2011.
[b]NA = not available.

large, warm system with degraded habitat. Spawner distribution appears to be seriously restricted in this population and it is probably the most vulnerable of any population in this ESU to increased temperatures.

The TRT's biological recovery criteria (Wainwright et al. 2008) describe a DSS to help evaluate the status of the ESU. One component of this DSS explicitly addresses diversity. Three of the diversity criteria are sensitive to abundance at the population (PD-1) and watershed (PD-3, PD-4) levels (see TRT Biological Recovery Criteria Analysis and Results subsection). The DSS is structured to provide high scores when spawner escapements are high and uniformly distributed among populations and strata. In addition, higher scores are generated when juveniles and spawners are widely distributed within each population.

Population and Life History Diversity

In the spatially and temporally varying environment inhabited by OCCS, diversity is important for species and population sustainability. Diversity allows OCCS to use a wider array of environments than they could without it (see reviews in Groot and Margolis 1991) and protects against short-term spatial and temporal changes in the environment. Genetic diversity provides the raw material for surviving long-term environmental changes; fish with different characteristics have different likelihoods of persisting, depending on local environmental conditions. The more diverse a population, the more likely it is that some individuals would survive and reproduce in the face of environmental variation (McElhany et al. 2000). As we see in this assessment of current status and future threats, OCCS as a species, population, or as individuals, regularly face changes in their freshwater, estuarine, and ocean environments due to natural and human causes.

Compared to other species of Pacific salmon, coho salmon exhibit moderate levels of life history diversity (reviewed by Groot and Margolis 1991, Weitkamp et al. 1995). Coho salmon also tend to have somewhat lower levels of genetic diversity among populations compared to other Pacific salmon species, particularly in the central portion of their range (reviewed by Weitkamp et al. 1995 and Ford et al. 2004). These patterns have been interpreted as evidence for relatively high levels of gene flow among coho salmon populations (Ford et al. 2004, Johnson and Banks 2008). In the extremes of their range in California and Alaska, coho salmon tend to have higher levels of population differentiation than in their central range, probably due to genetic drift in relatively small populations (Olsen et al. 2003, Bucklin et al. 2007). Even within the OCCS ESU, the geographically central populations had higher levels of genetic diversity compared to populations at the northern and southern boundaries of the ESU, apparently due to overlapping migration patterns from strong northern and southern populations (Johnson and Banks 2008).

Most studies of molecular genetic variation within and among Oregon coho salmon populations utilized either allozymes (reviewed by Weitkamp et al. 1995) or microsatellite loci (e.g., Ford et al. 2004, Johnson and Banks 2008). These types of genetic markers (particularly microsatellites) are typically interpreted under an assumption that the dominant forces driving their evolution are mutation, drift, and migration. In contrast, Johnson and Banks (2008) examined patterns of variation at several genes that play a role in olfaction, and are therefore potential candidates for evolution by natural selection. Their study focused on several of the

Lakes populations and the lower Umpqua River. The authors found notably higher levels of population differentiation at one of the olfactory loci than was observed at microsatellite loci, suggesting that natural selection is increasing patterns of genetic differentiation among these populations at some specific genes. This result suggests that despite the relatively high rates of genetic flow among OCCS populations, there is potential for local adaptation among these populations.

Within the OCCS ESU, there is substantial genetic and geographic variation and structure, with genetic similarities clustering into a few geographic units. In the workgroup analyses, they designated these clusters as biogeographic strata that represent genetic and geographic similarities, as well as identifiable diversity among them (Lawson et al. 2007). Biological recovery criteria (Wainwright et al. 2008) were based on the principle that preserving sustainable populations in each of these biogeographic strata would conserve major genetic diversity in the ESU, as well as spread risks to the maintenance of genetic and geographic diversity due to catastrophes.

Providing access to a diversity of productive habitat types allows expression of phenotypes that may not otherwise occur. Although it is unknown which particular life history type will be successful in a given year, the expectation is for some types to be more successful than others. Consequently, by ensuring that a wide range of productive habitats are accessible to coho salmon, the opportunity is provided for greater expression of life history diversity, which in turn should increase the chances that at least some coho salmon life history types will be successful (Wainwright et al. 2008). In a more recent analysis, Greene et al. (2010) "strongly suggest that life history diversity can increase production and buffer population fluctuations, particularly over long time periods. Our findings provide new insights into the importance of biocomplexity beyond spatio-temporal aspects of populations, and suggest that maintaining diverse life history portfolios of populations may be crucial for their resilience to unfavorable conditions such as habitat loss and climate change."

As an example, extensive loss of access to habitats in estuaries and tidal freshwater may have been an important factor in reducing population diversity in OCCS. The Oregon coastal drainages supporting independent OCCS populations all terminate in tidally influenced freshwater/brackish/saltwater wetland and estuarine habitats (e.g., Good 2000). Recent sampling in coastal rivers from northern California to Alaska indicates that coho salmon juveniles are often present in these lower river/tidal freshwater/estuarine habitats (e.g., Koski 2009). Migrant trapping studies indicate that a substantial number of coho salmon fry emigrate downstream from natal streams into tidal reaches and therefore use lower river wetland/estuarine habitats (e.g., Chapman 1962, Miller and Sadro 2003, Koski 2009). In the past, observations of spring or early summer downstream migration of coho salmon fry were thought to represent a passive displacement in response to increased stream flows, competitive interactions, or capacity limitations. However, in recent studies summarized by Koski (2009) there is evidence that downstream migrations of coho salmon may be associated with specific life history strategies that contribute to resiliency in the face of fluctuating environmental conditions.

More recently, working in the Coos River, Bass (2010) reported that "widespread estuarine wetland losses have likely reduced the rearing capacity of coastal basins and decreased resiliency by diminishing the expression of subyearling migrant life histories within and among coho salmon populations." Bass (2010) also reported nine coho salmon jacks PIT (passive integrated transponder) tagged as subyearling reservoir residents in spring 2008 and detected in the fall of 2009. For the 2007 broodyear, there were 33 returns from 1,191 coho tagged as presmolts (age 0) and 38 returns from 742 fish tagged as smolts.[9] This suggests that these "nomad" early downstream migrants can survive and contribute to the spawning population. Bass (2010) concludes: "By affecting coho salmon nomads to a greater extent than other life history types, estuary loss may reduce the potential productivity boost and resilience component that this life history may contribute to its natal and neighboring populations."

In the recent past, the effect of hatchery releases had a significant effect on life history diversity in the OCCS ESU. ODFW has significantly reduced hatchery releases of coho salmon, therefore the effect of hatchery fish on native population diversity should be abating, although there is little information about the duration of hatchery genetic effects on naturally spawning populations. Because of the significant reduction in hatchery releases of coho, the hatchery fraction of spawners observed on the spawning grounds has been substantially reduced (ODFW 2009b). This should lead to improvement of diversity in naturally produced OCCS in those populations once dominated by hatchery fish.

Since 1990 there have been years with extremely low escapements in some systems and many small systems have shown local extirpations, presumably reducing diversity due to loss of dependent populations. For example, Cummins Creek on the central coast had no spawners observed in 1998, indicating the potential loss of a brood cycle. These small systems are apt to be repopulated by stray spawners, most likely from larger adjacent populations during periods of higher abundance (Lawson et al. 2007), and recent local extirpations may represent loss of genetic diversity in the context of normal metapopulation function.

Current status of diversity shows improvement through the waning effects of hatchery fish on populations of OCCS. In addition, recent efforts in several coastal estuaries to restore lost wetlands should be beneficial. However, the loss of diversity brought about by legacy effects of freshwater and tidal habitat loss, coupled with the restriction of diversity from very low returns over the past 20 years, led us to conclude that diversity is lower than it was historically.

Harvest Impacts

Historical harvest rates on OPI area coho salmon were in the range of 60% to 90% from the 1960s into the 1980s (Table 3). Modest harvest reductions were achieved in the late 1980s, but rates remained high until a crisis was perceived and most directed coho salmon harvest was prohibited in 1994. Subsequent fisheries have been severely restricted (ODFW 2005c, 2009a) and most reported mortalities are estimates of indirect (noncatch) mortality in Chinook salmon (*Oncorhynchus tshawytscha*) fisheries and selective fisheries for marked (hatchery) coho salmon. Estimates of these indirect mortalities are somewhat speculative and there is a risk of underestimation (Lawson and Sampson 1996, PFMC 2009).

[9] K. Nordholm, Oregon State University, Dept. Fisheries and Wildlife, Corvallis, OR. Pers. commun., February 2011.

Amendment 13

PFMC (1998) adopted Amendment 13 to its Salmon Fishery Management Plan in 1998. This amendment was developed as part of the Oregon Plan for Salmon and Watersheds. It specified an exploitation rate harvest management regime with rates for OCN dependent on marine survival (as indexed by hatchery jack:smolt ratios) and parental and grandparental spawning escapements. Exploitation rates ranged from 13% to a maximum of 35%. Amendment 13 was reviewed in 2000 and the harvest rate matrix was modified to include an 8% category under conditions of extremely poor marine survival, as was observed in the late 1990s. At the same time, the maximum exploitation rate was increased to 45% and the grandparental escapement criterion was dropped. Exploitation rates were calculated to allow a doubling of spawners under conditions of moderate-to-good ocean survival.

Risk assessment was conducted for Amendment 13 (PFMC 1998) and the 2000 Amendment 13 Review (PFMC 2000) using the Nickelson/Lawson coho salmon habitat-based life cycle model (Nickelson and Lawson 1998). The models were augmented to include a management strategy evaluation that simulated the fishery management process, including errors in spawner assessment, prediction, and harvest management. In general, exploitation rate management with a 35% cap showed a lower risk of pseudoextinction than managing for an escapement goal of 200,000 spawners, but higher risk than a zero harvest scenario. Starting from the very low escapements of 1994, basins on the north coast had higher extinction risks than those on the Mid-North and Mid-South coasts.

Mark-selective fisheries

Beginning in 1998, most adult hatchery-origin coho salmon in the OPI area were marked with an adipose fin clip. This marking allowed the implementation of mark-selective fisheries, with legal retention only of marked fish. Unmarked fish were to be released unharmed. Recreational mark-selective fisheries have been conducted on the Oregon coast in each year since 1998, with quotas ranging from 9,000 to 88,000 marked fish.

Commercial troll fisheries targeting Chinook salmon were also operating. In 2007 a mark-selective commercial troll fishery targeting hatchery coho salmon was implemented with a quota of 10,000, and a similar fishery was implemented in 2008 with a quota of 21,240. Actual catch in these fisheries was about half of the quota in each year. A concern with these fisheries is the high ratio of unmarked to marked fish that was encountered.

The mark-selective coho and commercial troll Chinook salmon fisheries catch and release coho salmon, resulting in incidental mortalities. In addition some coho, so-called drop-offs, encounter the gear but escape or are eaten by predators. Estimates of noncatch mortalities from hook and release and drop-offs are difficult because they are, by their nature, unobserved. Field studies in the 1990s (NRC 1997) and a literature review and meta-analysis resulted in the adoption by the PFMC of hooking mortality rates of 13% for recreational fisheries and 26% for commercial fisheries. In addition, drop-off mortalities were assumed to equal 5% of the number of fish brought to the boat. These rates are used by the PFMC for a coho Fisheries Resource Allocation Model to estimate mortalities in council-managed fisheries. Postseason estimates of OCN exploitation rates based on that modeling have ranged from 0.07 to 0.15 since the curtailment of directed coho salmon fishing in 1994. The BRT considers that these rates may be underestimates and that actual mortalities may have been greater (Lawson and Sampson 1996).

Freshwater fisheries

A few small freshwater fisheries have been allowed in recent years, based on the provision in Amendment 13 that terminal fisheries can be allowed on strong populations as long as the overall exploitation rate for the ESU does not exceed the Amendment 13 allowable rate and population escapement is not reduced below full seeding of the best available habitat. The difference between these fisheries and the mark-selective fisheries in the ocean is that the freshwater fisheries are directed take on a listed species. NMFS has approved these fisheries with the condition that the methodologies used by ODFW to predict population abundances and estimate full seeding levels are presented to the PFMC for review and approval.

Despite these uncertainties, there is no doubt that harvest-related mortalities have been reduced substantially since harvest was curtailed in 1994. This reduction is reflected in positive short-term trends in spawner escapements (Figure 5 and Figure 6). Harvest management has succeeded in maintaining spawner abundance in the face of a continuing downward trend in productivity of these stocks. Further harvest reductions can have little effect on spawning escapements. Future remedies must be found outside of harvest management until the decline in productivity is reversed (Lawson 1993).

Artificial Propagation

As of 2009, there are only three coho hatchery stocks in propagation within the OCCS ESU. All other hatchery programs have been terminated. (For more discussion, see Artificial Propagation−Membership in the ESU subsection).

In previous OCCS status reviews, coho salmon hatchery programs were a major concern throughout the ESU. High numbers of hatchery coho that could not be differentiated from naturally produced fish were released in most populations, hatchery broodstocks were intermixed among stocks of different populations, multiple life stages of juvenile hatchery coho were stocked into wild production areas, and hatchery origin (stray) coho were common in natural spawning areas throughout the ESU. However, since the early 1990s the State of Oregon has reformed hatchery practices due to a variety of genetic, ecological, and economic factors. This has lessened the risks of hatchery programs to wild coho populations in the ESU. These management changes have been described in detail in previous BRT assessments (i.e., Good et al. 2005) and are summarized below.

Releases of hatchery coho salmon in the ESU have declined from a peak of approximately 35 million fish in 1981 to approximately 260,000 smolts in 2009 (Figure 9), (Oregon 2005, ODFW 2009a, 2009b). In the early 1990s, hatchery coho were released in 17 of 19 ESU populations. In 2009 hatchery coho salmon were released in 3 of 19 ESU populations (Nehalem, Trask, and South Umpqua). In the early 1990s, ODFW managed 16 different brood stocks throughout the ESU. In 2009 there were only three brood stocks still in propagation (ODFW 2009b).

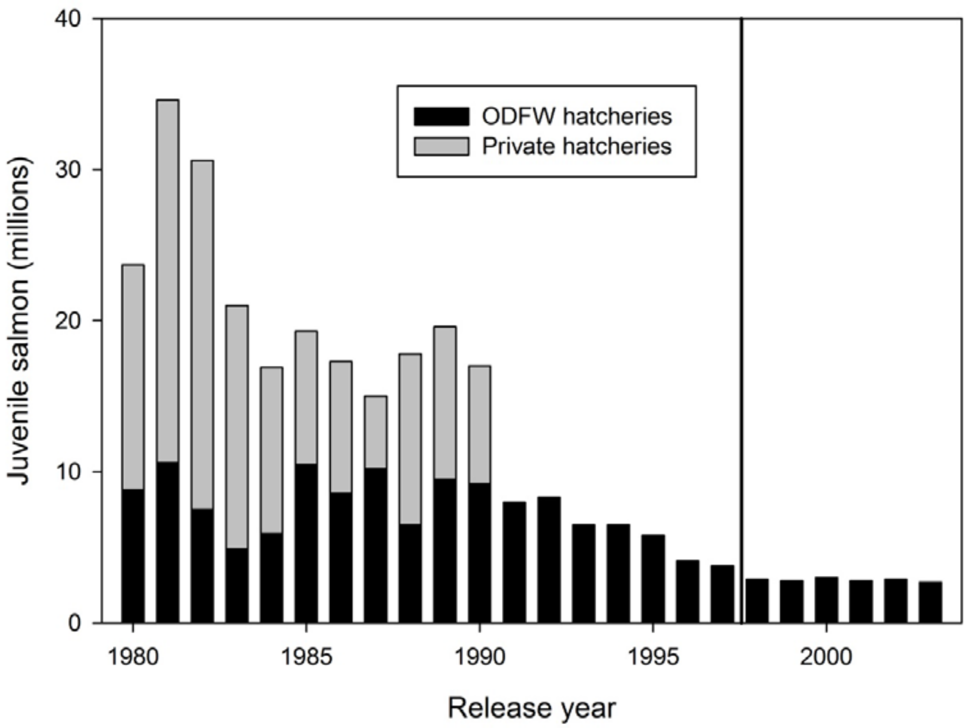

Figure 9. Releases of juvenile hatchery coho (all age classes) in the OCCS ESU by hatchery type from 1980 to 2003. The 1980 through 1984 release years are missing unfed fry release data. Vertical line marks the beginning of the Oregon Plan. Further reductions occurred from 2004 to 2009, as described in text.

Since 1997 all hatchery coho released have been adipose fin clipped in order to differentiate between hatchery-origin and natural-origin fish in mark-selective fisheries and evaluate straying of hatchery fish into natural spawning areas. External marking of all hatchery fish helped to resolve uncertainties about the magnitude of these interactions on the spawning grounds that had previously been assessed by evaluating fish scale data. In the 1990s many populations had proportions of hatchery fish in the natural spawning populations in excess of 40%, with the north coast populations having the highest proportion of hatchery spawners (Figure 10, Table 6).

By the early 2000s, stray rates had decreased in most populations due to the elimination of hatchery programs and reductions in the number of fish released. Most populations are now below Oregon's stray rate standard of no more than 10% hatchery coho on the spawning grounds (Oregon 2007). The notable exceptions are the Salmon River and North Umpqua River populations, where stray rates were still greater than 50%. However, in broodyear 2006, the Salmon and North Umpqua hatchery coho programs were eliminated entirely in order to decrease the straying problems. The percentage of hatchery fish spawning naturally in these two populations showed a substantial decrease beginning in the spawning season of 2009–2010.

Since the previous status review assessments in 1997 and 2003, new information and analyses are available that help inform the BRT of the potential risks associated with hatchery programs and the conservation of the ESU. Interactions between hatchery and wild fish are

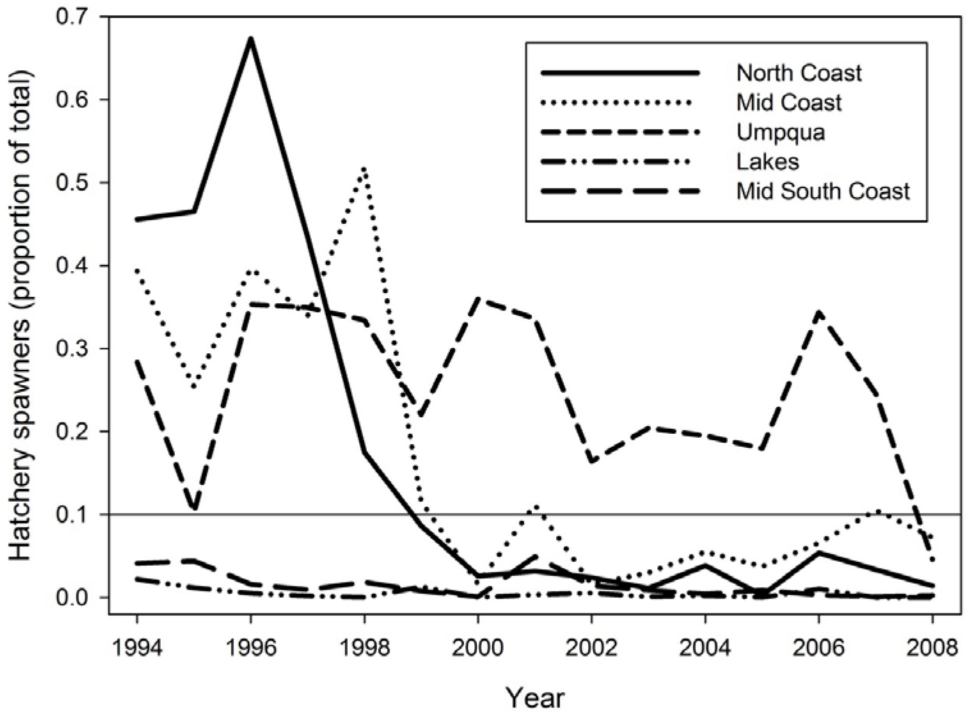

Figure 10. Proportion of hatchery origin coho salmon in each stratum of the OCCS ESU, 1994–2008. Data from ODFW 2009a.

generally considered to have negative outcomes for the wild fish. A large body of literature documents reduced spawning success, freshwater survival, and production of wild fish when hatchery fish are present (NRC 1997, Flagg and Nash 1999, Flagg et al. 2000, ISG 2000, Einum and Fleming 2001, IMST 2001, Chilcote 2003, Hoekstra et al. 2007, Araki et al. 2008, Naish et al. 2008). Analyses of the specific effects of hatchery coho salmon on wild coho salmon on the Oregon coast have all concluded the existing hatchery programs were detrimental to the survival and productivity of this ESU (Nickelson 2003, Oosterhout et al. 2005, Buhle et al. 2009). The recent management changes by the State of Oregon are therefore expected to largely alleviate the detrimental effects of hatchery programs on wild coho salmon.

Overall, the reduction in hatchery activity is expected to benefit wild runs throughout the ESU. For example, Buhle et al. (2009) used data on natural spawning abundance, hatchery releases, and the proportion of hatchery fish in spawning populations to fit a model that allowed estimation of the impacts of hatchery releases on natural OCCS productivity. Their model found a significant negative effect of hatchery releases and naturally spawning hatchery fish; they estimated the reductions in hatchery production since the mid-1990s accounted for approximately 27% of the increase in wild OCCS seen in the 1997 to 2000 brood cycles. These results indicate that at least some of the benefits from reduced hatchery production have already been observed in the recent abundance trends. To the degree that past hatchery practices led to genetic deterioration of wild salmon stocks, additional benefits from the reduced levels of hatchery production may continue to accrue in the future as these populations adapt back to wild conditions. In addition, the two populations that have only recently seen reductions in hatchery releases (North Umpqua and Salmon rivers) may also experience nearer term gains in productivity due to the recent elimination of hatchery coho salmon released in those watersheds.

TRT Biological Recovery Criteria Analysis and Results

The biological recovery criteria developed by the TRT (Wainwright et al. 2008) are framed within the context of a DSS. At the highest level, the DSS is structured to represent the hierarchical population structure of the ESU. Populations are grouped into biogeographic strata, which in combination make up the ESU. The DSS framework is organized into two categories, persistence and sustainability, which imply different levels of risk.

The persistence analysis assesses the ability of the ESU to persist (i.e., not go extinct) over a 100-year period without artificial support. This includes the ability to survive prolonged periods of adverse environmental conditions that may be expected to occur at least once during the 100-year time frame. This analysis has three population-level criteria that measure population productivity, probability of persistence, and abundance relative to critically low thresholds.

The sustainability analysis assesses the ability of the ESU to maintain its genetic legacy and long-term adaptive potential for the foreseeable future. Sustainability implies stability of habitat availability and other conditions necessary for the full expression of the population's (or ESU's) life history diversity into the foreseeable future. The criteria within the DSS (Table 7, Figure 10) are used to evaluate population diversity using objective measures of spawner abundance, artificial influence, spawner and juvenile distribution, and habitat capacity. In addition, ESU-level diversity that includes genetic diversity (a function of genetic structure, effects of selection, effects of migration, and effects of introgression), phenotypic and habitat diversity, and small populations was evaluated. The BRT then used recent observations of these population metrics to inform its assessment of risk to the OCCS ESU.

In practice, application of the DSS began with evaluating a number of primary biological criteria that are defined in terms of logical (true/false) statements about biological processes essential to the persistence or sustainability of the ESU. Evaluating these primary criteria with respect to available observations results in a truth value in the range from –1 (false) to +1 (true). Intermediate values between these extremes reflect the degree of certainty of the statement, given available knowledge, with a value of 0 indicating complete uncertainty about whether the statement is true or false. These primary criteria are then combined logically with other criteria at the same geographic scale, then combined across geographic scales to result in an evaluation of ESU-wide criteria. Thus the end result is an evaluation of the biological status of the ESU as a whole, with an indication of the degree of certainty of that evaluation.

Metrics for the DSS are derived from data provided by ODFW[10] from various survey and monitoring studies. Data include: spawner survey data (peak counts and area under the curve [AUC] estimates), estimates of wild and hatchery fish on the spawning grounds, distribution of spawners and summer juveniles, and estimates of habitat capacity. These contribute to a set of objectively measurable criteria. ESU-level diversity was more difficult to evaluate because

[10] K. Moore, ODFW, Salem, OR. Pers. commun., December 2010.

Table 7. Biological recovery criteria definitions (Wainwright et al. 2008).

ESU-level criteria

EP. ESU persistence: ESU will persist over the next 100 years. All biogeographic strata are persistent (see stratum persistence, SP).

ES. ESU sustainability: ESU is self-sustaining into the foreseeable future.

- **ES-1.** All strata sustainable: All biogeographic strata are sustainable (see stratum sustainability, SS).
- **ES-2.** ESU-level diversity: ESU has sufficient broadscale diversity to maintain its ecological and evolutionary functions into the foreseeable future.
- **ED-1.** Genetic diversity: ESU-level genetic diversity is sufficient for long-term sustainability of the ESU.
  - **ED-1a.** Genetic structure: Genetic diversity within the ESU is comparable to healthy coho salmon ESUs and forms the basis for life-history diversity.
  - **ED-1b**. Effects of selection: Human-driven selection is not sufficient to decrease genetic diversity.
  - **ED-1c.** Effects of migration: Genetic diversity is not compromised by changes in movements of fish.
- **ED-2.** Phenotypic and habitat diversity: ESU-level phenotypic and habitat diversity are sufficient for long-term sustainability of the ESU.
  - **ED-2a.** Phenotypic diversity: Phenotypic diversity is present within the ESU at levels comparable to healthy ESUs or the historical template.
  - **ED-2b.** Habitat diversity: Habitats are sufficiently productive, diverse, and accessible to promote phenotypic plasticity.
- **ED-3.** Small populations: Dependent populations within the ESU are not permanently lost.

Biogeographic stratum–level criteria

SP. Stratum persistence: Most of the historically independent populations in the stratum are persistent (see population persistence, PP).

SS. Stratum sustainability: Stratum is self-sustaining (re: diversity and functionality) into the foreseeable future.

SD. Stratum diversity: Most of the historically independent populations in the stratum are at present sustainable (see population sustainability, PS).

SF. Stratum functionality: All of the historically independent populations in the stratum are functional (see population functionality, PF).

Population-level criteria

PS. Population sustainability: Population is able to sustain itself into the future. Requires both population persistence (PP) and population diversity (PD).

PP. Population persistence: Population will persist for the next 100 years.

- **PP-1.** Population productivity: Productivity at low abundance is sufficient to sustain the population through an extended period of adverse environmental conditions.
- **PP-2.** Probability of persistence: Population has a high likelihood of persisting over the next 100 years, as estimated from PVA models.
- **PP-3.** Critical abundance: Population abundance is maintained above levels where small-population demographic risks are likely to become significant.

PD. Population diversity: Population has sufficient diversity and distribution to ensure continued fitness in the face of environmental change.

- **PD-1.** Spawner abundance: Population has sufficient naturally produced spawners to prevent loss of genetic variation due to random processes over a 100-year time frame.
- **PD-2.** Artificial influence: Abundance of naturally spawning hatchery fish will not be so high as to be expected to have adverse effects on natural populations.
- **PD-3.** Spawner distribution: On average, the historically occupied watersheds in the population’s range have spawners occupying the available spawning habitat (see watershed spawner occupancy, W-Sp).
- **PD-4.** Juvenile distribution: On average, the historically occupied watersheds in the population’s range have juveniles occupying the available juvenile habitat (see watershed juvenile occupancy, W-Ju).

PF. Population functionality: Habitat quality and quantity are adequate to support sufficient abundance to maintain long-term genetic integrity of the population.

Watershed-level criteria

W-Sp. Watershed spawner occupancy: Spawners occupy a high proportion of the available spawning habitat within the watershed.

W-Ju. Watershed juvenile occupancy: Juveniles occupy a high proportion of the available juvenile habitat within the watershed.

objectively measurable criteria were not available, so scores were produced using a formal process by an expert panel (Wainwright et al. 2008). ESU-level diversity was not reevaluated for this report.

The DSS was run with data through the 2009 spawning run. ODFW provided the data used in the update. In the process of compiling data for the 4 years since the TRT analysis, several inconsistencies were discovered and reconciled. For this reason, the DSS results reported in Figure 11, Figure 12, and Table 8 are not directly comparable to the results presented in Wainwright et al. (2008). Table 9 is presented for historical comparison but was not used by the BRT in its deliberations. Table 10 summarizes VSP attributes (McElhany et al. 2000) related to DSS results.

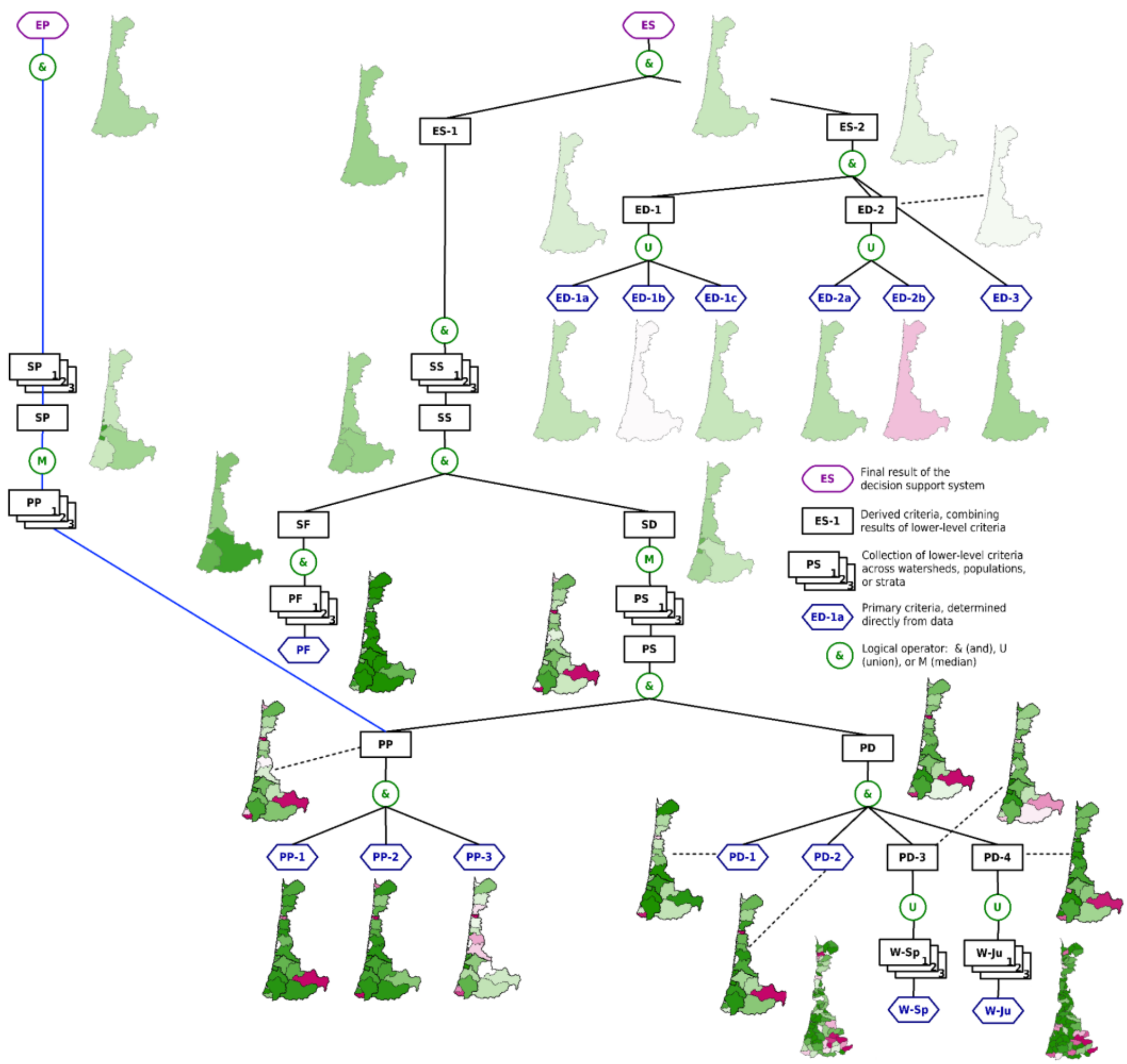

Figure 11. Decision tree for the biological recovery criteria. Flow lines show the logical connections from the primary criteria (blue hexagons) through intermediate levels (black rectangles) to ESU-wide evaluations (magenta ovals) for ESU persistence (EP) and sustainability (ES). Colored maps show results at each geographic scale, with dark red indicating poor conditions and dark green indicating good conditions Criteria abbreviations are as in Table 7.

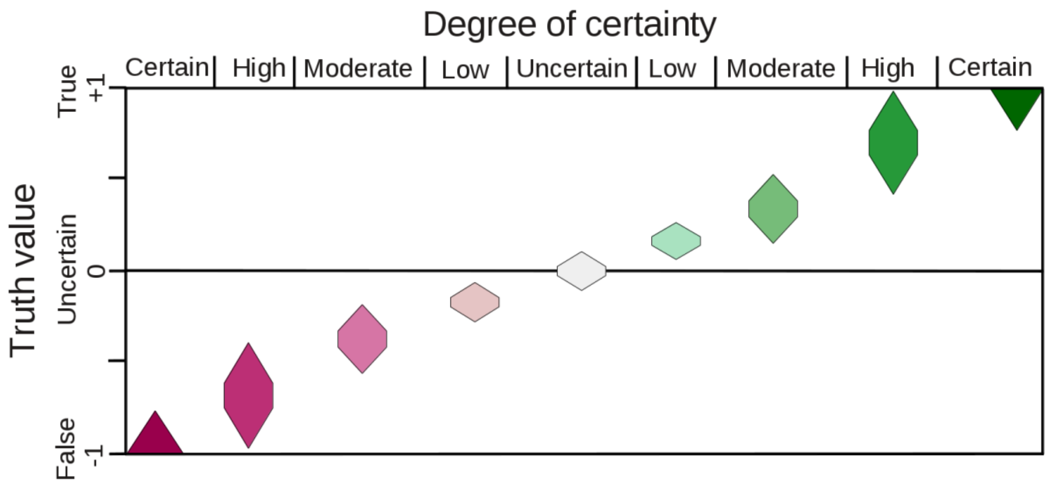

Figure 12. Truth value colors showing degree of certainty interpretation for colors found in Figure 11. (Reprinted from Wainwright et al. 2008.)

Two criteria were not updated. Persistence probability, PP-2, based on four population viability analysis (PVA) models, was not updated because sensitivity analysis presented in Wainwright et al. (2008) showed that DSS results were not very sensitive to small changes in individual model results. The main utility of the PVA model runs is to evaluate relative vulnerabilities of the populations. These relative vulnerabilities are unlikely to change with the addition of a few more years of data. The population functionality criterion, PF-1, based on habitat quantity, was not updated because it would have required a major analysis of recent habitat data. The BRT considered that this criterion was not sensitive to small changes in habitat conditions and was also not particularly informative. Habitat issues were addressed with more rigor in new analyses outside the structure of the DSS. A 10-year time series of habitat survey data was analyzed for evidence of trends in habitat quality, providing a much more informative metric than the habitat quantity measure currently used for PF-1 (see the habitat complexity discussion). In the future it may be possible to incorporate a habitat trend index in the DSS.

The critical abundance criterion, PP-3, in Wainwright et al. (2008), was discovered to have been evaluated by the TRT using the wrong data set. It was originally calculated using AUC spawner data rather than peak-count data as specified in the criterion. The updated critical abundance values are based on peak counts (Table 8). AUC estimates are always higher than peak counts because they include fish present on the spawning grounds over a longer period of time. Peak counts are simply the highest number observed at any one time. The object of the criterion was to evaluate the likelihood of depensation due to low spawner numbers. If too few fish are present on the spawning grounds at any one time, then the probability that individuals will be able to locate mates is reduced. This effect, termed depensation, is thought to become a problem at spawner densities below four fish per mile (Wainwright et al. 2008). Therefore, peak counts are more suitable than AUC estimates for evaluating this effect. For comparison with results presented in Wainwright et al. (2008), DSS values based on AUC spawner estimates are also presented (Table 9).

The DSS result for ESU persistence was 0.34 (Table 8). Recall that a value of 1 (Figure 12) would indicate complete confidence that the ESU will persist for the next 100 years, a value

Table 8. Decision support system results by criterion and independent population, stratum, and ESU. Critical abundance data (PP-3) from peak counts. See Figure 12 for color representation of truth value for each stratum.

		EP	**ED-1a**	**ED-1b**	**ED-1c**	**ED-2a**	**ED-2b**	**ED-3**	**ED-1**	**ED-2**	**ES-1**	**ES-2**	**ES**
OCCS ESU		0.34	0.28	–0.01	0.26	0.30	–0.20	0.40	0.18	0.05	0.42	0.13	0.24
Stratum		**SP**	**SD**	**SF**	**SS**								
North Coast		0.27	0.33	0.53	0.39								
Mid-Coast		0.25	0.36	0.47	0.40								
Lakes		0.88	0.64	0.38	0.47								
Umpqua		0.40	0.26	0.88	0.45								
Mid-South Coast		0.24	0.37	0.68	0.48								
Stratum	**Population**	**PP-1**	**PP-2**	**PP-3**	**PD-1**	**PD-2**	**PF-1**	**PD-3**	**PD-4**	**PP**	**PD**	**PS**	**PF**
North Coast	Necanicum River	0.95	–0.44	0.06	0.10	0.35	0.18	0.85	0.96	–0.26	0.36	–0.15	0.18
North Coast	Nehalem River	0.80	0.92	0.51	1.00	0.66	1.00	0.46	0.63	0.69	0.62	0.65	1.00
North Coast	Tillamook Bay	0.90	0.50	0.15	0.35	0.42	1.00	0.35	0.60	0.36	0.40	0.37	1.00
North Coast	Nestucca River	0.82	0.74	–0.08	0.26	0.92	0.89	0.66	0.77	0.18	0.51	0.28	0.89
Mid-Coast	Salmon River	–0.51	–1.00	–1.00	–0.14	–1.00	0.20	–0.14	1.00	–1.00	–1.00	–1.00	0.20
Mid-Coast	Siletz River	0.91	0.66	0.09	0.24	0.67	0.98	0.54	0.91	0.34	0.46	0.38	0.98
Mid-Coast	Yaquina River	0.97	0.76	0.20	0.46	0.69	1.00	0.80	1.00	0.47	0.66	0.54	1.00
Mid-Coast	Beaver Creek	0.97	0.62	0.90	0.08	0.86	0.14	1.00	1.00	0.79	0.44	0.56	0.14
Mid-Coast	Alsea River	0.63	0.96	–0.29	0.51	0.97	1.00	0.47	0.88	–0.03	0.65	0.13	1.00
Mid-Coast	Siuslaw River	0.89	0.98	–0.14	1.00	0.81	1.00	0.65	0.85	0.17	0.80	0.35	1.00
Lakes	Siltcoos River (Lake)	0.81	1.00	0.86	0.47	0.99	0.32	1.00	1.00	0.88	0.76	0.81	0.32
Lakes	Tahkenitch Lake	0.69	0.70	1.00	0.21	0.95	0.32	1.00	1.00	0.78	0.56	0.64	0.32
Lakes	Tenmile Lake	0.96	0.98	1.00	0.85	0.98	0.59	1.00	–0.36	0.98	–0.05	0.20	0.59
Umpqua	Lower Umpqua R.	0.68	0.86	0.48	0.96	0.42	1.00	0.66	0.59	0.63	0.59	0.60	1.00
Umpqua	Middle Umpqua R.	0.73	0.84	0.00	0.32	0.35	1.00	0.27	0.48	0.26	0.33	0.28	1.00
Umpqua	North Umpqua River	–0.96	0.52	0.22	0.97	–0.96	0.67	–0.37	–0.87	–0.95	–0.95	–0.95	0.67
Umpqua	South Umpqua River	0.92	0.94	0.26	0.28	0.50	1.00	–0.05	0.41	0.54	0.11	0.23	1.00
Mid-South Coast	Coos Bay	0.92	0.86	0.33	1.00	0.94	1.00	0.95	1.00	0.58	0.97	0.74	1.00
Mid-South Coast	Coquille River	0.96	0.96	0.72	1.00	0.98	1.00	0.67	0.81	0.86	0.83	0.85	1.00
Mid-South Coast	Floras Creek	0.99	0.92	–0.38	0.18	0.81	0.87	0.27	1.00	–0.11	0.40	0.01	0.87
Mid-South Coast	Sixes River	0.52	–1.00	–0.71	–0.87	0.17	0.38	–0.17	–0.58	–1.00	–0.83	–1.00	0.38

Table 9. Decision support system results based on using area under the curve (AUC) spawner density estimates for the critical abundance (PP-3) criterion for comparison with Wainwright et al. (2008).

		EP	**ED-1a**	**ED-1b**	**ED-1c**	**ED-2a**	**ED-2b**	**ED-3**	**ED-1**	**ED-2**	**ES-1**	**ES-2**	**ES**
OCCS ESU		0.70	0.28	–0.01	0.26	0.30	–0.20	0.40	0.18	0.05	0.53	0.13	0.28
Stratum		**SP**	**SD**	**SF**	**SS**								
North Coast		0.69	0.54	0.53	0.53								
Mid-Coast		0.55	0.54	0.47	0.50								
Lakes		0.92	0.64	0.38	0.47								
Umpqua		0.66	0.36	0.88	0.53								
Mid-South Coast		0.91	0.74	0.68	0.70								
Stratum	**Population**	**PP-1**	**PP-2**	**PP-3**	**PD-1**	**PD-2**	**PF-1**	**PD-3**	**PD-4**	**PP**	**PD**	**PS**	**PF**
North Coast	Necanicum River	0.95	–0.44	0.54	0.10	0.35	0.18	0.85	0.96	–0.22	0.36	–0.11	0.18
North Coast	Nehalem River	0.80	0.92	0.85	1.00	0.66	1.00	0.46	0.63	0.85	0.62	0.72	1.00
North Coast	Tillamook Bay	0.90	0.50	0.80	0.35	0.42	1.00	0.35	0.60	0.68	0.40	0.50	1.00
North Coast	Nestucca River	0.82	0.74	0.60	0.26	0.92	0.89	0.66	0.77	0.70	0.51	0.58	0.89
Mid-Coast	Salmon River	–0.51	–1.00	–1.00	–0.14	–1.00	0.20	–0.14	1.00	–1.00	–1.00	–1.00	0.20
Mid-Coast	Siletz River	0.91	0.66	0.46	0.24	0.67	0.98	0.54	0.91	0.62	0.46	0.51	0.98
Mid-Coast	Yaquina River	0.97	0.76	0.82	0.46	0.69	1.00	0.80	1.00	0.84	0.66	0.74	1.00
Mid-Coast	Beaver Creek	0.97	0.62	1.00	0.08	0.86	0.14	1.00	1.00	0.82	0.44	0.57	0.14
Mid-Coast	Alsea River	0.63	0.96	0.15	0.51	0.97	1.00	0.47	0.88	0.39	0.65	0.48	1.00
Mid-Coast	Siuslaw River	0.89	0.98	0.19	1.00	0.81	1.00	0.65	0.85	0.49	0.80	0.60	1.00
Lakes	Siltcoos River (Lake)	0.81	1.00	1.00	0.47	0.99	0.32	1.00	1.00	0.92	0.76	0.83	0.32
Lakes	Tahkenitch Lake	0.69	0.70	1.00	0.21	0.95	0.32	1.00	1.00	0.78	0.56	0.64	0.32
Lakes	Tenmile Lake	0.96	0.98	1.00	0.85	0.98	0.59	1.00	–0.36	0.98	–0.05	0.20	0.59
Umpqua	Lower Umpqua R.	0.68	0.86	0.85	0.96	0.42	1.00	0.66	0.59	0.78	0.59	0.66	1.00
Umpqua	Middle Umpqua R.	0.73	0.84	0.34	0.32	0.35	1.00	0.27	0.48	0.54	0.33	0.40	1.00
Umpqua	North Umpqua River	–0.96	0.52	0.76	0.97	–0.96	0.67	–0.37	–0.87	–0.94	–0.95	–0.95	0.67
Umpqua	South Umpqua River	0.92	0.94	0.75	0.28	0.50	1.00	-0.05	0.41	0.86	0.11	0.31	1.00
Mid-South Coast	Coos Bay	0.92	0.86	0.94	1.00	0.94	1.00	0.95	1.00	0.90	0.97	0.94	1.00
Mid-South Coast	Coquille River	0.96	0.96	0.93	1.00	0.98	1.00	0.67	0.81	0.95	0.83	0.89	1.00
Mid-South Coast	Floras Creek	0.99	0.92	0.85	0.18	0.81	0.87	0.27	1.00	0.92	0.40	0.58	0.87
Mid-South Coast	Sixes River	0.52	–1.00	–0.39	–0.87	0.17	0.38	–0.17	–0.58	–1.00	–0.83	–1.00	0.38

Table 10. VSP attributes related to the DSS criteria and results.

VSP Factor	DSS	Criterion	Status	Comments
Abundance	(PP-2)[a]	Probability of persistence (multiple PVA models)	Most (18 of 21) truth values > 0.25	
	PP-3	Critical abundance (mean spawner densities in low years)	Many (9 of 21) truth values > 0.25	
	PD-1	Spawner abundance (harmonic mean sufficient to avoid genetic risks)	Many (14 of 21) truth values > 0.25	
Growth rate (productivity)	PP-1	Population productivity (geometric mean of natural return ratio in low years)	Most (19 of 21) truth values > 0.25	
	PP-2	Probability of persistence (multiple PVA models)	Most (18 of 21) truth values > 0.25	
Spatial structure and connectivity	PD-3	Spawner distributions (>4 fish per mile in half of watersheds)	Many (17 of 21) > 0.25	South Coast has most low scores
	(PD-4)[a]	Juvenile distributions (pools with ≥1 fish)	Most (18 of 21) truth values > 0.25	Questions about usefulness of this criterion
	ED-1c[b]	Barriers to migration/connectivity	Low-moderate changes compared to historical template	
Diversity	PD-1	Spawner abundance (harmonic mean)	Most (14 of 21) truth values > 0.25	
	PD-2	Artificial influence (% hatchery fish spawning)	Most (19 of 21) truth values > 0.25	Improved since last assessment
	ED-1[b]	Genetic diversity (genetic structure, effects of selection, barriers to migration)	Low-moderate changes compared to historical template	
	ED-2[b]	Phenotypic and habitat diversity	Moderate changes compared to historical habitat diversity, low-moderate to phenotypic diversity	
	ED-3[b]	Small populations not permanently lost	Moderately certain small populations not lost	
Threats	(PF-1)[a]	Habitat sufficient (habitat model)	Most (18 of 21) truth values > 0.25	Usefulness of criterion questioned

[a]Population criteria (PP-2, PD-4, and PF-1) were not reevaluated in 2009 or 2010.
[b]ESU-level criteria were not reevaluated in 2009 or 2010.

of −1 would indicate complete certainty of failure to persist, and a value of 0 would indicate no certainty of either persistence or extinction. The BRT therefore interpreted a value of 0.34 as indicating a moderate certainty of ESU persistence over the next 100 years. The DSS result for ESU sustainability, ES, was 0.24, indicating a low-to-moderate certainty that the ESU is sustainable for the foreseeable future.

The overall ESU persistence and sustainability scores summarize a great deal of variability in population and stratum-level information on viability. For example, although the overall persistence score was 0.34, the scores for individual populations ranged from −1 (Salmon River, Sixes River) to +0.98 (Tenmile Lakes), and approximately two-thirds (13/21) of the populations had persistence scores greater than 0.25 (Table 8). The stratum-level persistence scores, SP, were calculated as the median of the population scores. Only the Lakes stratum had a very high certainty of stratum persistence (0.88) followed by the Umpqua (0.40). The three remaining strata and persistence scores range narrowly from 0.24 to 0.27. Population sustainability scores, PS, ranged from −1.0 in two populations to a high of 0.85 in the Coquille River. The stratum scores for sustainability, SS, were less variable (Table 8), in the narrow range of 0.39 to 0.48.

The data set adjustment from AUC counts to peak counts for critical abundance lowered the persistence score substantially. Persistence is evaluated using three factors, while sustainability uses seven (including the three persistence factors). As a result, persistence is much more sensitive to changes in a single factor than is sustainability, so this score is considerably lower than was reported in Wainwright et al. (2008) while sustainability is slightly higher.

Spawning escapements in some recent years have been higher than has been seen in the past 60 years. This is attributable to a combination of management actions and environmental conditions. In particular, harvest has been strongly curtailed since 1994, allowing more fish to return to the spawning grounds (Figure 5). As shown in Figure 10, hatchery production has been reduced to a small fraction of the wild production (ODFW 2009a). Nickelson (2003) found that reduced hatchery production led directly to higher survival of naturally produced fish. Ocean survival, as measured by smolt to adult survival of OPI area hatchery fish, generally started improving for fish returning in 1999 (Table 4). In combination, these factors have resulted in the highest spawning escapements that have been seen since 1950, although total abundance before harvest peaked at the low end of what was observed in the 1970s (Figure 6).

Higher spawner abundance in recent years has resulted in higher scores for population diversity, PD. Three of the population diversity factors are directly or indirectly related to abundance. The spawner abundance criterion is based on long-term harmonic mean abundance, so it will increase in periods of high abundance. Spawner distribution, PD-3, and juvenile distribution, PD-4, measure the distribution of coho salmon among watersheds within the populations. At higher spawner abundance, coho salmon tend to spread out through a greater area of habitat. This leads to a similar expansion of juvenile distribution. The criterion for evaluation of juvenile distribution was considered by some members of the BRT to be uninformative, so this criterion was given a lower weight in the BRT deliberations (but not in the computation of DSS results). In evaluating the DSS, both of the distribution criteria score higher during periods of high abundance unless all habitat is already occupied or unoccupied habitat is unsuitable or inaccessible. With minor exceptions, these scores increased from those in the TRT analysis, indicating that habitat was available for range expansions within most populations. Consequently, the peak in spawner abundance in the early 2000s, combined with the reduction in hatchery production, resulted in strong scores for population diversity (Table 8).

Factors for Decline and Threats

Introduction

The BRT utilized the results of the DSS and information on population abundance, growth rates (productivity), spatial structure, and diversity to inform its assessment of the current OCCS ESU biological status. Current information on harvest and hatcheries also was included. In addition, the BRT evaluated current and future threats to the ESU that may or may not be manifest in its current biological status. The BRT categorized these threats according to ESA Section 4(a)1:

- The present or threatened destruction, modification, or curtailment of its habitat or range;
- Overutilization for commercial, recreational, scientific, or educational purposes;
- Disease and predation; and
- Other natural or man-made factors affecting its continued existence.

To the degree possible, the BRT attempted to characterize threats whose effects are likely to already be reflected in the current biological status of the ESU and those that are likely to become manifest in the future.

As a starting point for reviewing threats, the BRT reviewed the factors for decline, which had been identified as part of the original ESA status review process for the ESU. For example, Table 11 lists freshwater habitat factors for decline that were identified in Oregon's OCSRI (1997) and subsequently discussed in NMFS (1997c). Other factors for decline were identified during the 1996 status review.

In the next step toward not only understanding what affected the ESU in the past, but identifying what affects the ESU now and may affect it in the future, NWR (NMFS 1997a) listed threats to the ESU as shown in Table 12. Threats were defined as:

> human activities or natural events (e.g., road building, floodplain development, fish harvest, hatchery influences, volcanoes) that cause or contribute to limiting factors. Threats may exist in the present or be likely to occur in the future.

Limiting factors were defined as:

> physical, biological, or chemical features (e.g., inadequate spawning habitat, high water temperature, insufficient prey resources) experienced by the fish at the population, intermediate (e.g., stratum or major population grouping), or ESU levels that result in reductions in VSP parameters (abundance, productivity, spatial structure, and diversity). Key limiting factors are those with the greatest impacts on a population's ability to reach its desired status (NMFS 1997a).

Table 11. Factors for decline and habitat limiting factors for OCCS (NMFS 1997c).

Modification or curtailment of range	Harvest	Disease and predation	Regulatory mechanisms	Other natural or man-made factors
Fish passage (hydro, tide gates, culverts)	Marine	Disease	NW Forest Plan	Droughts
Water withdrawal	Recreational	Predation	Forest practices	Floods
Land use and management			Dredge and fill	Ocean conditions
Logging			Ag practices (sedimentation, temperature)	Artificial propagation
Agricultural activities				
Estuary loss			Logging practices (sedimentation, temperature)	
Wetland loss				
Riparian area/ quality loss			Urban growth	
Channel complexity loss				
Floodplain connectivity loss				
Splash dams/ log drives				
Gravel/placer mining				
Forest and ag conversion to urban				
Urbanization				

Table 12. Threats to OCCS ESU identified by NMFS NWR (NMFS 1997c).

Human threats	Natural threats
Agriculture: Instream wood, water temperature, substrate sediment	Drought
Forestry and private lands: Instream wood, water temperature, substrate sediment	Floods
Gravel mining: Particular concern on the southern Oregon Coast where the Umpqua and Coquille River basins have significant sediment deficits	Wildfire
Water withdrawals or diversions: Current concern on the southern Oregon coast; future concern on mid-coast as urban areas grow Drought interaction	Tsunami
Urbanization: Floodplain functions, instream wood, substrate sediment, storm water	

Because the list of threats has changed over the years, Table 13 compares the list of threats and limiting factors between those identified in 2003 (Good et al. 2005) and those considered by the BRT in this review. The BRT considered these factors for decline, limiting factors, and threats that had been previously identified, then reviewed additional information that has become available since 1997. The BRT utilized a threats matrix (see Appendix A, Table A-1) to summarize the major human and natural threats facing the OCCS ESU at this time and in the future.

Table 13. Threats to OCCS ESU identified by NMFS NWR (NMFS 1997c) and the 2009/2010 BRT.

Threats previously identified by NWR	**Threats identified by BRT 2009/2010**
Agriculture and forestry	Agriculture and forestry
Instream wood	Stream complexity (includes disturbance, roads, splash damming, stream cleaning)
Water temperature	Water temperature
Substrate sediment	Substrate sediment analyzed in stream complexity section
Estuary and wetland habitat loss not identified as a major threat	Estuaryand wetland habitat loss, due to effect on life history diversity and numbers
Beaver dam loss not identified as a major threat	Beaver dam loss analyzed in stream complexity section
Fish passage restriction not identified as a major threat	Fish passage restriction
Gravel mining:	Gravel mining:
Umpqua River and Coquille River basins	Umpqua River, Coquille River, and Tillamook River basins
Water withdrawals or diversions:	Water withdrawals or diversions:
Mid-South Coast, Umpqua strata	Mid-South Coast, Umpqua strata
Drought interaction	Drought interaction
	Global climate change
Urbanization	Land use conversion, urbanization
Floodplain functions	Floodplain functions
Instream wood	Instream wood addressed in stream complexity analysis
Substrate sediment	Substrate sediment addressed in stream complexity analysis
Storm water	Storm water
Disease and parasites not identified as a major threat	Disease and parasites
Artificial propagation	Artificial propagation
Harvest	Harvest
Predation not identified as a major threat	Predation
Global climate change not identified as a major threat	Global climate change
Marine productivity not identified as a major threat by NWR, was identified by 2003 BRT	Marine productivity
Drought, floods, and wildfire	Drought, floods, and wildfire addressed under global climate change
Tsunami	Tsunami not addressed
NIS not previously identified	NIS introductions: ecosystem impacts of NIS, effects of NIS fish, and effects of NIS plants on stream complexity establishment not identified as a threat

Other Natural and Man-made Factors

This portion of the BRT review started with the discussion of other natural and man-made factors for decline. These include the effects of ocean conditions and marine productivity, which have been recognized as significant issues for the OCCS ESU since the 1993 status review (Weitkamp et al. 1995), and the effects of global climate change on freshwater and marine habitats.

Past ocean conditions and marine productivity

Evidence has accumulated to demonstrate 1) recurring, decadal-scale patterns of ocean and atmosphere climate variability in the North Pacific Ocean (Mantua et al. 1997, Zhang et al. 1997, Overland et al. 2009, Schwing et al. 2009), and 2) correlations between these oceanic changes and salmon population abundance in the Pacific Northwest and Alaska (Hare et al. 1999, Mueter et al. 2002, Francis and Mantua 2003, Lawson et al. 2004). There seems to be little doubt that survival rates in the marine environment can be strong determinants of population abundance for Pacific salmon and steelhead. It is also generally accepted that for at least two decades, beginning about 1977, marine productivity conditions were unfavorable for the majority of salmon and steelhead populations in the Pacific Northwest (in contrast, many populations in Alaska attained record abundances during this period). Good et al. (2005) cited evidence that an important shift in ocean and atmosphere conditions occurred around 1998 that was expected to persist for several years. However, that change has not persisted.

One indicator of the ocean-atmosphere variation for the North Pacific is the Pacific Decadal Oscillation (PDO) index. Since 1900 the PDO has shown a number of multidecade periods of predominantly positive (1926−1947, 1977−1998) or negative (1948−1976) values (Figure 13), which correspond roughly to periods of low (positive PDO) or high (negative PDO) West Coast salmon returns (Mantua et al. 1997). There was a sharp transition to negative values in 1999, followed closely by a positive transition in 2003 and a negative transition in 2007. Negative PDO values are associated with relatively cool ocean temperatures (and generally high salmon productivity) off the Pacific Northwest, and positive values are associated with warmer, less productive conditions. Wide fluctuations in many salmon populations in recent years may be largely a result of these shifts in ocean conditions.

Although these climate-related facts are relatively well established, much less certainty can be attached to predictions about what this means for the viability of listed salmon and steelhead. For several reasons, considerable caution is needed to project into the future. First, empirical evidence for cycles in PDO, marine productivity, and salmon abundance extends back only about a century, or about three periods of two to four decades in duration. These periods form a very short data record for inferring future behavior of a complex system. Thus as with the stock market, the past record is no guarantee of future performance. Second, the past decade has seen particularly wide fluctuations not only in climatic indices (e.g., the 1997–1998 El Niño was in many ways the most extreme ever recorded, and the 2000 drought was one of the most severe on record) but also in abundance of salmon populations. In general, as the magnitude of climate fluctuations increases, population extinction risk also increases. Third, as anthropogenically caused global climate change occurs in the future, it will affect ocean productivity and likely change the dynamics of ocean variation as well as ecosystem processes

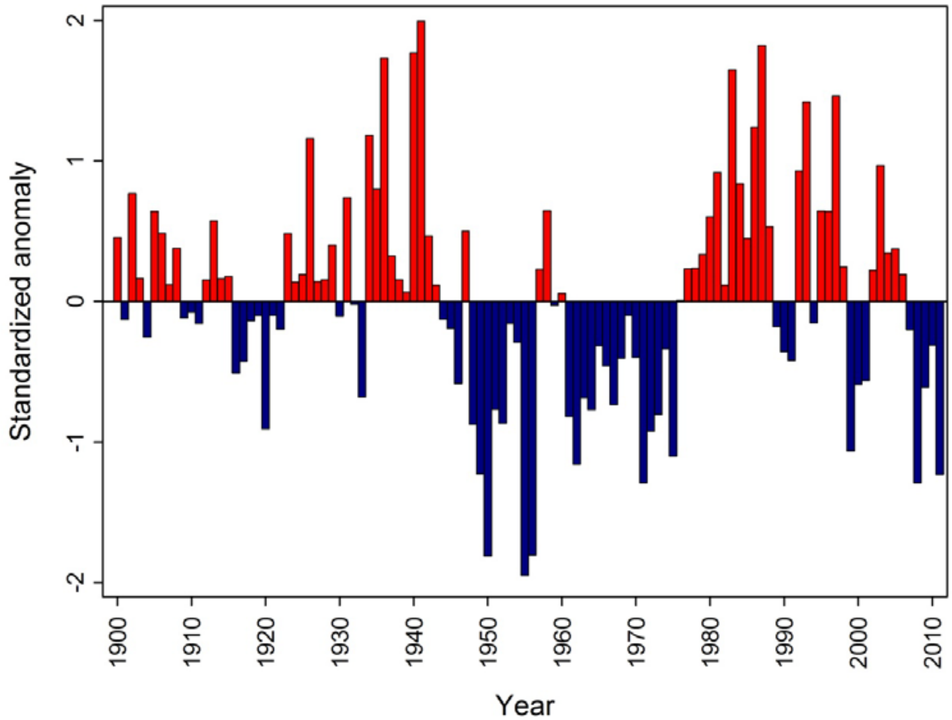

Figure 13. The PDO index from 1900 through 2009. Values shown are standardized deviations from the long-term (1900–1993) mean. Data from University of Washington (http://jisao.washington.edu /pdo/PDO.latest).

(Overland et al. 2009). Finally, changes in the pattern of ocean/atmosphere interactions do not affect all species (or even all populations of a given species) in the same way (Peterman et al. 1998).

Ocean ecosystem conditions

As ocean temperatures warm, empirical and theoretical studies show that marine fish and invertebrates tend to shift their distributions towards higher latitudes and deeper water at observed and projected rates of 30 to 130 km/decade towards the pole and 3.5 m/decade to deeper waters (Cheung et al. 2009). Although this change may occur gradually, anomalously warm conditions may allow temporary range expansions. For example, during the 1997−1998 El Niño, warm water fishes invaded Oregon waters, including striped marlin (*Tetrapturus audux*), Dorado (*Coryphaena hippurus*), and Pacific barracuda (*Sphyraena argentea*) (Pearcy 2002). In Alaska the summers of 2004 and 2005 were unusually warm and southern fish species including thresher shark (*Alopias vulpinus*), blue shark (*Prionace glauca*), opah (*Lampris guttatus*), and large numbers of Pacific sardines (*Sardinops sagax*) were recorded, often extending the northern limit of the species' known ranges (Wing 2006).

One species that is actively undergoing a substantial range expansion is the Humboldt or jumbo squid (*Dosidicus gigas*) (Field et al. 2007, Zeidberg and Robison 2007). Like most squids, Humboldt squid are ecological opportunists with a short life span (typically <2 years) and high fecundity, allowing their abundances to fluctuate greatly on short time scales; they have been likened to locusts of the marine realm, reaching plague proportions and creating famine (Rodhouse 2001). Even among squid species, Humboldt squid stand out; they have the highest growth rates (Mejia-Rebollo et al. 2008) and fecundity (up to 13 million eggs per female, Keyl et al. 2008) of any squid, are tolerant of water of a wide range of temperatures and oxygen levels, including water typically considered hypoxic (Gilly et al. 2006, Zeidberg and Robison 2007), and can move horizontally nearly 200 km in a week (Gilly et al. 2006). Humboldt squid typically make diel migrations between surface waters at night and depths in excess of 250 m during the day, although large numbers of squid have been observed at the surface during the day, indicating considerable plasticity in their behavior (Olson et al. 2006).

The normal range of Humboldt squid is the eastern tropical Pacific Ocean, extending as far north as southern California (≈30°N, Keyl et al. 2008), although they have been sporadically reported off the California coast throughout the last century (Field et al. 2007). In their current range expansion, they were first reported north of their normal distribution during the 1997 El Niño when they were observed in Monterey Bay (Zeidberg and Robison 2007) and off Oregon (Pearcy 2002). Reports of squid north of their typical range resumed in 2000 (Zeidberg and Robison 2007, Keyl et al. 2008) and Humboldt squid were reported from British Columbia and Alaska in 2004 (Cosgrove 2005) and again in Alaska in 2005 (Wing 2006).

Numerous long-term coastal sampling programs provide excellent documentation of this dramatic spread, in particular the apparent explosion of squid during summer 2009. For example, the joint U.S.-Canada Pacific hake *(Merluccius productus)* acoustic-trawl survey has documented a rapid increase in the number and frequency of Humboldt squid caught in trawls since the survey began in 1977 (Holmes et al. 2008). The first confirmed catch occurred in 2003 and, by the 2007 survey, the range and abundance of squid had greatly expanded (Holmes et al. 2008). In the 2009 survey, catches of Humboldt squid were extremely large and frequent: 44% of hauls in 2009 included at least one Humboldt squid.[11] Similarly, the NWFSC Predator (Emmett et al. 2006) and Stock Assessment Improvement Program (Auth 2008) research cruises, both of which sample with large trawls (mouth ≈25 m wide x 20 m deep) at night, first caught Humboldt squid in 2006 and 2004, respectively. In summer 2009, these studies caught Humboldt squid in 14% (n = 84 hauls) and 19% (n = 85) of their hauls, respectively, with the highest catches in late summer.[12]

A recent analysis of factors contributing to the collapse of Sacramento fall Chinook salmon includes one section on Humboldt squid (Lindley et al. 2009). The authors conclude that Humboldt squid likely had limited impact on Sacramento Chinook salmon due to limited spatial overlap; most squid were beyond the continental shelf while most juvenile salmon were on the shelf. However, in 2005 and 2009 squid were caught off the Washington and Oregon coasts by research programs targeting juvenile salmon, suggesting overlap of squid and juvenile salmon within the range of OCCS.

Humboldt squid are a "voracious, opportunistic predator" (Gilly and Markaida 2007), capable of feeding on a wide range of prey. Prey items identified in squid stomachs collected off the coasts of California and Oregon included commercial (e.g., Pacific hake, northern anchovy

[11] D. Chu, NWFSC, Seattle, WA. Pers. commun., November 2009.
[12] R. Emmett, NWFSC, Newport Research Station, Newport, OR. Pers. commun., December 2009.

[*Engraulis mordax*], Pacific sardine, rockfishes [*Sebastes* spp.], and flatfishes [Pleuronectiformes]) and noncommercial (e.g., northern [*Stenobrachius leucopsarus*] and blue [*Tarletonbeania crenularis*] lantern fishes) fish species (Field et al. 2006), with perhaps the biggest impact on hake populations (Holmes et al. 2008). Fishes found in squid stomachs were up to 42 cm in length, with greater than 10% of the total biomass ingested consisting of prey at least 35 cm in length. Squid have also been observed attacking larger fish (up to 50 cm) when the prey are confined, such as skipjack (*Katsuwonus pelamis*) and yellowfin (*Thunnus albacares*) tunas caught together with squid in purse seines (Olson et al. 2006). A Chinook salmon jack (262 mm fork length) was caught in the lower Columbia River estuary in October 2009 with what appeared to be a squid bite mark,[13] indicating that squid may attempt to take small salmon.

For OCCS, the timing of Humboldt squid presence in our area is of particular concern, because squid abundances typically peak during late summer (August, September) and are considerably lower in early summer (June, July) (Field et al. 2007, Gilly and Markaida 2007, Litz et al. 2011). The best predictor of Columbia River coho year-class success is the number of juveniles caught off the Washington and Oregon coasts in September of the previous year (Van Doornik et al. 2007, Wainwright and Weitkamp in prep.). This relationship indicates that the individuals that reside in local waters throughout the summer are the ones that return as adults. Unfortunately, these individuals will likely overlap with the highest abundances of Humboldt squid and therefore face high predation risk. By contrast, spring Chinook salmon originating from Pacific Northwest streams are present in local waters in early summer, then move northwards towards Alaska by mid-summer (Trudel et al. 2009). Because of their migratory patterns, these fish will likely experience much lower habitat overlap with Humboldt squid unless squid densities are also high farther north.

It is not clear whether this most recent population explosion is long lasting or transient, why it has occurred, or whether it includes a northern expansion of squid spawning areas (currently Gulf of California, the Costa Rica Dome, Gilly and Markaida 2007). No direct data or information indicated that the presence of Humboldt squid will lead to reduced abundance of OCCS, although the BRT was concerned that squid may potentially be a risk to the salmon or the ecosystem on which they depend. Overall, the BRT concluded that the presence of this warm water species off the Oregon coast and further north beyond its previously known range is a sign that the coastal ocean ecosystem on which coho salmon depend is in an unpredictable state of flux.[14]

Effects of climate change on the OCCS ESU

Recent climate change has had widespread ecological effects across the globe, including changes in phenology, changes in trophic interactions, range shifts (both in latitude and elevation and depth), extinctions, and genetic adaptations (reviewed by Parmesan 2006) and these changes have influenced salmon populations (ISAB 2007, Crozier et al. 2008a, Mantua et al. 2009). Although these changes have undoubtedly influenced the observed VSP attributes for the OCCS ESU, we cannot partition past climate effects from other factors influencing the status of the ESU. Continuing climate change poses a threat to aquatic ecosystems (Poff et al. 2002) and

[13] Reported by BRT member L. Weitkamp, December 2009.
[14] By 2011 Humboldt squid had nearly disappeared from Oregon waters.

more locally to Pacific salmon (Mote et al. 2003). Recently, Wainwright and Weitkamp (in prep.) reviewed the major potential physical effects of climate change in western Oregon, presented an approach to integrating these effects across life history stages, and applied that approach to evaluate potential effects on the OCCS ESU. Here we summarize their findings.

The coho salmon life cycle extends across three main habitat types: freshwater rivers and lakes, estuaries, and marine environments. In addition, terrestrial forest habitats are also essential to coho salmon because they determine the quality of freshwater habitats by influencing the types of sediments in spawning habitats and the abundance and structure of pools in juvenile rearing habitats (Cederholm and Reid 1987). Wainwright and Weitkamp (in prep.) begin by considering these four habitats, how physical climate change is expected to affect them over the next 50 years, and how salmon will respond to those effects during specific life history stages (Figure 14). Climate conditions have effects on each of these habitats, thus affecting different portions of the life cycle through different pathways. This leads to a very complex set of potential effects to assess.

Wainwright and Weitkamp (in prep.) recognized that, while we have quantitative estimates of likely trends for some of the physical climate changes, we do not have sufficient understanding of the biological response to these changes to reliably quantify the effects on salmon populations and extinction risk. For this reason, the analysis is qualitative: summarizing likely trends in climate, identifying the pathways by which those trends are likely to affect salmon, and assessing the likely direction and rough magnitude of coho salmon population response. Their assessment is summarized in Table 14 and discussed below.

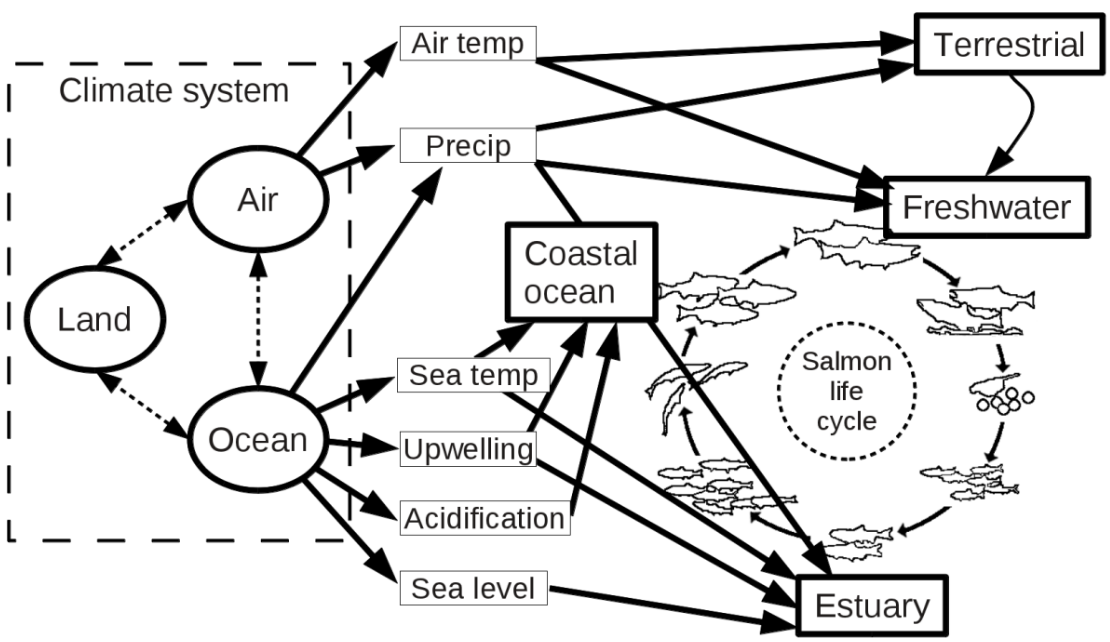

Figure 14. Conceptual diagram of multiple pathways by which climate influences the salmon life cycle. The climate system affects four habitats (terrestrial, freshwater, coastal ocean, and estuary) vital to salmon life stages, which in turn influence salmon reproduction, growth, and mortality. (Printed from Wainwright and Weitkamp in prep.)

Table 14. Summary of effects of physical climate changes on OCCS by habitat type. Strength and direction of effects are rated from strongly positive (+ +) through neutral (0) to strongly negative (– –). (Modified from Wainwright and Weitkamp in prep.)

Physical change	Certainty of change	Processes affecting salmon	Effect on salmon	Certainty of effect
		Terrestrial		
Warmer, drier summers	High	Increased number and intensity of fires, increased tree stress and disease affect large woody debris, sediment supplies, riparian zone structure	0 to – –	Low
Reduced snowpack	High	Increased growth of higher elevation forests affect large woody debris, sediment, riparian zone structure	+ to 0	Low
		Freshwater		
Reduced summer flow	High	Less accessible summer rearing habitat	–	Moderate
Earlier peak flow	High*	Potential migration timing mismatch	0 to – Umpqua: 0 to – –	Moderate
Increased floods	Moderate*	Redd disruption, juvenile displacement, upstream migration	0 to – Umpqua: – to – –	Moderate
Higher summer stream temperature	Moderate	Thermal stress, restricted habitat availability, increased susceptibility to disease and parasites	– to – –	Moderate
		Estuarine		
Higher sea level	High	Reduced availability of wetland habitats	– to – –	High
Higher water temperature	High	Thermal stress, increased susceptibility to disease and parasites	– to – –	Moderate
Combined effects		Changing estuarine ecosystem composition and structure	+ to – –	Low
		Ocean		
Higher ocean temperature	High	Thermal stress, shifts in migration, range shifts, susceptibility to disease and parasites	– to – –	Moderate
Intensified upwelling	Moderate	Increased nutrients (food supply), coastal cooling, ecosystem shifts; increased offshore transport	+ + to 0	Low
Delayed spring transition	Low	Food timing mismatch with outmigrants, ecosystem shifts	0 to –	Low
Intensified stratification	Moderate	Reduced upwelling and mixing lead to reduced coastal production and reduced food supply	0 to – –	Low
Increased acidity	High	Disruption of food supply, ecosystem shifts	– to – –	Moderate
Combined effects		Changing composition and structure of ecosystem, changing food supply and predation	+ to – –	Low

*Effects are strongest and most certain in higher elevation snow-fed basins.

The impacts of change in terrestrial forest habitats on salmon are indirect through effects on hydrology, water quality, and physical habitat structure. While there is widespread agreement on the atmospheric changes likely to affect forests (warmer, drier summers and reduced snowpack), the effects of resulting changes in forests are uncertain. Also, the subsequent effects on salmon are of low certainty and could range from slightly positive to strongly negative.

For freshwater habitats, climate change is expected to reduce summer flows and raise summer stream temperatures throughout western Oregon. These trends would reduce the amount of available summer rearing habitat for coho salmon and increase thermal stress, which would result in reduced growth and increased susceptibility to disease and parasites, and thus have a weak to strong negative effect on freshwater salmon production. Furthermore, increased water temperature will necessarily increase the consumptive demands of juvenile coho and their predators (Petersen and Kitchell 2001). It remains to be seen how the food web as a whole will respond to these changes in energetics. In addition, reduced winter and spring snowpack is expected to result in earlier peak flows and increased flooding in snow-fed rivers. For this ESU, changes in snowpack are likely to have a significant effect on salmon only in the lower main stem and upper portions of the Umpqua River basin (north and south Umpqua populations); the effect would be negative for those populations.

In estuaries, the main expected physical changes are rising sea level and warmer water temperatures. The effect of these changes on salmon is likely to be negative, due to losses of intertidal wetland habitats and increased thermal stress during migrations. As in freshwater, increased water temperature will increase the metabolic demand of coho salmon and their predators (Petersen and Kitchell 2001). Changing temperatures may also lead to unpredictable changes in estuarine community composition, which might have either positive or negative effects on salmon.

In the ocean, a number of anticipated physical changes could affect OCCS. These include higher water temperature, intensified upwelling, delayed spring transition, intensified stratification, and increasing acidity in coastal waters. Of these, only intensified upwelling would be expected to benefit coastal-rearing salmon; all the other effects would likely be negative. Increasing temperature and acidity have a high certainty of occurring and both would likely have negative effects on coho salmon. As with freshwater and estuarine ecosystems, increased water temperature will also lead to increased metabolic demand for coho salmon and their predators (Petersen and Kitchell 2001). However, the local physical interactions of wind patterns and stratification on upwelling and the timing of the spring transition for our coast are difficult to predict with certainty, so the response of salmon to these other ocean factors is of low certainty. In combination, all of these physical effects are likely to result in changes in the species composition and structure of the coastal ecosystem, with unpredictable consequences for coho salmon.

While we have noted some expected positive effects, negative effects of climate change predominate for each habitat and life history stage (Table 14). While many of the individual effects of climate change on OCCS are expected to be weak or are uncertain, we need to consider the cumulative impacts across the coho salmon life cycle and across multiple generations. Because these effects are multiplicative across the life cycle and across generations, small effects at individual life stages can result in large changes in the overall dynamics of populations. This means the mostly negative effects predicted for individual life history stages will most likely result in a substantially negative overall effect of climate change on OCCS over the next few decades. Despite large uncertainties surrounding specific effects at individual life stages, expectations for increasing air and water temperatures, drier summers, higher incidence of flooding, and altered estuarine and marine habitats lead us to expect increasingly frequent years with low survival, resulting in an overall increase in risk to the ESU from climate change over the next 50 years.

Ecosystem impacts of nonindigenous species

OCCS and other salmonids traverse large geographic areas from freshwater to estuarine and ocean habitats during their life cycle. During this time, they encounter numerous nonindigenous species (NIS) (Sanderson et al. 2009). Boersma et al. (2006) verified many invertebrate and plant species introduced into the Pacific Northwest (and more specifically the OCCS ESU) that have documented effects. In the OCCS ESU, the majority of NIS are plants and fishes. The mechanisms of impact by these NIS are predation, competition, hybridization, infection (disease and parasites), and habitat alteration (Mack et al. 2000, Simberloff et al. 2005).

The presence of NIS fishes poses one of the greatest threats to the persistence of healthy native fish populations (Rahel 2002). Sanderson et al. (2009) report high to moderate densities of NIS fishes in the ESU. Effects of these fishes include predation by channel catfish (*Ictalurus punctatus*), American shad (*Alosa sapidissima*), brook trout (*Salvelinus fontinalis*), smallmouth bass (*Micropterus dolomieu*), and largemouth bass (*Micropterus salmoides*). Other NIS fishes alter habitats and ecosystem function. For example, NIS warmwater fishes in Tenmile Lakes alter planktonic community structure. This affects the summer rearing and residency of OCCS juveniles (ODEQ 2007). Additional discussion of NIS fishes is found in the Predation subsection.

Plant and animal NIS invade and displace native plant species and communities when human disturbance such as timber harvest (USFS 2005) or climate change (Wainwright and Weitkamp in prep.) occurs. Brazilian elodea (*Egeria densa*), eurasian water milfoil (*Myriophyllum spicatum*), reed canary grass (*Phalaris arundinacea*), giant and Japanese knotweed (*Polygonum* spp.) [reed canary grass and giant knotweed effects are discussed in the Stream Complexity subsection], cordgrasses (*Spartina anglica*, *S. densiflora*, and *S. patens*), Japanese eelgrass (*Zostera japonica*), evergreen and Himalayan blackberry (*Rubus laciniatus*, *R. armeniacus*), purple loosestrife (*Lythrum salicaria*), and New Zealand mudsnail (*Potomopygyrus antipodarum*) are already found in the ESU (Boersma et al. 2006). These plants and invertebrates alter habitats and ecosystem function. For example, purple loosestrife displaces sedges and cattails (*Typha* spp.). This causes shifts in local nutrient availability and affects detrital foodwebs (Blossey et al. 2001). New Zealand mudsnails blanket streambeds, can consume a majority of gross primary production, and outcompete other macroinvertebrates such as larval mayflies, stoneflies, and caddis flies, which are important freshwater salmon prey (Hall et al. 2003, Kerans et al. 2005).

Potential new invasions that have been identified include zebra mussel (*Dreissena polymorpha*), Chinese mitten crab (*Eriocheir sinensis*), and hydrilla (*Hydrilla verticillata*). A list of potential invasive species is identified in the Oregon Conservation Strategy's discussion of the Coast Range Ecoregion (ODFW 2011) and in USFS (2005) Invasive Plant Program documents. Some of these have already caused significant damage to aquatic and terrestrial ecosystems in other parts of the United States. Of present concern, however, is the collapse of a major food source of OCCS in some estuaries. During outmigration of OCCS smolts in the Mid-Coast stratum, one of their major food sources is mudshrimp (*Upogebia pugettensis*). These intertidal benthic invertebrates are among the most dramatically affected by recent introduced species invasions and associated hydrological and geochemical alterations of the estuaries (Dumbauld et al. 2010).

Some members of the BRT were concerned that invasions by NIS present a risk to the salmon or the ecosystem on which they depend. Though some NIS species already established in the ESU may be reflected in current biological status, invasions by NIS and the subsequent ecosystem changes brought about by their establishment may constitute a future threat to the species.

The Present or Threatened Destruction, Modification, or Curtailment of its Range

Fish passage

The foundation of ESU viability is built on the ability of populations to function in an integrated manner and persist across the landscape. This integration includes dispersal within and 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). Dams are not the only barrier to migration; barriers can include smaller scale features such as road culverts that block seasonal passage to and from mainstem rivers and also intertributary and intratributary movement.

Any habitat modification that prevents annual or seasonal movement across the landscape can affect coho salmon populations. Ebersole et al. (2006) described 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 also provide improved summer survival and subsequently higher densities of juvenile salmonids prior to the overwinter period. Culverts, tide gates, and temperature blockages can affect access of juveniles to these small systems.

The BRT utilized the original factors for decline summarized by NMFS (1997a) in 1997 as a starting point for its assessment. A discussion of fish passage was included in this assessment because it was part of the original factors for decline discussion and some members of the BRT felt that it may be an important source of risk to the OCCS ESU. Fish passage was not identified in the State of Oregon's Oregon Coastal Coho Assessment (Oregon 2005) as either a primary or secondary limiting factor for any population of OCCS.

There are wholesale blockages of fish passage by hydropower projects, which is considered a major factor for decline in many salmonid ESUs. The effect of hydropower development in the ESU was reviewed in the Oregon Coast Coho Assessment (Oregon 2005).[15] Oregon's conclusion was:

> Generally, within the majority of the ESU, impacts from hydroelectric projects are insignificant or nonexistent. Specifically, within the Umpqua Basin, Oregon coastal coho have been prevented from reaching 36 miles of spawning and rearing habitat by hydroelectric projects. Impacts to downstream reaches include alteration of flows and interruption of natural sediment and large woody debris regimes (ODFW 2005e).

As discussed, fish passage problems are not just related to blockage by hydropower projects. As part of the Oregon Coastal Coho Assessment, Dent et al. (2005) gathered data on stream crossings from private industrial landowners (60% of data captured), federal (USFS 100% of data captured) and state (90% of data captured) agencies, and data sets with a complete census of state and county roads (100% of data captured). Fish passage status was provided by the landowners and crossing status was not verified. These crossings were binned in three categories: high intrinsic potential for coho salmon, low intrinsic potential, and non-coho. Their results show that for the entire ESU, 11% of low intrinsic potential streams, 10% of high intrinsic potential streams (these are probably minimal estimates because ODF did not have complete coverage of intrinsic potential streams for this analysis[16]), and 16% of non-coho streams have limited access. They estimated that of all the stream miles with limited access, approximately 18% were high intrinsic potential streams. The percent of stream crossings in high intrinsic potential areas are: 60% of the total number of stream crossings pass fish, 14% limit fish passage, and 26% are unknown. Dent et al. (2005) concluded:

> Our analysis suggests that a relatively small percent of coho habitat remains inaccessible (10−11%) and that Oregon Plan fish passage restoration projects have improved access to 6−11% of coho habitat. However, our analysis also suggests that a significant proportion of the coho habitat has unknown access status (28−31%). With unknown access to approximately one-third of the habitat, fish passage cannot be eliminated as a risk to coho at this point in time.

As part of the review of the Oregon Coastal Coho Assessment, NMFS NWR commented that fish passage through culverts, tide gates, and bridges presented a higher risk to OCCS than the Oregon (2005) assessment.

> The technical document (Oregon 2005) concludes that nearly a third of the area has an unknown passage status, that fish passage restoration projects have not been tested at high flows, that fish passage projects have rarely been monitored to test whether they are actually passable, and that fish passage cannot be eliminated as a risk to coho at this point in time (NMFS 2005a).

Since 2005 a concerted effort was made to improve the fish passage barrier geographic information system (GIS) layer, which was released in late 2009 (Oregon Fish Passage Barrier Data Set [OFPBDS], ODFW 2009d). This layer includes bridges, cascades, culverts, dams,

[15] Peer reviewer 2 (reference Appendix D) reports that future hydropower projects may occur in this ESU on the Siletz River at the old town site of Valsetz (Pacific 2009)
[16] G. Reeves, USFS, Corvallis, OR. Pers. commun., February 2011.

debris jams, fords, natural falls, tide gates, and weirs. The OFPBDS data set, however, does not include dikes, levees, or berms. Barriers in the data set may have information on their passability to salmonids. This information may designate complete or partial blockage to fish passage, complete passability, or an unknown passage status. The data set comes mainly from ODFW, Oregon Department of Transportation, and BLM (2009). It does not include barrier data from ODF, Oregon Water Resources Department (OWRD), soil and water conservation districts, watershed councils, tribes, and other originators; so the barriers shown in Figure 15 do not include those from private timber or agricultural lands, which contain 81% of the high intrinsic potential habitat in the ESU (Burnett et al. 2007). In addition, it reflects only a handful of the tide gates in the ESU; a newly available data set (Mattison 2011) documents approximately 350 tide gates, which we know from Bass (2010) are at least partial barriers to fish passage.

An aspect of fish passage that has been ignored until recently is the blockage of fish access to tidal stream, marsh, and swamp habitat by dikes and levees in estuarine and freshwater tidal areas. It is possible however, to utilize the presence of a tide gate as a proxy for blockage of habitat. Giannico and Sauder (2005) reviewed the effect of tide gates on migratory behavior of salmonids and found that tide gates had direct effects on salmonid movements through abrupt changes in salinity, elevated water velocities and turbulence, and a total physical barrier to fish passage during the time the gate is completely closed. Bass (2010) found that in Coos Bay, Oregon, tide gates have the potential to restrict movement of OCCS subyearlings and smolts when compared to a nongated channel. At a minimum, tide gates in the OCCS ESU act as partial barriers to fish passage and were, for the most part, previously unaccounted for in past analyses. Few tide gate locations are included in the 2009 OFPBDS, so are not shown in Figure 15. A noncomprehensive database of tide gates in the OCCS ESU is now available (Mattison 2011).

Fish passage barriers have not been identified as a major limiting factor for OCCS by ODFW, however, within the OCCS ESU, of barriers that are in the data set, nearly half (49%) are of unknown status. The incompleteness of the information in the 2009 OFPBDS and new information regarding the effect of tide gates on fish passage for OCCS led some members of the BRT to consider that, because information is unavailable on a large portion of important OCCS habitat (low gradient agricultural lands, low elevation private timber lands), and where information is available, 49% of crossings are of unknown status, fish passage may continue to be a significant information gap in identification of habitat problems in the ESU. In the absence of information regarding blockages due to dikes and levees from ODF, OWRD, soil and water conservation districts, watershed councils, and tribes, it is clear that there is a substantial uncertainty as to the true effect that fish passage barriers present to OCCS.

For the purposes of this risk assessment, the current biological status probably reflects, for the most part, the status of fish passage in the ESU. Improved passage status information in the database would allow a better assessment of the effect of fish passage on OCCS and therefore on the potential effects of any future increase in road building (culverts) or protection of low lying areas from higher flood elevations and sea level rise (tide gates). Future effects of fish passage barriers depend on the success of Oregon Plan programs to address fish passage problems, the anticipated effects of land use changes in the ESU, and the response to anticipated sea level rise.

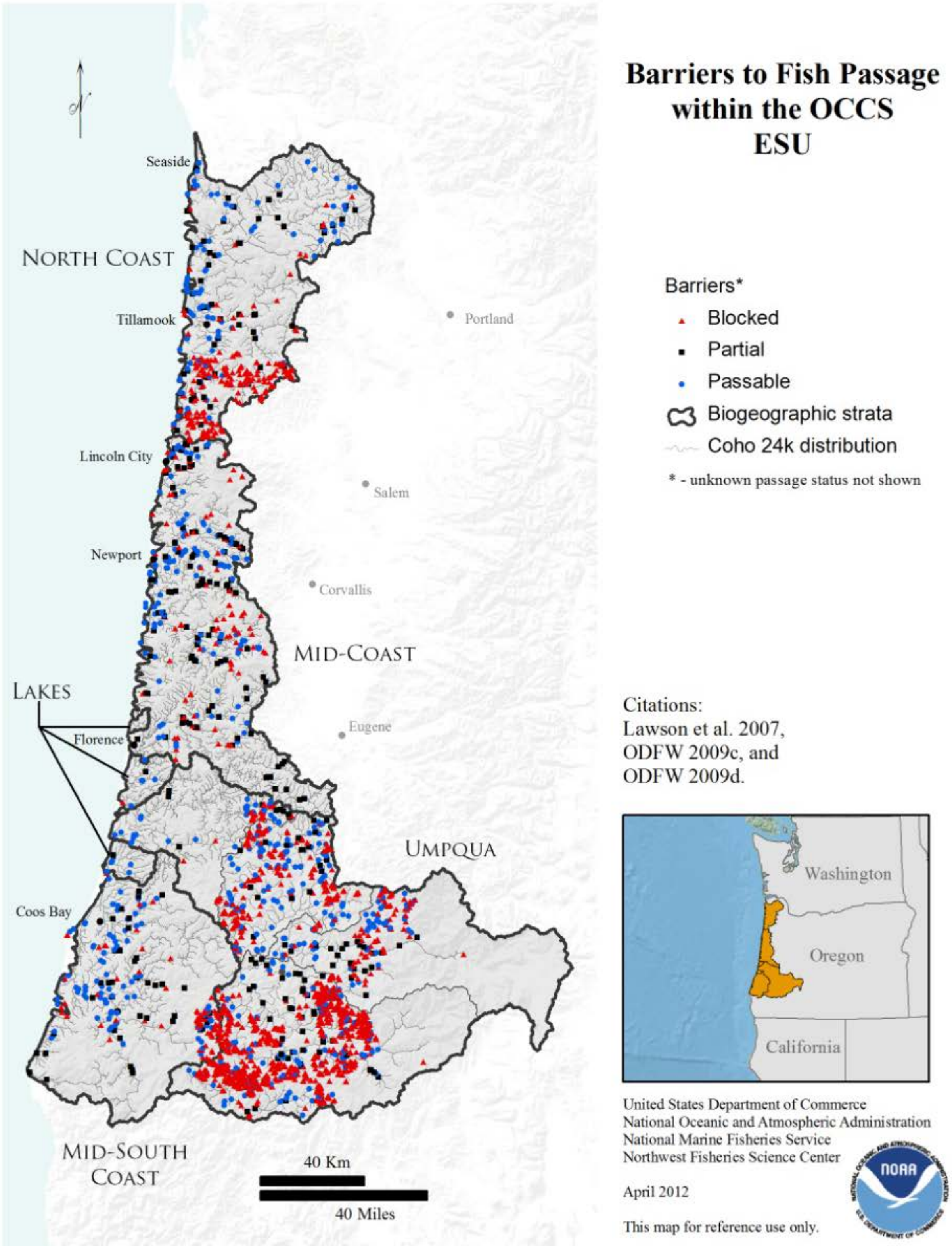

Figure 15. Barriers to fish passage. The database does not include barrier data from the ODF, OWRD, soil and water conservation districts, watershed councils, tribes, and other originators. It also includes approximately 28 tide gates.

Water availability

In its discussion of water availability as a threat, the BRT noted that the State of Oregon in the Conservation Plan (Oregon 2007) identified water availability as a primary limiting factor for the Middle Umpqua and South Umpqua OCCS populations. The Mid-South Coast stratum was identified as an area where water availability and water withdrawal is a problem as well, but was not identified as a primary limiting factor (Oregon 2007). Instream water availability problems can present limitations to OCCS through several mechanisms. May and Lee (2004) found that juvenile coho salmon abundance in pools decreased by 59% during the summer, with significantly higher losses occurring in gravel-bed versus bedrock pools. This means that gravel-bedded streams that have water withdrawals would likely have a higher potential impact on summer juvenile coho abundance than those in bedrock-dominated reaches. In addition, if connectivity is reduced due to the removal of water, then growth rates can be altered, which in turn has an effect on survivorship. For example, Kahler et al. (2001) found that juvenile coho salmon moved out of smaller and shallower habitat units, and fish that moved among habitat units grew faster than fish that remained in the same habitat unit (Kahler et al. 2001).

Ebersole et al. (2006) found that tributary streams that were naturally nearly dry in midsummer supported high densities of spawning coho salmon in the fall, and juveniles rearing there exhibited relatively high growth rates and emigrated as larger smolts. They also reported that improved winter growth and survival of juvenile coho salmon utilizing tributary habitats underscore the importance of maintaining connectivity between seasonal habitats and providing a diversity of sheltering and foraging opportunities, particularly where mainstem habitats have been simplified by human land uses.

What this means for water withdrawals is that where and when they occur in the landscape are critical to coho salmon. So water withdrawals that affect tributaries, particularly those that are gravel bedded, are most sensitive to changes in flow. If future water withdrawals are concentrated in tributaries of main river systems and are in gravel-bedded systems, this could lead to a decline in the summer abundance and overall survivorship of coho smolts. Another less investigated aspect of water availability is the effect of surface erosion on surface runoff. If climate change contributes to an increase in flooding, this flooding is often accompanied by mass wasting. As small valleys fill with coarse alluvium from such mass wasting, this could result in less surface flow and more subsurface flow.[17]

For most subbasins in the Umpqua stratum, water withdrawal for irrigation is a major consumptive use and, during the summer months from August to October, there is no natural stream flow available for new water rights (PUR 2007) except in the lowermost reaches of the Umpqua River (INR 2005) main stem. At times in the South Umpqua population, the flow is below one cfs in systems as large as Days Creek due to other consumptive uses (PUR 2007). EPA has placed all of the subbasins in the South Umpqua on the 303(d) list for flow modification; the North Umpqua and lower Umpqua River are on the list as well (PUR 2007).

OWRD has initiated instream water rights and leasing to mitigate loss of instream flow. The Oregon Watershed Enhancement Board (OWEB) 2007−2009 Oregon Plan update (Oregon

[17] Peer reviewer 7 (reference Appendix D). Pers. commun., September 2010.

2009b) reports that OWRD places a high priority on monitoring and protecting instream water rights statewide. Fifty-six percent of those streams regulated by OWRD during the 2007 water year were regulated on behalf of instream rights. Leases provide a mechanism for temporarily changing the type and place of use for a certificated water right to an instream use. The leased water remains in channel and benefits stream flows and aquatic species while leased and the instream use counts as use under the right for purposes of avoiding forfeiture (PUR 2007). However, the effectiveness of instream water rights protection does not provide certain instream flow for fish and wildlife because virtually all of these existing rights for instream flow have priority dates after 1955; they are fairly junior to other water rights in most basins and therefore do not often affect water deliveries (INR 2005).

In a landscape already significantly affected by instream water availability issues, increased demand (Kline et al. 2003), temperature rise, and the anticipated changes in precipitation patterns (see discussion of Effects of climate change on the OCCS ESU above) could have substantial effects on the OCCS ESU. In the Umpqua River stratum, the South Umpqua and Middle Umpqua populations are the most likely to be significantly affected by global climate change and temperature rise due to their interior position on the landscape. The Middle and Lower Umpqua populations are also the most subject in the ESU to downstream flow effects from the anticipated shift from a snow melt hydrology to rain hydrology that will affect stream flow timing and temperature due to shifts in precipitation in the Cascades. They are also the most likely to be affected by increased demand due to population growth (Kline et al. 2003).

Some water availability problems, such as the effect of summer rearing limitations experienced in the Umpqua River stratum, are probably already reflected to a large degree in the current biological status. However, future impacts to water availability from the effects of population growth, global climate change, or even shifts in shorter term climate variability are not reflected in current biological status and may constitute a future threat.

Land management—stream habitat complexity

Freshwater habitat complexity has been defined as the number of habitat units per length of stream (Quinn and Petersen, 1996), the number of pools per channel width (Montgomery et al. 1995), and the amount of wood and other obstructions that control specific channel features such as the amount of instream cover juvenile salmonids have during specific times of the year (Quinn and Petersen 1996). Freshwater habitat complexity is identified as a key limiting factor to the recovery of OCCS by ODFW (OCSRI 1997, Anlauf et al. 2009). Stream complexity has been identified as a factor for decline since 1997 (NMFS 1997a, OCSRI 1997). The State of Oregon also specifically identified it as a primary limiting factor for the purposes of the Oregon Coastal Coho Conservation Plan (Oregon 2007). Table 15 shows which populations ODFW considers to be limited by stream complexity. Thirteen of the 21 independent populations are considered stream complexity limited. The BRT's habitat subcommittee decided stream complexity was such an important component to any risk assessment of habitat that it would pursue analyses based on techniques and data sets utilized by ODFW.

Table 15. Primary and secondary limiting factors for independent populations in the OCCS ESU (Oregon 2005).

Population	Primary limiting factor	Secondary limiting factor
Necanicum	Stream complexity	None identified
Nehalem	Stream complexity	Water quality
Tillamook	Stream complexity	Water quality
Nestucca	Stream complexity	None identified
Salmon	Hatchery impacts	Stream complexity
Siletz	Stream complexity	None identified
Yaquina	Stream complexity	Water quality
Beaver	Spawning gravel	Stream complexity
Alsea	Stream complexity	Water quality
Siuslaw	Stream complexity	Water quality
Lower Umpqua	Stream complexity	Water quality
Middle Umpqua	Water quantity	Stream complexity, water quality
North Umpqua	Hatchery impacts	Stream complexity
South Umpqua	Water quantity	Stream complexity, water quality
Siltcoos	NIS	Stream complexity, water quality
Tahkenitch	NIS	Stream complexity, water quality
Tenmile	NIS	Stream complexity, water quality
Coos	Stream complexity	Water quality
Coquille	Stream complexity	Water quality
Floras	Stream complexity	Water quality
Sixes	Stream complexity	Water quality

Legacy splash damming, log driving, and stream cleaning—From an historical view, the stream complexity narrative begins with activities associated with the impacts of timber harvest. Three that have been identified are splash damming, log driving, and stream cleaning.

Splash damming and log driving are no longer practiced on Oregon coastal streams and rivers, but were used extensively during the deforestation phase of timber harvest (Maser and Sedell 1994). Splash dams were used to hold back enough water so that the logs that had been harvested and yarded to the pool behind the dam would sluice down the stream channel carrying the logs. This practice was well documented by Benner (1992) in the Coquille Basin. Often before the release took place, the downstream channel would be cleared as much as possible of impediments. These included instream boulder fields and debris jams. Figure 16 shows sites identified by Maser and Sedell (1994) and Miller (2010) for splash dams and log drives in the OCCS ESU. Legacy effects from these activities are still affecting geomorphic processes and landscape and local scale stream complexity in the OCCS ESU (Montgomery et al. 2003). These activities have contributed to loss of wood or boulders that acted to hold back gravel in the channel, loss of large trees that act as key constituents of log jams, and incision of stream channels and loss of floodplain connectivity (Montgomery et al. 2003).

Another aspect to the simplification of the complexity of streams in the OCCS ESU is that of stream cleaning activities formerly practiced by ODFW. Information presented in the Elliott Forest Watershed Analysis (ODF 2003) presents a picture of the effect of stream cleaning in Oregon coastal streams.

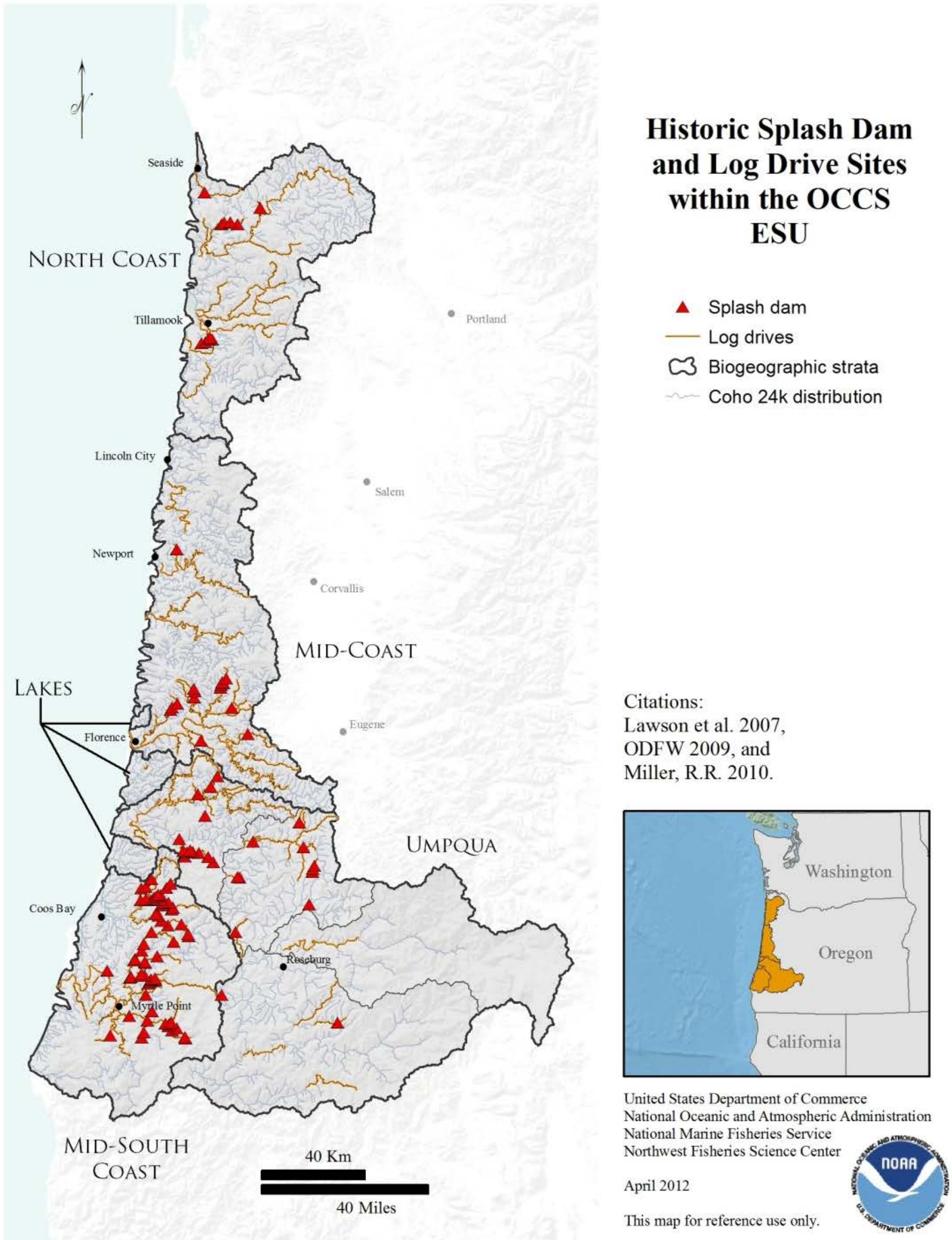

Figure 16. Historic splash dam and log driving locations.

> Damage caused to streams and rivers by early logging operations (splash dams, slash disposal in streams, log drives, etc.) often resulted in substantial logjams. In some cases, these jams could be a mile or more in length, and undoubtedly prevented or impeded anadromous fish passage. Largely as a result of these spectacular cases, in the 1930s the Oregon Game Commission began to require loggers to prevent woody debris from entering streams ….
>
> While the early stream surveys often called for clearing debris, its removal was effected in two ways. First, the Oregon Game Commission employed a stream improvement crew that drove throughout the region identifying obstructions and contacting land managers about their removal. This program lasted for 20 years, from about 1956−1976, according to ODFW files. The second tactic was to include stream cleaning, and specifically logging debris removal, in timber sale contracts. It appears to have continued until at least the mid-1980s ….
>
> Both kinds of stream cleaning were often done by running bulldozers up and down the stream (this technique also applied to log yarding from the 1950s into the 1970s). Notations … often identified the number of Cat D6 or D8 hours required for each job (although this also included winching logs out of streams). Without a doubt, stream cleaning had a widespread impact on aquatic habitat and the effects are still seen today in the amounts and distribution of wood in stream channels.

ODFW ended this practice, but legacy effects from the loss of large amounts of wood in the stream system endure. It is not surprising, therefore, that despite the number of instream complexity projects undertaken by watershed councils, ODFW, USFS, BLM, and private landowners (OWEB 2009), according to ODFW (Anlauf et al. 2009), “All monitoring areas are low in key pieces of wood relative to reference conditions.”

Beavers in OCCS habitat—Beavers are an important species to proper watershed functioning in coastal Oregon streams. They are considered a keystone species (Naiman et al. 1988) that provide significant coho salmon rearing habitat, primary productivity, nutrient retention/cycling, floodplain connectivity, and stream flow moderation (Reeves et al. 1989). Beavers and associated ponded habitats occur throughout the OCCS ESU and can be found from the headwaters to the estuarine environment. Found in estuarine and freshwater tidal marshes, beavers can build dams in the upper portions of sloughs and provide physical habitat for juvenile coho (Miller and Sadro 2003). Beavers are well known for damming smaller, lower gradient (less than 3%) streams with unconfined valleys (valley widths greater than four channels widths) (Retzer et al. 1956, Suzuki and McComb 1998, Pollock et al. 2003) as well as the floodplains of larger river systems (Murphy and Koski 1989, Pess et al. 2005). The potential benefits of beavers and their associated habitats to juvenile coho salmon thus may be dependent on their location within the landscape.

Historically, beavers were abundant throughout North America, with estimates ranging between 55 million and 400 million (Seton 1929, Pollock et al. 2003). Their pelt and castoreum was considered of great value and they were overexploited for centuries (Rosell et al. 2005). To reverse the effects of this overexploitation, beavers were protected and in the 1920s reintroduction programs were initiated. Their U.S. population is currently estimated to be 6 to 12 million (Naiman et al. 1986). While populations have increased, their abundance levels are typically 3 to 10% of their historic levels and have been so for some time (Pollock et al. 2003).

Several lines of evidence point to the importance of beaver ponds and side channels as principal habitat features for coho salmon (e.g., Naiman et al. 2000, Pollock et al. 2003). When evaluating habitat for OCCS using HLFM version 7 (Nickelson 1998), reaches with beaver ponds have rearing capacities an order of magnitude higher than reaches without beaver ponds (Beechie et al. 1994). As Pollock et al. (2004) report:

> Watershed-scale restoration activities designed to increase coho salmon production should emphasize the creation of pond and other slow-water environments; increasing beaver populations may be a simple and effective means of creating slow-water habitat.

The Pollock et al. (2004) study focused, for the most part, on sites in the Puget Sound Region; however, the BRT noted that areas where beaver pond density is highest typically have the same physical characteristics regardless of the ecological region—lower gradient (less than 2%), unconfined valley bottoms, in smaller watersheds (drainage areas typically less than 10 km^2). Smaller, lowland, rain-dominated Puget Sound watersheds have the same basic physical and hydrological characteristics as the smaller Oregon coast watersheds, thus the relationships we see with respect to beaver pond densities in Puget Sound should also hold true for the Oregon coast.

Beavers have been recognized as important to OCCS recovery by the State of Oregon in the Oregon Plan (OCSRI 1997) and the Oregon Coast Coho Conservation Plan (Oregon 2007). Notably, the Fisheries Section of ODFW has long recognized the importance of beavers to recovery of OCCS (ODFW 2005h) and is actively working to stress their importance to other sections of their agency as well as other state agencies (ODFW 2009b).

The BRT discussed the importance of beavers to coho salmon on the landscape and considered whether there have been changes that would lead BRT members to consider what loss of beavers would be expected over the next few decades. Two studies were discussed in regard to the present and future status of beavers in the OCCS ESU.

The first was a MidCoast Watersheds Council study (MCWC 2010), which attempts to address anecdotal evidence for major declines in large winter-persistent beaver ponds over the past 2−3 decades. In order to examine the issue, the MidCoast Watersheds Council engaged in a study to quantify trends, not on beaver populations, but the presence and habitat metrics of beaver dams and ponds. This study covered streams in the Upper Five Rivers (Alsea River), Tillamook Basin, Upper Yaquina River, and the rest of the Yaquina Basin. The results show that the Mid-Coast region included in the study has experienced widespread declines in numbers of beaver dams and ponds. Currently, the majority of dams are low and ponds are small and ephemeral. Only 5 of 40 streams surveyed in the Yaquina survey had healthy reaches of beaver habitat, with large, stair-stepped ponds. All five of these streams have difficult public access, with gated roads or no roads.

Another study was pursued in the Tillamook population (Biosurveys 2009); the entire basin was snorkeled with 320 miles of stream, from head of tide to end of coho salmon distribution in five river basins: Tillamook, Trask, Wilson, Kilchis, and Miami, plus small tributaries (tribs) entering the bay directly. These surveys also recorded beaver dams. The same 360 miles snorkeled in 2006 was repeated in 2007. The beaver dam summary was as follows.

Year	Tillamook	Trask	Wilson	Kilchis	Miami	Bay tribs	Total
2006	99	16	7	0	9	5	136
2007	70	23	13	0	7	0	113

The interannual comparison shows a decline in beaver ponds, most importantly in the Tillamook River. The Tillamook River has the proper morphology for extensive beaver colonization and a historical legacy of their presence (Coulton et al. 1996) in many reaches where they are currently absent. Because of limited stream morphology, the remaining Tillamook Basin rivers (Wilson, Kilchis, Miami, and Trask) have limited potential for broad colonization of beaver except for the Devils Lake Fork of the Wilson River. As expected, most beaver activity was found in the low stream gradients and sedimentary geologies. No active beaver dams were found in the Kilchis Basin, which is generally high gradient or highly disturbed.

In the past, ODFW has been able to track the harvest of beaver populations because all trapping required a permit and a harvest report. However, because of a change in the application of state regulations, no permit or harvest report is presently needed for trapping of nuisance animals on private land, making assessment of beaver harvest difficult (ODFW 2005h). As of 2005, an analysis of the data collected in aquatic habitat surveys showed no significant trend in beaver dams in the entire ESU from 1998 to 2003 (ODFW 2005h). Some monitoring areas such as the Umpqua River showed a very low percent of habitat that contains beaver pools.

Based on these limited sources of information, the BRT concluded that there is some evidence for continued concern in regard to beaver abundance, but of very uncertain extent or scope. Due to the limited data set, we cannot conclude that there is an overall trend and would recommend a more extensive monitoring effort be pursued to identify short-term and long-term trends throughout the OCCS ESU. If beaver abundance has in fact declined or does not trend upward in the form of beaver dam density throughout the ESU, the BRT would consider this to be a significant threat to the availability of high quality habitat for OCCS.

The BRT had no information on beaver population trends over time in the ESU, therefore the habitat subcommittee examined the ODFW stream monitoring habitat data for the Oregon coast to gain a better understanding of the overall trends in beaver dam density in the different strata from 1998 to 2009. The habitat subcommittee found that the densities of beaver dams in the surveyed streams for each stratum averaged less than 1 (0.53 dams/km ±0.33) beaver dam per kilometer, with the exception of the North Coast stratum (1.31 ±1.4) (Figure 17). The trend in beaver dam density over this time period is flat (Figure 17). In addition, the density of beaver dams throughout this time period is considerably less than what typically occurs in protected or remote areas throughout North America (Pollock et al. 2003).

There are also some stratum-specific trends that relate to overall habitat condition. For example, the occurrence of beaver dams in the Umpqua stratum was 36% of those streams surveyed between 1998 and 2009, while the occurrence level was 82% for all other strata with the exception of the Lakes stratum (18%). Thus not only were beaver dam densities the lowest (0.09 dams/km ±0.16) in the Umpqua of any strata, they typically were nonexistent in many of

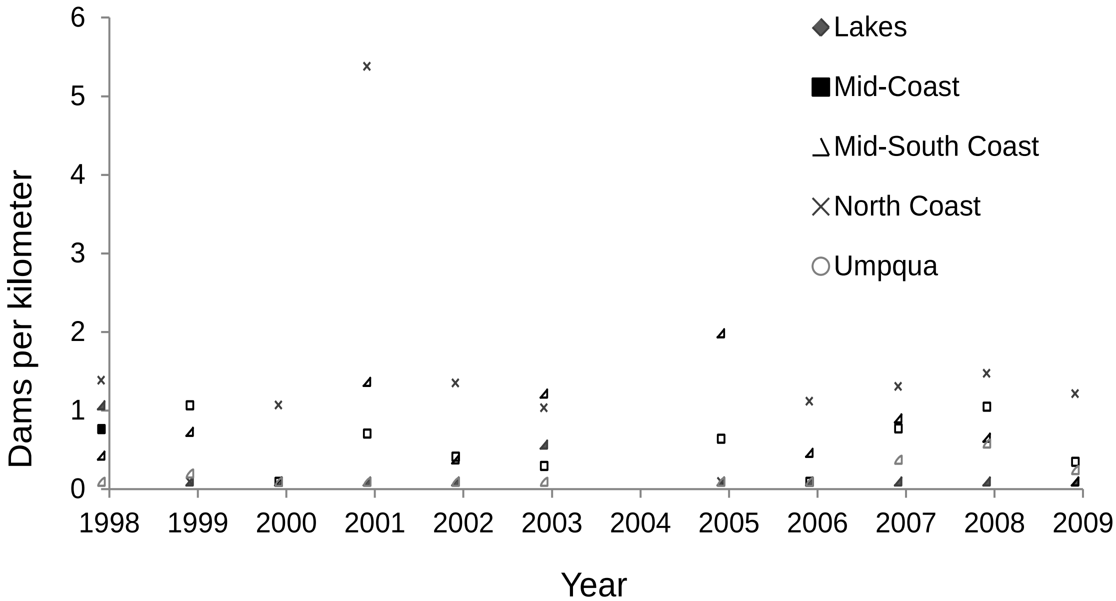

Figure 17. Beaver dam density in the OCCS ESU from 1998 to 2009. Data from Anlauf et al. 2011.

the streams surveyed. Possible causes for the consistently low numbers of beaver dams across the ESU could be natural population fluctuations, forest succession, disease (tularemia), trapping, increased cougar predation, reduced food supply, reduced supply of building materials, or a combination of all.

Pollock et al. (2003) summarized beaver dam density from pristine, remote, and protected areas in the North America and found the average to be 24.9 dams/km (±21.9). A low level of beaver dam density is typically correlated with lower abundance levels of beavers (Pollock et al. 2003). Managed and recovering beaver dam density typically ranges between 2 and 6 dams/km and an average of approximately 3 dams/km (Pollock et al. 2004). Assuming habitat preference, that is, the types of stream characteristics that beavers prefer (less than 4% stream channel gradient, unconfined valleys [greater than 4 channel widths] [Suzuki and McComb 1998, Pollock et al. 2004]), then the density of beaver ponds will vary as a function of the number of beaver colonies and beavers in those areas.

Because the number of empirical studies that assess beaver abundance on the Oregon coast is limited, a brief analysis was pursued based on the published literature. Pollock et al. (2003) identified that remote or protected beaver populations (i.e., where no trapping is occurring and they are either protected with regulations or due to their remoteness) have a density that ranges from 0.4 to 0.9 colonies/km^2, while recovering or managed populations have a range from 0.1 to 0.4 colonies/km^2. The number of beavers per colony ranges from 4 to 8 (Jenkins 1979, Pollock et al. 2007). This means the range of the number of beavers is 1.6 to 6.4 per km^2. Assuming a watershed size of 500 km^2, then the estimate for a beaver population in a pristine or protected area would be between 800 and 3,200 beavers, while in a managed or recovering area the same watershed would be between 200 and 1,600. Pollock et al. (2004) estimated the number of beavers in a pristine environment in a west Cascade watershed at 236 to 473 colonies and a population estimate that ranged from 946 to 3,782 beavers in any given year.

As of 2004, nuisance beavers may be removed by landowners or their agents without permits from ODFW (ODFW 2009b), and trapping is open in its entirety in all the coastal counties, including BLM and USFS lands, with the exception of Curry County (ODFW 2008a). The regulations state the following regarding the Coast Range:

> Attention Coastal Beaver Trappers. ODFW requests your continued cooperation in protecting beaver dams in coastal areas important to coho salmon rearing. If you are not familiar with this program, which was initiated in 1998, please contact your local ODFW biologist (ODFW Furbearer Regulations 2008b, page 2).

Thus while trapping is not promoted and beavers are acknowledged as an important part of the coastal area, only beaver dams are protected in some manner and not the population of beavers that create and maintain their existence (ODFW 2008a). Therefore the range of beaver colonies and the number of beaver in the OCCS ESU would likely fall into the managed, not recovering, and not protected category. This is also evident in the low density of beaver dams per kilometer from 1998 to 2009.

The effect of past declines in beaver dams in the OCCS ESU is probably manifest in the current biological status of the species, because beaver-created habitat degrades rapidly in the absence of active beaver populations (Naiman et al.1988). The combination of one agency promoting the importance of beavers with the lack of any protection for beavers on private lands and minimal or no requirements for monitoring the take of beavers makes it extremely difficult to predict the abundance of beavers in the future compared to current levels. Despite this uncertainty, the BRT was concerned that lack of protection for beavers could result in a potential decline of this important habitat-forming species, with continued low levels of beaver dams and potentially resultant declines in the abundance of high quality OCCS habitat. The BRT concluded that a lack of protection of beavers and degraded beaver dam density levels is an ongoing threat to OCCS that is not fully manifest in the current biological status of the species.

There is an important point to consider: even if beaver populations and beaver ponds increase in the Oregon coast, other ecological constraints need to be incorporated into the management actions of beaver ponds. One particular concern is the invasion of reed canary grass in ponded areas. The cycle of beaver impoundment and abandonment disrupts the native community and provides an ideal environment for reed canary grass, which once established tends to exclude development of herbaceous communities and limits vegetation species richness (Perkins and Wilson 2005). Strong management actions to control reed canary grass are most likely needed to reestablish native flora that is considered beaver food, which may be relatively expensive and a slow process (Healy and Zedler 2010, Hoffman 2010).

Roads—A number of studies have found negative correlations between road density and coho salmon productivity. Bradford and Irvine (2000) found that the rate at which individual coho populations declined between 1988 and 1998 in the Thompson River, British Columbia, was related to the extent of agricultural and urban land use and the density of roads in the watershed. An increase in road density was correlated to an increase in coho salmon population decline. The road densities in the Bradford study ranged 0−2 km/km^2 compared to 1.5−4 km/km^2 in the OCCS ESU (Figure 18). The road densities for the OCCS ESU in Figure 18 are an underrepresentation of actual road densities in the ESU because industrial forest land roads are not included in the data set.

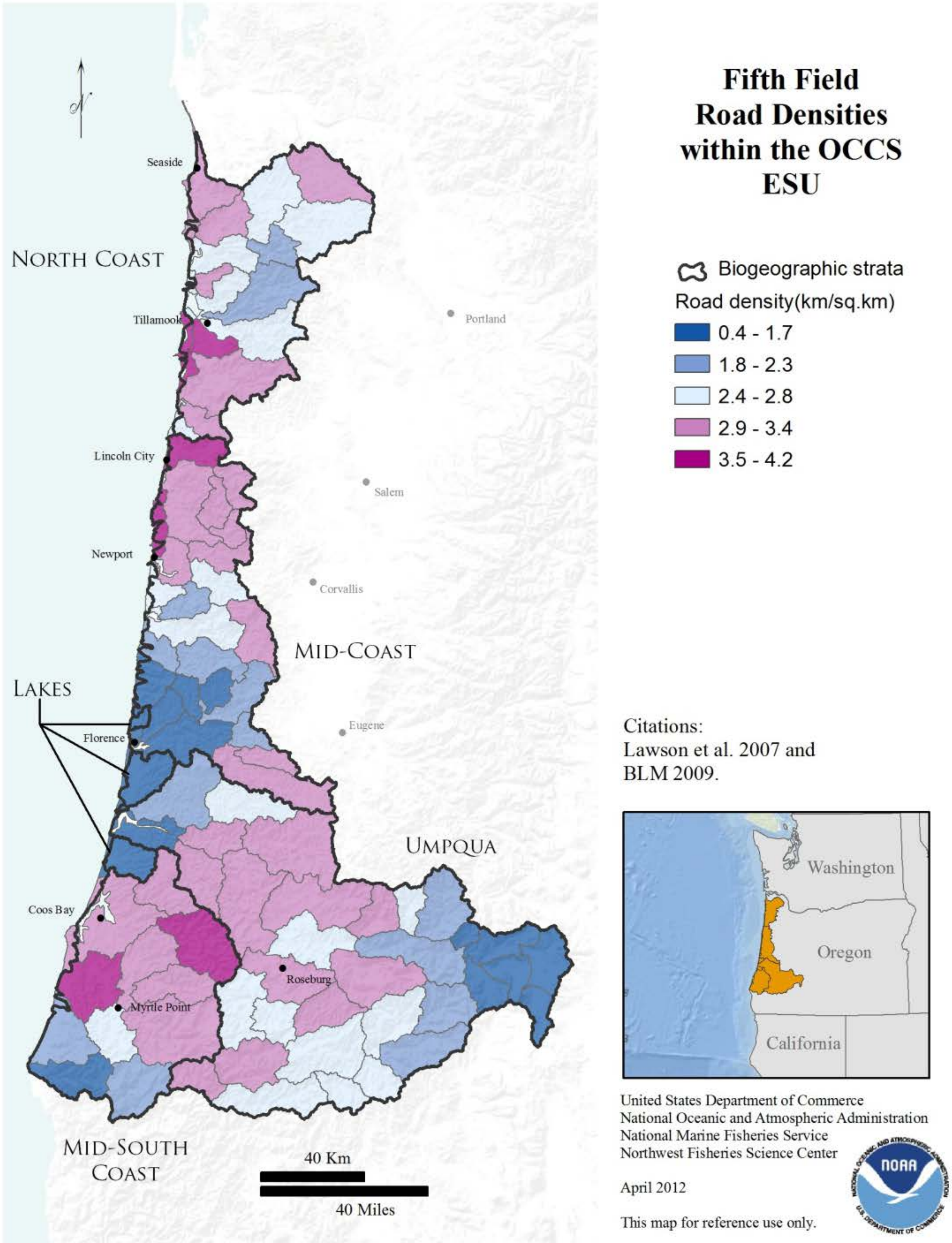

Figure 18. Active road densities by fifth-field hydrologic unit. These include only partial road densities from private lands.

Sharma and Hilborn (2001) found that lower valley slopes, lower road densities, and lower stream gradients were correlated with higher smolt density in 14 western Washington streams between 1975 and 1984. The results suggest a decrease of 500 smolts/km for each 1 km/km^2 increase in road density. If road densities affect Oregon streams similarly, they could have a significant effect on OCCS smolt production in much of the ESU (Figure 18).

Pess et al. (2002b) also found a negative relationship between road density and the number of fish-days[18] for coho salmon over time in the Snohomish River basin, Washington. Most of the negative correlation was the result of urbanization and impervious surface. Urbanization can lead to an increase in impervious surface area and increase stream flooding frequency and magnitude (Hollis 1975). The preurbanized 10-year recurrence interval flow event can occur every 2–5 years in urbanized areas of the Puget Sound region (Booth 1990), which can lead to declines in adult coho salmon (Moscrip and Montgomery 1997).

In a study of the tributaries of Elk River, Oregon, Burnett et al. (2006) found that density of large wood in pools was negatively correlated with road density. Road density was also negatively correlated with forest cover, and at the scale they examined, may integrate the impacts of timber harvest associated with the road network.

At the Oregon Coast Coho Salmon Symposium, Chris Frissell of PRC presented information on known road densities throughout the OCCS ESU and related those to the properly functioning condition defined for bull trout in the Columbia Basin (see New Comments section). His hypothesis was that with the high road densities that are known and included in the BLM roads GIS layer and the probable road densities that are not known,[19] road density in the OCCS ESU is probably very high and constitutes risk to OCCS as he has shown that road densities affect bull trout.

The effects of current road densities may not yet be reflected in the biological status, as existing and legacy roads can contribute to continued stream degradation over time through restriction of debris flows, sedimentation, restriction of fish passage, and loss of riparian function. Future land management actions in forest, agriculture, and urban settings with their resultant additions to the roads network have the potential to contribute to future reductions in OCCS populations and could constitute a future threat.

Nonindigenous plant species—Another aspect of human disturbance that can affect stream habitat complexity has been identified in the Oregon Conservation Strategy (ODFW 2011) and in the USFS Pacific Northwest Region Invasive Plant Program (NMFS 2005b, USFS 2005). Invasive nonnative species can be powerful disrupters of native plant and animal communities. Two examples of how exotic plants can affect stream complexity are those of giant knotweed (*Polygonum sachlinense*) and reed canary grass. Giant knotweed displaces regenerating alder and conifer trees in riparian areas (Urgenson et al. 2009) and reed canary grass prevents regeneration of willow and alder, species that may affect physical stream complexity but are also food items for beaver use (Perkins and Wilson 2005, Healy and Zedler 2010).

[18] Fish-days were calculated by multiplying the live fish observed on each survey date by the number of days between surveys. These values were then summed for the entire observation period to generate a relative index of spawner abundance at a reach for any given year.
[19] Industrial forest land road density data sets are not generally available and therefore not included in the GIS layer.

Human landscape disturbance—The condition of aquatic ecosystems and associated fish populations are a function, at least in part, of the characteristics of the surrounding landscape (Frissell et al. 1986, Naiman et al. 2000, Gregory et al. 2008). Timber harvest and associated roads have extensively altered aquatic ecosystems throughout the Pacific Northwest (Everest and Reeves 2007). A consequence of the effects of timber harvest activities is that the behavior of ecosystems is altered, which in turn has consequences for fish populations and their habitat (Reeves et al. 1993). There is a negative association between the amount of in-channel large wood and percent of area intensively logged in a watershed (Murphy and Koski 1989, Bilby and Ward 1991, Montgomery et al. 1995). Burnett et al. (2006) found that that the mean density of large wood in Elk River (on the southern Oregon coast) was positively related to the area in larger trees in the catchment. Reeves et al. (1993) examined watersheds in the Oregon Coast Range and found that the diversity of the fish assemblage and the amount of large wood was significantly greater in streams in which less than 25% of the watershed was clear-cut, compared to watersheds in which more than 25% of the area was clear-cut. This pattern was observed in other areas for other land uses including agriculture (Berkman and Rabini 1987) and urbanization (Scott et al. 1986).

The condition of aquatic habitat and fish populations is also directly correlated with the density of roads in a watershed, which in turn is generally directly related to the amount and intensity of land management activities (Lee et al. 1997). Roads are sources of sediment either as surface erosion or as mass erosion (Furniss et al. 1991). They also can alter water delivery by increasing the drainage network, particularly in the upper portions of the network. Sharma and Hilborn (2001) examined 14 streams in Washington and found that smolt density was inversely correlated with the density of roads. Logging activities involve the creation and maintenance of roads and logging has been linked directly to increased sediment levels in streams (Platts et al. 1989).

Despite the connection between human disturbance and fish habitat and population performance, the ONCC TRT (Wainwright et al. 2008) was unable to include habitat condition directly in its biological recovery criteria. Therefore, habitat condition was not included in the DSS analysis because at the time there was no uniform measure of habitat quality over the entire ESU. ODFW habitat surveys were available, but the density and distribution of on the ground surveys made them unsuitable for fine-scale analysis needed for biological recovery criteria.

Recent public availability of Landsat imagery and the development of tools for analysis now make it possible to analyze human disturbance patterns on a fine temporal and spatial scale. Satellite images have the potential for measuring properties of large landscapes at a relatively fine scale. In an analysis conducted for the BRT, satellite annual vegetation maps of the OCCS ESU were updated through 2008 and analyzed for patterns of disturbance for the time period 1986 to 2008. The scale of resolution of these analyses is approximately 100 m, so disturbances as small as 1 ha can theoretically be detected. This made it possible to detect individual disturbance events from the satellite images and map new disturbances on an annual basis. Intensity of disturbance can also be measured, so low-intensity disturbances (i.e., thinning) can be distinguished from high-intensity disturbances (i.e., clear-cut). Fires were also mapped, but fire has had a small role in shaping habitat in the OCCS ESU over the past 23 years (for more information on methods, see Appendix B and Kennedy et al. 2010).

Human disturbance was widespread over the ESU and predominantly of high intensity (Figure 19). Disturbance patterns varied over space, time, and land ownership. Some river systems have experienced higher disturbance than others (Figure 20). The time series of cumulative disturbance, derived from Landsat images, is shown for four major river systems in the OCCS ESU in Figure 21. Disturbance in these systems spans the range observed in the ESU, from a low of 10% (Upper Nehalem) to 50% (Siletz). Most disturbance is in the high category, with a lesser amount of low intensity disturbance, and the proportion of high to low disturbance is fairly constant through the time period. Three patterns of disturbance are evident (Figure 21). Constant rates of annual disturbance occurred in the Siletz and Alsea river systems. The Necanicum River basin showed little harvest activity early in the time period, with an exponential increase in more recent years. By contrast, the North Umpqua River basin showed an overall low rate of disturbance, with most of the activity early in the time period.

Changes in the regulatory environment have largely driven patterns of land disturbance over the past two decades. Disturbance in four land ownership categories in the Alsea River basin is shown in Figure 22. Prior to 1990, there were high rates of disturbance on federal lands (BLM and USFS). With implementation of the Northwest Forest Plan, federal logging activity decreased to very low levels. Logging on private lands proceeded at a steady pace through the late 1990s, then increased. This general pattern is evident throughout the ESU, with logging on National Forest and BLM land decreasing after 1989 and activity on private lands increasing.

Caution is needed in interpreting the implications of the vegetative disturbance on OCCS and their freshwater habitat. Other researchers have found relationships between landscape characteristics and the condition of habitat of coho salmon (Pess et al. 2002a) and other species of Pacific salmon (e.g., Steel et al. 2004). Hicks and Hall (2003) noted that discerning the effects of timber harvest on fish and fish habitat in the Oregon Coast Range was particularly difficult because of the inherent variability in rock types and associated stream features. The use of Landsat imagery to assess the rate of vegetative change and the extent of disturbance does not assess the impacts of timber harvest on populations and habitat of coho salmon. However, it can provide insights into the potential effects and the extent of the impacts across the ESU, which in turn has implications for the assessment of the status of the ESU.

Because of limits on time and resources, the BRT was unable to conduct an analysis to determine whether there is a relationship between the rates of vegetation disturbance and changes in the abundance of coho salmon and habitat conditions. However, this information on rates of disturbance can be a potential indicator of current and, at least in part, future habitat conditions. The diversity of the salmonid assemblage and the amount of large wood in the channel is related to the amount of timber harvest in watersheds in the Oregon Coast Range. Once large wood is removed from upslope areas, recruitment to streams is reduced, perhaps for hundreds of years. Legacy effects of logging will be felt as long as existing wood in streams is decaying faster than it is being replaced (Reeves et al. 1993).

There is recognition in the scientific literature about the importance of periodic disturbances for creating and maintain fish habitat (Naiman et al. 1992, Reeves et al. 1995, Rieman et al. 2006). Timber harvest can alter the disturbance process and ultimately have negative rather than positive consequences to fish populations and habitat (Reeves et al. 1995, Bisson et al. 2009, Cover et al. 2010). Naturally occurring disturbance events were important

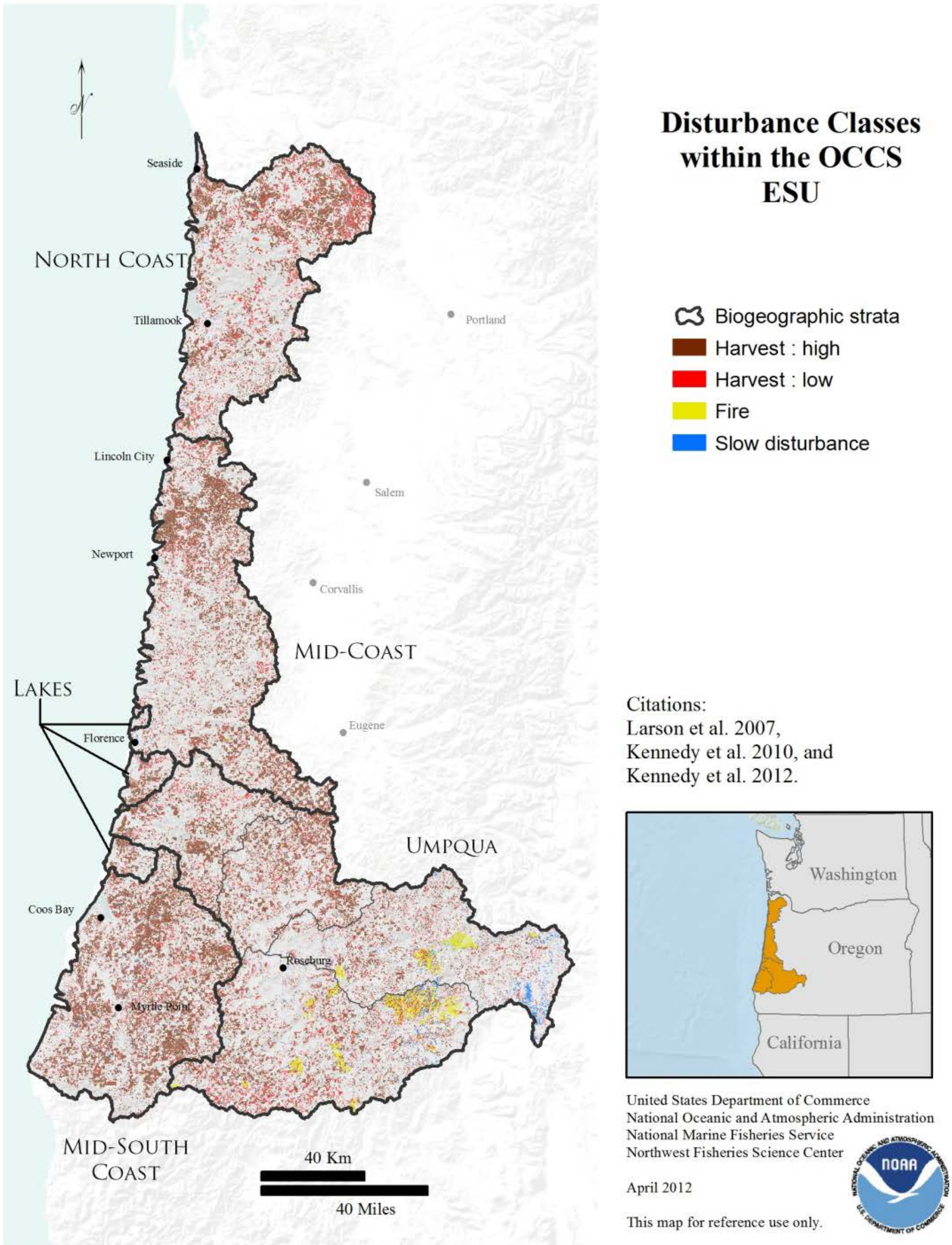

Figure 19. Distribution and intensity of vegetation disturbance from 1986 to 2008, based on analysis of Landsat imagery.

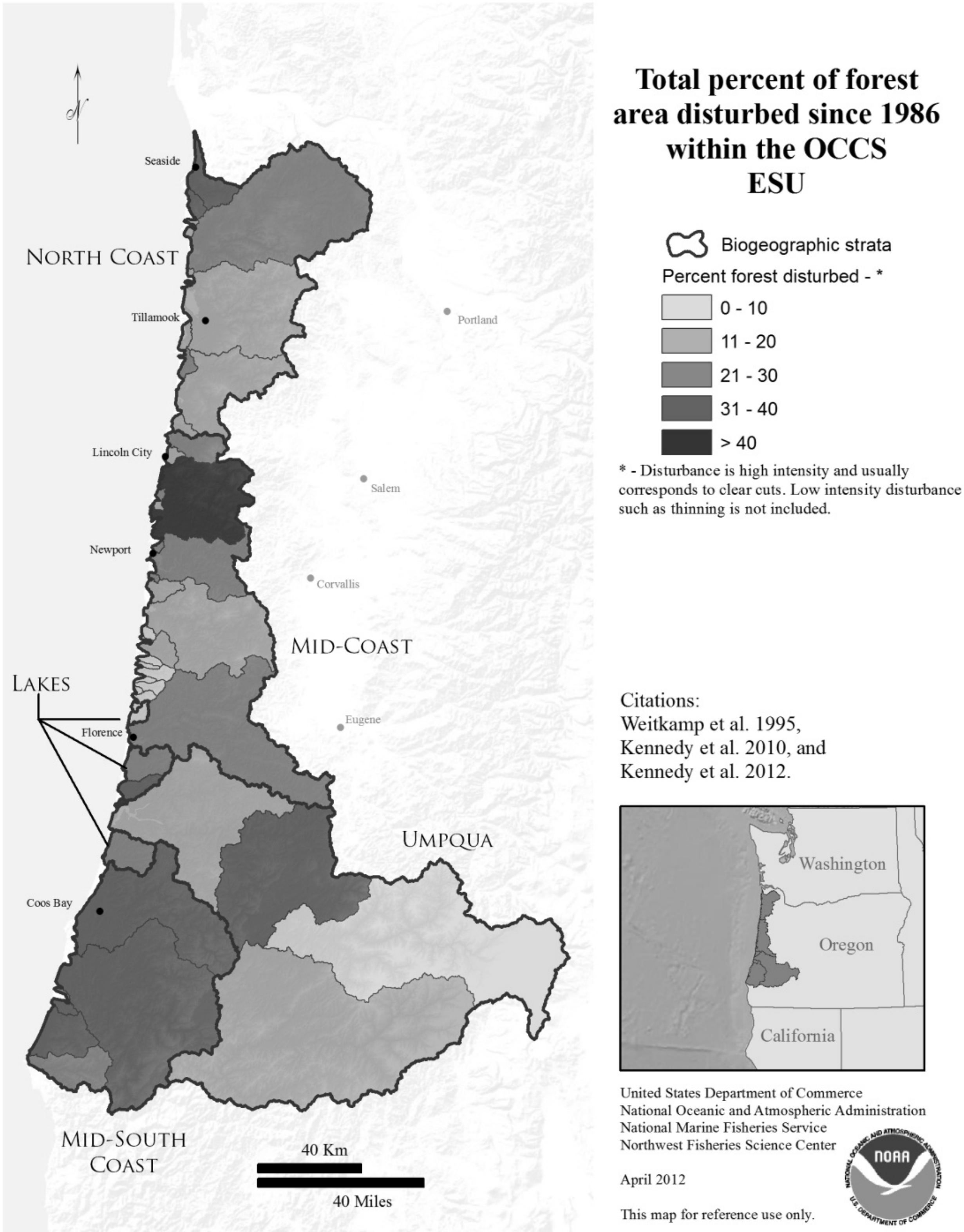

Figure 20. Ranking of river basins and the Umpqua subbasins by cumulative percent vegetation disturbance from 1986 to 2008.

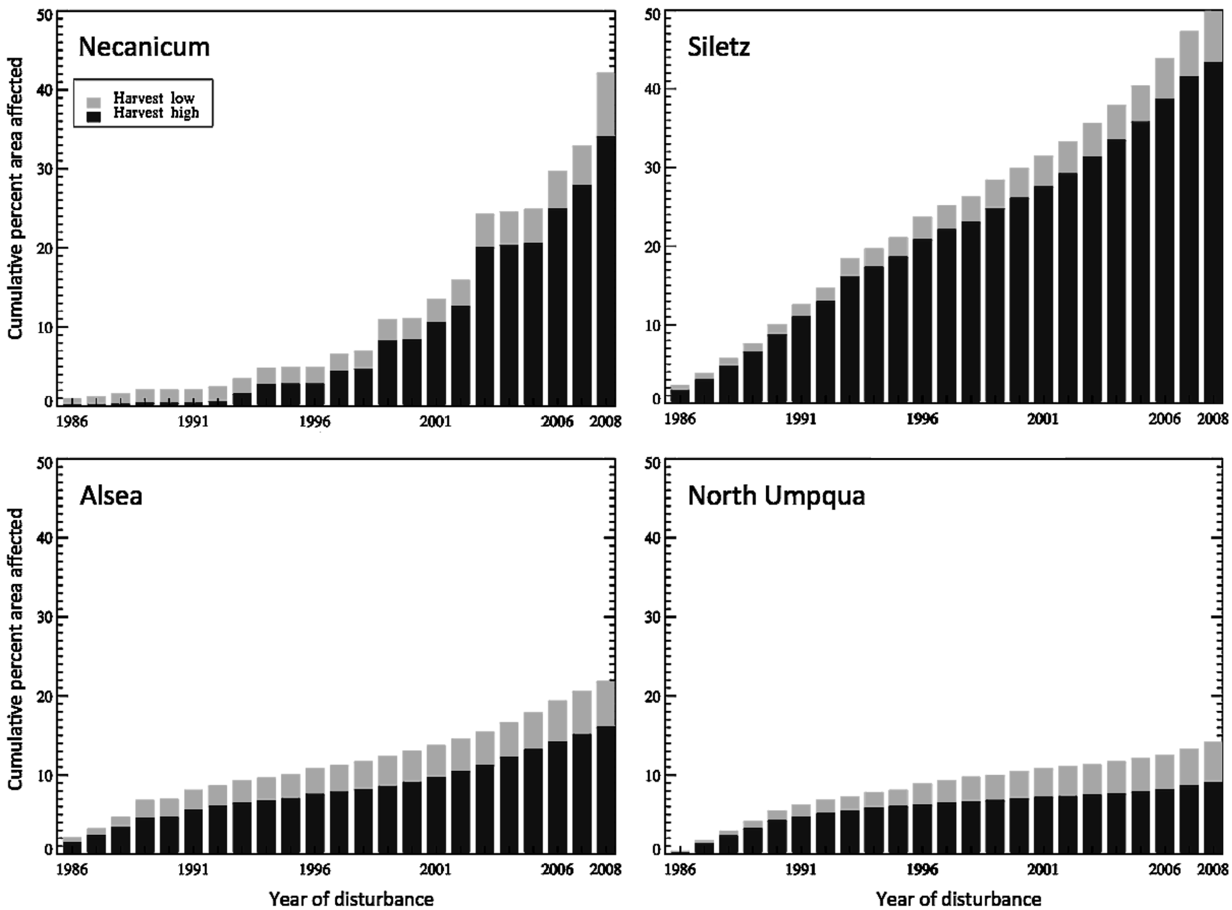

Figure 21. Time series of cumulative area of vegetation disturbance for four river basins in the OCCS ESU. High disturbance (dark gray) is usually clear-cut logging, while low disturbance (light gray) is related to forest thinning. Data from Kennedy et al. 2010 and Kennedy et al. 2012.

sources of sediment and large wood, the basic structural components of habitat. This is particularly true in the Oregon Coast Range (Reeves et al. 1995, May and Gresswell 2003, Reeves et al. 2003, Bigelow et al. 2007). Disturbances associated with timber harvest, primarily landslides and debris flows, have less large wood associated with them than those that occurred naturally (Hicks et al. 1991, Lancaster et al. 2003). The loss of wood results in decreased habitat quantity and quality (Reeves et al. 1995, Cederholm et al. 1997).

Burnett et al. (2007) suggested widespread recovery of coho salmon in the OCCS ESU is unlikely unless habitat improved in areas of high intrinsic potential on private lands. Timber harvest effects on fish and habitat are likely most pronounced on private and state lands. Requirements for management of riparian zones on these lands are less than on federal lands. Current forest practice regulations reduce the size of the streamside riparian area to less than that needed to maintain the full suite of ecological processes provided by riparian areas and allow for removal of trees from within this zone, which further reduces ecological effectiveness. Additionally, there is no requirement for protection on small intermittent streams, which are important sources of wood (May and Gresswell 2003, Reeves et al. 2003, Bigelow et al. 2007) on private lands. These streams are given consideration on a portion of each stream on state

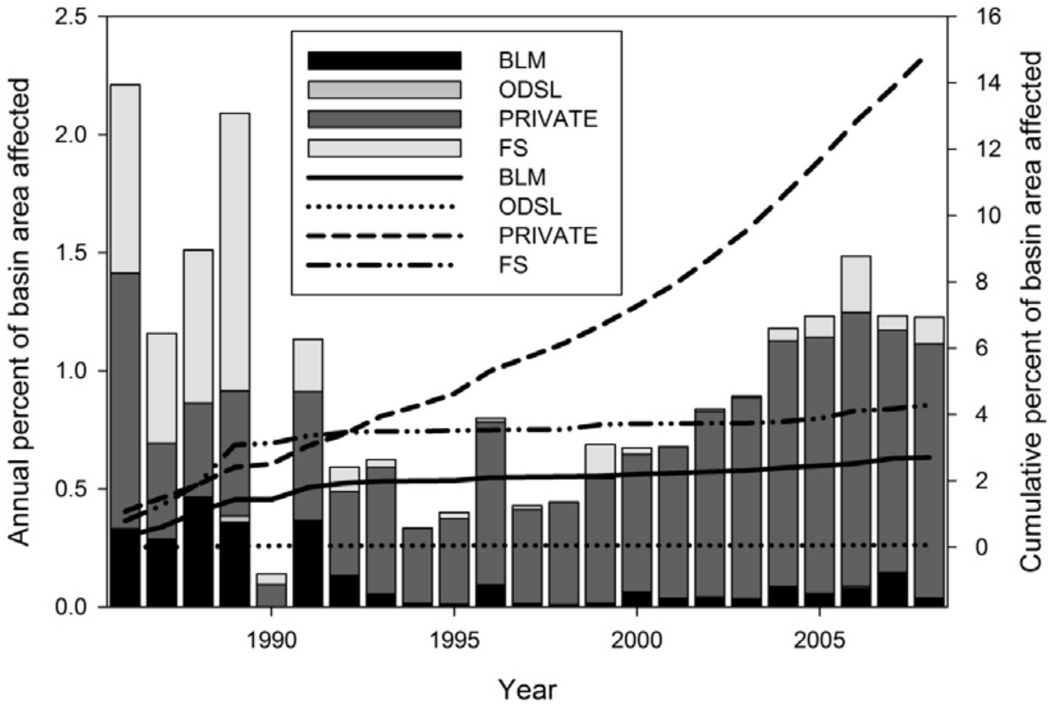

Figure 22. Total area (hectares) of vegetation disturbance in the Alsea River area of the OCCS ESU by four land ownership categories from 1986 to 2008. BLM = Bureau of Land Management timberlands, ODSL = Oregon state timberlands, Private = industrial and nonindustrial private timberlands, and FS = U.S. Forest Service. Data from Kennedy et al. 2010 and Kennedy et al. 2012.

lands. Botkin et al. (1995) and the Independent Multidisciplinary Science Team (IMST 1999) found these regulations insufficient to improve or recover habitat that is currently degraded.

The recent availability of Landsat images, along with the development of tools for analysis, allowed a comprehensive, uniform picture of human disturbance patterns that was previously unavailable. This analysis showed that disturbance has been widespread in the ESU, some basins experienced much higher disturbance than others, rates of disturbance are relatively constant, and the most intense disturbance has moved from federal to private lands, presumably in response to policy changes. The BRT thought that human landscape disturbance is captured somewhat in the current biological status, but the effects of human landscape disturbance constitute an ongoing threat to OCCS.

Loss/gain of large wood for future habitat conditions—Large wood is a key component of habitat complexity for coho salmon in the OCCS ESU. This wood is recruited from riparian areas immediately adjacent to the stream and from upslope sources, primarily along smaller, nonfish-bearing streams (Reeves et al. 2003). Currently wood is lacking in many streams in the OCCS ESU because of past management activities.

Burnett et al. (2007) examined the current and future condition of riparian areas along streams with coho salmon within the entire ESU. Thirty-six percent of the stream length available to coho salmon were classified as high intrinsic potential (see Glossary). The vast majority of that (81%) was primarily on nonindustrial private lands. Forty-four percent of the riparian areas along streams with high intrinsic potential are currently either nonforested or recently logged; 10% have stands that are dominated by large (50−75 cm quadratic mean diameter) or very large (>75 cm quadratic mean diameter) trees. The large and very large trees are the size that creates more complex habitat conditions (Abbe and Montgomery 1996) and are found almost entirely on federal lands, which have a relatively small proportion of the high coho salmon intrinsic potential streams (Burnett 2007).

The percentage of buffers with large and very large trees is projected to increase to at least 75% on federal lands and 60% on state lands in 100 years under current policies. Less than 25% of the buffers in private ownership will have vegetation in these size classes at the end of that time. As a result, Burnett et al. (2007) concluded that widespread recovery of habitat in high intrinsic potenital streams, a key element of future OCCS habitat recovery, is unlikely unless there are greater improvements on private lands.

The likelihood of complex stream habitat recovery for coho salmon in the ESU is potentially further limited because of the lack of or limited requirements to consider nonfish-bearing streams on private and state lands, respectively, in current management policies. Reeves et al. (2003) found that 65% of the large wood in Cummins Creek, a small watershed in a federally designated wilderness area on the central Oregon coast, originated in areas outside of the stream-adjacent riparian zone. Bigelow et al. (2007) found that wood delivered in debris torrents in nonfish-bearing streams was a key component of habitat in a sandstone watershed on the central Oregon coast. Thus the potential of landscape and local scale stream complexity in habitat for coho salmon in the ESU to improve is likely to be less than what Burnett et al. (2003) concluded, because current policies guiding the management of riparian areas on state and private lands have limited or no management requirements for this important potential source of wood. The BRT thought that the loss of large wood from streams in the OCCS ESU is captured to some degree in the current biological status, but the effects of continued loss and lack of replacement of large wood in areas that can contribute to stream complexity constitutes an ongoing threat to OCCS.

In-channel habitat complexity—Since the original status review, the ESU has experienced an increase in abundance and productivity that largely reflects improved marine survival conditions. This increase may have reduced short-term risks to the ESU; however, the BRT was also concerned that freshwater habitat may not be sufficient to maintain the ESU at times when marine conditions are poor. The BRT also noted that the criteria in the DSS do not take advantage of some important habitat monitoring data. To address the latter deficiency and generally evaluate trends in freshwater habitat, the BRT in collaboration with ODFW conducted some additional analyses of trends in the freshwater habitat attributes of this ESU.

From 1998 to the present, ODFW has monitored wadeable streams to assess freshwater rearing habitat for the OCCS ESU during the summer low flow period (Anlauf et al. 2009). The goal of this program is to measure the status and trend of habitat conditions throughout the range of the ESU through variables related to the quality and quantity of aquatic habitat for coho salmon: stream morphology, substrate composition, instream roughness, riparian structure, and winter rearing capacity (Moore et al. 2008). In 2009 ODFW and NMFS scientists independently analyzed these data to ask the question: Has juvenile coho habitat changed over the past 11 years? These analyses reached somewhat different conclusions. In particular, the Anlauf et al. (2009) analysis generally indicated that there were no significant trends in habitat attributes (either positive or negative) across the ESU, while the 2009 NMFS analysis indicated declining trends in some measures of habitat quality across several regions of the ESU.

To better understand and resolve the discrepancies between these analyses, NMFS and ODFW formed a joint Habitat Trends Working Group (HTWG). The HTWG determined that the differences in results were caused primarily by the use of two different data sets and the use of slightly different statistical models of data analysis. In particular, the Anlauf et al. (2009) analysis focused on data only from within the spawning and rearing distribution of OCCS, while the NMFS team used a data set that also included habitat sites upstream from these areas. In addition, the Anlauf et al. (2009) analysis used a statistical model with some parameters that were supported by the survey design but that the NMFS analysis found as currently unsupported by the data.

The HTWG (Appendix C) estimated trends for five habitat metrics—1) Aquatic and Riparian Effectiveness Monitoring Program (Reeves et al. 2004, 2006) channel score, 2) summer parr capacity, 3) winter parr capacity, 4) percent of riffle that is sand/silt/organics, and 5) volume of large woody debris (LWD) per 100 m (Table 16). The first three metrics are multivariate measures of habitat complexity and capacity, and the latter two are univariate metrics of habitat condition that are reasonably well understood to indicate quality of coho salmon rearing habitat. The HTWG used three statistical models to evaluate trends. Two models were forms of a maximum likelihood–based analysis of variance linear model and the third was a similarly structured linear model in a Bayesian framework (see Appendix C for model details). The analyses were conducted on two nonoverlapping data sets, one limited to data collected within coho rearing habitat and the other from habitat monitoring locations upstream and inaccessible to coho salmon, but potentially important nonetheless as determinants of downstream habitat-forming processes (e.g., source areas for wood and sediment). For the data from areas upstream of coho rearing habitat, only the two univariate metrics (LWD and percent sand) were evaluated. Overall, the predicted spatial and temporal patterns in multiple metrics of fish habitat were very similar between the multiple models (Appendix C). Therefore, to simplify the presentation, here we discuss just the results from the Bayesian modeling framework, since this approach provides more explicit information on the relative certainty/uncertainty of trends in the data.

The results from the coho rearing areas are summarized in Table 17. Trend estimates are mixed and vary among metrics and population strata. Positive indications of habitat condition change include trends of decreasing fine sediment levels in the North Coast and Mid-Coast, increasing wood volume in the Mid-Coast and Mid-South Coast, and increasing habitat complexity (as indexed by channel score) in the North Coast and Mid-Coast. However, the monitoring data also show that habitat conditions are declining in some regions for some metrics. For example, fine sediment levels are increasing in the Mid-South Coast and wood volume is decreasing in the North Coast and Umpqua. Similarly, habitat complexity, in this case as indexed by channel score and summer and winter parr capacity, is declining in the Mid-South Coast (winter parr capacity and channel score) and the Umpqua (winter and summer parr capacity).

Table 16. Graphical representation of the maximum likelihood analysis and Bayesian analysis trend results. Arrow style indicates strength of trend: black vertical arrow represents greater than 90% Bayesian probability or significance ($P < 0.05$) of trend; light gray vertical arrow represents greater than 65% Bayesian probability of trend; horizontal gray arrow represents lower (<65%) Bayesian probability of trend or no significant trend detected (maximum likelihood). Upward pointing arrow indicates a positive trend and downward pointing arrow indicates a negative trend.

	Stratum											
	North Coast			**Mid-Coast**			**Mid-South**			**Umpqua River**		
	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities
Winter parr	↔	↔	↔	↔	↔	↔	↔	↔	⬇	↔	↔	⇩
Summer parr	↔	↔	⇧	↔	↔	⬆	↔	↔	⬆	↔	↔	⇩
Channel score	↔	↔	⇧	↔	↔	⇧	⬇	⬇	⬇	↔	↔	↔
Wood volume	⬇	⬇	⬇	↔	↔	⇧	↔	↔	⬆	↔	↔	⬇
% fine sediment in riffles	↔	↔	⬇	↔	↔	⇩	⬆	⬆	⬆	↔	↔	↔

Table 17. Trend analysis summary for five habitat metrics by region, including only sites designated as coho spawning/rearing habitat. The Bayesian posterior mean (estimate) is given for each metric-region combination. As a measure of uncertainty, the probability of a negative trend (proportion of the posterior distribution < 0) is given for each estimate. AREMP = Aquatic and Riparian Effectiveness Monitoring Program, LWDVOL = volume of LWD, RIFSNDOR = percent of fine sediment/organic matter in riffles.

Metric		**North Coast**	**Mid-Coast**	**Mid-South Coast**	**Umpqua**
Winter parr	Estimate	0.000	0.000	–0.025	–0.013
	Pr < 0	0.550	0.640	0.950	0.790
Summer parr	Estimate	0.009	0.027	0.030	–0.028
	Pr < 0	0.320	0.080	0.090	0.880
AREMP	Estimate	0.006	0.006	–0.025	0.003
channel score	Pr < 0	0.170	0.180	1.000	0.360
LWDVOL	Estimate	–0.062	0.016	0.020	–0.026
	Pr < 0	1.000	0.110	0.090	0.940
RIFSNDOR	Estimate	–0.010	–0.004	0.026	0.002
	Pr < 0	0.970	0.760	0.000	0.400

Across the OCCS ESU in the coho salmon rearing areas, habitat conditions show positive and negative trends. These trends vary in magnitude and are supported to varying degrees by the data (e.g., probability of trend being non-zero). In contrast to the coho rearing areas, trends in upstream areas were more pronounced. In particular, LWD declined substantially in all regions. Trends in sediment were mixed, with increases in the Mid-Coast and Mid-South strata and declines in the North Coast and Umpqua strata (Table 18). However, the fundamental take home message from the data analysis is that the data do not indicate a clear, consistent pattern of habitat improvement or degradation over the ESU for these metrics and this time frame.

The BRT was impressed with the ODFW habitat monitoring program and believes that it is a valuable source of information on freshwater habitat trends on the Oregon coast. The results from the HWTG were encouraging in that they resolved some clear discrepancies between earlier analyses. The BRT concluded that the results paint a complex picture of habitat trends along the Oregon coast. Some trends, such as the increase in habitat complexity and summer parr capacity in three of the four regions, were clearly encouraging, but the declining trends in winter parr capacity (believed to be a limiting life stage for coho salmon production) in two strata concerned the BRT. Other trends, such as the declines in LWD in the North Coast and Umpqua strata and in upstream areas in all strata, appear more troubling. The North Coast trend in LWD may be a result of large debris dams that formed during the 1996 floods and have been actively redistributed over the past several years, reducing overall LWD densities. While the North Coast stratum experienced a large decline in LWD frequency in the past decade, it is important to note that it also had the largest amount of LWD relative to the other strata.

Summary of stream habitat complexity—Stream habitat complexity at landscape and local scales has been identified as a factor for decline (OCSRI 1997, NMFS 1997a), a key limiting factor (OCSRI 1997, Anlauf et al. 2009), and a primary limiting factor (Oregon 2007) for OCCS. Complex stream habitats are diverse and dynamic. Complexity is maintained

Table 18. Trend analysis summary for two habitat metrics by region, using only sites that are not designated as coho salmon spawning/rearing habitat. The Bayesian posterior mean (estimate) is given for each metric-region combination. As a measure of uncertainty, the probability of a negative trend (proportion of the posterior distribution < 0) is given for each estimate.

Metric		**North Coast**	**Mid-Coast**	**Mid-South Coast**	**Umpqua**
RIFSNDOR	Estimate	–0.008	0.009	0.009	–0.007
	Pr < 0	0.94	0.1	0.04	0.93
LWDVOL	Estimate	–0.041	–0.031	–0.058	–0.027
	Pr < 0	1	0.95	1	0.97

through connection to the surrounding landscape and it has been well established that a century and a half of land use activities have simplified Oregon coastal streams (Reeves et al. 1993, 1995, Burnett et al. 2007). Because of its importance to the status and recovery of the species, the BRT considered multiple aspects of this issue. These included legacy effects of splash dams, log drives, and stream cleaning; beaver status and management; road densities and their effects on coho smolt densities; disturbance, large wood in riparian zones; and trends in landscape and local stream complexity across the ESU (Naiman et al. 1988, Maser and Sedell 1994, Bradford and Irvine 2000).

The BRT habitat subcommittee analyzed the complexity of available freshwater habitat using multidimensional stream complexity metrics developed by ODFW's Oregon Plan coho salmon monitoring program (HLFM winter and summer parr capacity). The subcommittee analyzed channel score and parr capacity metrics that were constructed from the ODFW stream habitat monitoring data sets. Newly available Landsat data were also analyzed to examine anthropogenic disturbance to the landscape of the OCCS ESU. Other impacts such as roads were discussed with reference to their effects on coho smolt densities from Washington and British Columbia (Bradford and Irvine 2000). Legacy effects of splash dams and stream cleaning and current and future condition of large wood in riparian areas were discussed with respect to the availability of wood for stream complexity. Indications as to the present and future status of beavers were examined through beaver studies that occurred in the ESU and an analysis based on published literature.

Even though splash damming, log drives, and stream cleaning are no longer practiced or endorsed by ODFW, legacy effects of these activities still affect the amount and type of wood and gravel substrate available and, therefore, stream complexity across the ESU (Miller 2010, Montgomery et al. 2003). Increasing complexity would indicate that these legacy effects are being mitigated as wood and gravel move into the stream channel. The resulting channel would be more hydraulically diverse, with pools, side channels, and backwater units that support higher summer and winter capacity for spawning and rearing. Eleven-year trends of stream complexity metrics were analyzed at the level of the stream, population, and stratum (HLFM version 7). Similar to the ODFW/Anlauf et al. (2009) trend analysis of individual habitat attributes, the subcommittee's analyses found that habitat complexity across the ESU exhibited no consistent trends over the period of consideration (1998−2008). There are exceptions, such as increasing summer parr capacity in the Mid-South or Mid-Coast strata. But for any metric or any stratum, no trends were identified.

To help understand these patterns, the BRT examined several other lines of evidence. Clear-cut logging removes wood from upslope, disturbs the riparian zone (Montgomery et al. 2003), reduces the amount of large wood available to the streams, and interferes with processes that generate complexity (Reeves et al. 2003, Burnett et al. 2007). Use of Landsat images allowed the BRT to look at patterns of clear-cutting and thinning from 1986 to 2009. Timber harvest and other land use activities were widespread throughout the ESU, with about 40% of the total forest area experiencing anthropogenic disturbance in the 23-year period. Timber harvest rates varied by basin, but there was no evidence of a general reduction in the pace of logging. The cumulative percentage of forest clearing by basin was highest in the Siletz Basin, followed by the Necanicum to the north, and Coos, Coquille, and mid-Umpqua to the south. The most striking change was a shift in impacts from National Forest land to private industrial land.

The patterns of simplification of stream habitat and reductions in salmon habitat capacity caused by forestry activities are consistent with other information (ODFW 2005b, 2009a) that indicate low levels of large wood (Burnett et al. 2006) and high levels of sediment (Lee et al. 1997) in streams of the Oregon Coast Range. The BRT considered the long-term (multiple decades) effects of logging activities and associated road building on stream conditions, the widespread occurrence of these activities, and lack of any sign that logging activities are abating as indications that these threats to habitat are pervasive and ongoing in the OCCS ESU.

Beavers are an important species to proper watershed functioning in coastal Oregon streams, and the loss of beavers and their dams has been identified by ODFW (OCSRI 1997, ODFW 2007) and many other authors as an important loss to stream complexity that significantly affects OCCS. Because ODFW has only aquatic habitat survey data from which to infer beaver populations and structures, knowledge of what could be a significant contributor to OCCS recovery is severely limited; however, continued loss of this important keystone species constitutes a continuing risk to stream complexity and impediment to habitat improvement.

In summary, habitat complexity across the ESU did not improve over the period of consideration (1998−2008). Road densities are high and affect stream quality through hydrologic effects like runoff and siltation and by providing access for human activities. Beaver activities, which produce highly productive coho salmon rearing habitat, appear to be reduced, and recovery of beaver populations could be impaired by their classification as a nuisance species. Stream habitat restoration activities may be having a short-term positive effect in some areas and passive efforts to restore landscape condition may be effective on much longer time periods than is considered here, but the quantity of impaired habitat and the rate of continued disturbance appears at this time to be countering the efforts to restore complex instream habitat.[20]

Some stream complexity problems such as the legacy effects of splash damming and stream cleaning are probably already reflected to a large degree in the current biological status.

[20] The effect of restoration projects not reflected in the ODFW data set is not discussed in this document, but is discussed in the Federal Register notice as conservation measures.

However, future impacts to stream complexity from large wood availability; disturbance from road building, logging, and other land use practices; and reduction of beaver populations are not reflected in current biological status and may constitute a future threat.

Land management—forest and agriculture conversion

The pressures of urban and rural residential land use affect aquatic ecosystems and salmonids through alterations of and interactions among hydrology, physical habitat structure, water quality, and fish passage. These alterations occur at local and especially watershed scales, and thus require study and management at multiple scales. Urban and rural residential development causes profound changes to the pathways, volume, timing, and chemical composition of stormwater runoff. These changes alter stream physical, chemical, and biological structure and potential, as well as the connectivity of streams with their watersheds (IMST 2010).

The BRT discussed several modeling studies undertaken to understand the potential for conversion of lower density land uses to higher density ones. These were modeling studies by Kline et al. (2003) (see Table 19) and Lettman et al. (2009) that looked at the potential for land use conversion based on land use regulations existing at the time of the study. Kline et al. (2003), as part of the Coastal Landscape Analysis and Modeling Study (CLAMS) Project, modeled the potential expansion of urban and suburban areas in most of the OCCS ESU (Figure 23). Land use is projected to change in the ESU; primary changes are expected to be from agriculture, forest, and rural residential to urban (Table 20). Figure 23 shows a possible scenario between 1994 and 2044 based on existing land use zoning and property ownership as of 1994

Table 19. Results of Kline et al. (2003) by biogeographic stratum in the OCCS ESU. The year 1994 is used as a baseline for analyses predicting change in land use to rural residential and urban land use in 50 years and 100 years.

	Land use %		
Stratum	**Other**	**Rural residential**	**Urban**
1994			
Lakes	91.87	7.18	0.96
Mid-Coast	97.91	1.88	0.21
Mid-South Coast	96.61	2.32	1.07
North Coast	97.31	2.51	0.18
Umpqua	97.15	2.48	0.36
2044			
Lakes	92.85	5.59	1.56
Mid-Coast	97.22	1.93	0.85
Mid-South Coast	95.69	2.67	1.64
North Coast	96.11	2.99	0.89
Umpqua	95.82	3.29	0.88
2094			
Lakes	89.95	6.47	3.58
Mid-Coast	95.58	3.21	1.21
Mid-South Coast	93.05	4.66	2.30
North Coast	91.97	5.89	2.14
Umpqua	93.32	4.97	1.71

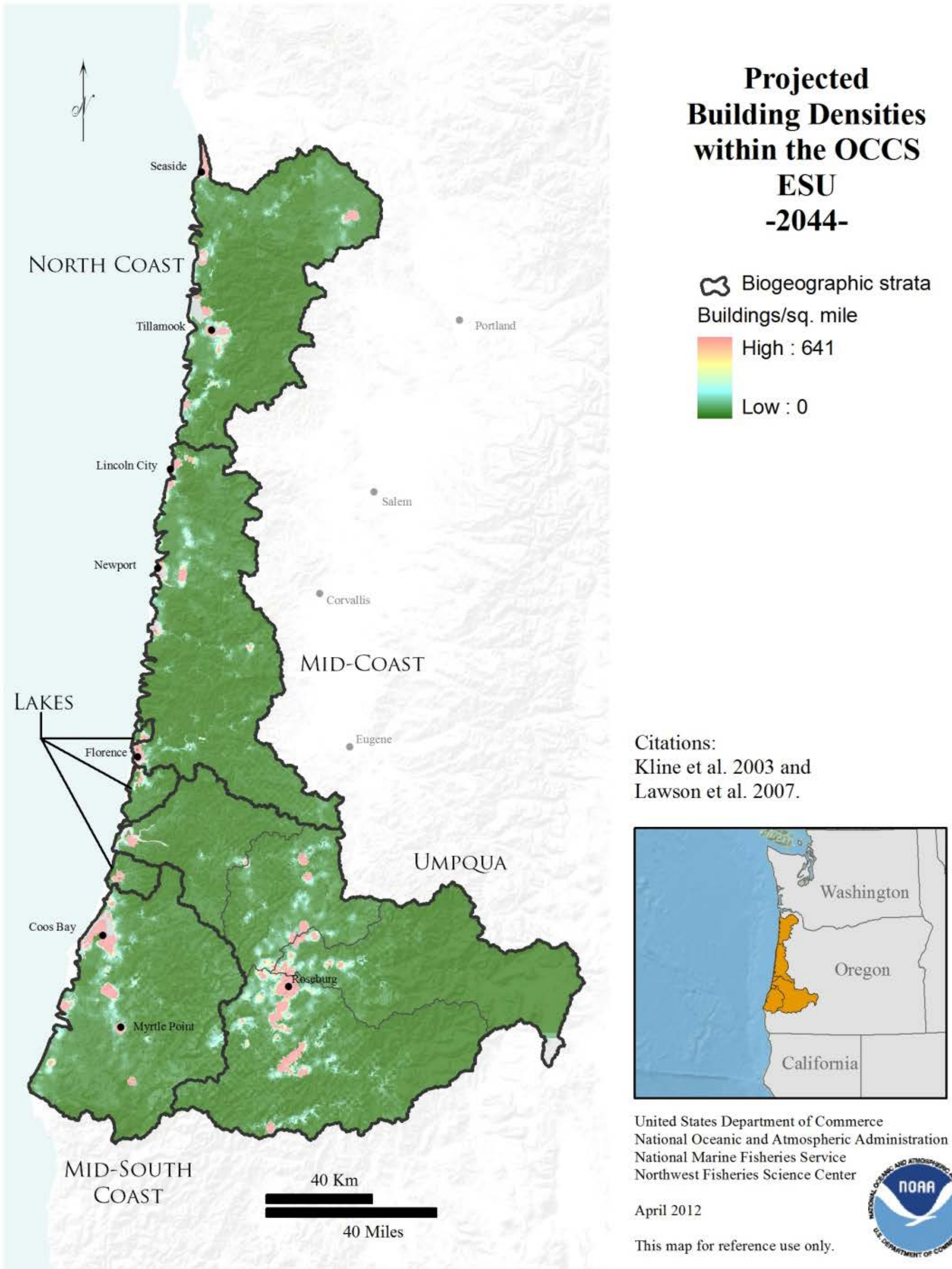

Figure 23. Projected building densities, 2044.

Table 20. Change in land use types predicted by Lettman et al. (2009). The baseline land use types are from 2005 information with 2035 as a 30-year projected change.

	Stratum														
	Lakes			Mid-Coast			Mid-South Coast			North Coast			Umpqua		
Land use type	**2005**	**2035**	**Area (ha)**	**2005**	**2035**	**Area (ha)**	**2005**	**2035**	**Area (ha)**	**2005**	**2035**	**Area (ha)**	**2005**	**2035**	**Area (ha)**
% wild land forest	81.56	81.09	–266	93.90	93.70	–1,065	85.41	85.20	–1,157	94.00	94.80	+4,145	87.88	87.63	–3,036
% mixed forest and agriculture	0	0	0	1.67	1.66	–56	4.80	4.80	0	0.59	0.50	–444	5.61	5.52	–1,093
% intensive agriculture	1.06	1.06	0	1.18	1.18	0	5.00	4.93	–352	2.82	2.58	–1,184	4.12	3.84	–3,400
% low density residential	5.64	6.07	+244	2.57	2.56	–56	3.33	3.44	+553	1.63	2.03	+1,977	1.60	1.98	+4,615
% urban	0.49	0.53	+23	0.44	0.67	+1,288	0.86	1.07	+1,057	0.59	0.71	+592	0.55	0.79	+2,914
% other	11.25	11.25	0	0.18	0.16	–112	0.59	0.58	–50	0.37	0.34	–148	0.24	0.24	0

(Kline et al. 2003). This model allows building densities to increase on any private lands, with some lands or ownerships (e.g., nonindustrial private) having greater likelihood of increases. By 2044 in this analysis, some change is anticipated in certain areas, particularly the area of the ESU near the urban centers along the Oregon coast. The Lakes stratum is anticipated for urban densities to nearly double, the Mid-Coast stratum to increase by a factor of 4, the North Coast to increase by a factor of 5, the Mid-South coast by a factor of 1.5, and the Umpqua stratum to increase by 2.5. This analysis did not include the entire Umpqua Basin, however.

While these increases are relatively large, they are still below the potential threshold effects of fundamentally altering the magnitude and frequency of flood events (Booth 1990, 1991). However, if urbanization is concentrated in distinct areas, as is typically the case, then watersheds with those areas could have increases that result in urbanized drainage areas of greater than 10−15% where the 1-year to 4-year flood event has a magnitude that is more similar to a 10-year flood recurrence interval (Moscrip and Montgomery 1997). This change in the hydrology of the streams could then result in decreases in coho salmon abundance levels from 2.5 to 4 times the levels typically seen in forested environments, particularly if urbanization also included alteration to wetland habitats directly associated with the stream network (Pess et al. 2003).

ODF also developed a model that predicts potential future land use changes in the ESU due to increased conversion of forest land to agriculture and urban/suburban uses (Figure 24, Table 20) (Lettman et al. 2009). The results of these projections show that under each of these scenarios, the most likely effects will be in the Mid-Coast, Mid-South Coast, and Umpqua River strata.

Human disturbances such as agriculture and urbanization can lead to a decrease in coho salmon habitat availability and quality (Berkman and Rabeni 1987, Beechie et al. 1994, Bradford and Irvine 2000). Beechie et al. (1994) found a decrease in tributary and off-channel habitats (e.g., wetlands, sloughs, and ponds) of up to 75%, almost all of which was due to deliberate modifications of the channel and floodplain. The vast majority of these impacts are related to the conversion of forested areas to agricultural and subsequently residential use. Maintained channelization can increase channel incision to the point where the streambed is disconnected from its floodplain (Booth 1990). Floodplain isolation reduces the amount of off-channel habitat available for adult salmonid spawning and juvenile rearing, which can lead to the downstream displacement of newly emerged salmonids to less desirable habitats (Seegrist and Gard 1972, Erman et al. 1988). Stream cleaning and riparian vegetation removal reduces the amount of in-channel wood, leading to a loss of pool habitat quantity (Montgomery et al. 1995, Collins et al. 2002), which can substantially reduce coho redd density (Montgomery 1999).

Urbanization can lead to an increase in impervious surface area and increase stream-flooding frequency and magnitude (Hollis 1975). The preurbanized 10-year recurrence interval flow event can occur every 2–5 years in urbanized areas of the Puget Sound region (Booth 1990), which can lead to declines in adult coho (Moscrip and Montgomery 1997). Urban watersheds also generate high concentrations of compounds that are toxic to salmon or alter their behavior in ways that could reduce survival (Scholz et al. 2000).

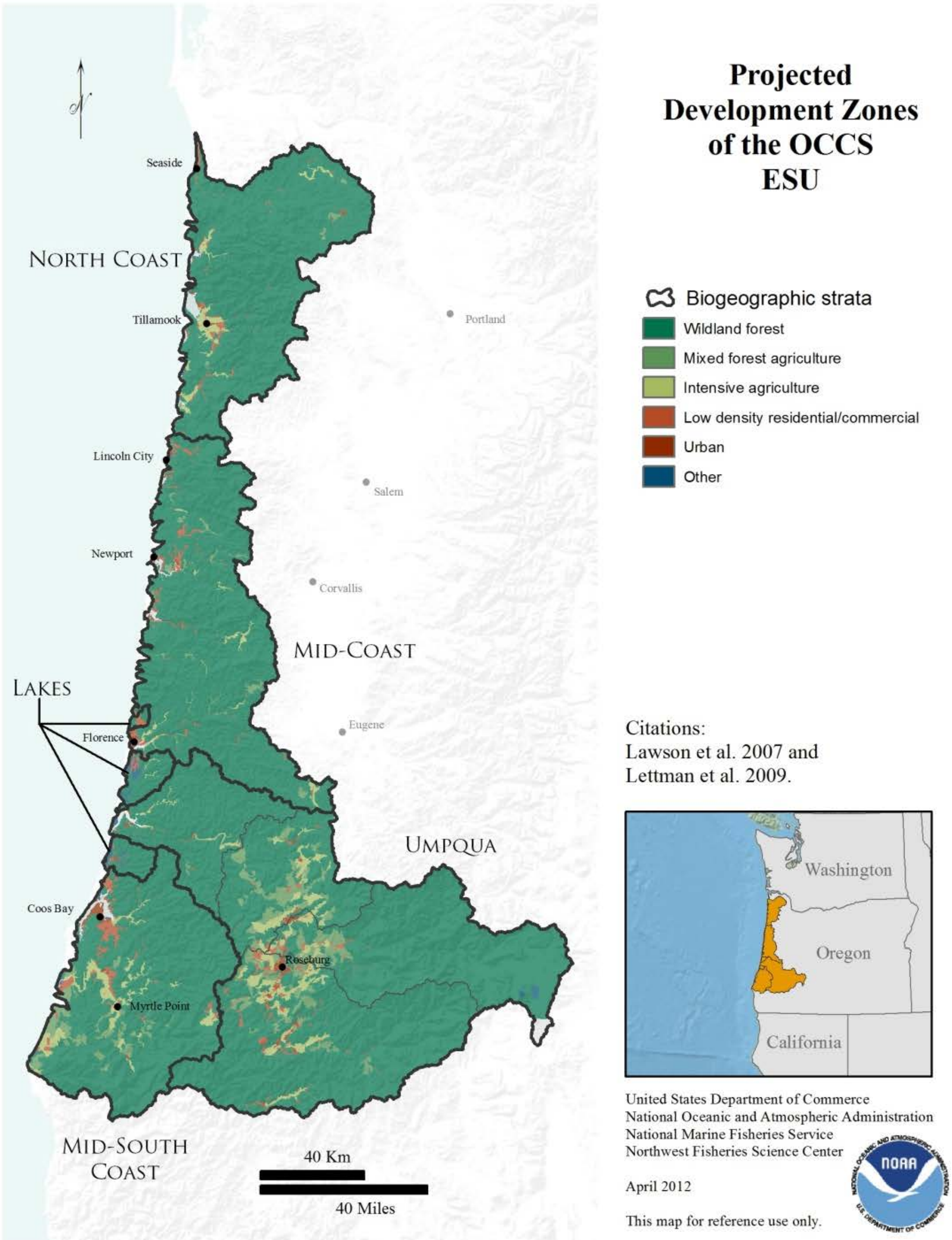

Figure 24. Projected development zones, 2035.

Conversion of lower intensity land use to higher intensity land use with a greater amount of impervious surface was identified by NMFS (NMFS 1997a) as a factor for decline in portions of the OCCS ESU. If urbanization, rural residential development, and loss of forest cover are concentrated in distinct areas, as is often the case due to land use zoning, then those watersheds would experience a change in the hydrology of the streams that would result in decreases in coho salmon abundance levels. IMST (2010) found that:

> In the Pacific Northwest, there is a growing understanding that aquatic habitat affected by existing development is important for salmonids (e.g., Pess et al. 2002b, Regetz 2003, MacCoy and Blew 2005, Sheer and Steel 2006, Burnett et al. 2007, Bilby and Mollot 2008). Projections of future land use and land cover in Oregon's coastal mountains show increasing rural residential and urban development within 328 foot (100 m) buffers surrounding high quality coho and steelhead habitat, with more rapid development projected for coho habitat (Burnett et al. 2007).

The BRT considered that the existing land use in the ESU was reflected in the current biological status of OCCS. Future conversions of lands to urban, suburban, and agriculture are dependent on many factors including economic conditions and land use planning and are therefore uncertain. Some BRT members thought that urbanization presented a smaller problem to OCCS compared to salmon in the Willamette Valley. Other BRT members, however, thought that urbanization and rural residential development retard advances in recovering important OCCS habitat in locations such as Tillamook Bay and Coos Bay. They also considered that conversion of agriculture and forests to urban and rural residential land uses results in a disproportional impact to high potential coho salmon habitat; the effects of conversion of land to uses with levels of impervious surface above 15% within a watershed were therefore considered a potential future threat with uncertain magnitude to OCCS populations.

Land management—loss/gain of estuarine and freshwater intertidal habitat

The Oregon coastal drainages supporting independent OCCS populations terminate in tidally influenced freshwater wetland and estuarine habitats (Figure 25) (e.g., Good 2000). In declaring critical habitat for OCCS, NMFS (2008) recognized that Oregon's estuaries and tidal freshwater wetlands provide habitat important to the migratory and rearing life stages of this ESU. IMST (2002) also highlighted the importance of estuaries for the "productive foraging environments for juvenile salmonids before they enter the ocean, refuge from predation, refuge from strong tidal and river currents, habitats of intermediate salinity for juvenile salmonids transitioning from fresh water to the ocean, migration corridors for adult salmonids returning from the sea, and at times, cooler water temperatures than [occur in] mainstem lowland rivers." ODFW has also cited the role of estuaries in providing foraging and growth opportunities for outmigrating coho smolts and the importance of stream and estuary ecotones for rearing coho juveniles[21] (Miller and Sadro 2003).

OCCS use of estuarine and freshwater intertidal habitat—The predominant life history pattern for coho salmon originating south of the central British Columbia coast is a 3-year cycle, including freshwater rearing for approximately 18 months followed by an equivalent

[21] R. Buckman, ODFW, Newport, OR. Pers. commun., January 2011.

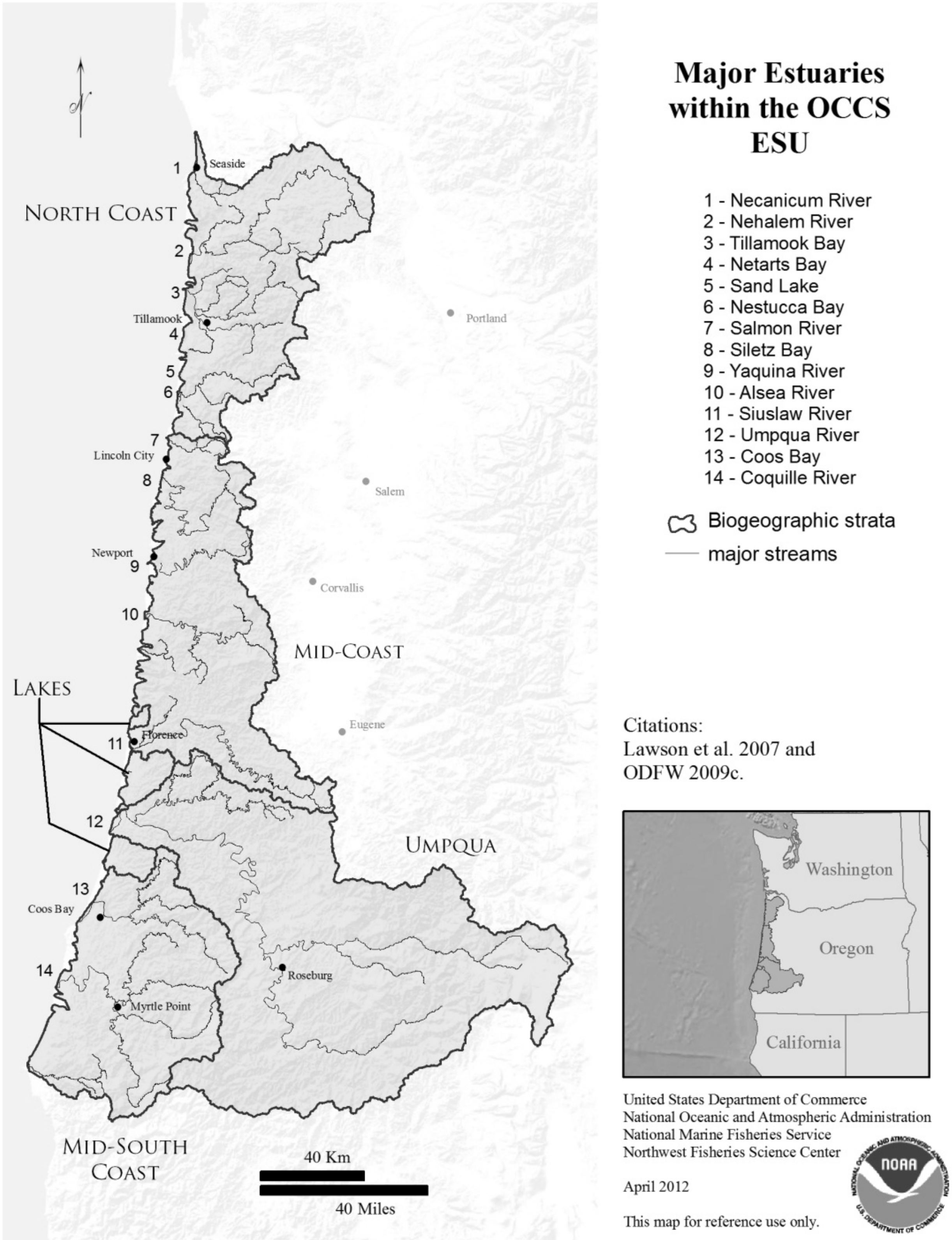

Figure 25. Locations of major estuaries.

period of ocean residence (Weitkamp et al. 1995). Several studies (Schreck 2002, Chapman et al. in prep., Power[22)] have focused on the use of estuaries and tidal freshwater habitats by yearling smolts emigrating to the ocean from natal rearing reaches. Koski (2009) reviewed results from several studies of downstream coho migration and rearing and discussed the importance of the stream and estuary ecotone as a rearing area.

The stream and estuary ecotone, defined as the transition zone from tidal fresh to tidal brackish waters, can serve as a transition area for smolts adapting to saltwater. This zone, characterized by low salinity, warm temperatures in the summer, and an abundance of food for juvenile salmonids, can serve as acclimation areas allowing coho salmon juveniles to adapt to the higher salinity levels associated with downstream subtidal reaches. Smolts outmigrating from upstream freshwater reaches may feed and grow in lower mainstem or estuarine habitats for a period of days or weeks prior to entering the nearshore ocean environment[23, 24] (Miller and Sadro 2003, Chapman et al. in prep.). Chapman et al. (in prep.) found that wild juvenile coho smolts rearing in several Mid-Coast Oregon estuaries prey almost exclusively on intertidal benthic invertebrates found on mudflats that are available only during high tides. In addition to serving as transition areas for outmigrating smolts, estuarine (brackish and freshwater) areas may provide more extended rearing opportunities for young of the year (age 0) coho juveniles (Miller and Sadro 2003).

Juvenile sampling studies in coastal rivers from northern California to Alaska indicate that coho salmon (age-0) juveniles are often present in these lower river and estuarine habitats, particularly in freshwater tidally influenced habitats (e.g., Jones et al. unpubl. manuscr.). Koski (2009) summarized information from recent studies indicating that downstream migrations of coho salmon may be associated with specific life history strategies that contribute to resiliency in the face of fluctuating environmental conditions. The relative contribution to adult returns from variations on an early downstream emigration pattern is not known for OCCS populations (Jones et al. unpubl. manuscr.).

Migrant trapping studies have shown that substantial numbers of coho salmon fry may emigrate downstream from natal streams into tidally influenced lower river wetland and estuarine habitats (e.g., Chapman 1962, Koski 2009, Bass 2010). Observations of spring or early summer downstream migration of coho salmon fry were originally thought to represent a passive displacement in response to increased stream flows, competitive interactions, or capacity limitations. Chapman (1962) used the term nomads to characterize coho salmon juveniles moving downstream between emergence and early fall, which is well before typical smolt migration in the spring. However, little direct quantitative information exists on the relative proportions of coho salmon juveniles that use this life history pathway, the survival rates and capacity relationships involved, and the relative contribution to adult returns.

At least three discrete life history strategies involving downstream coho fry and presmolt migrations into lower river habitats have been identified in the literature (e.g., Sandercock 1991, Koski 2009):

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[22] J. Power, EPA, Newport, OR. Pers. commun., May 2011.
[23] See footnote 21.
[24] See footnote 22.

- Late fall migration into side-channel or pond habitats connected to lower mainstem reaches from mainstem summer rearing habitats. For example, juveniles following this pattern had relatively high growth and overwinter survival rates in the freshwater side-channel habitats in lower Clearwater River, a major tributary to the Queets River on the Washington coast (Peterson 1982). In Winchester Creek (a relatively short tributary draining into the South Slough of Coos Bay, Oregon), some of the coho salmon juveniles that had emigrated downstream and reared over the summer in the brackish portion of the creek migrated into off-channel beaver pond habitats to overwinter (Miller and Sadro 2003). Similarly, Wallace and Allen (2009) determined that coho salmon juveniles rear through the summer in the tidal freshwater portions of Humboldt Bay tributaries. A portion of those juveniles emigrate into side-channel habitats for overwintering.
- Lower mainstem and estuarine summer rearing followed by upstream migration for overwintering. Skeesick (1970) documents upstream movements of coho salmon juveniles into overwintering habitats in three Oregon coastal streams including Munsell Creek in the tidal portion of the Siuslaw River. Koski (2009) also cites a number of studies that demonstrate fall movement of coho salmon juveniles into habitats with conditions conducive to overwintering survival.
- Lower mainstem and estuarine rearing followed by subyearling outmigration to ocean. A substantial number of subyearling coho in the Salmon River (Oregon coast) migrate downstream through the summer and early fall and rear in estuarine and freshwater tidal marsh habitats. Some of these juveniles may enter the ocean as subyearlings. Scale analyses conducted in one system, the Salmon River, indicated that the annual proportions of adult coho returning to the Salmon River that entered the ocean as subyearlings varied from 1% to 18% between 1993 and 2003 (Jones et al. unpubl. manuscr.). Future otolith analyses may provide estimates of relative adult return contributions from subyearlings that migrate directly to sea from upstream natal habitats versus those that may rear for an extended period in intertidal habitats prior to entering the ocean.

The relative contributions of these alternative life history pathways to either current or historical adult coho returns to Oregon coastal populations is not known. For example, numerous historic studies reporting age structure of adult coho salmon from scales very rarely find individuals that did not spend at least 1 year in freshwater prior to ocean entry (reviewed in Weitkamp et al. 1995). Few systematic surveys exist of the relative density and timing of juvenile coho rearing in upper and lower estuarine habitats for Oregon coastal drainages. Examples of Oregon coast stream and estuary ecotones cited by Koski (2009) include the upper 3 km of Winchester Arm of South Slough of Coos Bay (Miller and Sadro 2003), Lint Slough (Garrison 1965), and the Salmon River (Cornwell et al. 2001). More recent work not reported in Koski has been done in the Salmon, Alsea, Siuslaw, Nestucca, and Yaquina rivers (Jones et al. unpubl. manuscr.), as well as Coos Bay (Bass 2010). ODFW has also sampled juveniles and smolts in these habitats in the Siletz, Yaquina, and Alsea river basins.[25]

Losses of intertidal habitat—Historical losses of tidal habitat are documented in two reports that summarize estimates of current and historical tidal wetland habitats within Oregon

[25] See footnote 21.

coastal drainages with independent coho salmon populations (Good 2000, Adamus et al. 2005). Because the two assessments used different techniques to determine losses, the estimated quantities of wetland habitat loss were also different, although the two analyses yield similar trends among coastal basins (Table 21 and Figure 26).

Both assessments indicate that the historical ratio of estuarine and tidally influenced wetlands to total drainage area for the Coquille, Coos, and Tillamook basins were relatively high in comparison with other Oregon coastal drainages. The Umpqua River represents the largest single drainage on the Oregon coast and includes four independent populations. Adamus et al. (2005) estimated the highest proportion of historical lower river wetlands habitat in the Umpqua River, while the river ranked fourth among Oregon coastal drainages in total estuarine habitat in the Good (2000) analysis.

Table 21. Population or population aggregates with the largest estimated area of intertidal marsh habitat.

Rank	Good (2000)	Adamus et al. (2005)
1	Coos Bay	Umpqua River
2	Coquille River	Coquille River
3	Tillamook Bay	Coos Bay
4	Umpqua River	Tillamook Bay

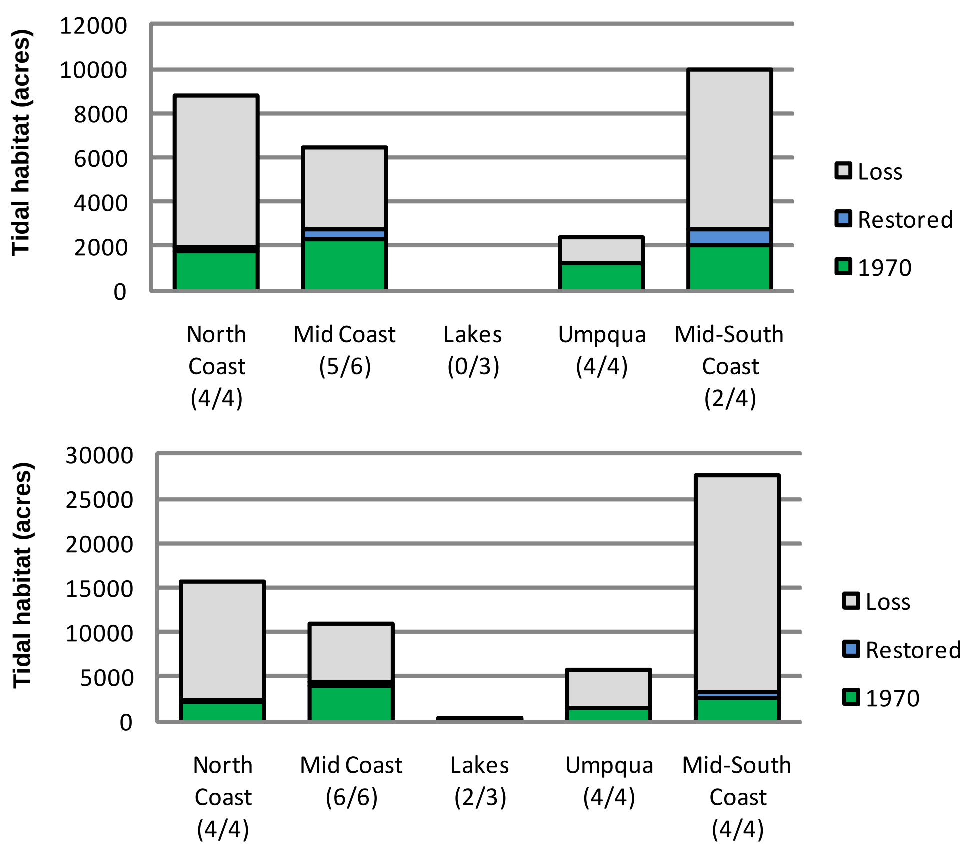

Figure 26. Tidal estuary gains in strata of the OCCS ESU. The top plot reflects the Good 2000 analysis and the bottom plot reflects the Adamus et al. 2005 analysis.

The amount of tidal wetland habitat available to support coho salmon migration, foraging, and rearing has declined substantially relative to historical estimates across all of the biogeographic strata (Table 21, Figure 26). The greatest historical losses (total area and proportional reduction) have occurred across populations in the North Coast and Mid-South Coast strata, driven by the relatively high proportional reductions in the largest estuaries. The time frame for contemporary estimates of tidal wetland areas differ between the two sources: Good (2000) reported values as of 1970, whereas Adamus et al. (2005) summarized wetland totals for the early 2000s. In addition to the direct losses, restriction of access to sections of tidal habitat and stream and estuary ecotone through the installation of tide gates (Bass 2010) has severely changed habitats available to outmigrating smolts relative to historical conditions. Overall, the results of recent coho salmon surveys imply that beyond the potential effects on the smolt rearing capacity of coastal basins, widespread estuarine and tidal freshwater wetland losses have also likely diminished the expression of subyearling migrant life histories within and among coho salmon populations.

Restoring and protecting intertidal habitat—Estuarine and tidal freshwater restoration projects have been carried out in several drainages in more recent years. Additional wetland habitat that has become potentially available to juvenile salmon through these OWEB, USFWS, and USFS projects are incorporated in Table 22. If aggregated across OCCS independent populations, recent restoration efforts have targeted a total area equivalent to 14−20% of current baseline of tidal habitat (Table 22). The largest increase has been in the Mid-South Coast stratum (Coos Bay and Coquille Bay), with a 28−37% aggregate increase in potential intertidal rearing habitat. The North Coast (11−14%) and Mid-Coast (11−19%) strata also had relatively large proportional increases. Intertidal habitat gains in a small basin, the Nestucca River, accounted for the change in the aggregate North Coast area total. Likewise, the Mid-Coast stratum increase was accounted for largely by changes in the Salmon River. These gains notwithstanding, the proportional change in the total amount of available intertidal habitat after adding in gains through recent restoration efforts is small relative to historical conditions (Table 22). In addition to the restoration actions, approximately 2,900 acres of existing high quality intertidal or adjacent riparian habitat has been afforded protection through OWEB fee title and conservation easement programs in recent years.[26]

Table 22. Summary of recent restoration versus current and historical estimates of intertidal marsh habitats aggregated across populations within OCCS ESU biogeographic strata.

	Recent restoration vs. current (%)		Recent restoration vs. historical (%)	
Biogeographic strata	**Good 2000**	**Adamus et al. 2005**	**Good 2000**	**Adamus et al. 2005**
North Coast	14	11	0.03	0.02
Mid-Coast	19	11	0.07	0.04
Umpqua	2	2	0.01	0.01
Mid-South Coast	37	28	0.07	0.03
Total	**20**	**14**	**0.05**	**0.02**

[26] M. Hulst, OWEB, Salem, OR. Pers. commun., 15 October 2010.

The OCCS biological recovery criteria directly considered the status of tidally influenced habitats at the population and ESU levels as an indication of remaining diversity (Wainwright et al. 2008). Two of the component criteria in the DSS are informed by measures of the relative status of tidally influenced habitats. The workgroup noted that while it was clear that estuarine habitat conditions have changed relative to historical conditions, it is difficult to determine the degree to which those changes have affected fish.

Future threats to OCCS from loss of estuarine and freshwater tidal habitat may also come in the form of sea level rise in Oregon's estuaries (NWF 2007, OCCRI 2010). Although recent restoration efforts have increased the amount of estuarine habitats suitable for coho rearing, it is uncertain whether gains can continue to be realized in light of the potential impact of sea level rise. If the human response to sea level rise is to raise the protection level of dikes and levees, then there would likely be widespread loss of tidal habitat because the opportunity for tidal marshes, swamps, and mudflats to move to higher ground will be blocked by protection structures and basin topography. Tidal marshes and mudflats are substantial contributors to the estuarine food chain in direct and indirect ways (Gray 2005, Chapman et al. in prep.). Loss of more tidal habitats through sea level rise could have a negative effect on feeding and rearing of OCCS in estuarine and tidal freshwater habitats.

The current biological status of Oregon coastal coho populations reflects the effects of estuarine tidal habitat loss relative to historical conditions, including the potential impacts of the associated diminished life history diversities. With an increasingly variable marine ecosystem, this loss of life history diversity may constitute a future threat, particularly for production from smaller tributaries associated with relatively large estuaries. It is difficult to quantify the potential impact of those losses, given the current uncertainty regarding the historical contributions from the various life history patterns.

Land management—loss/gain of freshwater wetland habitat

Determining the freshwater wetland losses outside estuaries in each population of the OCCS ESU is not possible with the data sets available at present. There have been estimates of estuarine wetland losses in several studies (Good 2000, Christy 2004, Adamus et al. 2005). All have differing estimates, probably from the use of differing data and methodologies. As an example of the severity of the losses, Christy (2004) found that the estimated total acres of wetlands in estuaries in the OCCS ESU that were converted to other uses from 1850 to 2000 was estimated at 43,672 acres. Of these losses, freshwater wetlands were highest with 34,276 acres, salt marsh losses were next with 9,383 acres, lake-associated wetlands were reduced by only 13 acres, and subtidal habitat had zero acres of loss. Of course, these numbers do not reflect any losses upstream of the estuaries.

For somewhat recent losses and gains to wetlands in the OCCS ESU, Table 23 (ODSL 2005) details the information available at the time. This analysis is not restricted to just estuarine wetlands, so is not comparable to Christy (2004), but shows continued wetland loss to filling activities as well as restoration of wetlands in counties occupied by the OCCS ESU.

More recent requests for information (2007−2008) from the Oregon Department of State Lands (ODSL) permit tracking system reported 12.5 acres of freshwater and estuarine wetlands

Table 23. Oregon Department of State Lands (ODSL) summary of wetland fill, compensatory wetland mitigation (CWM), enhancement and restoration, and OWEB-funded restoration projects (nonmitigation) authorized/completed from 1 July 2000 to 30 June 2004 in acres. (Reprinted from ODSL 2005.)

County	Wetland fill permitted by ODSL (county area)	CWM required by ODSL* (county area)	OWEB-funded wetland restoration projects (nonmitigation) (ESU)
Clatsop	5.2	4.3	
Coos	9.0	20.1	
Douglas	41.2	90.1	30 (Dawson Creek)
Lane	48.7	44.4	30 (Enchanted Valley)
Lincoln	0.3	0.4	70 (Lint Slough)
Tillamook	0.6	1.8	
Totals	**105.0**	**161.2**	**130**

* Creation, enhancement, or restoration.

lost, 9.6 acres gained, and 46.21 acres enhanced in the counties of the ESU.[27] There are still wetland losses occurring, and some wetland gains being made, but probably not at the scale that historical freshwater wetlands (just in estuaries) were lost. Substantial development of data and historical reconstructions are necessary before the true magnitude of wetland losses throughout the OCCS ESU are understood.

The results of coho salmon habitat surveys (ODFW 2009b), however, imply that loss of wetlands throughout the ESU has had a significant effect on rearing capacities of coastal basins, not just in estuaries. These losses may originate from, to name a few, stream incision and loss of connection with the floodplain, filling and diking of wetlands for agriculture and urban development, and loss of beaver-engineered wetlands due to trapping and disease. This, in addition to estuarine losses, may also have diminished the nomad life history in OCCS populations due to loss of slow water rearing areas.

Although it is apparent that wetland losses in estuaries have slowed and in some basins reversed, losses in freshwater wetlands upstream of the estuaries in the ESU are difficult to quantify. Some information about recent losses is available through the ODSL permit tracking system, but studies of historic freshwater wetland losses are either too large scale for usefulness, or restricted to the Willamette and Klamath basins (Morlan 2000). Many of the freshwater wetlands important to coho salmon are not inventoried because they are outside the wadeable stream restriction for the ODFW aquatic habitat surveys. Because wetlands are so important to coho rearing (Nickelson 1998, Burnett et al. 2003), lack of information regarding these off-channel and slow water areas constitutes a risk in making future management decisions without a robust understanding of OCCS life cycle and utilization of these habitats. The BRT considered that freshwater wetland losses were probably reflected in the current biological status of the species. Because the potential magnitude of future freshwater wetland losses is poorly understood, the scale of the future threat to the OCCS ESU is uncertain.

[27] J. Vaughn, ODSL, Salem, OR. Pers. commun., December 2009.

Land management—mining

Mining in general and gravel mining in particular were identified as factors for decline by NMFS (1997c). Until recently, gravel mining, particularly in the Umpqua River and Tillamook River basins, has been a serious concern in the past to fishery managers and remains a concern in the Coquille River. Providing for fisheries in gravel mining operations has been the subject of substantial effort for protection of all anadromous salmonids in the Umpqua stratum. At this point in time, there are no active instream gravel mining operations in the Umpqua; however, there are continuing operations in the Tillamook and Nehalem basins in the North Coast stratum. There is a concern that if ESA protections are removed, instream gravel mining operations could become a serious threat to the OCCS ESU in the future. The BRT considered that the effects of mining were probably reflected in the current biological status of the species. However, because the potential for future gravel mining activities are poorly understood, the scale of the future threat to the OCCS ESU is uncertain.

Land management—water quality degradation

Water quality has long been identified as a factor for decline (NMFS 1997a) and a limiting factor for recovery (Oregon 2005) for OCCS. Water quality is made up of many facets that were presented in NMFS (1997c), ODEQ (2005), and Oregon (2005). Table 15 lists the 15 populations where water quality is an important limiting factor.

In 2005 the Oregon Department of Environmental Quality assessed the situation in the OCCS ESU:

> Water quality improvements in an area like the coastal coho ESU—where the problems largely relate to nonpoint source pollution and flow and channel modification—take time. At this time, we are not able to demonstrate an improving trend in water quality, but there are some indications that improvements will occur. One sign of progress is reflected in the on-the-ground efforts of landowners and others and the partnerships being forged to conduct total maximum daily loads implementation activities (ODEQ 2005).

For the purposes of this status review, the focus is on temperature limitations within the ESU because of temperature's important effect on coho salmon success in freshwater. For an overview of water quality status of the OCCS ESU streams, Figure 27 shows a substantial amount of the streams and rivers in the ESU as water quality limited. Category 5 shows impairment by one or more pollutants and Category 4 shows that the reach is impaired but has an approved total maximum daily loads management plan. The mileage of impairments in the OCCS ESU is difficult to assess because impairments of stream reaches may be different and overlap. However, as illustrated in Figure 28 (ODEQ 2007), the temperature impairments in the OCCS ESU are 40% of OCCS distribution stream miles.

It can be argued that water temperature is the primary source of water quality impairment in the OCCS critical habitat. Welsh et al. (2001) found in the Mattole River, California, that juvenile coho were not found in streams with mean weekly average temperatures greater than 18°C, and that all streams in their study area with mean weekly average temperatures below 14.5°C held juvenile coho. Temperatures above about 15°C stress salmon in a variety of ways.

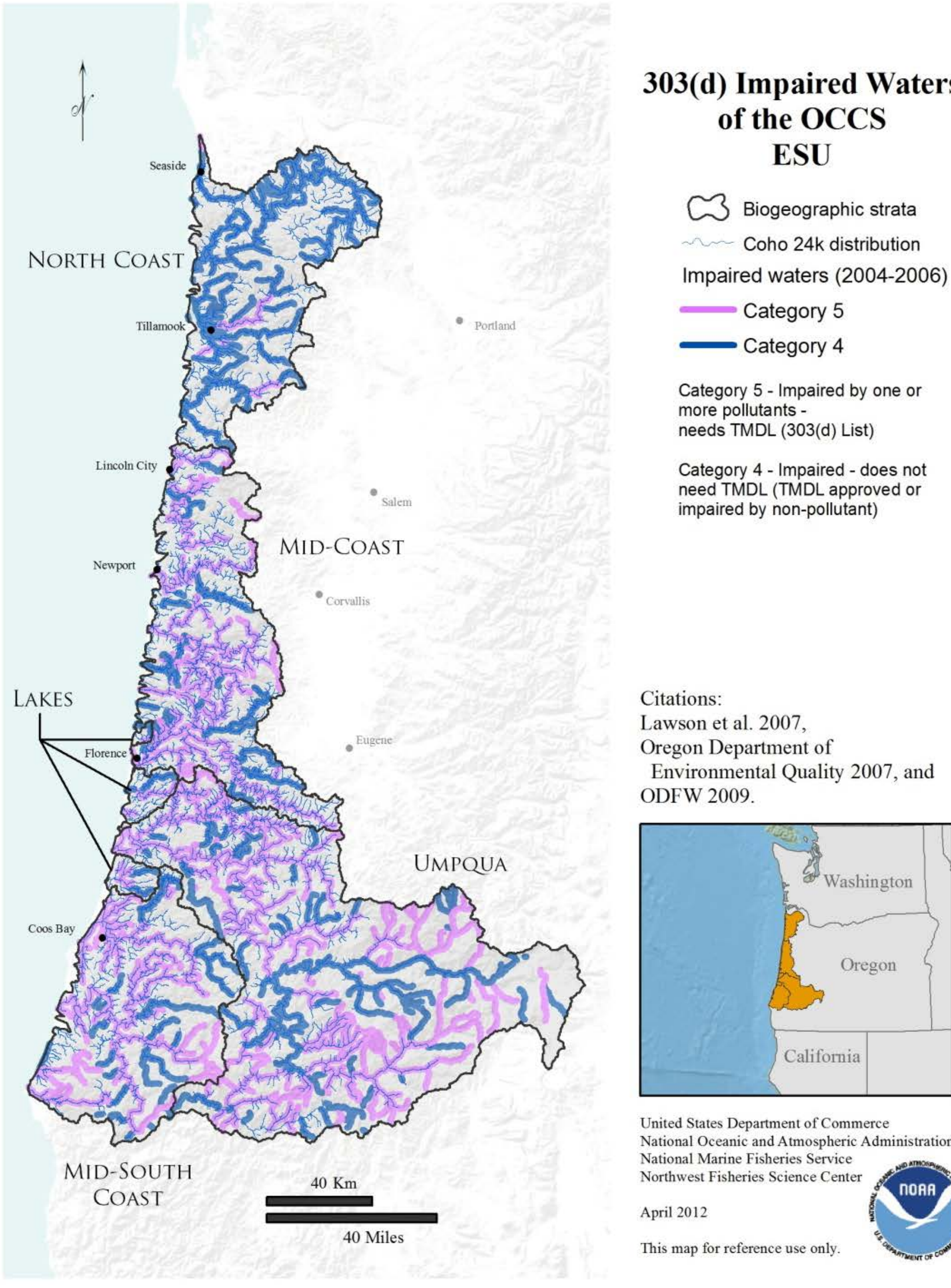

Figure 27. EPA 303(d) water quality impaired waters.

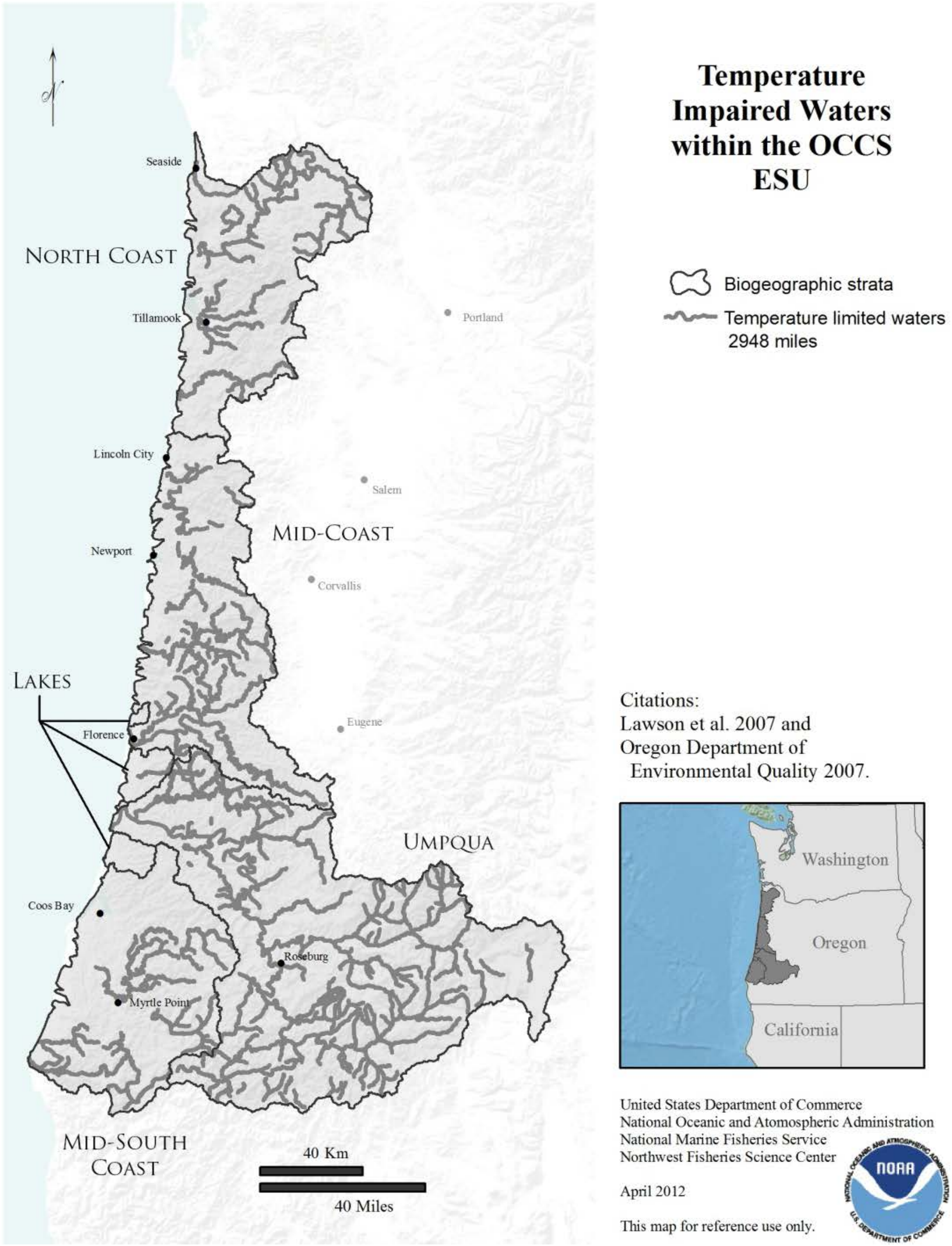

Figure 28. EPA 303(d) listed streams with temperature impairment.

At higher temperatures, metabolic rates are higher so fish must forage more actively to maintain growth rates. This adds stress if the food supply is limited and forces fish to be more exposed to predators. Dissolved oxygen is lower at higher temperatures, further stressing the fish. Many of the diseases to which salmon are susceptible occur at higher rates as temperatures increase (Marcogliese 2001, 2008). These include fungal infections such as *Columnaris*, which can cause mortality in juveniles and returning adults.

Temperature has been negatively correlated with coho salmon survival and abundance in freshwater (Lawson et al. 2004, Crozier et al. 2008b). Temperature effects operate through a wide variety of mechanisms; beaver pond wetlands tend to moderate water temperatures, parasites are more virulent at higher temperatures (Lawson et al. 2004), and life cycle timing can be disrupted at higher temperatures, potentially leading to a mismatch between smolt outmigration timing and onset of upwelling in spring (Crozier et al. 2008b). Higher temperatures in the summer limit the quantity of stream habitat that is available for juvenile salmon rearing, while high temperatures in the fall can block adult migrants from reaching spawning grounds (Ebersole et al. 2006). The broad conclusion is that the rising temperatures anticipated with global climate change will have an overall negative effect on the status of the ESU. If 40% of the OCCS ESU is already temperature impaired (ODEQ 2007), just the effects of climate change in the absence of threats from other human activities like forestry and agriculture pose a significant risk to those systems already impaired, and increase the likelihood of temperature impairment in the rest of the aquatic systems in the ESU. The BRT considered that the effects of current water quality impairment were probably reflected in the current biological status of the species. Because of the expected effects of global climate change on OCCS habitat, water quality was considered a significant future threat to the OCCS ESU.

Disease or Predation

Disease and parasitism

In its assessment of OCCS, ODFW (2005b) asserted that disease is not an important consideration in the recovery of OCCS. However, Jacobson et al. (2003, 2008) identified *Nanophyetus salmincola* as a potentially important source of early marine mortality. Cairns et al. (2005) have also shown that “the direct effects of temperature associated with increased metabolic demand can be exacerbated by other factors, including decreased resistance to disease and increased susceptibility to parasites.” Jacobson (2008) reports that annual parasite prevalences of *N. salmincola* in yearling coho salmon caught in ocean tows off the coast of Oregon were 62−78%. Yearling coho had significantly higher intensities of infection and higher infection in wild versus hatchery juveniles, presumably due to the greater exposure to metacercaria in natal streams. Parasite prevalences and intensities of yearling coho salmon caught in September were significantly lower (21%) than those caught in May or June in 3 of 4 years of data. This suggests parasite-associated host mortality during early ocean residence for yearling coho salmon. Pearcy (1992) hypothesized that ocean conditions (food and predators) are very important to marine mortality, especially soon after the juveniles enter the ocean. This is the time period that Jacobson et al. (2008) observed the loss of highly infected juveniles. Jacobson hypothesizes that high levels of infection may lead to behavioral changes in the fish and thus make the juveniles more susceptible to predation.

The issue that Cairns et al. (2005) investigated is the influence of summer stream temperatures on black spot infestation of juvenile coho salmon in the West Fork of the Smith River, Oregon, in the OCCS ESU. Their studies show that, "although other environmental factors may affect the incidence of black spot, elevated water temperature is clearly associated with higher infestation rates in the West Fork of the Smith River stream network." This may be an important issue for coho salmon juveniles, as many of the streams they inhabit are already very close to lethal temperatures during summer months (see Figure 28) and, with the expectation of rising stream temperatures due to global climate change, changes in metabolic rates may act as a stressor that may result in higher infection rates in coho salmon.[28] Changes in infection rates of juvenile coho by parasites as well as new parasites associated with invasive species may become an increasingly important stressor for freshwater and marine survival.

Parasitism and disease was not considered an important factor for decline in early status reviews for OCCS. However, some of the studies discussed above suggest that it may become more important as temperatures rise due to global climate change and may become a very important risk for juveniles in the early ocean entry stage of the life cycle. The BRT considered that the effects of disease and parasitism were probably reflected in the current biological status of the species. However, because of the expected temperature effects of global climate change on OCCS freshwater habitat, disease and parasitism was considered a potential future threat to the OCCS ESU.

Predation

Due to the visibility of predators and their interactions with resource users in freshwater and salt water, predators are often mentioned by stakeholder groups as a serious threat to OCCS populations (ODFW 2005g). Fresh (1997) concluded that predation was probably not a primary factor in OCCS population declines. IMST (1998) examined the question of predation and concluded that salmon have evolved with predators and that, despite the presence of many kinds and large numbers of predators, coho salmon have persisted over many millennia. It notes that there is variability in predators over time depending on ocean conditions, the size of the predator, and availability of salmon juveniles. It also concluded that when populations are low, however, predation can have a significant effect on extinction risk.

Birds and marine mammals—Cormorants (*Phalacocrorax* spp.), terns (*Sterna* spp.), brown pelicans (*Pelecanus occidentalis*), sooty shearwaters (*Puffinus griseus*), common murres (*Uria aalge*), mergansers (*Mergus* spp.), gulls (*Larus* spp.), belted kingfishers (*Megaceyle alcyon*) grebes and loons (*Gavia* spp.), herons (Family Ardeidae) ospreys (*Pandion haleaetus*) and bald eagles (*Haliaeetus leucocephalus*) all prey on juvenile salmonids in the OCCS ESU to one degree or another (IMST 1998). In the Columbia River estuary just adjacent to the OCCS ESU, terns and double-crested cormorants have been shown to affect juvenile salmonid survival significantly (Collis et al. 2002, Roby et al. 2003, Antolos et al. 2005). However, river basins in the OCCS ESU do not have dredge spoil islands to attract large tern and double-crested cormorant colonies. Neither do they have an extended time period of juvenile salmonid outmigration similar to the Columbia River system.

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[28] Although Bisson and Davis (1976) found that elevated temperatures reduced infestation rates by *Nanophyetus* on juvenile Chinook salmon.

Predation by avian predators may, however, be important in the loss of salmonid juveniles in some populations in the ESU. In a study of steelhead outmigrants in the Nehalem River, Schreck et al. (2002) observed substantial mortality of juvenile steelhead in the estuary, presumably from predation by double-crested cormorants, Caspian terns, and harbor seals (*Phoca vitulina*). More recently, Johnson et al. (2010) observed mortality rates on naturally reared steelhead juveniles as high as 53% in the Alsea River, mainly in the lower estuary and presumably from avian predation and harbor seals. Neither study, however, demonstrated direct evidence of predation by any particular predator. Bass (2010) in Coos Bay was able to demonstrate predation on OCCS juveniles by double-crested cormorants by utilizing PIT tag detections of deposits below the rookery. These were smolts that he had tagged for a study on the effect of tide gates on juvenile coho salmon movement.

The common murre is the most abundant seabird in the OCCS ESU, but does not appear to have a significant impact on juvenile salmonids in the nearshore at present. The common murre breeding population on the north coast of Oregon has been severely affected by bald eagle predation. They have abandoned their nesting sites on the north coast rocks. Murres therefore are not feeding on juveniles from those coho salmon populations in the large concentrations that they would if they were breeding on the nearshore rocks.[29] Other species have shown substantial increases in population levels, particularly Caspian terns and double-crested cormorants in the lower estuary of the Columbia River (Collis et al. 2002). However, outside of the Columbia River system, Adkins and Roby (2009) report that there were 2,384 breeding pairs of double-crested cormorants nesting at 22 colony sites along the Oregon coast. This is similar to the 1992 estimate of 1,850 breeding pairs in 13 colonies (Carter et al. 1995), so there is no reason to believe that substantially higher abundance in double-crested cormorant populations has contributed significantly to OCCS population declines in recent years.

Because of the increasing abundance and visibility of marine mammal predators since the passage of the Marine Mammal Protection Act, there is a perception among users of the estuarine and marine environment that reducing predation by harbor seals and California sea lions (*Zalophus californianus*) is important for the restoration of OCCS (Smith et al. 1997). Botkin et al. (1995) concluded that marine mammal predation on anadromous fish stocks in northern California and southern Oregon was only a minor factor for their decline. NMFS (1997d) also examined the issue and determined that marine mammal predation in some northwest fisheries has increased on the Pacific Coast. This predation may significantly affect salmonid abundance in some local populations when other prey are absent and physical habitat conditions lead to the concentration of adults and juveniles in restricted areas or stocks.

IMST (1998) concluded "that the California sea lion, Pacific harbor seal, Caspian tern, and cormorant populations along the Oregon coast have all increased in recent years, coinciding with historic lows in salmon abundance. Predation by these species may be a factor in the lack of recoveries of some depressed stocks, but there is no compelling scientific evidence that predation has been a primary cause for decline of salmonids." In the 2005 Oregon State Coho Assessment, ODFW (2005h) reported that there is little new evidence that allows analysis beyond the summary statements made by NMFS (1997d) and IMST (1998). It said the result of

[29] R. Lowe, U.S. Fish and Wildlife Service, Oregon Coastal Refuges, Newport, OR. Pers. commun., September 2010.

future investigations is "not likely to change the general conclusion that, while negative effects can occur in specific situations where other prey is in unusually low abundance, local predator numbers are high and restrictions in passage or reduction in habitat quality have all increased predation success, natural predation by pinnipeds or seabirds has not been a significant cause in the decline of salmonid stocks at the ESU scale."

A recent study by Brown et al. (2005) reports that though the abundance of harbor seals has increased since the passage of the Marine Mammal Protection Act, the Oregon and Washington harbor seal population grew rapidly until the 1990s but appears to be stable around an equilibrium with variability due to ocean conditions. Whether or not the harbor seal population is growing, Schreck et al. (2002) (Nehalem River) and Johnson et al. (2010) (Alsea River) implicate harbor seals as well as birds in significant (29−66%) loss of juvenile steelhead. Avian and mammalian predation may not have been a significant factor for decline when compared with other factors, but this more recent work shows that it may be important to recovery actions in certain populations and specific situations within the OCCS ESU.

Nonindigenous fish—In contrast to mammalian and avian predators, OCCS have not evolved with NIS fish. Fish predation can be a significant source of mortality of coho salmon juveniles, particularly in lake and slow water systems. Largemouth bass and smallmouth bass are particularly efficient predators of juvenile coho salmon[30] (Bonar et al. 2004).

Lake-rearing coho salmon represent life history diversity that is essential to the resilience of OCCS (Lawson et al. 2007). While river populations exhibited wild swings in abundance during the low return years of the 1990s, lakes produced consistent returns during that time period. However, the change in productivity of the Tenmile Lakes system in the 1970s shows the effect of NIS fish on OCCS. High abundance was observed from 1955 to 1973 when adult spawners ranged from about 5,700 to 42,000 adults. The Tenmile Lakes escapement from 1974 to 1999 after introduction of NIS warm water fishes and treatment with rotenone to rid the lake of them fell to an average of only 3,453 (777 to 7,581) (Zhou 2000). Current returns in the Tenmile Lakes system remain substantially lower than returns prior to the introduction of NIS fish. For Siltcoos and Tahkenitch lakes, which had introductions of these warm water game fish in the 1930s, it is impossible to discern changes due to lack of data. The effects of these NIS fish are not consistent across the landscape of the OCCS ESU; the North Coast and Mid-Coast monitoring areas have some introduced fish species, but they do not have much in the way of lakes and slow water like the Lakes, Umpqua, and Mid-South Coast strata. Also, higher summer temperatures in these more southerly systems favor NIS fish (ODFW 2005f). This effect is expected to increase with rising temperatures in the lakes and slow water areas of the Oregon coast.

EPA (2009) commented that NIS fish are capable of ecosystem changing effects as well of those of predation. NIS warm water fishes pose a future threat to coho rearing due to ecosystem change as well as predation if anticipated temperature rise associated with global climate change occurs. Peer reviewer 2 (reference Appendix D) commented that predation and competition, particularly in light of the warming water temperatures from global climate change, could significantly affect the lakes and slow water rearing life history of OCCS, not only by NIS

[30] Reported by BRT member L. Kruzic, December 2009.

fish but by native invasions as well (Reeves et al. 1998). As water temperatures increase, NIS warm water and other native fish will be at an even greater advantage over OCCS in lake and slow water situations due to predation, competition, and ecosystem alterations.

For this analysis on the current status of the effect of predation, effects of current populations of NIS warmwater fish are probably reflected in the OCCS current biological status of these populations. However, in anticipating future conditions, as water temperatures increase there is greater risk to OCCS in lake and slow water situations due to predation, competition, and ecosystem alterations. This effect on the slow water and lake life histories of OCCS may present a significant threat to diversity of the species.

Summary of Factors for Decline and Threats

As described above, the BRT analysis started with the list of major threats previously identified by the NMFS NWR and revised it to include discussion of emerging issues such as global climate change. Some threats, in particular hatchery production and harvest, have been greatly reduced over the last decade and appear to have been largely eliminated as significant sources of risk. Other factors, such as habitat degradation and water quality, were evaluated to be ongoing threats that appear to have changed little over the last decade. Changes to freshwater and marine habitat due to global climate change were considered threats likely to become manifest in the future. A summary of the threats considered by the BRT is found in Table 24.

Table 24. BRT summary comments on threats.

2009 BRT threats	Comments
Agriculture and forestry	
Splash dams	Legacy effects on stream complexity.
Human landscape disturbance	Effects constitute an ongoing threat.
Roads	Existing and legacy roads can contribute to continued stream degradation, could constitute a future threat.
Stream cleaning	Legacy effects on stream complexity.
Substrate sediment	See stream habitat complexity analysis.
Stream habitat complexity	Habitat complexity and summer parr capacity are decreasing in the Umpqua but increasing in the other strata. Winter parr capacity is trending flat in the North Coast and Mid-Coast, but declining in the Mid-South and Umpqua strata. For the percent of fine sediment in riffles, there appear to be declines in the North and Mid-Coast, a positive trend in the Mid-South, and little change in the Umpqua strata. Large wood volume appears to be declining in the North Coast and Umpqua, while increasing in the Mid-Coast and Mid-South strata. LWD trends in upstream areas declined substantially in all strata. Trends in sediment were mixed, with increases in the Mid-Coast and Mid-South and declines in the North Coast and Umpqua strata.
Water temperature, water quality degradation	Because of the expected effects of global climate change (especially temperature), water quality was considered a significant future threat.
Estuary, wetland habitat loss	With an increasingly variable marine ecosystem and sea level rise, loss of life history diversity may constitute a future threat.
Water availability	Future impacts to water availability from effects of population growth; global climate change may constitute a future threat.
Beaver dam loss due to effect on local stream complexity	Lack of protection of beavers and degraded beaver dam density levels is an ongoing threat.
Fish passage restriction	Incomplete data in important OCCS habitat is a significant information gap to some BRT members.
Gravel mining	Because the potential for future gravel mining activities are poorly understood, the scale of the future threat is uncertain.
Land use conversion-urbanization, rural residential Floodplain functions Instream wood Substrate sediment Storm water	Future conversions of lands to urban, suburban, and agriculture use are dependent on many factors and are therefore uncertain. Some BRT members felt that urbanization did not present a significant future threat. Other members believed that urbanization and rural residential development retards advances in recovering important OCCS habitat in locations like Tillamook Bay and Coos Bay. Some members considered that conversion of agriculture and forests to urban and rural residential land uses results in a disproportional impact to high potential coho salmon habitat. The effects of conversion of land to uses with levels of impervious surface above 15% within a watershed were considered a potential future threat with uncertain magnitude.

Table 24 continued. BRT summary comments on threats.

2009 BRT threats	Comments
Disease and parasitism	Because of the expected temperature effects of global climate change on OCCS freshwater habitat, disease and parasitism was considered a potential future threat.
Predation	May be important to recovery actions in certain populations and specific situations.
Harvest	Further harvest reductions would have little effect on spawning escapements. Future remedies must be found outside of harvest management until the decline in productivity is reversed.
Artificial propagation	Reduction in hatchery activity is expected to significantly benefit wild runs throughout the ESU.
Global climate change	It was noted that there are some expected positive effects; however, negative effects of climate change predominate for each habitat and life history stage. While many of the individual effects of climate change on OCCS are expected to be weak or are uncertain, we must consider the cumulative impacts across the coho salmon life-cycle and across multiple generations. Because these effects are multiplicative across the life cycle and across generations, small effects at individual life stages can result in large changes in the overall dynamics of populations. This means the mostly negative effects predicted for individual life history stages will most likely result in a substantially negative overall effect of climate change on OCCS over the next few decades. Despite large uncertainties surrounding specific effects at individual life stages, expectations for increasing air and water temperatures, drier summers, higher incidence of flooding, and altered estuarine and marine habitats, lead us to expect increasingly frequent years with low survival, resulting in an overall increase in risk to the ESU resulting from climate change over the next 50 years.
Marine productivity	BRT concluded that the coastal ocean ecosystem on which coho salmon depend is in an unpredictable state of flux.
Drought	Addressed in global climate change.
Floods	Addressed in global climate change.
Wildfire	Addressed in global climate change.
Tsunami	Not considered.
NIS	May affect stream complexity; invasions by NIS and subsequent ecosystem changes may constitute a future threat. Effects of NIS fish are expected to increase with rising temperatures in lakes and slow water areas.

Overall Risk Assessments

The BRT's determination of overall risk of extinction to the OCCS ESU used the three categories of high risk, moderate risk, and low risk (not at risk). The high and moderate risk levels were defined by the NMFS NWR in its status review request as follows:

> Moderate risk: A species or ESU is at moderate risk of extinction if it exhibits a trajectory indicating that it is more likely than not to be at a high level of extinction risk. A species/DPS [distinct population segment] may be at moderate risk of extinction due to projected threats or declining trends in abundance, productivity, spatial structure, or diversity. The appropriate time horizon for evaluating whether a species or DPS is more likely than not to be at high risk depends on the various case-specific and species-specific factors. For example, the time horizon may reflect certain life history characteristics (e.g., long generation time or late age-at-maturity) and may also reflect the time frame or rate over which identified threats are likely to impact the biological status of the species or DPS (e.g., the rate of disease spread). The appropriate time horizon is not limited to the period that status can be quantitatively modeled or predicted within predetermined limits of statistical confidence.

> High risk: A species or ESU with a high risk of extinction it is at or near a level of abundance, productivity, or spatial structure that place its persistence in question. The demographics of a species/DPS at such a high level of risk may be highly uncertain and strongly influenced by stochastic or depensatory processes. Similarly, a species/DPS may be at high risk of extinction if it faces clear and present threats (e.g., confinement to a small geographic area, imminent destruction, modification or curtailment of its habitat, or disease epidemic) that are likely to create such imminent demographic risks.

Quantitative and qualitative conservation assessments for other species have often used a 100-year time frame in their extinction risk evaluations (McElhany et al. 2000, Morris and Doak 2002) and the BRT adopted this time scale as the period over which it had confidence in evaluating risk. In particular, the BRT interpreted the high risk category as a greater than approximately 5% risk of extinction within approximately 100 years, and the moderate risk category as a greater than 50% risk of moving into the high risk category within 30–80 years. Beyond the 30–80 year time horizon, the projected effects on OCCS viability from climate change, ocean conditions, and trends in freshwater habitat become very difficult to predict with any certainty. The overall extinction risk determination reflected informed professional judgment by each BRT member, based on the quantitative and qualitative information reviewed in this report. This assessment was guided by the results of the risk matrix analysis (see below), supplemented by results from the DSS (Table 7), and integrating information about demographic risks with expectations about likely interactions with threats and other factors.

Risk Matrix Approach

In previous NMFS status reviews, BRTs have used a risk matrix as a method to organize and summarize the professional judgment of a panel of knowledgeable scientists. This approach, described in detail by Wainwright and Kope (1999), has been used for more than 10 years in Pacific salmonid status reviews (e.g., Good et al. 2005, Hard et al. 2007), as well as in reviews of Pacific hake, walleye pollock (*Theragra chalcogramma*), Pacific cod (*Gadus macrocephalus*) (Gustafson et al. 2000), Puget Sound rockfishes (Stout et al. 2001a, Drake et al. 2010), Pacific herring (*Clupea pallasii*) (Stout et al. 2001b, Gustafson et al. 2006), eulachon (*Thaleichthys pacificus*) (Gustafson et al. 2010), and black abalone (*Haliotis cracherodi*) (VanBlaricom et al. 2009).

In this risk matrix approach, the collective condition of individual populations is summarized at the ESU level according to four demographic risk parameters: abundance, growth rate/productivity, spatial structure and connectivity, and diversity (Table 25). These viability parameters, outlined in McElhany et al. (2000), reflect concepts that are well founded in conservation biology and are generally applicable to a wide variety of species. They describe demographic risks that individually and collectively provide strong indicators of extinction risk. The summary of demographic risks and other pertinent information obtained by this approach is then considered by the BRT in determining the species' overall level of extinction risk.

After reviewing all relevant biological information for the species, including the threats currently impacting the ESU or reasonably certain to impact the ESU in the future, each BRT member assigned a risk score (Table 25) to each of the four parameters. The scores were tallied (means, modes, and range of scores), reviewed, and the range of perspectives discussed by the BRT before making its overall risk determination. Although this process helped to integrate and summarize a large amount of diverse information, there was no simple way to translate the risk matrix scores directly into a determination of overall extinction risk. For example, an ESU with a single extant population might be at a high level of extinction risk because of high risk to spatial structure and connectivity, even if it exhibited low risk for the other parameters. Another species might be at risk of extinction because of moderate risks to several parameters.

To allow individuals to express uncertainty in determining the overall level of extinction risk facing the species, the BRT adopted the likelihood point method, often referred to as the FEMAT (Forest Ecosystem Management Assessment Team) method because it is a variation of a method used by scientific teams evaluating options under the Northwest Forest Plan (FEMAT 1993). In this approach, each BRT member distributes 10 likelihood points among the 3 extinction risk categories, reflecting his or her opinion of how likely that category correctly reflects the true species status (Table 26). Thus if a member were certain that the species was in the low risk (not at risk) category, he or she could assign all 10 points to that category. A reviewer with less certainty about the species' status could split the points among two or even three categories. This method has been used in all status reviews for anadromous Pacific salmonids since 1999, as well as in reviews of Puget Sound rockfishes (Stout et al. 2001a, Drake et al. 2010), Pacific herring (Stout et al. 2001b; Gustafson et al. 2006), Pacific hake, walleye pollock, Pacific cod (Gustafson et al. 2000), eulachon (Gustafson et al. 2010), and black abalone (VanBlaricom et al. 2009.)

Table 25. Risk Matrix template used by the BRT to capture comments and assessment of risk. Risks for each VSP parameter are ranked on a scale of 1 (very low risk) to 5 (very high risk).*

Risk assessment matrix	
Risk category	**Risk score**
Abundance Comments:	
Growth rate/productivity Comments:	
Spatial structure and connectivity Comments:	
Diversity Comments:	

*The rankings are defined as follows.

1. Very low risk: It is unlikely that this factor contributes significantly to risk of extinction, either by itself or in combination with other factors.
2. Low risk: It is unlikely that this factor contributes significantly to risk of extinction by itself, but there is some concern that it may, in combination with other factors.
3. Moderate risk: This factor contributes significantly to long-term risk of extinction, but does not in itself constitute a danger of extinction in the near future.
4. High risk: This factor contributes significantly to long-term risk of extinction and is likely to contribute to short-term risk of extinction in the foreseeable future.
5. Very high risk: This factor by itself indicates danger of extinction in the near future.

Table 26. FEMAT voting sheet. Each of 13 BRT members allocated 10 likelihood points among the three status categories. The numbers allocated across the categories should add up to 10 points. Low risk category = not at risk.

ESU is at high risk	**ESU is at moderate risk**	**ESU is at low risk**	**Total = 10 pts.**

In its May 2010 draft report, the BRT reported the risk assessment matrix analysis and the overall extinction risk assessment that was conducted under two different sets of assumptions. First, the BRT evaluated extinction risk based on the demographic risk parameters (abundance, growth rate/productivity, spatial structure and connectivity, and diversity) recently exhibited by the ESU, assuming that the threats influencing ESU status would continue unchanged into the future. This case in effect assumed that all of the threats evaluated in the previous section of the report were already fully manifest in the current ESU status and would in aggregate neither worsen nor improve in the future. In the 2010 draft report, the BRT also evaluated extinction risk based on the demographic risk criteria currently exhibited by the ESU, taking into account consideration of predicted changes to threats that the BRT evaluated to be not yet manifest in the current demographic status of the ESU. In effect, this scenario asked the BRT to evaluate whether threats to the ESU would lessen, worsen, or remain constant compared to current conditions.

In the time since the completion of the last risk assessment in 2010, the BRT considered additional information on the potential magnitude and trajectory of threats including climate change, changes in ocean conditions, and trends in freshwater habitat. The BRT also further refined the time horizon used to evaluate whether the OCCS ESU was at moderate risk of extinction. Considering this new information, the BRT felt it unnecessary and potentially confusing to conduct the risk assessment under multiple sets of assumptions. For the final risk assessment reported here, therefore, each BRT member evaluated all the available information on current demographic status and threats to come to a single overall conclusion on the degree of extinction risk.

Summary of Risk Conclusions

The mean risk matrix scores for each demographic risk factor fell between the low risk (not at risk) (2) and moderate risk (3) categories (Table 27), indicating that the BRT as a whole did not consider any of the demographic risk parameters as likely to contribute substantially to a high risk of short-term extinction when considered on its own. The overall assessment of extinction risk to the OCCS ESU, taking into account both the demographic risk parameters and an evaluation of threats, indicated considerable uncertainty about its status; most likelihood points were evenly split between moderate risk and low risk at 47% each, and a small minority of points amounting to 6% indicated high risk.

That lack of a strong mode in the overall assessment of risk (6%, 47%, 47%) and the large range in the demographic risk scores (Table 27) were indicative of considerable uncertainty

Table 27. Assessment of the risk associated with each of four demographic factors. Risks for each demographic factor are ranked on a scale of 1 (very low risk) to 5 (very high risk).

	Abundance	Growth rate/ productivity	Structure and connectivity	Diversity
Mean	2.21	2.63	2.33	2.67
Median	2	3	2	2.75
Minimum	1	2	1	2
Maximum	4	3.5	4	4

within and among BRT members about the current level of risk facing the ESU. This uncertainty was due largely to the difficulty in balancing the clear improvements in some aspects of the ESU's status over the last approximately 15 years against persistent threats driving the longer term status of the ESU, which probably have not changed over the same time frame and are predicted to degrade in the future. Both issues are discussed in more detail below. In addition, the BRT noted that accurately predicting the long-term trend of a complex system is inherently difficult, and this also led to some uncertainty in the overall risk assessment.

The BRT concluded that some aspects of the ESU's status have clearly improved since the initial status review in the mid-1990s (Weitkamp et al. 1995). In particular, the BRT assigned a relatively low mean risk score to the abundance factor, noting that spawning escapements were higher in some recent years than they had been since 1970 (Figure 5). Recent total returns (preharvest recruits) were also substantially higher than the low extremes of the 1990s, but still mostly below levels of the 1960s and 1970s (Figure 5). The BRT attributed the increased spawner escapements largely to a combination of greatly reduced harvest rates, reduced hatchery production, and improved ocean conditions (see New Data and Updated Analysis section). Even with the recent increases, however, abundance remains at approximately 10% of estimated historical abundance (≈150,000 current compared to ≈1.5 million historical; see discussion in the Current Biological Status subsection).

The BRT noted that compared to the mid-1990s, the ESU contained relatively abundant wild populations throughout its range, leading to a relatively low risk associated with spatial structure (Table 27). The BRT also discussed the observation that the recent natural origin spawning abundance of the OCCS ESU was higher than that observed for other listed salmon ESUs, although some members noted that the fifteenfold variability in abundance since the mid-1990s brings into question how heavily to weigh abundance as an indicator of status. Finally, the BRT noted that hundreds of individual habitat improvement projects over the last approximately 15 years had likely benefited the ESU, although quantifying these benefits is difficult.

The BRT discussed some ongoing positive changes that are likely to become manifest in abundance trends for the ESU in the future. In particular, hatchery production continues to be reduced with the cessation of releases in the North Umpqua River and Salmon River populations, and the BRT expects that the near-term ecological benefits from these reductions would result in improved natural production for these populations in future. In addition, the BRT expected that reductions in hatchery releases that have occurred over the past decade may continue to produce some positive effects on the survival of the ESU in the future, due to the time it may take for past genetic impacts to become attenuated.

Despite these positive factors, the BRT also had considerable concerns about the long-term viability of the ESU. The BRT continued to be concerned that there had been a long-term decline in the productivity of the ESU from the 1930s through the 1990s (Figure 5). Despite some improvements in productivity in the early 2000s, the BRT was concerned that the overall productivity of the ESU remains low compared to what was observed as recently as the 1960s and 1970s (Figure 5 and Figure 6). The BRT was also concerned that the majority of the improvement in productivity in the early 2000s was likely due to improved ocean conditions, with a relatively smaller component due to reduced hatchery production (Buhle et al. 2009).

The BRT noted that the legacy of past forest management practices combined with lowland agriculture and urban development has resulted in a situation in which the areas of highest habitat capacity (intrinsic potential) are now severely degraded (see Land management—stream habitat complexity subsection). The BRT also noted that the combined ODFW/NMFS analysis of freshwater habitat trends for the Oregon coast found little evidence for an overall improving trend in freshwater habitat conditions since the mid-1990s and evidence of negative trends in some strata (Appendix C). The BRT was also concerned that recent changes in the protection status of beavers, which through their dam building activities create coho habitat, could result in further negative trends in habitat quality. The BRT was therefore concerned that when ocean conditions cycle back to a period of poor survival for coho salmon, the ESU may rapidly decline to the low abundance seen in the mid-1990s.

Some members of the BRT observed that the reduction in risks from hatchery and harvest are expected to help buffer the ESU when marine survival returns to a lower level, likely resulting in improved status compared to the situation in the mid-1990s. Others noted that potential declines in beavers, observed negative trends in some habitat features, and the potential for more severe declines in marine productivity could result in even lower abundance levels than during the last period of poor ocean conditions. On balance, the BRT as a whole was uncertain about whether the long-term downward trajectory of the ESU's status has been arrested and uncertain about the ESU's ability to survive another prolonged period of low ocean survivals.

Finally, the BRT was also concerned that global climate change will lead to a long-term downward trend in freshwater and marine coho salmon habitat compared to current conditions (see Effects of climate change on the OCCS ESU subsection). There was considerable uncertainty about the magnitude of most of the specific effects climate change will have on salmon habitat, but the BRT was concerned that most changes associated with climate change are expected to result in poorer and more variable habitat conditions for OCCS than exist currently (Table 14). Some members of the BRT noted that changes in freshwater flow patterns as a result of climate change may not be as severe in the Oregon coast as in other parts of the Pacific Northwest, while others were concerned by recent observations of extremely poor marine survival rates for several West Coast salmon populations. The distribution of overall risk scores reflects some of this uncertainty.

Significant Portion of its Range Question

The BRT concluded that, when future conditions are taken into account, the OCCS ESU as a whole is at moderate risk of extinction. The BRT therefore did not explicitly address whether the ESU was at risk in only a significant portion of its range.

Glossary

abundance. The number of fish in a population. See also **population**.

artificial propagation. Hatchery spawning and rearing of salmon, usually to the smolt stage.

AUC. For *area under the curve*. A statistical technique for estimating an annual total number of spawners from periodic spawner counts. See also **spawner**.

barrier. A blockage such as a waterfall, culvert, or rapid that impedes the movement of fish in a stream system.

BLM. For *U.S. Bureau of Land Management*.

BRT. For *biological review team*. The team of scientists who evaluate scientific information for National Marine Fisheries Service status reviews.

catastrophic events. Sudden events that disastrously alter large areas of landscape. These can include floods, landslides, forest fires, and volcanic eruptions.

channel gradient. The slope of a stream reach.

CLAMS. For *Coastal Landscape Analysis and Modeling Study*. A cooperative project between the Oregon State University Department of Forestry and the U.S. Forest Service Pacific Northwest Forest Science Laboratory.

comanagers. Federal, state, and tribal agencies that cooperatively manage salmon in the Pacific Northwest.

critical habitat. 1) specific areas within the geographical area occupied by the species at the time of listing, on which are found those physical or biological features that are essential to the conservation of the listed species and that may require special management considerations or protection, and 2) specific areas outside the geographical area occupied by the species at the time of listing that are essential for the conservation of a listed species. If a species is listed or critical habitat is designated, ESA section 7(a)(2) requires federal agencies to ensure that activities they authorize, fund, or carry out are not likely to jeopardize the continued existence of such a species or to destroy or adversely modify its critical habitat (NMFS 2008).

delisting. Taking a species off the endangered species list.

demographic risk. Risks to a small population resulting from population processes such as depensation or chance events in survival or reproductive success.

density effects. Survival of juvenile salmon may be influenced by their density. Survival is usually higher when density is low.

dependent populations. Populations that rely on immigration from surrounding populations to persist. Without these inputs, dependent populations would have a lower likelihood of persisting over 100 years.

depensation. The effect where a decrease in spawning stock leads to reduced survival or production of eggs through either 1) increased predation per egg given constant predator pressure, or 2) the Allee effect (a positive relationship between population density and the reproduction and survival of individuals) with reduced likelihood of finding a mate.

DNA. For *deoxyribonucleic acid*. A complex molecule that carries an organism's heritable information. The two types of DNA commonly used to examine genetic variation are mitochondrial DNA (mtDNA), a circular molecule that is maternally inherited, and nuclear DNA, which is organized into a set of chromosomes. See also **electrophoresis**.

DPS. For *distinct population segment*. A population or group of populations of a vertebrate species that is discrete from other populations and significant to the biological species as a whole. See also **ESU**.

DSS. For *decision support system*. A computer application that assists users in using data and models to solve problems. It typically links and analyzes many pieces of data or models at a variety of scales, producing results that aid in decision making rather than replacing human judgment.

ecoregion. An integration of physical and biological factors such as geologic history, climate, and vegetation.

electrophoresis. The movement of charged particles in an electric field. This process has been developed as an analytical tool to detect genetic variation revealed by charge differences on proteins or molecular weight in DNA. See also **DNA**.

endangered species. A species in danger of extinction throughout all or a significant portion of its range. See also **ESA** and **threatened species**.

EPA. For *U.S. Environmental Protection Agency*.

ESA. For *U.S. Endangered Species Act*. Passed by Congress in 1973, its purpose is to provide a means to conserve the ecosystems on which endangered species and threatened species depend. See also **endangered species** and **threatened species**.

escapement. Usually refers to adult fish that escape from fisheries and natural mortality to reach the spawning grounds.

estuarine habitat. Areas available for feeding, rearing, and smolting in tidally influenced lower reaches of rivers. These include marshes, sloughs and other backwater areas, tidal swamps, and tide channels.

ESU. For *evolutionarily significant unit*. An ESU represents a distinct population segment of Pacific salmon under the Endangered Species Act that 1) is substantially reproductively isolated from conspecific populations and 2) represents an important component of the evolutionary legacy of the species. See also **DPS**.

exploitation rate. The proportion of adult fish from a population that die as a result of fisheries.

extinction. The loss of a species or ESU; may also be used for the extirpation of local populations.

factors for decline. These are factors identified that caused a species to decrease in abundance and distribution and become threatened or endangered.

fecundity. The number of offspring produced per female over her lifetime.

fish-day. Fish-days are calculated by multiplying the live fish observed on each survey date by the number of days between surveys. These values are then summed for the entire observation period to generate a relative index of spawner abundance at a reach for any given year (Pess et al. 2002b).

fourth-field and fifth-field hydrologic units. In the U.S. Geological Survey, hydrologic units have been divided at different scales. The area of a fourth-field hydrologic unit is 440,000 acres and a fifth-field hydrologic unit is between 40,000 and 250,000 acres.

freshwater habitat. Areas available for spawning, feeding, and rearing in freshwater.

fry. Young salmon that have emerged from the gravel and no longer have a yolk sack.

full seeding. Having enough spawners to fully occupy available juvenile habitat with offspring.

functionally independent population. A high-persistence population whose dynamics or extinction risk over a 100-year time frame is not substantially altered by exchanges of individuals with other populations (migration). Functionally independent populations are net donor populations that may provide migrants for other types of populations. This category is analogous to the independent populations of McElhany et al. (2000).

fuzzy logic. A system of logic in which a statement can be true, false, or any of a continuum of values. A type of logical analysis that allows a system to process imprecise information in evaluating conditions.

gene conservation group. Management area defined by Kostow (1995) to conserve genetic diversity in Oregon Coast coho salmon. See also **monitoring area**.

genetic bootstrap support. A measure of the confidence in a particular branch in a genetic tree. Specifically a large number of trees are created using randomly drawn sets of loci sampled from the data with replacement. The bootstrap value for a node is the proportion of the trees that have all the samples contained on that node.

gradient. The slope of a stream segment.

habitat quality. The suitability of physical and biological features of an aquatic system to support salmon in the freshwater and estuarine system.

hatchery. A facility where artificial propagation of fish takes place.

historical abundance. The number of fish produced before the influence of European settlement.

HLFM. For *habitat limiting factors model.*

HTWG. For *Habitat Trends Working Group*. A joint group formed by NWFSC and ODFW and composed of scientists from each agency, with contributions by statisticians from the EPA and Oregon State University.

hydrology. The distribution and flow of water in an aquatic system.

IMST. For *Independent Multidisciplinary Science Team*. A scientific advisory body to the Oregon legislature and governor on watershed, forestry, agriculture, and fisheries science issues.

independent population. A population that historically would have a high likelihood of persisting in isolation from neighboring populations for 100 years.

integrated hatchery. In this case, the Cow Creek hatchery program where wild coho salmon are regularly taken into the hatchery program's broodstock. Typically more than 10% of the broodstock annually is of wild fish origin. In some years, 100% of the broodstock is wild fish.

intrinsic potential. A modeled attribute of streams that includes the channel gradient, valley constraint, and mean annual discharge of water. Intrinsic potential in this report refers to a measure of potential coho salmon habitat quality. This index of potential habitat does not indicate current actual habitat quality.

isolation. The degree to which a population is unaffected by migration to and from other populations. As the influence of migration decreases, a population's isolation increases.

jack. A male coho salmon that matures at age-2 and returns from the ocean to spawn a year earlier than normal.

juvenile. A fish that has not matured sexually.

keystone species. A species that plays a pivotal role in establishing and maintaining the structure of an ecological community. The impact of a keystone species on the ecological community is more important than would be expected based on its biomass or relative abundance.

Landsat. For *land remote-sensing satellite.* The satellites supply global land surface images and data.

life history. The specific life cycle of a fish from egg to adult. Life history includes changes experienced from birth through death and variation in traits such as the size and age at maturity and fecundity. Traits such as juvenile growth rate and age at ocean emigration are aspects of coho salmon life history.

limiting factors. Factors that limit survival or abundance. They are usually related to habitat quantity or quality at different stages of the life cycle. Harvest and predation may also be limiting factors.

listed species. Species included on the List of Endangered and Threatened Species, authorized under the Endangered Species Act and maintained by the U.S. Fish and Wildlife Service and NMFS.

lowland habitat. Low-gradient stream habitat with slow currents, pools, and backwaters used by fish. This habitat is often converted to agricultural or urban use.

LWD. For *large woody debris*. Large piece of woody material such as a log or stump that intrudes into a stream channel.

marine survival rate. The proportion of smolts entering the ocean that return as adults.

metacercaria. Tiny cysts that contain the intermediate stages of parasites.

microsatellite. A class of repetitive DNA used for estimating genetic distances.

migrant. A fish that is born in one population but returns to another population to spawn.

migration. Movement of fish from one population to another.

migration rate. The proportion of spawners that migrate from one population to another. See also **stray rate**.

monitoring area. ODFW's monitoring areas are similar to but not identical to gene conservation groups. Additional information online at http://nrimp.dfw.state.or.us/crl /default.aspx?pn=AIProjOrPlnSalWtrshd. See also **gene conservation group**.

naturally produced fish. Fish that were spawned and reared in natural habitats, regardless of parental origin. See also **wild fish**.

NIS. For *nonindigenous species*.

NRR. For *natural return ratio*. The ratio N/T, where N is naturally produced spawners in one generation and T is total (hatchery produced + naturally produced) spawners in the previous generation.

OCCS. For *Oregon coast coho salmon.*

OCN. For *Oregon coast natural coho salmon*. Often used by ODFW to distinguish from hatchery-raised fish and includes fish from the Southern Oregon/Northern California Coasts Coho Salmon ESU in Oregon.

OCSRI. For *Oregon Coastal Salmon Restoration Initiative*. Now the Oregon Plan for Salmon and Watersheds. A plan established by the State of Oregon in 1997 to restore salmon runs, improve water quality, and achieve healthy watersheds and strong communities throughout the state.

ODF. For *Oregon Department of Forestry.*

ODFW. For *Oregon Department of Fish and Wildlife.*

OFPBDS. For *Oregon Fish Passage Barrier Data Set*.

ONCC TRT. For *Oregon and Northern California Coast Technical Recovery Team*.

OPI. For *Oregon Production Index*.

OWEB. For *Oregon Watershed Enhancement Board*.

OWRD. For *Oregon Water Resources Department.*

parasite prevalence. The number of hosts infected with one or more individuals of a particular parasite species (or taxonomic group) divided by the number of hosts examined for that parasite species.

parr. The life stage of salmonids that occurs after fry and is generally recognizable by dark vertical bars (parr marks) on the sides of the fish.

PDO. For *Pacific Decadal Oscillation*. A long-term pattern of Pacific Ocean climate variability, with events lasting 20 to 30 years and oscillating between warm and cool regimes.

persistent population. One that is able to persist (i.e., not go extinct) over a 100-year period without support from other populations. This includes an ability to survive prolonged periods of adverse environmental conditions, which may be expected to occur at least once in the 100-year time frame.

PFMC. For *Pacific Fishery Management Council*.

PIT tag. For *passive integrated transponder tag*. An injectable, internal, radio-type tag that allows unique identification of a marked fish passing within a few inches of a monitoring site.

population. A group of fish of the same species that spawns in a particular locality at a particular season and does not interbreed substantially with fish from any other group. See also **abundance**.

population classification. The grouping of populations into functionally independent, potentially independent, and dependent classes.

population dynamics. Changes in the number, age, and sex of individuals in a population over time, and the factors that influence those changes. Five components of populations that are the basis of population dynamics are birth, death, sex ratio, age structure, and dispersal.

population identification. Delineating the boundaries of historical populations.

population structure. This includes measures of age, density, and growth of fish populations.

potentially independent population. High-persistence population whose population dynamics are substantially influenced by periodic immigration from other populations. In the event of the decline or disappearance of migrants from other populations, a potentially independent population could become a functionally independent population.

production. The number of fish produced by a population in a year.

productivity. The rate at which a population is able to produce fish.

recovery. The reestablishment of a threatened or endangered species to a self-sustaining level in its natural ecosystem (i.e., to the point where the protective measures of the ESA are no longer necessary).

recovery domain. The area and species for which a TRT is responsible.

recovery plan. A document identifying actions needed to make populations of naturally produced fish comprising the OCCS ESU sufficiently abundant, productive, and diverse so that the ESU as a whole will be self-sustaining and will provide environmental, cultural, and economic benefits. A recovery plan also includes goals and criteria by which to measure the ESU's achievement of recovery, and an estimate of the time and cost required to carry out the actions needed to achieve the plan's goals.

recovery scenario. Sequence of events expected to lead to recovery of Oregon coast coho salmon.

run timing. The time of year (usually identified by week) when spawning salmon return to the spawning beds.

salmonid. Fish of the family Salmonidae, including salmon, trout, and char.

significant. Biological significance refers to an effect that has a noteworthy impact on health or survival.

smolt. A life stage of salmon that occurs just before the fish leaves freshwater. Smolting is the physiological process that allows salmon to make the transition from freshwater to salt water.

smolt capacity. The maximum number of smolts a basin can produce. Smolt capacity is related to habitat quantity and quality.

spawner. Adult fish on the spawning grounds. See also **AUC**.

spawner survey. Effort to estimate the number of adult fish on spawning grounds. It uses counts of redds and fish carcasses to estimate escapement and identify habitat. Annual surveys can be used to compare the relative magnitude of spawning activity between years.

species. Biological definition: A group of organisms formally recognized by the scientific community as distinct from other groups. Legal definition: refers to joint policy of the USFWS and NMFS that considers a species as defined by the ESA to include biological species, subspecies, and DPSs.

stray rate. As used in this document, stray rate refers to the number of spawning adults that return to a stream other than their natal stream within a basin. See also **migration rate**.

sustainability. An attribute of a population that persists over a long period of time and is able to maintain its genetic legacy and long-term adaptive potential for the foreseeable future.

threatened species. A species not presently in danger of extinction, but likely to become so in the foreseeable future. See also **endangered species** and **ESA**.

TRT. For *technical recovery team*. The TRT establishes biologically based ESA recovery goals for listed salmonids within a given recovery domain. Members serve as science advisors to the recovery planning phase.

USFS. For *U.S. Forest Service*.

valley constraint. The valley width available for a stream or river to move between valley slopes.

viability. The likelihood that a population will sustain itself over a 100-year time frame.

viability criteria. A prescription of a population conservation program that will lead to the ESU having a negligible risk of extinction over a 100-year time frame.

VSP. For *viable salmonid population*. An independent population of any Pacific salmonid (genus *Oncorhynchus*) that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a long time frame (McElhany et al. 2000).

warm water fish. Spiny-rayed fish such as sculpins, minnows, darters, bass, walleye, crappie, and bluegill that generally tolerate or thrive in warm water.

wild fish. Fish whose ancestors have always lived in natural habitats, that is, those with no hatchery heritage. See also **naturally produced fish**.

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Appendix A: Threats Matrix and DSS Criteria

This appendix consists of two tables: Table A-1, the Oregon Coast Coho Salmon (OCCS) Biological Review Team's (BRT) threats matrix; and Table A-2, the descriptions, metrics, data sets, and sources for decision support system (DSS) criteria.

Table A-1. Threats matrix used by OCCS BRT. This was used as a memory tool to keep track of information presented.

Stratum	**Marine productivity/ocean conditions**	**Global climate change**	**Artificial propagation**	**Restriction of fish passage**	**Water withdrawal**	**Disturbance**	**Riparian loss (not addressed)**	**Splash damming, log drives legacy**	**Wetland, beaver dam loss**	**Estuary destruction**	**Stream complexity**	**Mining**	**Water quality degradation**	**Land use conversion, urbanization**	**Harvest**	**Disease and parasitism**	**Predation**	**Urban growth**	**Agricultural practices**	**Dredge and fill**	**Forest practices**	**Northwest Forest Plan**
North Coast																						
Mid-Coast																						
Lakes																						
Umpqua River																						
Mid-South Coast																						
Whole ESU																						
ESA listing factor	Other natural or human factors		Destruction, modification, curtailment of OCCS habitat or range												Over-utilization	Disease and predation		Regulatory mechanisms				

Table A-2. Descriptions, metrics (including time periods), data sets, and sources for DSS criteria.

Criterion	Description	Metric	Data set	Source
PP-1	Population productivity	Geometric mean of the natural return ratio for broodyears with spawner abundances below the median of the last 4 generations (12 years)	Spawner abundance by population, hatchery fish spawning naturally by population	ODFW random spawning surveys (Jacobs et al. 2002) as reported by ODFW[a]
PP-2	Probability of persistence	Results of four population viability models	Various	Models and associated data sets are described in detail in Wainwright et al. (2008)
PP-3	Critical abundance	Average peak spawner density in the lowest 3 of the last 12 years	Spawner counts	ODFW random spawning surveys (Jacobs et al. 2002) as reported by ODFW[a]
PD-1	Spawner abundances	Long-term harmonic mean of naturally produced spawners (both 3-year-old adults and 2-year-old jacks)	Annual surveys conducted by ODFW using various methodologies from 1950 to the present	ODFW[a]
PD-2	Artificial influence	Six-year (2 generations) mean of annual estimates of the proportion of naturally produced fish in spawning surveys	Survey estimates of hatchery fish spawning naturally	ODFW random spawning surveys[a]
PD-3	Spawner distribution	Average occupancy rate of watersheds during the most recent 12 years, analyzed by fifth field hydrologic units	ODFW random spawning surveys	Surveys (Jacobs et al. 2002) as reported by ODFW[a]
PD-4	Juvenile distribution	Average occupancy rate of surveyed reaches with at least two pools during the most recent 12 years, analyzed by fifth field hydrologic units	ODFW summer juvenile surveys	ODFW[a]
PF	Population functionality	Estimated smolt capacity for each basin as estimated by the ODFW Habitat Limiting Factors Model version 6.0 (Nickelson et al. 1992, Nickelson 1998)	ODFW habitat survey data	ODFW[b]
ED-1 ED-2 ED-3	ESU-level sustainability criteria	Expert opinion	None	Lawson et al. 2007

[a] K. Moore, Oregon Dept. Fish and Wildlife, Corvallis (ODFW), OR. Pers. commun., 9 September 2009.
[b] T. Nickelson, ODFW, Corvallis, OR. Pers. commun., 5 January 2006.

Appendix B: Disturbance

The condition of aquatic ecosystems and associated fish populations are a function, at least in part, of the characteristics of the surrounding landscape (Frissell et al. 1986, Naiman et al. 2000). Timber harvest and associated roads have extensively altered aquatic ecosystems throughout the Pacific Northwest (Everest and Reeves 2007). A consequence of these effects of timber harvest activities is that the behavior of ecosystems is altered, which in turn has consequences for fish populations and their habitat (Reeves et al. 1995). There is a negative association between the amount of in-channel large wood and percent of area in a watershed intensively logged (Murphy and Koski 1989, Bilby and Ward 1991, Montgomery et al. 1995). Burnett et al. (2006) found that the mean density of large wood in Elk River on the southern Oregon coast was positively related to the area in larger trees in the catchment. Reeves et al. (1993) examined watersheds in the Oregon Coast Range and found that the diversity of the fish assemblage and the amount of large wood was significantly greater in streams in which less than or equal to 25% of the watershed was clear-cut, compared to watersheds in which more than 25% of the area was clear-cut.

The condition of aquatic habitat and fish populations is also directly correlated with the density of roads in a watershed, which in turn is generally directly related to the amount and intensity of land management activities (Lee et al. 1997). Roads are sources of sediment as either surface erosion or mass erosion (Furniss et al 1991). They also can alter water delivery by increasing the drainage network, particularly in the upper portions of the network. Sharma and Hilborn (2001) examined 14 streams in Washington and found that smolt density was inversely correlated with the density of roads. Logging activities involve the creation and maintenance of roads, and logging has been linked directly to increased sediment levels in streams (Platts et al. 1989).

The Oregon Northern California Coast Technical Review Team (ONCC TRT) (Wainwright et al. 2008) was unable to include habitat data directly in its biological status criteria, because at the time there was no uniform measure of habitat quality over the entire evolutionarily significant unit (ESU). Habitat surveys by the Oregon Department of Fish and Wildlife (ODFW) were available, but the density and distribution of on-the-ground surveys make them unsuitable for fine-scale analysis. ODFW and this biological review team (BRT) have used these habitat surveys to provide a general assessment of stream complexity (see Appendix C).

Satellite images have the potential for measuring properties of large landscapes at a relatively fine scale. Recent public availability of Landsat imagery and the development of tools for analysis now make it possible to analyze disturbance patterns on a fine temporal and spatial scale. In an analysis conducted for the BRT, satellite annual vegetation maps of the Oregon Coast Coho Salmon (OCCS) ESU were updated through 2008 and analyzed for patterns of disturbance for the time period 1986–2008. The scale of resolution of these analyses is approximately 100 m, so disturbances as small as 1 ha theoretically can be detected. This made it possible to detect individual disturbance events from the satellite images and map new disturbances on an annual basis. Intensity of disturbance can also be measured, so low-intensity disturbances (i.e., thinning) can be distinguished from high-intensity disturbances (i.e., clear-cut). Fires were also mapped, but fire has had a small role in shaping habitat in the OCCS ESU over the past 23 years.

Yearly maps of forest disturbance were derived from Landsat Thematic Mapper (TM) satellite imagery. Useful Landsat TM images have been acquired continuously for the entire conterminous United States (and portions of the world) since 1984; they form the basis for many land cover and land cover change maps used in natural resource disciplines (Cohen and Goward 2004). Time series of Landsat TM imagery were assembled for all areas that intersected the footprint of the OCCS ESU, then processed with LandTrendr, a change detection package developed at Oregon State University (Kennedy et al. 2010, Kennedy et al. 2012). Image preparation includes basic atmospheric correction (using the COST approach, Chavez Jr. 1996) followed by radiometric normalization of all scenes within one time series (using the MADCAL algorithms, Canty et al. 2004) and a semiautomated cloud-screening approach with human supervision (Kennedy et al. 2010, Kennedy et al. 2012.). After image preparation, the time series of a spectral index (here the Normalized Burn Ratio, van Wagtendonk et al. 2004) for each pixel (30 × 30 m footprint) was extracted, and temporal segmentation algorithms were used to identify periods of both stability and change in each pixel's trajectory (Kennedy et al. 2010, Kennedy et al. 2012) The segmentation approach utilizes information from nearly every year in the satellite record (with occasional gaps caused by persistent cloud cover), increasing the signal-to-noise ratio of the data and improving the ability to distinguish subtle change from random noise.

Disturbance events detected in the segmentation phase were then used to create maps of forest disturbance. In each pixel, the magnitude of spectral change in a disturbance event was converted to an estimate of percent vegetation cover loss by relating spectral change to percent vegetation cover using a statistical model of cover developed from photo-interpreted plots (Cohen et al. 2010). Pixels with disturbance magnitude less than 10% vegetation loss were removed, groups of pixels with the same disturbance year and with size greater than 1 ha were retained as disturbance patches, and mean percent cover change across all pixels in the patch was recorded. Separately, a mask distinguishing forest from nonforest was created from summary results obtained during segmentation, capturing all areas where forest existed at any point in the 23-year period (1985 to 2008). Pixels in nonforest areas were removed from further consideration.

Of the remaining disturbance patches, 300 disturbed in 2002, 2003, or 2004 were randomly selected for change labeling. A trained interpreter viewed each patch on 2005 digital air photos and assigned a change-type label as either a clear-cut (less than 15% tree cover remaining) or a thinning (tree removal but more than 15% tree cover remaining). The labeled values were compared to the mean patch-level percent cover change calculated earlier and distributions of cover change were evaluated for clear-cuts and thins. The modal percent cover change for clear-cuts and thins was 55% and 22.5% change, respectively, with the intersection of the distributions occurring at 35% cover change. This value was used to separate patches into "harvest high" or "harvest low" categories for all subsequent analyses. Separately, a point-based validation tool was used to corroborate disturbance mapping across all years. At 274 points randomly distributed throughout the Coho ESU, the LandTrendr algorithms correctly identified disturbance or lack of disturbance in 89.7% of plots; omission and commission error were balanced at 5.4% and 4.7%, respectively.

Disturbance was widespread over the ESU, and predominantly of high intensity (Figure B-1). Disturbance patterns varied over space, time, and land ownership. Some river systems have experienced higher disturbance than others (Figure B-2). The Siletz, Necanicum, and Tahkenitch have had up to 50% of basin area disturbed in the analysis period, while North Umpqua has had less than 10% disturbance.

The time series of cumulative disturbance, derived from Landsat images, is shown for four major river systems in the OCCS ESU in Figure B-3. Disturbance in these systems spans the range observed in the ESU, from a low of 10% (Upper Nehalem) to 50% (Siletz). Most disturbance is in the high category, with a lesser amount of low intensity disturbance, and the proportion of high to low disturbance is fairly constant through the time period. Three patterns of disturbance are evident (Figure B-3). Constant rates of annual disturbance occurred in the Siletz and Alsea river systems. The Necanicum River basin showed little harvest activity early in the time period, with an exponential increase in more recent years. By contrast, the North Umpqua River basin showed an overall low rate of disturbance, with most of the activity early in the time period.

Changes in the regulatory environment have largely driven patterns of land disturbance over the past two decades. Disturbance in four land ownership categories in the Alsea River basin is shown in Figure B-4. Prior to 1990, there were high rates of disturbance on federal lands (BLM and USFS). With implementation of the Northwest Forest Plan, federal logging activity decreased to very low levels. Logging on private lands proceeded at a steady pace through the late 1990s, then increased. This general pattern is evident throughout the ESU, with logging on National Forest and BLM land decreasing after 1989 and activity on private lands increasing.

The recent availability of Landsat images, along with the development of tools for analysis, allowed us a comprehensive, uniform picture of disturbance patterns that was heretofore unavailable. Preliminary analysis showed that disturbance has been widespread in the ESU, some basins experienced much higher disturbance than others, rates of disturbance are relatively constant, and the most intense disturbance has moved from federal to private nonindustrial lands in response to policy changes. Short-term fluctuations in disturbance rates often can be attributed to economic conditions.

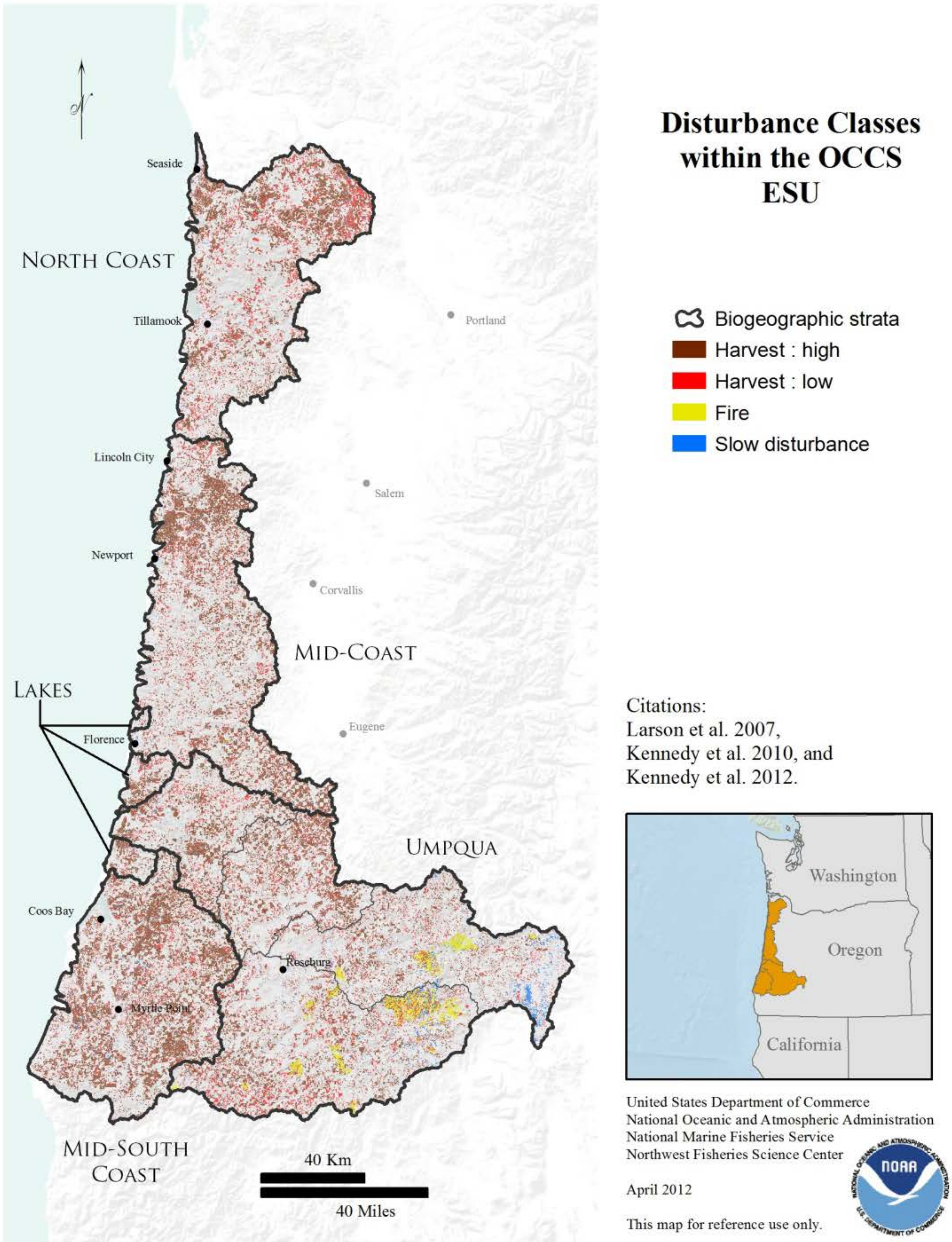

Figure B-1. Distribution and intensity of vegetation disturbance from 1986 to 2008, based on analysis of Landsat imagery.

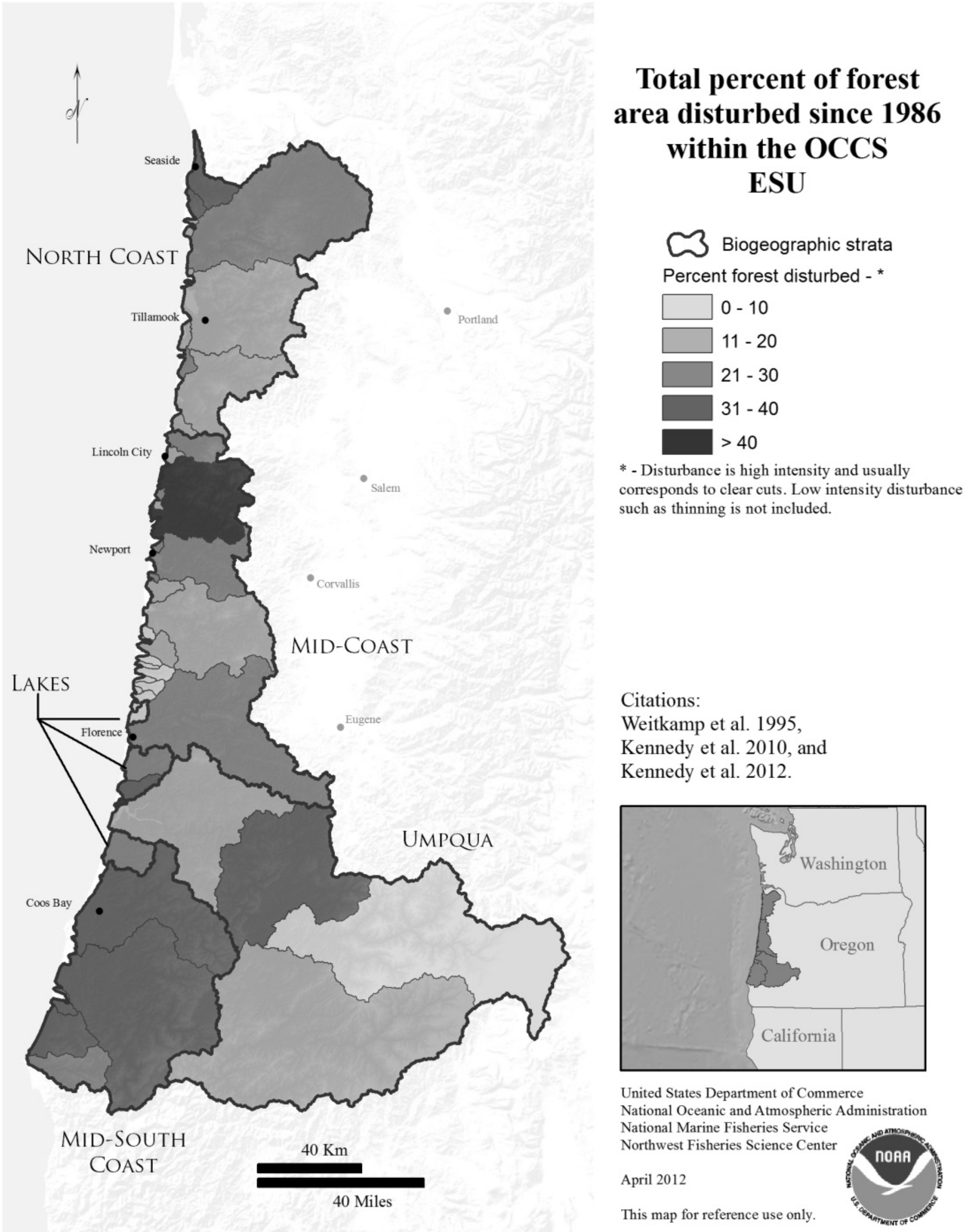

Figure B-2. Ranking of river basins and the Umpqua subbasins by cumulative percent vegetation disturbance from 1986 to 2008.

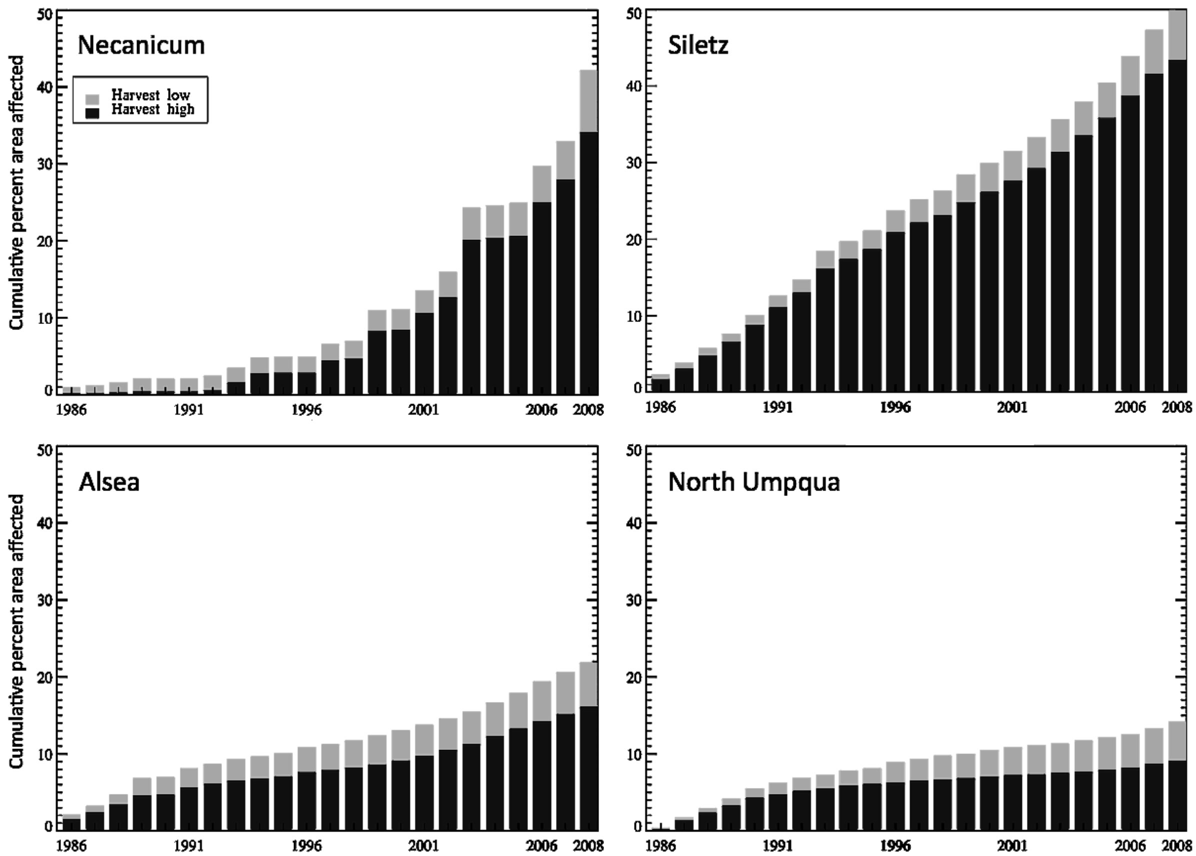

Figure B-3. Time series of cumulative area of vegetation disturbance for four river basins in the OCCS ESU for years 1986 to 2008. High disturbance (dark gray) is usually clear-cut logging, while low disturbance (light gray) is related to forest thinning. Data from Kennedy et al. 2010 and 2012.

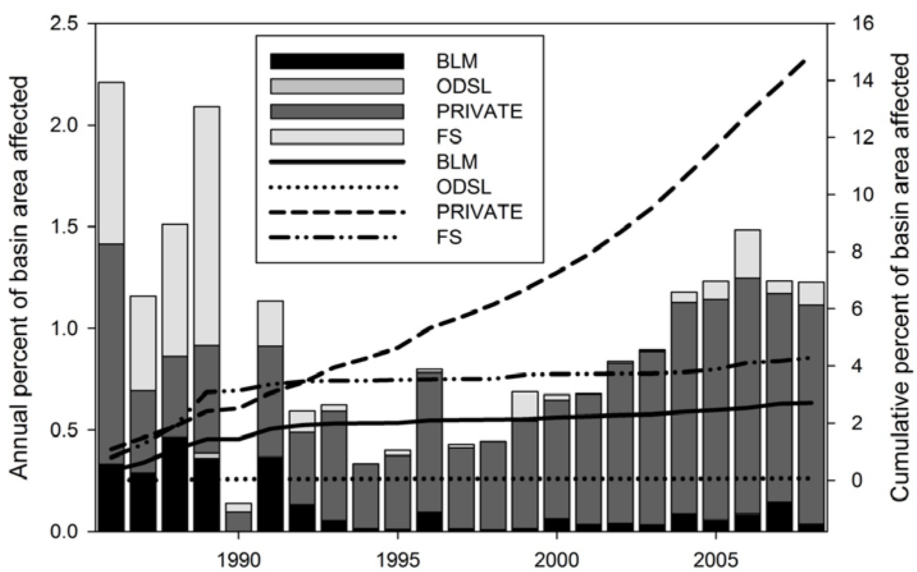

Figure B-4. Total area (hectares) of vegetation disturbance in the Alsea River area of the OCCS ESU by four land ownership categories from years 1986 to 2008. BLM = Bureau of Land Management timberlands, ODSL = Oregon state timberlands, Private = industrial and nonindustrial private timberlands, and FS = U.S. Forest Service. Data from Kennedy et al. 2010 and 2012.

Appendix C: Summary of Findings of the BRT-ODFW Habitat Trends Working Group

Introduction

From 1998 to the present, the Oregon Department of Fish and Wildlife (ODFW) has monitored wadeable streams to assess freshwater rearing habitat for the Oregon Coast Coho Salmon (OCCS) evolutionarily significant unit (ESU) during the summer low flow period (Anlauf et al. 2009). The goal of this program is to measure the status and trend of habitat conditions throughout the range of the ESU through variables related to the quality and quantity of aquatic habitat for coho salmon (*Oncorhynchus kisutch*): stream morphology, substrate composition, instream roughness, riparian structure, and winter rearing capacity (Moore 2008). In 2009 scientists from ODFW and scientists of the OCCS Biological Review Team (BRT) independently analyzed this data to ask the question: Has juvenile coho habitat changed during ODFW's monitoring program over the past 11 years? These analyses reached different conclusions, necessitating that the discrepancies be resolved.

To address differences in the BRT's analysis of habitat trends based on ODFW data and ODFW's original analysis (Anlauf et al. 2009), NWFSC and ODFW formed a joint Habitat Trends Working Group (HTWG) composed of scientists from each agency (Eric Ward, Chris Jordan, and Mike Ford, NWFSC; Kara Anlauf, Kim Jones, Jeff Rodgers, Julie Firman, and Kelly Moore, ODFW). Statisticians from the U.S. Environmental Protection Agency (John Van Sickle) and Oregon State University (Don Stevens and Lisa Madsen) also contributed to the HTWG analysis. There were several important differences to address: in the BRT and ODFW analysis, different data sets were analyzed, different trend models considered, and different estimation approaches were used. The goal of the HTWG was to reconcile these differences.

The first discrepancy between the BRT's analysis and the ODFW analysis (Anlauf et al. 2009) that caused conflicting results was that different subsets of the ODFW habitat monitoring program were used. The ODFW approach focused only on sites designated as coho spawning or rearing habitat (first through third order wadeable streams and below barriers, Anlauf et al. 2009). In contrast, the BRT's analysis included sites within and outside of the area recognized as spawning and rearing for coho salmon habitat. The HTWG agreed that a common data set should be used in the joint analysis, and that initially only spawning or rearing sites within the coho ESU be included. A further selection was made on which sites were included. The ODFW monitoring program surveys 25% of sites annually, 25% of sites every 3 years, 25% of sites every 9 years, and 25% of sites only once. The HTWG analysis excluded sites with only 1 visit, yielding 530 data points over the period 1998–2008. These data are distributed among 4 regions or strata within the OCCS ESU (North Coast, Mid-Coast, Mid-South, Umpqua), with 133 unique sites within those regions (Figure C-1).

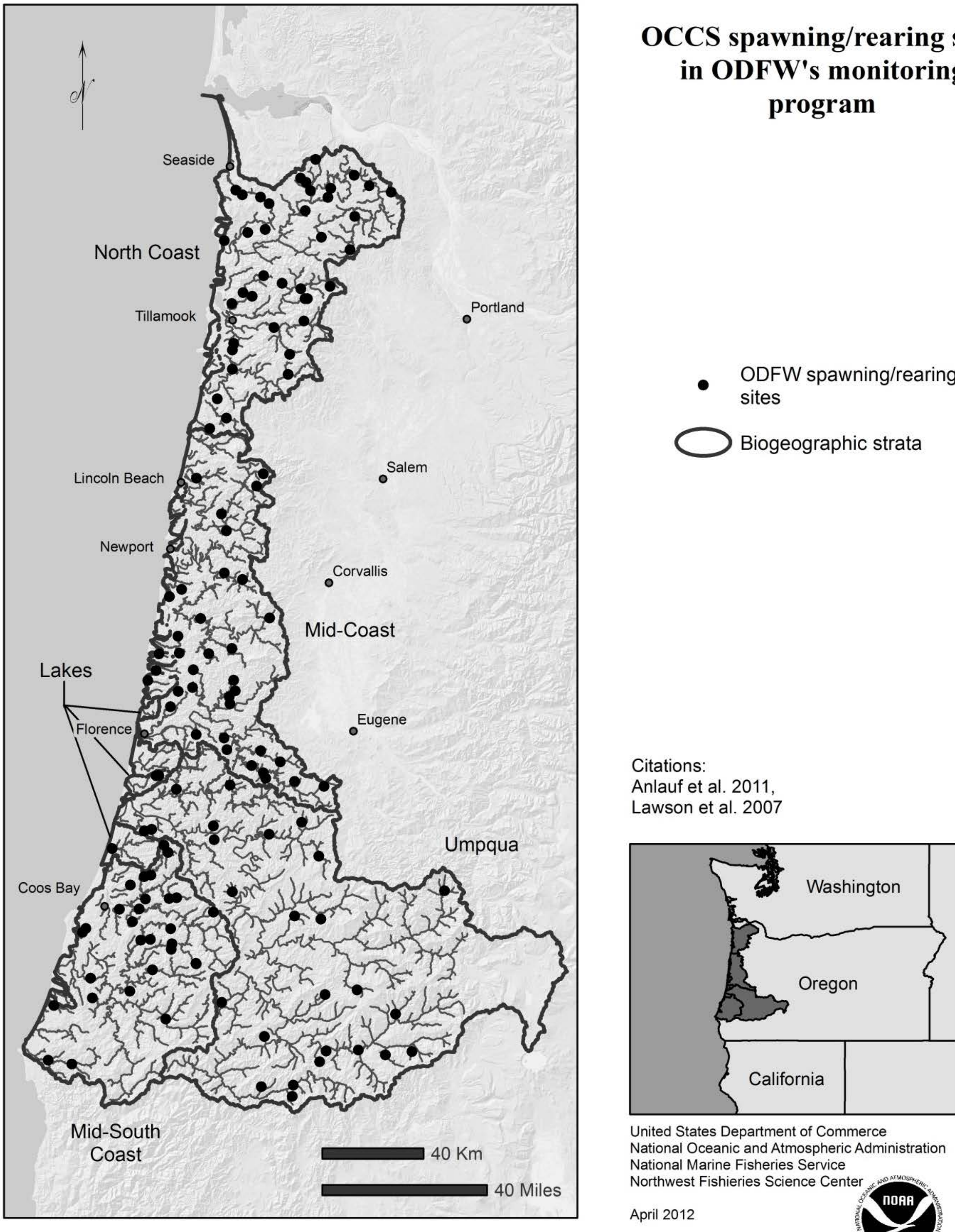

Figure C-1. Map of Oregon coho spawning and rearing sites.

A second potential discrepancy in the BRT and ODFW analysis was that the BRT adopted a Bayesian statistical approach, while ODFW had used maximum likelihood methods. We would expect the results to be identical, because when using normal linear models with noninformative priors, the Bayes posterior is approximately normal, centered on the maximum likelihood estimate (Gelman et al. 2006). After analyzing identical data sets with different estimation methods, the HTWG concluded that when the same trend model is applied to identical data sets, the point estimates (maximum likelihood estimates, posterior means) are identical using both the Bayesian and maximum likelihood approaches.

A third potential source of differences between the BRT and ODFW analysis was the choice of trend model used. In its analysis, ODFW advocated the inclusion of interaction terms consistent with the study design. The BRT's approach was based on model selection, allowing the model with the most data support to be used for trend analysis. The HTWG considered both of these approaches.

Methods

In the BRT's original analysis, three measures of habitat complexity were assessed for trends: winter parr capacity, summer parr capacity, and channel score (Aquatic and Riparian Effectiveness Monitoring Program [AREMP]). In addition to winter parr capacity, ODFW also examined trends in large woody debris (LWD) and fine organic sediment (Anlauf et al. 2009). The HTWG agreed that the three measures of complexity would be reanalyzed, in addition to the volume of LWD and fine organic sediment in riffles. These five variables were then transformed using the methods shown in Table C-1 to make each approximately normally distributed.

The linear trend models used by the BRT and ODFW are similar, with only subtle differences. The model used in the BRT's analysis can be described as

$$X_{hijk} = \alpha_h + \beta_h t + s_j + \varepsilon_{hijk} \quad (1)$$

where X_{hijk} is the estimated response for site *j*, region *h*, and year *i*. The parameters α_h and β_h represent region-specific fixed effects (intercept and slope, respectively). The intercept at each site is treated as a normally distributed random effect, $s_j \approx \text{Normal}(0, \sigma_s)$, and residual error is included as $\varepsilon_{hijk} \approx \text{Normal}(0, \sigma)$. Biologically, these parameters mean that after accounting for variability between sites within a region, there are regional specific trends and differences in mean values (Figure C-2), reflecting shared overall patterns within sites in a given region.

Table C-1. Individual habitat metrics used by the HTWG for the trend analysis, the HTWG codes, and the transformations used to achieve normality.

Habitat metric	**HTWG code**	**Transformation**
Winter parr (ODFW)	Winter parr	ln(x)
Summer parr (ODFW)	Summer parr	x^(1/4)
Channel score (USFS)	Channel score	x
% of riffle that is sand/silt/organics	RIFSNDOR	asin(sqrt(x/100))
Volume of LWD per 100 m	LWDVOL	ln(x+1)

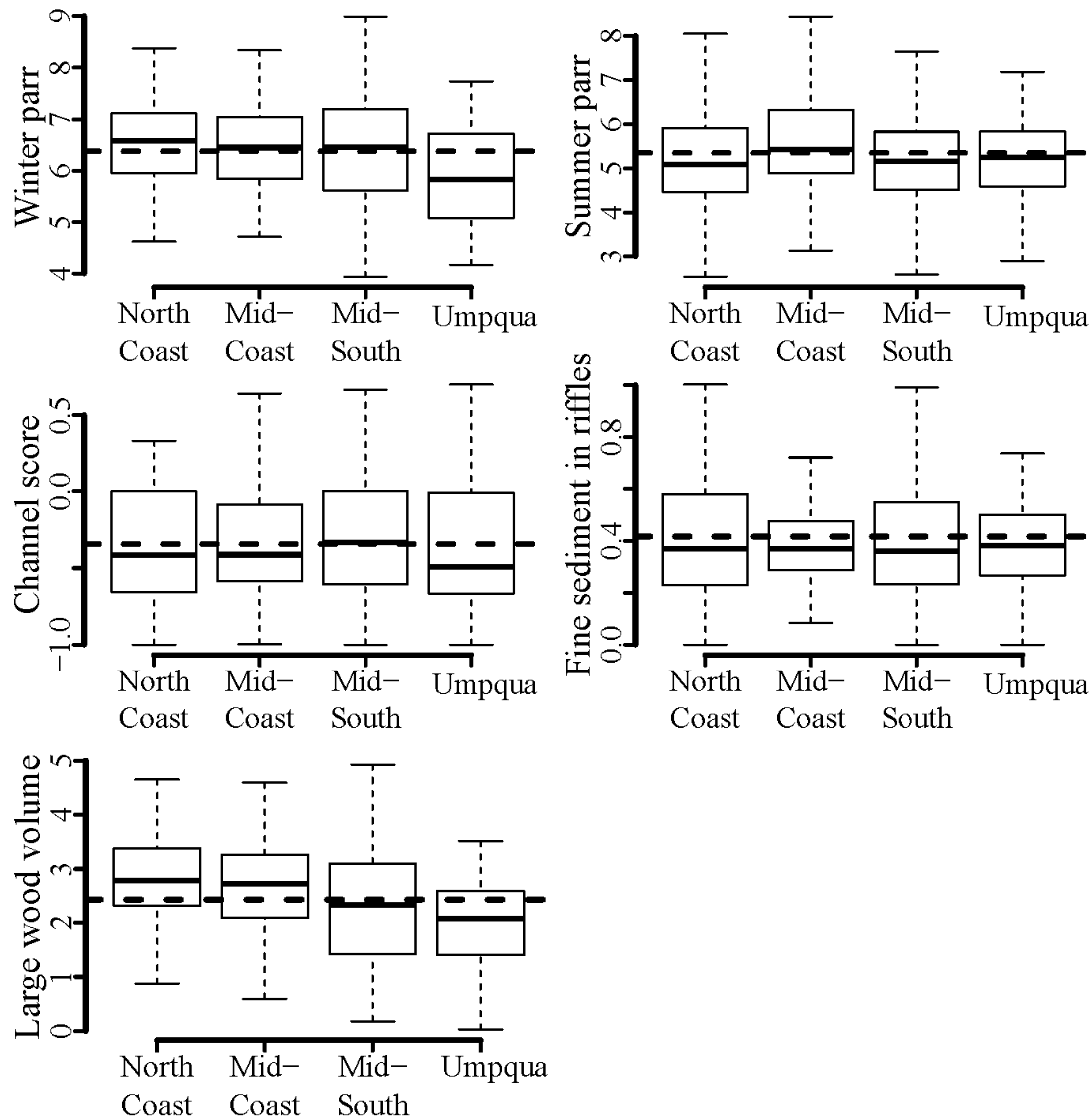

Figure C-2. Box plots of habitat complexity (Winter parr capacity, Summer parr capacity, Channel Score), percent fine sediment in riffles (RIFSNDOR), and volume of LWD by region for all sites (n = 530).

The linear trend model used by ODFW is identical to that used by the BRT, with the inclusion of two additional terms,

$$X_{hijk} = \alpha_h + \beta_h t + s_j + \gamma_i + s_j{*}\gamma_i + \varepsilon_{hijk} \qquad (2)$$

where γ_i is a year-specific random effect, and $s_j{*}\gamma_i$ represents a site-by-year interaction that allows each combination of site-year to be unique. Including year-specific random effects for all sites in all regions makes the assumption that all regions are similarly affected by large-scale environmental processes, regardless of whether the trends are shared (Figure C-3a) or region-specific (Figure C-3b). Including the year-specific interaction term is justified by the survey

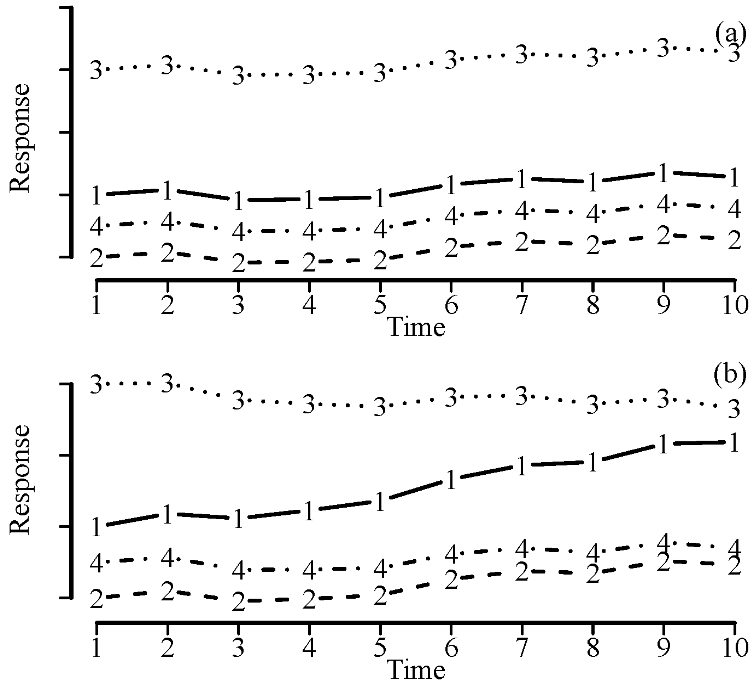

Figure C-3. Illustration of year random effects for four hypothetical populations with a shared trend (panel a) and different trends (panel b). The response variable is simulated from arbitrary values and has no scale.

design; by revisiting some sites multiple times in the same year, the survey hopes to separate process variation from observation or measurement error. If data from the revisits are not substantially different from the long-term trend (Figure C-4a), it may be difficult to partition the total variance into observation and process components. If there are unique differences between years at the same site (Figure C-4b), including this additional variance will be warranted (and supported by model selection criteria).

As a first analysis, the HTWG used maximum likelihood methods to conduct significance on each of the five transformed metrics. As a second step, the HTWG repeated the BRT's model selection exercise in a Bayesian statistical framework. Bayesian statistics offer advantages in the ability to communicate uncertainty as probabilities; one obvious disadvantage and source of criticism is the need to specify prior distributions on model parameters. Because the class of statistical models considered by the HTWG are normal linear models and well known noninformative priors exist (Gelman 2006), we were able to avoid introducing subjectivity into the trend analysis. For each metric, four candidate models were considered, reflecting a range of models between the models used in the original ODFW and BRT analyses. These models all included region-specific trends and site-specific intercepts; their differences were whether or not site-by-year interactions and random effects in years were included (Table C-2). The data support for each of the regression models was evaluated independently for each metric using the Deviance Information Criterion (DIC, Spiegelhalter et al. 2002).

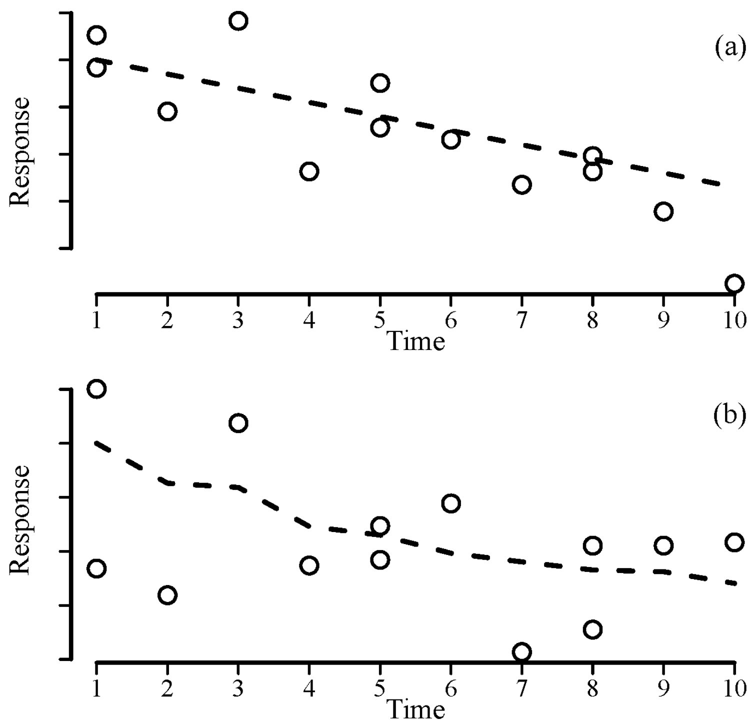

Figure C-4. Illustration of multiple site visits with no process variation (panel a) and process variation (panel b) representing environmental stochasticity. Each survey was simulated to have repeat visits in years 1, 5, and 8 (two surveys each). The response variable is simulated from arbitrary values and has no scale.

Table C-2. Differences between DIC scores between four competing models of changes in habitat metrics over time. In addition to the two models used in the ODFW and BRT reports, two intermediate models are considered. Delta DIC values represent the difference between each model and the best model (zeroes).

	Model 1 (BRT)	**Model 2**	**Model 3**	**Model 4 (ODFW)**
Site-specific intercepts	**x**	**x**	**x**	**x**
Regional trends	**x**	**x**	**x**	**x**
Year random effect		**x**		**x**
Site: year interaction			**x**	**x**
Habitat metric	**ΔDIC**	**ΔDIC**	**ΔDIC**	**ΔDIC**
Winter parr	**0.0**	7.3	3729.6	4460.1
Summer parr	**0.0**	2.4	3576.0	3266.9
Channel score	**0.0**	8.7	2346.4	3891.0
RIFSNDOR	**0.0**	7.1	4705.8	4714.0
LWDVOL	12.5	**0.0**	5777.1	4113.9

Results

For each of the three measures of complexity and for fine sediment in riffles, the HTWG's model selection exercise found little data support for including either the year random effect term or the site-by-year interaction term (Table C-2). The exception was the model selection comparison for LWD volume; this metric suggested that year-specific random effects be included. This difference is important, because it suggests that unlike the complexity measures, there are large scale patterns across regions in the OCCS ESU that affect how wood volume is accumulated (or removed). To examine how the choice of model may affect trend estimates, the HTWG compared maximum likelihood estimates between the ODFW and BRT models. Both the trend estimates themselves and the results of significance testing are identical (Table C-3 and Table C-4), with two statistically significant results found (percent fine sediment organics in riffles in the Mid-South and large wood volume in the North Coast).

The advantage of examining the distributions of trend in a Bayesian setting is that we can quantify the proportion of the trend estimate that is negative or positive (e.g., Pr [X < 0] = 0.03%, Figure C-5). Using winter parr capacity as an example, while none of the trend estimates meet the requirements to be considered "statistically significant," there is strong support for negative trends in the two southern regions (95% in the Mid-South and 79% in the Umpqua). Although these negative annual rates of decline are small, they translate into reductions of winter parr capacity of 22% and 12%, respectively, over the 10-year time series.

Trend estimates for the other measures of complexity and habitat metrics are largely mixed. There is evidence of increasing trends in summer parr and channel score (AREMP) in three of the four regions (Figure C-5). For the percent of fine sediment in riffles, there appear to be declines in the North and Mid-Coast, a positive trend in the Mid-South, and little change in the Umpqua. Large wood volume appears to be declining in the North Coast and Umpqua, while increasing in the Mid-Coast and Mid-South regions.

The approaches taken by the BRT and ODFW are similar in that they both center on the use of the same likelihood function that relates parameters to data. The HTWG also considered exploring other approaches that are not likelihood based. One of these alternative approaches considered individual regressions on data from each site to graphically estimate the distribution of the trends for each metric. It is difficult to compare trends across sites, because some sites have few visits (n = 2–3) and others have been visited in each year (n = 10); sites with many visits will have much more precise estimates of trends. However, it is possible to overlay these trend estimates from individual sites on the posterior distributions of the regional trends from the Bayesian model to examine agreement between the site-level trends and regional trends.

Table C-3. Maximum likelihood estimates of trend parameters by region for two measures of habitat complexity (winter parr, summer parr) and two habitat metrics (% fine sand or organic material in riffles and the volume of LWD). Estimates are from the trend models used by ODFW and BRT (in parentheses); estimates found to be statistically significant ($P < 0.05$) are in boldface.

	North Coast	Mid-Coast	Mid-South	Umpqua
Winter parr	0.000 (−0.002)	−0.005 (−0.004)	−0.026 (−0.025)	−0.016 (−0.013)
Summer parr	0.014 (0.009)	0.027 (0.027)	0.031 (0.030)	−0.022 (−0.027)
RIFSNDOR	−0.009 (−0.010)	−0.003 (−0.003)	**0.027 (0.026)**	0.003 (0.002)
LWDVOL	**−0.063 (−0.062)**	0.015 (0.016)	0.019 (0.020)	−0.028 (−0.025)

Table C-4. Graphical representation of the maximum likelihood analysis and Bayesian analysis trend results. Arrow style indicates strength of trend: black vertical arrow represents greater than 90% Bayesian probability or significance ($P < 0.05$) of trend; light gray vertical arrow represents greater than 65% Bayesian probability of trend; horizontal gray arrow represents lower (<65%) Bayesian probability of trend or no significant trend detected (maximum likelihood). Upward pointing arrow indicates a positive trend and downward pointing arrow indicates a negative trend.

	Stratum											
	North Coast			Mid-Coast			Mid-South			Umpqua River		
	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities	Maximum likelihood (BRT)	Maximum likelihood (ODFW)	Bayesian probabilities
Winter parr	↔	↔	↔	↔	↔	↔	↔	↔	⬇	↔	↔	⇩
Summer parr	↔	↔	⇧	↔	↔	⬆	↔	↔	⬆	↔	↔	⇩
Channel score	↔	↔	⇧	↔	↔	⇧	⬇	⬇	⬇	↔	↔	↔
Wood volume	⬇	⬇	⬇	↔	↔	⇧	↔	↔	⬆	↔	↔	⬇
% fine sediment in riffles	↔	↔	⬇	↔	↔	⇩	⬆	⬆	⬆	↔	↔	↔

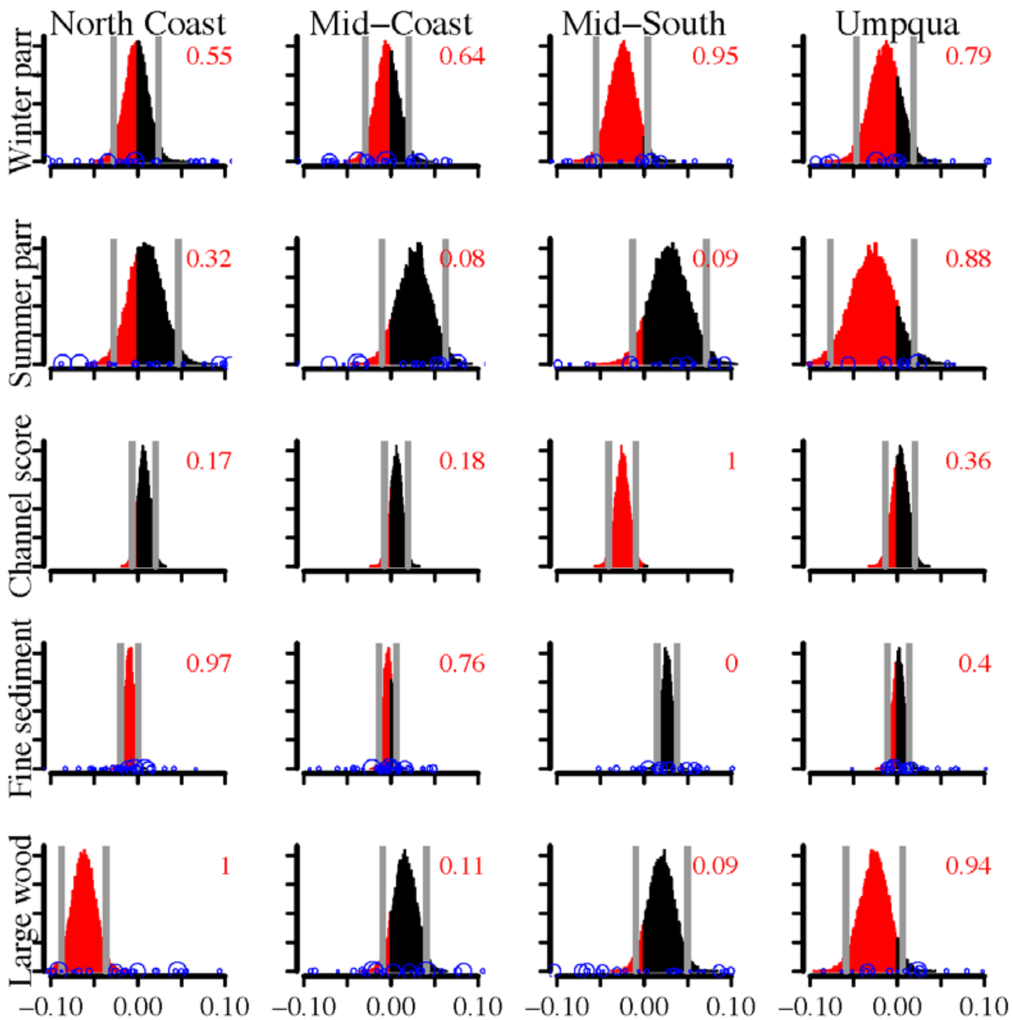

Figure C-5. Posterior distributions of trend estimates for three measures of habitat complexity (winter parr, summer parr, channel score) and two habitat metrics (fine sediment, i.e., percent fine sand and sediment in riffles, and large wood, i.e., volume of LWD). Each probability distribution is colored to represent the region less than zero (red) and greater than zero (black). Ninety-five percent credible intervals are shown with vertical gray lines and trend point estimates at the level of individual sites are shown in blue circles (whose sizes correspond to the number of data points). The proportion of each posterior trend estimate less than zero is quantified in red in the upper left corner of each panel.

Ideally, the posterior trend estimate will be driven by the sites that have the most data, with outliers representing sites with few site visits. For many of the metrics, there is agreement in the posterior trend for the region and the individual site trends within the region (Figure C-5). For some combinations of regions and metrics, there appears to be some discrepancy (e.g., summer and winter parr in the North Coast, LWD pieces in all regions). This discrepancy is not a failure of the estimation or the modeling approach used, but suggests that the HTWG may want to consider further hypotheses in future analyses. For instance, rather than assuming all sites within a region share the same trend, the HTWG might consider alternative hypotheses based on trends being shared among populations.

The HTWG was interested in comparing trends in habitat metrics inside and outside coho salmon spawning/rearing habitat. The BRT's analysis focused on all sites, and sites beyond spawning/rearing habitat may be different (they are generally located higher in watersheds). The HTWG repeated the trend analysis on the two raw metrics (percent sand and organic sediment in riffles, large wood volume) for sites not considered to be coho spawning or rearing habitat (n = 887). The biggest difference between sites inside and outside spawning habitat is that sites not designated as spawning/rearing habitat have negative declines in large wood volume for all regions (Figure C-6).

Summary

The HTWG's analysis expanded on the original BRT analysis, and there are important differences between the trends originally estimated by the BRT and those estimated by the HTWG. The primary source of these differences was the data set used for the analysis. After restricting the data to sites designated as coho spawning/rearing habitat and correcting several errors in the database, the HTWG is confident that results are insensitive to the choice of model or estimation approach used. In addition to changing some of the estimates reported by the BRT, the results from the HTWG also change the estimates reported in ODFW's analysis (Anlauf et al. 2009). An example of a result that changed is that rather than the AREMP channel score declining in three of the four regions, there is now evidence of increasing trends in three of the

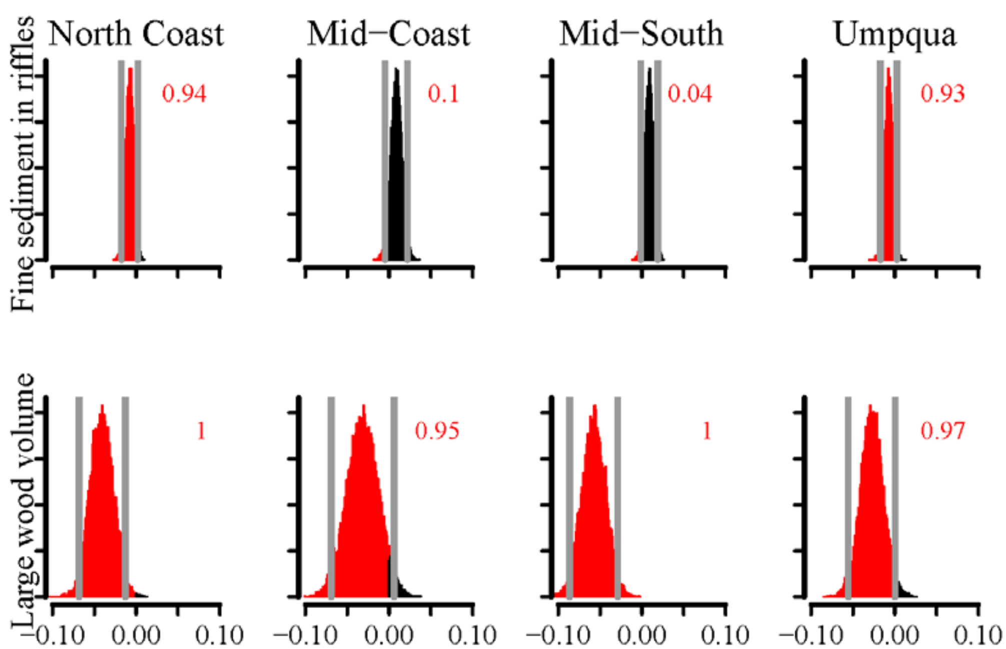

Figure C-6. Posterior distributions of trend estimates for two habitat metrics (percent fine sand or organic material in riffles and volume of LWD). Estimates are presented for each region. The data set analyzed is all sites that are not designated as coho spawning or rearing habitat. Each probability distribution is colored to represent the region less than zero (red) and greater than zero (black). Ninety-five percent credible intervals around each estimate are shown with vertical light gray lines. The proportion of each posterior trend estimate less than zero is quantified in red in the upper right corner of each panel.

four regions (Table C-5). Some of the original results initially reported by the BRT remain unchanged, however. There appears to be little or no change in winter parr capacity in the two northern regions (North Coast, Mid-Coast), and negative declines in the two southern regions (Mid-South = –2.5% and Umpqua = –1.3%). Other measures of habitat complexity trend positive or negative, depending on region, and there is no strong positive increase in complexity for all regions.

Table C-5. Bayesian trend estimates (means) applying the BRT's trend model to the HTWG's data set. The original trend estimates and proportion of the posterior less than zero (Pr < 0) are given in parentheses.

	Channel score		**Summer parr**		**Winter juveniles**	
Stratum	**Slope**	**Pr < 0**	**Slope**	**Pr < 0**	**Slope**	**Pr < 0**
North Coast	0.006 (–0.02)	0.17 (1.00)	0.009 (–0.02)	0.32 (0.80)	0.0 (–0.02)	0.55 (0.90)
Mid-Coast	0.006 (–0.01)	0.18 (0.88)	0.027 (0.02)	0.08 (0.24)	0.00 (0.00)	0.64 (0.48)
Mid-South Coast	–0.025 (–0.02)	1.00 (1.00)	0.03 (–0.08)	0.09 (1.00)	–0.025 (–0.03)	0.95 (0.99)
Umpqua River	0.003 (0.00)	0.36 (0.48)	–0.028 (–0.08)	0.88 (1.00)	–0.013 (–0.03)	0.79 (0.99)

Appendix D: Peer Reviews and Public Comments with the BRT's Responses

This appendix has four subsections: Comments of Eight Peer Reviewers, Comments of the Oregon Department of Fish and Wildlife, Comments of the MidCoast Watersheds Council Coordinator, and Comments of Douglas County. Our Oregon Coast Coho Salmon Biological Review Team (OCCS BRT) responses to these comments are contained in brackets and placed within the text of the original comments. Some of the peer reviewer comments were edited for clarity and some style and format changes were made to those and the public comments, including inserting additional paragraph indents, to accommodate the insertion of responses. For the original text of the three public commentators, due to the length and public availability of their comments, we refer the reader to http://www.regulations.gov/#!docketDetail;dct=FR%252BPR%252BN%252BO%252BSR%252BPS;rpp=25;po=0;D=NOAA-NMFS-2010-0112.

Comments of Eight Peer Reviewers

Reviewer 1

Here are some comments on the status of Oregon coastal coho. I read most of the report and found it comprehensive, well written and up to date on the literature. Nice job by the authors.

I thought that the Climate Change subsection was excellent, covering potential changes in both salmon physiology and change in habitats.

I did not see a section on specific recommendations for management and I think that this would be logical and desirable. For example, this review could state possible ways that climate change could be ameliorated: protection of cold water refugia, enhancement of wetlands and uplands, better riparian protection of "non-fish bearing" streams, beaver reintroduction, reduction of irrigation and promoting instream water rights during critical periods, removing catch limits on invasive species such as bass, etc. Some of these are covered in the text, but not emphasized as action items for managers or researchers.

[**Response:** The goal of this document is documentation of the OCCS BRT deliberations. It is not appropriate, therefore, to address action items for managers or researchers here. That is the function of the recovery planning process and should be included in a recovery plan.]

The subsection on predation is rather dated. Reference could be made to impacts of Caspian terns in the Columbia (Roby and Collis). What about predation in estuaries (Steve Johnson, Alsea), and cormorants in the Nehalem Bay? Paragraphs on Humboldt squid, however, were excellent.

[**Response:** We updated our discussion of predation with a look at new (Johnson et al. 2010) and somewhat older (Schreck et al. 2002) literature as a recent population assessment of double-crested cormorants (*Phalacrocorax auritis*) in the evolutionarily significant unit (ESU) and other sources. We did conclude that the significant increases in avian predation appear to be restricted to the Columbia River System and does not affect the OCCS ESU as much as it does the Columbia River ESUs because of where the birds (Caspian terns [*Hydroprogne caspia*] and double-crested cormorants) are nesting and foraging during coho salmon (*Oncorhynchus kisutch*) smolt outmigration.

Reviewer 2

Please find my review of the suggested section of the Oregon Coast Coho status review (pages 59–85). Overall very well written, nice analysis and interpretations—very complementary to work by ODFW and other agencies. My comments are referenced by the draft's page number, paragraph, and sentence.

Comments on section pertaining to "The Present, or Threatened Destruction, Modification, or Curtailment of its Range" (draft pages 59–85).

Conclusions quoted regarding present impacts of hydropower should be expanded to consider future development as well. I know there are possible plans for hydroelectric dams to be placed in some coastal rivers, such as the Siletz River near the former town site of Valsetz. Also the development of small hydro may come into play in the future as the region develops alternative energy sources. This is becoming an issue in other parts of western North America (e.g., British Columbia) (page 59, paragraph 1).

[**Response:** Mention of this possibility has been included in the revised document.]

Should also mention the U.S. Forest Service database on barriers and fish passage (contact Bruce Hansen: bhansen@fs.fed.us.) (page 59, paragraph 3.)

[**Response:** The ODFW 2009 fish barrier database includes the U.S. Forest Service data.]

In the Umpqua and other basins where nonnative centrarchid (bass, bluegill, and relatives) invasions are underway (e.g., Coquille and likely other systems), increases in temperature will shift the balance toward favoring these species at the expense of native salmonids. "Native" invasions are also likely to occur and be important, as pointed out by Reeves et al. and other work on interactions between minnows and salmonids in relation to water temperature. These points may be covered elsewhere in the document. Threats are also possible from invaders that have yet to reach the northwest, such as northern pike (*Esox lucius*) or Asian carp (*Hypophthalmichthys harmandi*). I hope species like this never reach habitats with Oregon Coast coho, but control of these invasions has proved less than complete in other areas. It is worth worrying about (page 61, paragraph 3).

[**Response:** In response to these comments, the BRT discussed this issue more fully; expanded discussion and literature citation is included in the revised document in the Ecosystem impacts of nonindigenous species subsection, Nonindigenous plant species subsection, and Nonindigenous fish subsection.]

Comment on “stream complexity:” This seems to deal mostly with local complexity and I assume larger-scale or “landscape” complexity is dealt with elsewhere (page 62).

[**Response:** Larger-scale or landscape complexity was treated in somewhat more depth in the revised BRT document; however, for this assessment, local complexity was the focus of our analyses.]

Useful to clarify that stream cleaning was a historical practice no longer endorsed by ODFW (page 63).

[**Response:** This has been clarified in the document.]

I think you mean to say inter-annual, not intra-annual (page 64, paragraph 5, line 5).

[**Response:** This was revised.]

Overall in this discussion I am not sure if we are seeing cycles or trends in beaver activity. The table seems to show three cases of declines, two of increases, and one of no change (zero in both years). Not sure if this is statistically or biologically “significant.” I don’t think there is good evidence for a trend of any kind.

[**Response:** We added the following statement in the Beaver in OCCS habitat subsection: “Due to the limited data set we cannot conclude that there is an overall trend and would recommend a more extensive monitoring effort be pursued to identify short-term and long-term trends throughout the OCCS ESU.”]

Also clarify that the work by Pollock et al. was done in Washington; useful for general comparisons, but specific numbers may not apply to Oregon coast.

[**Response:** We included a paragraph that addresses this comment. “The Pollock et al. (2004) study focused, for the most part, on sites in the Puget Sound region; however, the BRT noted that the areas where beaver pond density is highest typically have the same physical characteristics regardless of the ecological region—lower gradient (less than 2%), unconfined valley bottoms, in smaller watersheds (drainage areas typically less than 10 km^2). Smaller, lowland, rain-dominated Puget Sound watersheds have the same basic physical and hydrological characteristics as the smaller Oregon coast watersheds, thus the relationships with respect to beaver pond densities in Puget Sound should also hold true for the Oregon coast.”]

I think the simple conclusion to this discussion on beaver is sufficient and I would clarify by saying “potential” declines. Probably the largest threat to beaver is how they are currently managed. The current situation does not lend itself well to management of beaver to benefit fish. The state agencies have organized a working group to address this issue, however (page 66).

[**Response:** We have captured this comment in revised text in the Beaver in OCCS habitat subsection.]

Can you term this section "Human Disturbance," as natural disturbance is usually viewed as a positive thing, so long as human activities have not rendered species excessively vulnerable to them (page 67)?

[**Response:** We have changed the subsection title.]

A more complete table caption would be helpful here (Table 15).

[**Response:** We have revised the caption to clarify the table.]

I think the conclusion here about complexity (rate of continued disturbance outpacing restoration) is likely correct, but we don't know for sure. Local "active" restoration activities are likely dwarfed by the larger human footprint on the landscape, but passive efforts to restore landscape condition (e.g., improved forest harvest practices) will likely take decades to yield detectable positive trends. Might be worth clarifying the issue here because passive restoration is much likely to have longer term and more widespread benefits in the future (page 73).

[**Response:** A short clarification of this issue is now included in the Stream habitat complexity summary subsection.]

In the section on land conversion, I noted there was little attention to increases in effects associated with urban or rural/residential development, such as septic drainage (nutrients), fertilizers, pharmaceutical and personal care products (PPCPs), or other contaminants (e.g., herbicides and pesticides). The potential impacts of these factors are more obvious and more studied within urban areas, but they are a growing and difficult to detect threat outside of urban zones, as land is increasingly developed for homes, hobby farms, and other similar uses. Clearly this is an area of great uncertainty—even in well studied urban areas—but it should be mentioned here as a potentially serious, yet virtually unknown threat. The State of Oregon Independent Multidisciplinary Science Team is producing a report that summarizes many of these issues. It might be advisable to contact Dr. Carl Schreck at Oregon State University for more information related to this (page 75).

[**Response:** The IMST (2010) report has just been released and our revision reflects its findings in the Land management—forest and agriculture conversion subsection.]

Another place to possibly emphasize factors identified directly above (page 84).

[**Response:** The Land management—water quality degradation subsection discussion focused on temperature issues in the ESU. A discussion that briefly mentions these issues is found in the Land management—forest and agriculture conversion subsection.]

I work a lot on impacts of temperature on salmonids and was hoping to see a bit more than a paragraph on the issue. It is well studied, but the short amount of text devoted to this issue versus hydrogeomorphic factors seems out of balance. Perhaps a sentence or two emphasizing the primacy of temperature as a component of habitat and threat to salmon. I believe temperature is the number one source of water quality impairment in Oregon (page 85).

[**Response:** New text has been added in the Land management—water quality degradation, Effects of climate change on the OCCS ESU, Water availability, and Land management—forest and agricultural conversion subsections. Additional emphasis of temperature is found in Wainwright and Weitkamp (in prep).]

Overall I found the section I reviewed and supporting material to represent a very useful addition to existing information that has been used to evaluate status and threats for Oregon Coast coho. Once again the BRT has produced an excellent product and one that would be the envy of just about any other team working on threatened fish, thanks to the team's work and the wealth of useful data collected by federal agencies and the state of Oregon.

A few clarifications and additions as suggested above would improve the document in my view and do not require major revisions. Thanks very much for the opportunity to review this.

Reviewer 3

I just finished with the BRT report and except for several mistakes in terms of incomplete sentences or repeated paragraphs, very short and not informative table headings, a few typos and other minor things, I think it is in general a good summary of the current situation for coastal coho salmon in Oregon.

I do think the emphasis given to the importance of estuarine habitat is moderate and adequate, given the information available in the literature. Based on my own experience working in creeks that drain into Coos Bay, I think that the early estuarine rearing life history strategy (nomadic) is rather common. What we do not know yet is whether it is genetically based (expressed by a particular genotype) or just one possible of various expressions of a flexible phenotype that is condition dependent (i.e., it becomes common under certain environmental conditions or at certain rearing densities). Regardless of the mechanisms behind it, this estuarine-based life history may serve as a bet hedging strategy that supports the population in years when stream conditions are poor.

In the fall of 2009, we detected nine coho salmon jacks PIT tagged as subyearling reservoir residents in spring 2008, suggesting that coho nomads do survive their early life history experience and contribute to the spawning population. By affecting coho salmon nomads to a greater extent than other life history types, estuary loss may reduce the potential productivity boost and resilience component that this life history may contribute to its population of origin and any metapopulation it may be part of. I have to check Art Bass' thesis, but he should have information in it about those nine jacks.

[**Response:** Text has been added in this section to capture these comments and to incorporate information from Bass (2010) into the text.]

We are going to look into if, and to what degree, "nomad" fish return as spawning adults in the next phase of our study. I have to talk Jessica Miller into reading some otoliths we have collected, but independently of this, this fall we could be getting the first PIT tagged "nomads" returning to spawn.

[**Response:** We have reported the most recent information as of March 2011 in the BRT document.]

In coastal streams with well-developed estuaries, the early estuarine rearing strategy coho display may be more important for the long-term viability of those populations than the opinion you shared with me (see ODFW comments below) reflects. If most or all of the thousands of coho fry that move into estuarine habitats early in their lives died before spawning, this would be a poorly adaptive behavior that natural selection forces would have eliminated from coastal populations (or at least it would be a very rare strategy).

[**BRT note:** In a written comment on the document, reviewer 3 requested more discussion of storm water and floodplain effects. **Response:** We added discussion of these issues in the Land management—forest and agriculture conversion subsection.]

Reviewer 4

Below please find my review of Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon, by Stout et al. As requested, I focused my review on the assessment of 1) the scientific validity of the status review, including any assumptions, methods, results, and conclusions; 2) quality of data used; 3) appropriateness of the analyses; 4) validity of results and conclusions; and 5) appropriateness of the scope of the assessment.

Scientific validity of the status review, including any assumptions, methods, results, and conclusions: The authors have compiled and examined a tremendous amount of information to build on the three previous assessments. The process has been ongoing for more than a decade and many scientists with extensive experience and knowledge contributed to the report. The BRT concluded that, when future conditions are considered, the ESU is at moderate risk of extinction (54% agreement on this risk classification). The authors provide a synopsis of previous assessments, then present new information used in the current assessment. Overall, the document synthesizes an extensive amount of information in a succinct manner and incorporates future threats into the assessment. In many cases, it is difficult to assess the document's scientific validity per se because much of the analyses presented are based on data or conclusions from previous studies or documents without sufficient information to assess the data quality or previous conclusions. I am not suggesting that complete syntheses of previous efforts be included in the current assessment. However, it is challenging to thoroughly assess the current conclusions independently, as they rely on much previous work.

The primary methods for the current assessment are the decision support system (DSS) based on previously developed criteria and the development of a risk matrix. It appears that the metrics and data sources for the criteria are presented in detail in Wainwright et al. (2008), which I only briefly examined. Overall, as presented, it is difficult to sort out which data sets were included or considered in each of the two analyses. It would be useful and informative to have a master table or appendix that clearly listed the metrics and associated data sets that were incorporated into the DSS and the criteria to which they were applied.

[**Response:** This has been included in Appendix A.]

It is also not clear how the “truth values” for the recovery criteria were determined. Were these based on the opinions of the BRT or some quantitative assessment? Although much of this information may be presented in Wainwright et al. (2008), it seems that inclusion of adequate information to understand the approach of the current DSS is appropriate.

[**Response:** More information on this is included in the TRT Biological Recovery Criteria Analysis subsection, but because of the length of the truth value discussion, we refer readers to Wainwright et al. (2008.)]

Table 6 and Table 7 provide great synthesis of the major conclusions. Similarly, a relatively minor amount of additional information presented as a summary table may help this important analysis stand-alone in this document more effectively.

[**Response:** This table is included in Appendix A.]

An example is the passage barrier data set. From the text, it appears that there are no data for private lands, although this is not clearly noted in Figure 14.

[**Response:** The caption has been revised to reflect the absence of private lands from the data set.]

How are these data used in the DSS? It is indicated that there are low to moderate changes compared to historical template for the passage criteria, although it is not clear what this means or how it was determined.

[**Response:** The historical template was determined by an expert panel made up of TRT members and discussed at length in Wainwright et al. (2008.) It is not included here due to length.]

Additionally, was the recent disturbance analysis included in the DSS or only considered in the ESU risk matrix?

[**Response:** The disturbance analysis was not included in the DSS, as it was not one of the recovery criteria that was developed by the TRT and discussed in Wainwright et al. (2008.) It was individually taken into account by each BRT scientist in voting with the Risk Assessment Matrix and the FEMAT voting sheet.]

The second primary synthesis is the ESU risk matrix, which is based on professional opinion. In this matrix (Table 22), the analyses were separated into status with and status without consideration of future trends, so that two matrices were developed. There was another risk matrix (Table A-1), but it is not clear how those categories were scored and if or how they were incorporated into the overall ESU risk matrix (i.e., Table 22).

[**Response:** The second risk matrix found in the appendix was actually mislabeled in the BRT document. That matrix was utilized as a memory aid for the BRT’s discussion of threats so the BRT scientists could keep notes on the many types and magnitude of threats to OCCS at the ESU, biogeographic stratum, or population level.]

If it was presented, it could be more explicit, as I could not locate a specific explanation when I looked back through the document. Also, was there consideration of incorporating the results from the DSS into the ESU risk matrix?

[**Response:** The results from the DSS were considered with other information discussed by the members of the BRT. The BRT then used this in voting using the ESU risk matrix.]

As many of the DSS criteria could be grouped into the five categories used for the risk matrix, it appears that the DSS results could be averaged into the final assessment, perhaps counting as one or more "professional" opinions. It seems that an objective inclusion of the data-based DSS assessment combined with professional opinion may reduce the spread of opinion and would strengthen the overall conclusions.

[**Response:** Indeed, it would seem logical to include the DSS in the risk matrix which is, after all, a kind of DSS itself. However, the DSS used by the BRT was created by the TRT for a somewhat narrower purpose. The risk matrix as implemented by the BRT is a standard approach for evaluating the status of marine species in the Pacific Northwest. As we learned in the TRT, building a DSS for this kind of evaluation is difficult and time consuming. While taking the approach suggested by the reviewer may have "strengthened the overall conclusions," it was not feasible, given the time available. Instead, we essentially used nested DSSs, with the TRT's DSS informing the expert opinion input to the risk matrix.]

I also wondered if there was any discussion of incorporating "future threats" into a DSS framework to provide a more quantitative assessment of future risk. The challenges associated with quantifying future threats associated with climate change were discussed and a rationale was presented for a qualitative approach, but given the range of certainty associated with various future actions, it seems that some type of quantitative assessment could be developed and incorporated.

[**Response:** Incorporating future threats into a DSS framework would be an interesting and potentially valuable approach, but outside the scope and ability of the BRT within court-mandated time constraints.]

Quality of data used: As noted, it is not really possible to assess the quality of the data incorporated in the assessment itself, as very few details on data acquisition and analytical methods are included here. The status report obviously cannot be a summary of all data collection methods ever used to compile information on coastal coho. However, as noted above, a table or appendix that clearly lists the data sets included in the status assessment, along with their sources, the period of data collection, and which analyses they were used for, would be very useful. Given the "truth values" associated with the DSS, it appears that there are opinions or decisions rendered regarding the quality of the data sets, which could be included as well. With such an approach, it would be much easier for reviewers or other researcher/managers to assess the data availability and, potentially, quality of particular data sets.

[**Response:** The DSS was thoroughly documented and justified in Wainwright et al. (2008). A table summarizing data sets and sources is now provided in Appendix A.]

Several of the comments below address [items] 3 through 5 listed above.

The scope of the document appears appropriate. I think the attempt to include some assessment of the extensive restoration activities throughout the region was a valuable and important component. The future threats assessment is important, although challenging, and synthesizes much of the available information. Given that the consideration of future threats effectively changed the conclusion from not at risk (low risk) to moderate risk, their inclusion is clearly an important aspect of the current assessment. Along those lines, as noted above, development of a more quantitative process to incorporate future threats into the methodology could prove valuable.

[**Response:** We agree that a more quantitative process for incorporating future threats would be desirable. Future threats must be evaluated in terms of effect on viability of the species. Likely effect depends on the likelihood of occurrence of the future conditions and the expected magnitude or strength of the effect. This is a highly subjective exercise that would require an advance in the state of the art of applying DSSs to conservation biology and resource viability assessment.]

Although possibly outside of the scope of this assessment, a clear identification of key data gaps would be an important outcome. Given that the adequacy of available information is a component of the risk assessment, future data collection efforts would be aided by a clear articulation of key gaps in knowledge that the BRT identified. Again, one can infer some of these by reading the text, but a clear summary of the combined opinion of the BRT as to which data gaps were the most relevant would be useful.

[**Response:** In response to this request, the BRT identified some of the most important data gaps such as beaver populations and fish passage and discussed them in the appropriate text.]

A more general comment related to organization: I had some trouble with the organization of the information. The initial section summarizes previous assessments, which is clear for the most part but would also be enhanced by a table with a timeline and relevant synthesis documents identified. The document then transitions from previous assessments into the current work. The New Contributions section includes basic information regarding the current approach. The populations are identified in a table but the biogeographical strata are not. It would be useful to include the recognized strata.

[**Response:** We have added additional text to enhance the clarity of the document and included the table of the timeline and a map of the biogeographic strata for a more informative discussion.]

This section is then followed by public comment and a qualitative summary of a symposium; it is not clear if or how this information was used in the current process. Perhaps these comments and the symposium summary could be included as an appendix or their purpose more clearly articulated in the document.

[**Response:** The public comments and information presented in the symposium were considered in the BRT's deliberations and as such are germane to presentation of information in the BRT document. Additional text was added to clarify the use of these two sources of information.]

Then the New Data and Updated Analyses section presents the information used in the current assessment. Again, it was not always clear which categories where considered quantitatively as data incorporated into the DSS and which were qualitative assessments considered by the BRT for the Risk Matrix. I was a bit confused by some of the redundancies between the main body of the text and the appendices. For example, much of the habitat complexity discussion (≈p. 70–72) is repeated in Appendix E. Typically, I think of an appendix as supplementary data that are referenced at the appropriate location in the text for those readers seeking additional detail (similar to Appendix F).

[**Response:** For the Draft BRT document, source documentation of analyses were included in the appendices of the document because they are not yet published, and we wanted to make these analyses available for comment. In the final technical memo, most of these documents are in review or in press, so will not be included. Two reports, however, one on landscape disturbance and the other prepared by the Habitat Trends Working Group, remains in the appendices because of the utility in understanding the Land use management—stream habitat complexity discussion.]

The data since 1994 presented in Table 2 and Figure 6 do not match as the text indicates they should. For example, year 2000 had 66,900 spawners and 72,200 recruits but in Figure 6, the number of recruits is much less? I realize that harvest levels changed at this time but it does not appear to explain the discrepancy.

[**Response:** Figure 6 was erroneously cited. That has been rectified in the document.]

A somewhat broader definition of life history in the glossary may be useful. A life history encompasses changes experienced from birth through death, including variation in life history traits, such as the size and age at maturity, fecundity. Arguably traits, such as juvenile growth rate, age at ocean emigration, are aspects of species' life history.

[**Response:** Text has been added to the glossary to capture this comment.]

Reviewer 5

I reviewed the document you provided and did not find anything surprising in the habitat-related sections. I did have a couple of comments on the figures that went with the document. I think Figure 16 is of active road densities and does not account for legacy roads that are not maintained and that may pose long-term risks. Figure 26 appears to be of 303(d) listed streams with documented temperature problems or simply streams with documented temperature problems (?), though this is not clear in the title.

[**Response:** Captions for these two figures have been changed to respond to your suggestion.]

Reviewer 6

Review of Oregon Coast Coho status review: Ocean conditions section.

While Good et al. cite evidence for an important shift in climate around 1998, there are no methods for generating climate forecasts that have proven to have skill at lead times of greater than about 1 year (almost all the skill is linked to ENSO monitoring, predictions, and teleconnections) (p. 51).

Periodic warming of the CCS [California Current System] related to major El Nino events or more regionally generated wind anomalies (like those in the spring and early summer of 2005) can cause range expansions for some species, but also range contractions or depth changes for others. Basically, episodic warm extremes in the CCS are caused by a failure of the typical spring/summer seasonal upwelling process to pump cold and nutrient-rich water into the surface layer of the coastal ocean, and to reduce the north-south transport of "subarctic" water over the continental shelf (p. 52, first paragraph).

Changes in ocean conditions appear to simultaneously impact the lower tropic levels that influence the forage base for coho, and at the same time alter the horizontal (and perhaps vertical) distribution of coho competitors and predators. This section on ocean conditions is completely focused on the competitors and predators, and would be improved with more discussion of regional winds, stratification, nutrient concentrations, phytoplankton and zooplankton production and species assemblages, and forage fish.

I really like the discussion of Humboldt squid range changes, and how it is used to illustrate the complex and apparently unpredictable nature of important aspects of the CCS marine ecosystem.

Third sentence from bottom is missing some text (p. 54).

Lead sentence should be in bold, missing a period at the end (p. 55, second paragraph).

Table 11 [now Table 14, revised to reflect these comments]:

Note that early spring peak flow only applies to snowmelt-dominated watersheds, and many Oregon coast coho streams are rainfall dominated, so would not experience snowmelt-related shifts in stream flow timing. Likewise, flood frequency and timing changes due to a warming climate are also dependent in part on the type of hydrology that characterizes different watersheds (e.g., rain-dominant, mixed rain/snow "transient," or snowmelt dominant). For most coho watersheds, the main factor that influences flood frequency and intensity is rainfall frequency and intensity. Generally speaking, rainfall intensity is expected to increase in a warming climate, but confidence in this at the regional scale is relatively low. See Mantua et al. 2010 for a recent analysis of projected changes in high and low flow statistics for Washington State watersheds:

Mantua, Tohver, and Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Clim. Change. DOI: 10.1007/s1058409845-2.

I think that the prediction for delayed spring transition as a consequence of climate change should be rated as "low confidence," given that there is only one study that has looked at this (Snyder et al.), and that was based on a single climate model's output. Bograd et al.'s analysis of trends in upwelling winds did identify a trend to a later spring transition in the northern part of the CCS over the 1967–2007 period, however, there was no attempt at attributing that trend to anthropogenic climate change. This relatively short period of record contains a great deal of natural variability at interannual to interdecadal time scales, so I would caution against overinterpreting the findings of their analysis.

For higher stream temperature, some discussion of potential benefits in winter is warranted. For instance, the experimental logging studies on Carnation Creek showed a strong stream temperature response to logging, and along with the temperature change came increasing growth rates for rearing coho. See:

Holtby, L. B. 1988. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 45:502–515.

Holtby, L. B., and M. C. Healey. 1986. Selection for adult size in female coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 43:1946–1959.

For marine impacts, I would include increased upper ocean stratification. Increased stratification will likely limit the ability of wind-driven upwelling to pump cold, nutrient-rich water into the well-lit surface layer, and this may reduce phytoplankton production and alter the phytoplankton species community. Increased upwelling, if it becomes too strong, may not be favorable for increased phytoplankton and zooplankton production if surface layer waters are swept offshore so rapidly and consistently that phytoplankton and zooplankton are not able to keep pace with the too-energetic physical system.

This table appears to blend the climate change certainty with the ecological impacts certainty, and this is problematic. For instance, increased acidity due to increasing atmospheric concentrations of CO_2 has already been documented, and is virtually certain to continue into the future. The ecological impacts are poorly understood, so that part can be rated with a lower certainty. Perhaps this distinction can be made explicitly in the table caption. For some physical changes, the certainty is much lower (e.g., changes in precipitation).

Appendix C, p. 17: The last sentence of the first full paragraph should be modified to better reflect the fact that no attempt has been made at attributing recent trends in various environmental factors related to coastal upwelling in the CCS to anthropogenic climate change. While the results of Snyder et al.’s modeling study and those of Bograd et al.’s analysis of 1967–2008 upwelling winds share some commonalities, together they do not constitute evidence that anthropogenic climate change is already altering coastal upwelling in the CCS. In order to make that statement, one needs to carry out a formal detection and attribution study, and no such study has been done.

The issue of twentieth century trends in springtime Cascade snowpack has received a great deal of attention in recent years, and the section here should reflect some of the more recent contributions to the literature. A key issue evaluated in these studies is the relative roles of natural low-frequency variability like that associated with the Pacific Decadal Oscillation (PDO) versus the “forced” part of climate change due to increasing concentrations of greenhouse gases and aerosols. Both appear to have played important roles in the observed snowpack trends between the 1930s, 1940s, and 1950s, and the late twentieth century/early twenty-first century.

Basically, it is an oversimplification to say that the observed declines in Cascade snowpack from the mid- to late-twentieth century are due to the regional impacts of global warming.

Recent studies worth citing (along with their abstracts) include:

Casola, J. H., L. Cuo, B. Livneh, D. P. Lettenmaier, M. Stoelinga, P. W. Mote, and J. M. Wallace. 2009. Assessing the impacts of global warming on snowpack in the Washington Cascades. Journal of Climate 22:2758-2772, doi: 10.1175/2008JCLI2612.1.

The decrease in mountain snowpack associated with global warming is difficult to estimate in the presence of the large year-to-year natural variability in observations of snow water equivalent. A more robust approach for inferring the impacts of global warming is to estimate temperature sensitivity λ of spring snowpack and multiply it by putative past and future temperature rises observed across the northern hemisphere. Estimates of λ can be obtained from a) simple geometric considerations based on the notion that as the seasonal-mean temperature rises by the amount δT, the freezing level and the entire vertical profile of snowpack should rise by the increment $\delta T/\Gamma$, where Γ is the mean lapse rate, b) regression of April 1 SWE measurements on mean winter temperatures, c) a hydrological model forced by daily temperature and precipitation observations, and d) use of inferred accumulated snowfall derived from daily temperature and precipitation data as a proxy for snow water equivalent. All four methods yield an estimated 20% loss of spring snowpack for a 1°C temperature rise. The increase of precipitation accompanying a 1°C warming can be expected to decrease the sensitivity to 16%. Considering various rates of temperature rise over the northern hemisphere, it is estimated that spring snow water equivalent in the Cascades portion of the Puget Sound drainage basin should have declined by 8–16% over the past 30 years due to global warming and it can be expected to decline by another 11–21% by 2050.

Hidalgo H. G., T. Das, M. D. Dettinger, D. R. Cayan, D. W. Pierce, T. P. Barnett, G. Bala, A. Mirin, A. W. Wood, C. Bonfils, B. D. Santer, and T. Nozawa. 2008. Detection and attribution of stream flow timing change in the western United States. Journal of Climate.

The Hidalgo et al. article applies formal detection and attribution techniques to investigate the nature of observed shifts in the timing of stream flow in the western United States. Previous studies have shown that the snow hydrology of the western United States has changed in the second half of the twentieth century. Such changes manifest themselves in the form of more rain and less snow, reductions in the snow water contents, and earlier snowmelt and associated advances in stream flow "center" timing (the day in the "water-year" on average when half the water-year flow at a point has passed). However, with one exception over a more limited domain, no other study has attempted to formally attribute these changes to anthropogenic increases of greenhouse gases in the atmosphere. Using the observations together with a set of global climate model simulations and a hydrologic model (applied to three major hydrological regions of the western United States, the California region, the Upper Colorado River basin, and the Columbia River basin), we find that the observed trends toward earlier "center" timing of snowmelt-driven stream flows in the western United States since 1950 are detectably different from natural variability (significant at the $P < 0.05$ level). Furthermore, the nonnatural parts of these changes can be attributed confidently to climate changes induced by anthropogenic greenhouse gases, aerosols, ozone, and land use. The signal from the Columbia dominates the analysis, and it is the only basin that showed detectable signals when the analysis was performed on individual basins. It should be noted that, although climate change is an important signal, other climatic processes have also contributed to the hydrologic variability of large basins in the western United States.

Stoelinga, M. T., M. D. Albright, and C. F. Mass. 2010. A new look at snowpack trends in the Cascade Mountains. Journal of Climate 23:2473–2491.

Stoelinga et al.'s study examines the changes in Cascade Mountain spring snowpack since 1930. Three new time series facilitate this analysis: a water-balance estimate of Cascade snowpack from 1930 to 2007 that extends the observational record 20 years earlier than standard snowpack measurements; a radiosonde-based time series of lower-tropospheric temperature during onshore flow, to which Cascade snowpack is well correlated; and a new index of the North Pacific sea level pressure pattern that encapsulates modes of variability to which Cascade spring snowpack is particularly sensitive.

Cascade spring snowpack declined 23% during 1930–2007. This loss is nearly statistically significant at the 5% level. The snowpack increased 19% during the recent period of most rapid global warming (1976–2007), though this change is not statistically significant because of large annual variability. From 1950 to 1997, a large and statistically significant decline of 48% occurred. However, 80% of this decline is connected to changes in the circulation patterns over the North Pacific Ocean that vary naturally on annual to interdecadal time scales. The residual time series of Cascade snowpack after Pacific variability is removed displays a relatively steady loss rate of 2.0% decade^{-1}, yielding a loss of 16% from 1930 to 2007. This loss is very nearly statistically significant and includes the possible impacts of anthropogenic global warming.

The dates of maximum snowpack and 90% melt out have shifted 5 days earlier since 1930. Both shifts are statistically insignificant. A new estimate of the sensitivity of Cascade spring snowpack to temperature of −11% per °C, when combined with climate model projections of 850-hPa [hectopascals] temperatures offshore of the Pacific Northwest, yields a projected 9% loss of Cascade spring snowpack due to anthropogenic global warming between 1985 and 2025.

[**Response:** Wainwright and Weitkamp (in prep.) was revised to reflect the changes suggested by this peer reviewer and now includes information and citations of the suggested scientific articles. These are also reflected in the BRT document in the Effects of Climate Change on the OCCS ESU subsection.]

Reviewer 7

Review of Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon (*Oncorhynchus kisutch*).

<u>General Comments</u>

The BRT is to be commended for producing an objective, professional assessment of the current status of Oregon coast coho salmon and the likelihood that this ESU will persist into the future. I found the presentation of data clear and the conclusions drawn from them reasonable.

In particular, I found myself agreeing with the report's overall risk analysis that Oregon coast coho face a moderate risk of extinction over the next century.

The inclusion of the potential impacts of climate change on coho habitat was helpful, as was the inclusion of other factors (e.g., human population growth and land use conversions) that will be likely to cause problems for the species. Given the overwhelmingly strong scientific evidence for climate change and the near certainty of population growth and land conversion along the Oregon coast—all of which have major implications for habitat quality—it would have been imprudent to ignore these factors. Additionally, it is quite probable that there will be interactions among these factors, many unforeseen at present, which could exacerbate habitat loss.

[**Response:** The underpinning of the analysis found in the BRT document is now found in detail in Wainwright and Weitkamp (in prep.) This manuscript has been significantly refocused and revised to respond to peer review and internal NWFSC review.]

If anything, I might have been tempted to emphasize the potential future spread of species not currently prominent in the area that could impact coho. Some of these are exotic plants and animals and some rare species will simply become more abundant along the Oregon coast as the climate gradually warms, but in any case the presence of more people and changes in land development patterns will favor a future with a different suite of species than the aquatic and riparian communities of today, and there really isn't much we can do about it. In some recent literature restoration ecologists have referred to this as a "no analogue future."[1]

[**Response:** In response to these comments, the BRT discussed this issue more fully and expanded discussion and literature citation is included in the revised document in the Ecosystem impacts of nonindigenous species subsection, Nonindigenous plant species subsection, and Nonindigenous fish subsection.]

I was impressed with the landscape-level analyses of overall habitat condition, which indicated that freshwater habitats for coho are gradually worsening (especially in the last decade, which surprised me a bit). This conclusion will surely draw the attention of policy makers. I wish there were a way in which future effects of restoration (again, on an ESU-wide basis) could be similarly quantified, but as the report points out, it is difficult to project future benefits with certainty of projects that take decades to mature. I've often felt there is a pressing need to determine whether habitat is currently being lost or damaged faster than it can be restored or rehabilitated, particularly because so much money is being spent on recovering salmon habitat based on the belief that long-term improvement can be achieved at very large spatial scales. Based on this assessment, covering an area that has received considerable restoration investment (Oregon Plan) and regulatory overhaul (Forest Practices Act and Northwest Forest Plan), I would have to conclude that the jury is still very much out.

[1] Hobbs, R. J., and V. A. Cramer. 2008. Restoration ecology: Interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. Annu. Rev. Env. Resour. 33:39–61.

I was pleased to read that ODFW has significantly altered its hatchery program to reduce impacts on naturally spawning coho populations. The state is probably at the forefront of acknowledging the potential evolutionary and ecological harm caused by unintended mixing of hatchery and wild stocks, and they are to be congratulated for taking positive steps to improve the situation.

Following are some comments specific to different sections of the report.

In the discussion of habitat changes prior to and including 1996, what was the impact of the (February?) 1996 flood? I would guess that this one event had a significant effect on coho habitat, but it isn't mentioned here (page 15).

[**Response:** We added some text to highlight this important event, but the Habitat Trends Working Group did not examine this issue as such. Unfortunately, the flood predated the habitat monitoring program that ODFW operates that did allow an assessment of habitat changes over the last 11 years.]

The categorization of threats to coho habitat doesn't really include anything about food web alterations, which may be an oversight. Although I realize we don't know as much about trophic changes as we do about physical habitat changes, food web alteration is an issue that deserves more attention, in my opinion. Are we seeing any fundamental shifts in food web structure (say, in estuaries due to reductions in FPOM [fine particulate organic matter]) that could affect productivity (pages 48 and 49)?

[**Response:** The BRT assessment did not focus on trophic level changes in lakes and estuaries, although we did mention the issue as a result of exotic fish and invertebrate species and temperature rise in a general sense.]

The discussion of water availability centers on water withdrawals, but I think there should be a few sentences on the effect of accelerated erosion on surface runoff. One of the predictions of climate change models is a slight increase in flooding, which is often accompanied by mass wasting. As valleys gradually fill with coarse alluvium, my guess is we will see less surface flow and more subsurface flow, and some streams that are currently perennial could become seasonally intermittent. This could become an issue in some of the smaller streams that will experience less runoff anyway over time. Perhaps someone already has considered this (page 60).

[**Response:** We added text to the Water availability subsection to highlight this issue.]

I thought the definition of stream complexity, as used in the report, was a little vague. In most cases it appeared to be used synonymously with pool abundance. A bit of clarification would help if pool habitat is really what was meant (page 62 and elsewhere).

[**Response:** We have attempted to clarify the definition of stream complexity with text changes wherever it is appropriate.]

Equating disturbance (in this case, logging) with habitat loss seems to be one of the main points of the discussion, but I would suggest some caution here. I agree that the evidence is strong that logging impacts have shifted from federal to state/private lands, but a more persuasive case should be made that private lands are where the problem now primarily resides. For example, long-term impacts to coho in the Alsea watershed study appear to be less than impacts to coastal cutthroat trout.[2] The whole "disturbance = habitat loss" issue as it applies to coho deserves additional discussion. My personal bias, of course, is that some types of disturbance are beneficial (pages 68 and 69).

[**Response:** This comment led to a major addition to text in the Human landscape disturbance subsection.]

I agree that temperature increases will exact a greater metabolic toll on parasitized or infected fish, but temperature may also affect the infection/infestation rate itself. In my prehistoric thesis research I found that elevated temperature actually reduced infestation rates by *Nanophyetus* on juvenile Chinook.[3] It is possible that climate change will promote the spread of some pathogens and reduce the problems caused by others. And we have a very poor understanding of the effects of new parasites and diseases associated with invasive species (page 85).

[**Response:** We have added some brief text to capture this comment.]

Explicitly mentioning that climate change will contribute to both storm and wildfire severity will help to show why it is such an important factor in projecting future habitat changes (page 96).

[**Response:** Wainwright and Weitkamp (in prep.) included a more complete discussion of this issue.]

That's it. Once again I'd like to compliment the BRT for its excellent work. If you have any questions, please don't hesitate to contact me.

Reviewer 8

I enjoyed the executive summary, but frankly, I was disappointed at the brevity of the section where I was hoping to find synthesis of the population structure work (p. 33).

Although Mike Ford's and one of Marc Johnson's papers were referenced, I could not find the area where these were discussed. Sorry my time is so short. Can you perhaps help lead me to where I might find further discussion of genetics work? Is this perhaps in Lawson et al. 2007?

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[2] Gregory, S. V., J. S. Schwartz, J. D. Hall, R. C. Wildman, and P. A. Bisson. 2008. Long-term trends in habitat and fish populations in the Alsea basin. *In* J. Stednick (ed.), Hydrological and biological responses to forest practices: The Alsea watershed study, p. 237–258. Ecological Studies 199, Springer, New York. This paper really should be cited in the report.

[3] Bisson, P. A., and G. E. Davis. 1976. Production of juvenile Chinook salmon, *Oncorhynchus tshawytscha*, in a heated model stream. Fish. Bull. 74(4):763–774.

I attach a more recent paper from Marc’s work you might not be aware of. This was Among top 25 Hottest articles in Earth and Planetary Sciences: Marine Genomics January to March 2010.

Not that you have to cite this, just seems that (Ford et al. 2004, Johnson and Banks 2008, 2009) provide evidence for structure, migration, distribution of diversity, etc., that have relevance to thoughts on history, contemporary population dynamics, and futures.

Bucklin et al. 2008 (also attached) provides an interesting comparison—evidence for deeper structure in California. This contrast could enable interesting comments related to the climate change section—genetic drift—potential futures for Oregon coho etc.

[**Response:** The Populations and life history diversity section was revised to capture these comments.]

Comments of the Oregon Department of Fish and Wildlife

These comments serve to illustrate some of ODFW's primary concerns with respect to the BRT report evaluating the status of the Oregon Coast Coho ESU. The technical nature of these comments is intended to establish a starting point for further discussion.

Interpretation of Productivity Trends

Throughout the report, the BRT makes generalizations regarding trends in coho salmon productivity that are not consistent with patterns of productivity observed over the last 12 years. A typical example (p. 36) states, "Harvest management has succeeded in maintaining spawner abundance in the face of a continuing downward trend in productivity of these stocks." It is more accurate to say that, based on the BRT 12-year moving average, coho productivity and abundance was generally maintained from the early 1970s until about 1999 and has increased significantly from 2000 until 2009.

On page 95, long-term trends are again characterized as "the current BRT was also concerned about the long-term downward trend in productivity of this ESU." Productivity, in this sense, is measured as the ratio of returning spawners to parental spawners. The Oregon Coast Coho Technical Recovery Team (TRT) workgroup developed a productivity criteria based on this "Natural Return Ratio" (Wainwright et al. 2008) that included just the three lowest abundance years over the evaluation period. This adjustment was an attempt to evaluate productivity somewhat independently of variations in parental abundance or ocean survival. In the BRT document, however, Natural Return Ratio (NNR) information (cited as Figure 8 but actually Figure 7) includes all years to create and is interpreted by the authors as a long-term downward trend since 1959.

The problem with this approach is illustrated by the 2007 return year. The abundance of parental spawners was relatively high (168,500) and would not have been included in the TRT's calculation of productivity. Relatively low spawner abundance in 2007 (51,700) is represented in the figure with a value for NNR of about –0.2. While low, spawner abundance in 2007 actually exceeded expectations, given the extremely poor ocean conditions this brood encountered on ocean entry (Oregon Coast Coho Conservation Plan Annual Report Card 2008).

The BRT report's characterization of declining productivity is inconsistent with data presented for the TRT Population Productivity (PP-1) criteria presented in Table 6. Comparing results from this most recent time series to the earlier TRT report (Wainwright et al. 2008) show an overall increase in productivity at the ESU scale. Productivity scores for some populations increased dramatically (Nehalem, Alsea) while others were stable or declined slightly from very high values in the preceding period. Overall, most populations (18 of 21) had truth values greater than 0.25 (BRT report Table 7). This would seem to be an important result; however, the text of the BRT report provides no discussion of the PP-1 scores or changes in population productivity.

[**Response:** This subsection has been extensively rewritten and updated and is found in the Current Biological Status subsection.]

Problems with the Persistence Analysis

The BRT report recalculated the scores for critical abundance (PP-3) using data peak adult counts from ODFW's spawning surveys. Peak counts are the highest number of fish observed at any one time. Unlike, area under the curve (AUC) counts, peak counts are not adjusted for observation bias or ability to see fish during high flow or turbid conditions. The BRT points out that the TRT criteria for critical abundance is defined in terms of peak counts. We agree, but ODFW is very concerned that use of the peak counts greatly overstates the risk of depensation and that, as a result, scores for PP-3 propagate to higher levels of analysis within the DDS to marginally reduce the value for ESU sustainability and greatly reduce the value for ESU persistence. The BRT acknowledges this effect, describing the 0.09 score for ESU persistence as "counterintuitive." This new approach has not been subject to a sensitivity test, nor have the results been reviewed by members of the TRT Workgroup.

The BRT used the average of raw peak counts in each population for the three lowest abundance years since 1997. This criteria becomes the metric for critical abundance (PP-3) in their Decision Support System model. Critical abundance (PP-3) dominates the DSS model, as it is additive as the lowest score among a group of criteria is moved up in the analysis to calculate population persistence (PP). This explains why ESU sustainability only improves slightly as used in the BRT Report and why ESU persistence goes all the way down to 0.09.

The BRT report correctly says that this "counterintuitive" result is directly related to using peak counts instead of adjusted AUC and that using AUC was a mistake. We believe that the mistake was in the definition of the criteria, perhaps in the shape of the truth curve, and, lastly, the fact that they used averaged counts for all surveys. Taking the average for all surveys is inconsistent with the concept that coho populations become restricted to only the highest quality habitat during extended periods of poor ocean survival (Nickelson and Lawson 1988).

Depensation due to low spawner numbers is a concern because it may increase the risk that fish cannot locate mates and that spawning will not occur. Because salmon have high fecundity, and because straying allows for recolonization of unoccupied habitats, the risk of depensation is a rare occurrence in salmon populations. Examples of depensatory effects at low numbers come primarily from isolated populations at very low abundance. Our data show that coastal coho maintain adequate numbers of spawners within core areas of each population and that sufficient straying occurs both within and among populations to assure recolonization.

Barrowman et al. (2003) examined spawner counts and smolt production from 14 independent watershed studies and concluded that depensation was rare, and that effects only occur when spawner density falls below about one female per mile. They also note that including streams with no fish at all results in a measure of depensation that occurs trivially.

ODFW's Life Cycle Monitoring Project provides estimates of spawner abundance and smolt production at several small to medium-sized basins within the Coastal Coho ESU. These data are comparable to that used by Barrowman et al. (2003) and provide an opportunity to examine the occurrence of depensation in Oregon coast coho. Spawner to spawner ratios were calculated for years when parental spawner abundance was at or below 20 fish per mile, 5 fish per mile, and 2 fish per mile. This range of abundance is the same as considered in the TRT critical abundance criteria.

Examination of these data (Table 1) demonstrates that average spawner to spawner ratios for each site always were greater than one, regardless of the number of parental spawners. This means that smolt production at low spawner abundance was sufficient to assure adequate abundance of spawners in successive generations over a range of ocean conditions. Also, smolt production measured as smolts per female spawner did not demonstrate any effect of depensation even at the lowest levels of spawner abundance (Suring et al. 2009, online at https://nrimp.dfw .state.or.us/crl/reports/Annpro/LCMRpt0608.pdf).

Population size and distribution within basins expand and contract through time and as ocean conditions vary. Persistence of populations during periods of poor marine survival is dependent on the amount and spatial distribution of higher quality habitats (Nickelson and Lawson 1998). Through the process of straying, these habitats become source areas to replenish spawners in lower quality habitats as marine conditions improve. The TRT's biological recovery criteria for spawner distribution (PD-3) is determined using minimum occupancy criteria for each fifth-field hydrologic unit (HUC5) within the geographic extent of each population. The spawner distribution criteria was designed to track patterns of occupation and to increase sensitivity to potential for local, catastrophic disturbance. Because of this spatial sensitivity, we believe that spawner distribution criteria (PD-3) combined with TRT spawner abundance criteria (PD-1) provide better metrics for assessing negative demographic effects at low abundance.

In summary, we believe that the use of peak count data fundamentally altered the results of the DDS analysis. In addition, we believe that negative depensatory effects on coastal coho is extremely unlikely, based on experience with other populations and because of the lack of any evidence of such effects in the life cycle basins or at the population scale. Table 2 shows the effect of recalculating the values for persistence and sustainability in the absence of depensatory effects. The ESU persistence truth value increases to 0.58, indicating a moderate to high level of certainty that ESU viability will be persistent over the long term. Values for ESU sustainability increase less dramatically, but solidify the interpretation that ESU viability is sustainable in the foreseeable future.

Table 1. Spawner replacement during periods of low abundance and poor marine survival measured by ODFW's Life Cycle Monitoring Project. Values for the ratio of spawners in one year to the number of spawners in the parent generation are shown in bold. Any value greater than 1.0 indicates that sufficient spawning occurred to at least replace the parent generation.

		Spawner density (adults/mile)		
Population	**Location**	**≤ 20**	**≤ 5**	**≤ 2**
Nehalem	NF Nehalem	**2.7**	—	—
Siletz	Mill Creek	**4.8**	**5.0**	—
Alsea	Cascade	**3.7**	**5.3**	**3.7**
L. Umpqua	NF Smith	**11.3**	**22.2**	—
L. Umpqua	Winchester	**6.4**	**6.4**	**8.8**
Average spawner/spawner all locations		**5.8**	**9.7**	**6.3**
Number of years		21	8	3

Table 2. DDS Results for ESU and stratum criteria assuming of absence of depensatory effects at the population scale.

ESU	Persistence	Sustainability		
	EP	ES		
	0.56	0.29		
Stratum	**Persistence**	**Diversity**	**Functionality**	**Sustainability**
	SP	**SD**	**SF**	**SS**
North Coast	0.53	0.43	1.00	0.63
Mid Coast	0.43	0.36	0.66	0.46
Lakes	0.93	0.76	0.65	0.69
Umpqua	0.78	0.10	1.00	0.35
Mid-South Coast	0.44	0.36	1.00	0.57

We can only speculate about the impact these different DSS scores would have made on the BRT review process. But, as it appears that the DSS analysis served as a starting point for subsequent discussion, it seems likely that this different interpretation may have had substantial effects on the outcome.

In informal discussions among some members of the Oregon Coast TRT Workgroup, it seems clear that the whole DSS approach should be revisited and revised. ODFW would welcome such an opportunity.

[**Response:** As was discussed in Stout et al. (2010), the critical abundance criterion was analyzed in Wainwright et al. (2008) inadvertently with AUC data instead of the chosen data set of the TRT: peak counts. This discrepancy was discovered when rerunning the DSS for the BRT analysis. The analysis found in Stout et al. (2010) is a correction, not a change.]

Assessment of Habitat Trends

It is clear that the results of the BRT analysis of habitat condition and trend were extremely important to the conclusions regarding ESU sustainability and persistence. However, several aspects of the habitat trend analysis appear problematic and ODFW requests a fuller discussion of this issue before a final status determination is made.

Our primary concern is that the BRT report placed too much emphasis on a Bayesian analysis of habitat trends that used a small subset of the available data. We cannot comment on the quality of analysis because insufficient detail was provided in the report. However, even if the analysis were correct and confidence in the Bayesian results is high, we have serious concerns about the applicability of the results to the ESU scale. The use of the ODFW Habitat Limiting Factors Model (HLFM) may also be inappropriate, particularly when applied to the full range of streams within the ESU. This is because the “fuzzy logic curves” used in the AREMP [Aquatic and Riparian Effectiveness Monitoring Program] model and the smolt production values used in the ODFW HLFM analysis are both derived from assessments made on small watersheds, primarily on federal lands. While both are useful tools for interpreting habitat data, application of these approaches to other ownerships, different channel types and habitat forming processes, and different ecoregions clouds interpretation.

We are also concerned that the BRT essentially ignored the results of the ODFW habitat assessment provided to the BRT based on undocumented statements about the importance of interactions between variables. Interactions among variables are likely to be important, but the BRT document failed to document the nature of those interactions, or to provide appropriate citations, for us to do an independent analysis.

We do know that the AREMP "truth curves" used to assess interactions between habitat variables were developed by federal scientists to help monitor the aquatic and riparian strategy of the Northwest Forest Plan. The system is calibrated to conditions on federal lands, as interpreted by federal scientists whose professional opinions are largely based on their experience on federal lands. The BRT does not document if or how this approach may have been adjusted to access the range of land use, land management, and channel form and process that comprise the data set used in ODFW's analysis.

The BRT report does reference a description of the AREMP (Reeves et al. 2004). That document states that a minimum of 50 sites per analysis area is necessary to meet the criteria of the AREMP sampling design and analysis. Based on what we can tell about the sample reaches from the map in the BRT appendix, there were only about 30–40 sites in each stratum that met this criteria. It appears at least 50 sites were included for the North Coast stratum, but 8–10 of those sites appear to be outside the range of coho distribution. Thus the analysis of habitat trend data appears to violate basic requirements of the AREMP design.

Additional comments on BRT Report Appendix E, Habitat complexity

<u>Introduction (page 36)</u>

1) Figure E-1 (sites used by BRT for trend analysis) does not appear to be the correct selection of a) sites within the range of coho and b) annual and 3-year sites. The map does not match that presented in Anlauf et al. (2009).
2) BRT states that "Anlauf et al. (2009) does not capture interactions among the various habitat attributes and does not adequately represent habitat complexity."
    a. Not true: HLFM represents peer reviewed integration of primary and off-channel pool habitat and wood complexity that represents the limiting factor for freshwater survival of coho salmon in Coast Coho ESU.
3) BRT stated that Anlauf et al. (2009) used sites that had been surveyed only once or twice.
    a. Not true: Only sites surveyed at least twice or more times were used in the analysis. Sites were pulled from the annual and 3-year site list as described in Anlauf et al. 2009.
    b. The BRT did not request clarification of the methods section in Anlauf et al. (2009).
4) Contrary to statement by the BRT (page 38), ODFW was not requested to rerun the analysis for any variable or data sets (pers. commun., K. Anlauf and K. Jones, ODFW). BRT did ask for the literature citations for the scientific basis of HLFM which was e-mailed to them (K. Jones, ODFW).

5) Contrary to the statement by BRT concerning the rerunning of the HLFM on first paragraph page 38, the ability to estimate summer habitat was present in the version of the HLFM published in 1989 and has remained in the model. The BRT did not review the literature supporting the HLFM model or interpret the published information correctly. Anlauf et al. 2009 did not present information on summer rearing capacity because summer habitat does not limit the populations of juvenile coho and the capacity is directly related to the amount of surface area in pool habitat.

6) Paragraph 2, page 38: The statistical validity of the AREMP model used by the BRT to combine variables for trend analysis is not supported in the document or referenced by published literature.

7) Page 38, paragraph 2: Statement by the BRT that Anlauf et al. (2009) did not stratify the analysis is not correct. The sample frame was designed to describe status and trends at the ESU and monitoring area scales; Anlauf et al. (2009) used the design strata to describe status and trends at these two scales. The sample size was too small (and the sample frame statistically inappropriate) to describe status or trends at the population or site scale as done by the BRT.

<u>Habitat complexity indices (page 38)</u>

1) Anlauf et al. (2009) used the HLFM model to describe habitat complexity for juvenile coho during the summer and overwinter life stages. It is a peer reviewed, quantitative model, which captures a combination of pool complexity and large wood. It considers habitat complexity relative to coho life history.

2) We do not agree that individual habitat variables are not useful for describing status and trends. It is essential to understand the trend in individual variables to be able to interpret the results of a metric that integrates variables. In addition, individual variables have a direct relationship to habitat requirements of each life stage of coho. For example, fine sediment can reduce the emergence success of egg to alevin stages in spawning areas (Everest et al. 1987) and the amount of pool habitat influences the carrying capacity for juvenile coho (Nickelson et al. 1992, Rosenfeld et al. 2000). An integrated metric may describe general conditions in a watershed (AREMP model), but it does not focus on habitat requirements for the different life stages of coho. To be appropriate to the review of viability of coho in coast populations, the integrated metric must be developed specifically for carrying capacity and quality of habitat for coho, as with the HLFM presented by Anlauf et al (2009).

3) Page 39: The validity of the BRT's use of the AREMP decision support model channel condition score to evaluate productivity of aquatic habitat for coho populations is not evident or forthcoming. Information is not presented to describe the values, weights, or logic statements of the individual variables relative to habitat productivity for coho salmon.

4) Page 39, third paragraph: The interpretation of the HLFM model by the BRT was not correct. The HLFM was designed to estimate spring, summer, winter, and spawning habitat for coho salmon from the onset (Reeves et al. 1989, Nickelson et al. 1992). The modeling of capacity of winter habitat was improved recently with the addition of a large wood metric. Also, winter rearing capacity scores for the status assessment were

generated from winter and summer surveys (Anlauf et al. 2009). Because the trend analysis required data from repeat survey sites, we used the summer values converted by a regression model ($R^2 = 0.879$) to reflect winter rearing capacity.

Trend modeling (page 39)

1) The sample frame and site selection were designed for application at the ESU and monitoring (referred to as the GCG unit by the BRT) scales. It is not statistically appropriate to use the site data to describe conditions at the population scale (number of sites per population unit, site weight issues).

Results (page 41)

1) First paragraph of Results section, first sentence: The ODFW did not rerun the HLFM in the trend model. No request was received from the BRT.
2) The BRT needs to provide the data sets, assumptions, and models used in its trend models. We are not able to evaluate the statistical rigor of its approach, use of the DSS model, variance partitioning, or its conclusions.
3) Page 42: the second paragraph is a repeat of the last paragraph on the previous page. The BRT should edit its report before release.
4) Table E-1: The BRT needs to provide statistical support to indicate that the probability estimates are significantly different from 0.
5) Last paragraph, second sentence: ODFW did not rerun the HLFM. The BRT needs to provide the data set and model they used when they apparently reran a trend model.

Conclusion

ODFW cannot evaluate the validity of the BRT's conclusions because of:

a. inaccuracies in its statements,
b. apparent lack of understanding of the underpinnings of the HLFM model,
c. incorrect data set,
d. concern over use of DSS for quantifying habitat relationship to productivity of life stages of coho salmon, and
e. unknown statistical rigor of the BRT's trend analyses.

[**Response:** National Marine Fisheries Service (NMFS) scientists and ODFW scientists formed a joint habitat group (Habitat Trends Working Group) to resolve these issues and come to consensus regarding appropriate analyses, data sets, data transforms, etc. Revised text is found in the In-channel habitat complexity subsection of the BRT document proper and a detailed presentation is included in Appendix C.]

Incomplete Discussion of Limiting Factors and Threats

Although it is difficult for us to determine the effect of the final BRT recommendations, we believe that, in some cases, the discussion of limiting factors and threats was often based on incomplete information or misunderstanding of available information. This was particularly true in the assessment of fish passage and estuary habitats. The fish passage issues are technical and may require additional analysis; we would like to discuss this further.

[**Response:** The report was revised to reflect information contained in several sources supplied by ODFW. In addition, we included information from a new data compilation that was, for the most part, absent from the most recent ODFW database. This is found in the Fish passage subsection.]

The following comments apply to coho use in estuary habitats.

The BRT hypothesizes that estuaries are critical habitat for the productivity and life history diversity of coho salmon. To date, these assertions of have not been born out in published studies. As described by the BRT (based on a synthesis paper by Koski 2009), juvenile coho are observed using tidal habitats in some estuaries of the Pacific Northwest. We (ODFW) have observed use of estuary habitats by yearling and subyearling coho in the spring, and subyearling coho in the summer (very limited) and winter. However, neither the Koski (2009) paper nor studies by ODFW have demonstrated a link of estuary rearing to adult contribution. There is no demonstrated evidence that estuary use is a critical life history strategy for coho for any independent populations.

Without question, estuary habitat has been significantly reduced over the past 100 years. However, the Oregon Watershed Enhancement Board (OWEB) has invested significant resources and had many successes in restoring, acquiring, and protecting estuary habitat since the inception of the Oregon Plan in 1997. The actions of OWEB will provide significant buffer to climate change and rising sea level.

Specific comments on BRT review:

Appendix D includes all the information that is sprinkled throughout the primary document (Scientific Conclusion of the Status Review). It is difficult to tell in the primary document whether the BRT concluded that estuaries were “critical” to the viability of coho:

1. Summary: The BRT states “However, little direct quantitative information exists on the relative proportions of coho salmon juveniles that use this life history pathway, the survival rates and capacity relationships involved, and the relative contribution to adult returns.” This is a true statement and in fact the literature suggests that the estuarine life history contribution is relatively small except possibly in small streams that feed directly into the estuary ecotone.
2. The Koski (2009) manuscript is a nice synthesis paper. It was published in Ecology and Society as part of the special edition on salmon in 2010. Koski’s basic statement regarding 0-age coho contribution to the adult population is: “documentation of age 0 coho migrating to the ocean and surviving to adults is limited.”

3. Even though we (ODFW, Jones and Cornwell) have documented that age-0 and age-1 coho use the estuary, we do not have evidence as to the contribution to the adult population. We suspect estuary rearing is a life history pattern, and in Salmon River, the abundance and quality of restored wetlands provides good prey resources as juvenile coho rear or migrate to the ocean. However, to suggest that the estuary life history pattern is critical to viability of the Salmon River population is an overstatement.
4. For the most part, our sampling in fresh and brackish wetlands and channels in 2003–2004 in the Siuslaw, Alsea, Yaquina, and Nestucca document that subyearling coho are present in the spring through June, then disappear. Juvenile coho clearly prefer freshwater wetland habitats (K. Jones, T. Cornwall, D. Hering, L. Borgerson, D. Bottom, S. Simenstad and A. Bierber. 2007. Patterns of coho salmon migration and residency in Oregon estuaries. Oregon AFS presentation).
5. The fact that Lisa Borgerson has identified 2-year-old adult coho returning to Salmon River does not necessarily imply that they used the estuary, only that they migrated to the ocean as subyearlings.
6. The BRT's reference to Bottom et al. (2005) regarding the life history of Chinook salmon as representative of coho use of estuarine habitat is not a valid template for life history of coho salmon. It is an inaccurate and misleading comparison.
7. The example from Skeesick (1970) is not valid. Skeesick (1970) focused on migrations of juvenile coho in and out of Spring Creek from the mainstem Wilson River (Tillamook basin), not Tillamook Bay. Skeesick also mentions observed migrations into Munson Creek, a tributary to the Siuslaw River, during high flow events. The theme of Skeesick's paper was the importance of small tributaries, not the use of estuarine habitat.
8. OWEB is aggressively acquiring freshwater and brackish wetlands and supporting extensive restoration of tidally inundated habitats. The BRT should acknowledge fully the restoration and protection of tidal-fluvial and estuarine habitat by OWEB. The actions by OWEB will serve as a hedge against global climate change.
9. The BRT's review of the relationship of coho viability and estuary habitat at the independent population scale is highly speculative and circumstantial (my opinion). That is not to denigrate the ecological importance of estuaries to the coast ecosystems and salmon populations, particularly Chinook and chum salmon, along with sea run cutthroat trout. The estuary environment may provide an alternative life history pathway for juvenile coho in some cases, but is unlikely to be a major factor in the viability of coast coho populations.

[**Response:** Additional text was included in the document to further explain the importance of estuaries to OCCS and to include information provided by ODFW and other sources. Both the BRT and ODFW are in agreement that there has been significant loss of estuary habitat along the Oregon Coast during the last 100 years. We acknowledge that there is some scientific disagreement between ODFW and the BRT regarding the severity of the effect of estuary loss on the viability of the OCCS ESU. However, the loss of estuary habitat is only one of many factors limiting the sustainability of this ESU. In contrast to the previous comments, Reviewer 3 stated that "the emphasis given to the importance of estuarine habitat is moderate and adequate, given the information available in the literature." The reviewer notes that they have observed juvenile

OCCS rearing in estuaries and feels that this life history strategy is fairly common. The reviewer also provided some specific scientific information to support the statement. This viewpoint is consistent with the BRT's position on the importance of estuaries to juvenile OCCS. The BRT revised the report's section on estuaries to include the information provided by the reviewer and by ODFW and OWEB.]

Incomplete Assessment of Restoration Efforts

ODFW believes that the BRT report underestimates both the variety and effectiveness of habitat and watershed process restoration efforts. Further discussion and evaluation is necessary to assist NMFS in its understanding of these efforts.

[**Response:** The SHAPE (Salmon Habitat Assessment Project Evaluator) analysis has been removed from the BRT document and protective efforts are assessed by the NMFS Northwest Region Office (NWR) and included in the Federal Register notice.]

Comments on "Restoration Projects" (page 90) of Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon (*Oncorhynchus kisutch*)

We (ODFW) are submitting the comments of behalf of the OWEB and a summary of recent habitat restoration monitoring conducted by ODFW.

The analysis presented in Appendix F compares restoration projects from a NOAA database of restoration projects with habitat concerns identified in watershed assessments. The analysis is creative and useful to assist in better targeting funding. To be effective, the analysis should include a complete disclosure of data limitations, embodied assumptions in the analysis, and limits on the interpretation of the data. The following comments are provided to help build a fuller understanding of the analysis and the limits of the analysis.

<u>Data Limitations</u>

1. Watershed assessments using the OWEB Watershed Assessment Manual were completed between 1999 (when the manual was completed) and 2007. This means that many if not most of the restoration projects funded and available in the database (see 3, below) were finished prior to completion of the assessments. To have a direct comparison between the issues identified in assessments with projects completed would require analysis of only those projects implemented for each watershed following completion of an assessment. It would be interesting to see if there was a chance of power of association following completion of assessments.
2. The restoration data used in the analysis came from the Pacific Northwest Salmon Habitat Project Database (PNSHPD). This database is populated from the OWEB. The last time the PNSHPD was populated by the OWRI [Oregon Watershed Restoration Inventory] was June 24, 2004 which included projects completed between 1995 and 2002. This limits the analysis to most projects completed prior to the completion of watershed assessments.
3. The OWRI contains data compiled since 1995 on all restoration projects funded by OWEB and a significant number of projects (approximately 70%) reported voluntarily by

others that do not use OWEB funding. The early reporting was dominated by voluntary projects primarily on private industrial forest lands. What this means is that there may be a reasonable disconnect between limiting factors for coho salmon and implemented projects.

4. OWEB funds projects for resources other than coho salmon. Projects that protect habitat for invertebrates and other species will not always align with priorities set for watershed functions.

Embodied Assumptions

1. Central to the analysis is the assumption that the two variables (habitat concerns and project types) have a temporal relationship, for example habitat concerns are known before projects are selected. Appendix F does not indicate that such a sorting was done. It appears that all projects funded in a geographic area were compared to habitat concerns regardless of when the projects were implemented and the concerns identified.

Limits on Data Interpretation

2. The data results state that “14 assessment units (28%) have a SHAPE score of between 0 and 0.5 and 25 assessment units (50%) have a SHAPE score of between 0.5 and 1.0.” This should indicate that more than 78% of the assessment units have a positive relationship between habitat concerns and project types. If the same data were calculated on a HUC basis, the percentage of positive scores would be even higher (approximately 86%).
3. There were only two watersheds where there was a relatively high negative correlation between habitat concerns and restoration actions. It appears that this fact alone should indicate that overall there is a good to fair correspondence.
4. Even a cursory evaluation of the watersheds that have a low score would show that these watersheds are dominantly federally managed and it would not be expected that restoration activities on a minority of the watershed would address the principal habitat concerns (the Upper Cow Creek watershed is 73% public land and the West Fork Cow Creek watershed is 53% public land). The OWEB database does not have any projects from the West Fork Cow Creek watershed.
5. There was no power analysis of the results suggesting that small sample sizes might affect results significantly. If there was any data selection for small samples, it was not described in the appendix.
6. An equally valid interpretation of the results could be that there is a very high correspondence between habitat concerns and restoration projects implemented despite the data limitations and lack of analytical precision.

[**Response:** The SHAPE analysis has been removed from the BRT document and protective efforts are assessed by the NMFS NWR and included in the Federal Register notice.]

Summary of ODFW effectiveness monitoring report, cited as “in prep.” in Anlauf et al. (2009) in materials submitted to the BRT in September 2009:

Tippery, S., K. K. Jones, K. J. Anlauf, C. H. Stein., M. J. Strickland. 2010. Effectiveness monitoring report for the Western Oregon Stream Restoration Program, 1999–2008. OPSW-ODFW-2010-6, Oregon Dept. Fish and Wildlife, Salem.

The report examined the location of the instream large wood projects relative to the distribution of coho and reaches of high intrinsic potential, the long-term (6 years) retention of large wood and complexity of habitat, and estimated the overwinter rearing capacity for juvenile coho. The OWEB database has additional information on riparian and passage projects.

This is the first extensive study of before-after effects of restoration treatments over a broad geographic scale. Previous studies of the WOSRP [Western Oregon Stream Restoration Program] projects (e.g., Jacobsen et al. 2007) reported retention and recruitment of large wood, but few other changes were significant. Here we demonstrated an increase in surface area of pools and sorting of substrate within the project areas in the majority of the projects. Of the 46 projects evaluated over a 6-year period, only 8 (17%) were considered to have "failed," losing large wood or pool area. Another 15 projects (30%) increased the amount of large wood, but generally did not experience other change. For 24 (53%) of the projects however, wood was maintained, additional pool area was created, gravel accumulated, and rearing capacity for juvenile coho increased. These projects, distributed throughout the coast and lower Columbia basins and including both large wood and wood/boulder projects, were considered successful 6 years following treatment. While 6 years may be marginally "long term," this study is a first attempt to quantify change in habitat following restoration more than 6 years after treatment.

All of the projects were placed in appropriate geomorphic settings and within the distribution of coho, maximizing the effectiveness of restoring fish habitat. We expect that the projects have started to improve fish habitat (habitat complexity and rearing capacity) at the population and ESU scales. The WOSRP and OWEB large wood projects treated 750 km of stream from 1995–2007 in western Oregon, of which 550 km were within the distribution of coho. If half of the OWEB projects are successful (as indicated by the WOSRP evaluation), up to 2% of coho habitat in the Coast Coho ESU has been significantly improved. While limited relative to the kilometers of stream potentially inhabited by coho, the impact is more pronounced, given the projects were placed within the stream reaches most productive for overwintering juvenile coho.

Although the amount of stream treated was not statistically detectable by ODFW's monitoring scale GRTS [Generalized Random Tessellation Stratified design] samples (we sample approximately 3% of coho habitat), the placement in potentially productive habitat and effectiveness of the projects most likely improves conditions for juvenile coho in the Coast Coho ESU.

[**Response:** NMFS scientists and ODFW scientists formed a joint habitat group (Habitat Trends Working Group) to resolve these issues and come to consensus regarding appropriate analyses, data sets, data transforms, etc. Revised text is found in the In-channel habitat complexity subsection of the BRT document proper and a detailed presentation is included in Appendix C. Other information presented here was taken into account by the NMFS NWR and included in the protective efforts section of the Federal Register notice.]

Comments of the MidCoast Watersheds Council Coordinator

21 July 2010

I am the Coordinator of the MidCoast Watersheds Council, based in Newport. The council does restoration work in the area from Cascade Head to Heceta Head, with an emphasis on improving watershed function and improving habitat for coho. I was a member of the OC Coho Stakeholders group assembled by the State of Oregon and NMFS that produced Oregon's OC Coho Native Fish Conservation plan. I provide here a set of comments on particular sections of the text, followed by three more general comments on coho status.

Wayne Hoffman
Coordinator
MidCoast Watersheds Council

Pp. 9–18. Review of BRT and Listing. The review of previous status evaluations is very useful, as is the history of listing decisions included in the Introduction. These are probably adequate for the purposes of this document, but I think it would be very useful to develop a document that integrates the status evaluations and listing decisions, with more detail provided on the latter than is present in the introduction. I think such a document would be very useful in understanding the evolution of thought on what constitutes an adequate status evaluation, and on what level of decline or risk supports listing decisions. I think such a document would be helpful to people managing other depleted, declining, or otherwise at-risk fish.

Pp. 20–21: Artificial Propagation. Membership in the ESU. This discussion could use some elaboration. More considerations than those mentioned here ought to go into decisions about inclusion or not in the ESU. Basically this says that the NF Nehalem and Trask hatchery stocks have not had naturally-spawning fish deliberately included in the brood stock for several generations, so they must be divergent. Conversely for Cow Creek, "wild" fish are included in the broodstock, so they must not be divergent.

If actual evidence for such divergence has been documented in Nehalem or Trask, it should be reported here: for example, has run timing diverged? Or other phenotypic or life cycle differences arisen? Average size? Frequency of jacks? For Cow Creek, has a lack of divergence been documented? If so, how?

I have concerns that we are still not adequately understanding how salmonid stocks evolve, and that as a result we tend to misinterpret the tools we use. On the one hand, we have abundant evidence that salmonid populations can respond with impressive speed to selection, and reasons to believe they are continuing to do so. On the other hand, the molecular tools we tend to use in studying salmon population structure (e.g., microsatellites) are assumed to be selectively neutral, such that their measures of divergence (or lack thereof) are independent of evolutionary changes resulting from strong selection.

In many cases, hatchery stocks do show evidence of rapid phenotypic divergence, and these ought to be documented. Dr. Michael Blouin and his coworkers have been reporting divergence in fitness of hatchery steelhead stocks so rapid that they are looking at epigenetic mechanisms to account for it. If the mechanisms they propose are in fact operating, inclusion of “wild” broodstock may not be adequate to prevent divergence in the Cow Creek stock, although microsatellites might not show it.

[**Response:** The Artificial propagation subsection has been revised with a link to the previously published (Federal Register notice) discussion of membership in the ESU and additional explanatory text.]

P. 24. Is there a mistake in the description of EPA comments? I do not understand what the phrase “on other OC coho salmon reviews” means in this context.

[**Response:** This has been revised for clarity.]

Pp. 30–88. New Data and Updated Analyses. This section (really the meat of the Status Review) would be improved if each subsection included (perhaps ended with) a listing of remaining information and analytical gaps that affect the listing recommendation. This would be particularly useful in that information used in previous assessments is not really summarized. For example, the Population Diversity subsection (Pp. 33–34) discussed Koski’s new information on estuarine rearing but did not indicate whether or not the BRT considered current information adequate on other life history diversity (e.g., lake rearing, occupancy of beaver ponds, nontidal wetlands, etc.). In effect this is done for the Mark-Selective Fisheries subsection (p. 35) with reference to catch-release and drop-off mortality, but not so much for the other subsections.

[**Response:** We revised many of the subsections to capture some of the information gaps that existed. These include beaver populations and fish passage on private lands.]

P. 59. Quote from NMFS NWR, “fish passage restoration projects have not been tested at high flows, … have rarely been monitored to test whether they are actually passable.” This is not true for the projects my organization (the MidCoast Watersheds Council) has been doing. We routinely check for coho upstream of our passage projects. This is often qualitative, that is, presence-absence, rather than abundance estimates. In addition we always monitor the replacement structures for compliance with the passage guidelines, which we think are good indicators of performance. If a replacement culvert retains a gravel substrate at appropriate gradient, without apparent barriers (subsurface flow, downstream lip, etc.) I think we can be safe to assume it is passable. I do not understand the need for testing “at high flows” because even if a structure is difficult to pass at highest flows, such flows are generally transient, and salmon needing to wait a few hours, or even a day or three to pass is hardly catastrophic. Salmon more commonly have to wait for proper flows at natural barriers (small waterfalls, beaver dams).

[**Response:** The Fish passage subsection has been substantially revised with additional information provided by ODFW and other sources.]

Pp. 63–66. Beavers. I agree with the general conclusion that “declines in beaver abundance is an ongoing threat to OC coho salmon” (p. 66), but I think this threat needs to be further analyzed and discussed.

First, it needs to be fully acknowledged that the habitat benefits beavers provide are landscape-context specific. Beavers occur within the ESU in a variety of contexts, from brackish estuarine marshes, to lakes, to large mainstem rivers, to smaller tributaries. The role of beavers in affecting coho habitat varies greatly among these landscape contexts. Beavers in estuarine marshes may build dams in upper portions of sloughs and both provide physical habitat for coho and may positively modify salinity by impounding freshwater inflow. Beavers in lakes and mainstem rivers generally do not build dams, and may have little effect, positive or negative, on coho habitat. Beavers in headwaters areas do build dams, and this is where their positive contributions to coho habitat are best understood. This is important because population trends in beavers and threats to beavers likely vary across the landscape. For example, recreational trappers are more likely to recognize opportunities and have access in headwaters areas than along main stems.

[**Response:** We added text to capture this comment in the Beaver in OCCS habitat subsection.]

Second, this section properly reviews the legal status of (non)protection for beavers, but needs also to address other potential causes for the documented declines in beaver dams. Two of these in particular need discussion. Cougar populations have increased substantially in the ESU in the past 2 decades, and cougars are probably effective predators of beavers, particularly in headwaters areas. This needs verification, but if true, may need to be treated as a fairly intractable increased threat to coho habitat.

[**Response:** We agree with the reviewer in part. Estimated cougar populations have increased since the 1970s over the entire State of Oregon from approximately 214 to more than 2,800 by 1992 (Keister and VanDyke 2002). However, nothing in the literature suggests that predation on beaver is a primary cause for reduction in beaver population. The majority of studies identify deer and elk as the primary food source for cougars (Ackerman et al. 1984).]

Second, we see in many areas colonization of riparian zones by reed canary grass (*Phalaris arundinacea*), which outcompetes the trees and shrubs that beavers prefer. Likely beaver populations fluctuated on a decadal cycle, with population crashes followed by recolonization and proliferation of food plants, then recolonization by beavers. Reed canary grass seems to break this cycle by preventing the recolonization and proliferation of the shrubs and trees. We see many former beaver ponds now as grass flats in successional stasis, in some cases with the stream channel down-cutting. We currently are experimenting with brute force control of the grass to reestablish beaver food, but expect this to be expensive and slow with current methods.

[**Response:** This is an excellent point. Reed canary grass is an issue with respect to beaver ponds in the Oregon coast (Perkins and Wilson 2005). In addition, more aggressive management actions are needed to deal with it, as evidenced by recent work that suggests plantings and natural vegetation alone cannot control it (Kim et al. 2006, Healy and Zedler 2010, Spyreas et al. 2010). We added text to capture this point in the Beaver in OCCS habitat subsection.]

Pp. 66–67. Roads. I am sure the negative relationships between road density and productivity are real, but that does not mean they are fully causal. Road density may be best thought of as a surrogate variable for landscape disturbance that depresses productivity on a long-term basis, and it is worthwhile to dig deeper and look at other aspects of the disturbance on a site specific basis, including direct effects of logging and yarding, stream cleaning, and so on, that likely correlate with road density. Clearly roads can affect productivity directly (e.g., through sediment inputs and passage impediments), but these may not necessarily correlate as tightly with road density as the cited studies imply. For example, obliterating a valley-bottom road and replacing it with two ridge top roads may improve productivity without reducing road mileage in the basin.

[**Response:** These are certainly good points regarding the relationship of road density and productivity that bear closer examination. This may be a task for the recovery planning process.]

Pp. 69–70. Loss/Gain of Large Wood. This discussion may overstate the issues with large wood recruitment from riparian areas. The statement, "the large and very large trees are the size of tree that creates more complex habitat conditions" is generally true, but wood size needed scales to stream size and power. I was informed recently that in consultation on thinning projects, NOAA has recently reported to the Siuslaw National Forest that in small headwaters streams, trees as small as 20 cm diameter are important for recruitment to the streams.

Pp. 70–75. Habitat Complexity. The conclusion here that habitat complexity is continuing to erode (so to speak) is certainly provocative, and I suspect very influential to the conclusions reached. I find this conclusion hard to evaluate because the descriptions of analyses lack necessary detail on the sampling protocol and particularly the sample sizes. I urge you to prepare a more complete report detailing these analyses.

I am familiar with the basic data collection method (ODFW Aquatic Habitat Inventory protocol), and am aware that the protocol gets updated periodically. It may well be that the particular parameters used are measured comparably through these updates, but this should be verified and, if so, stated. I know that some parameters are very sensitive to flow rates at the time of survey. For example, an area that will classify as "glide" at moderate flows may break out into multiple pools and riffles at minimum flows. The logistics of matching flows across multiple surveys in different years would be daunting, and I am not aware of any attempts to do this. Of course this is one of the things that with adequate sample size should even out, but this inflates the necessary sample size. I am also aware of flow-related inconsistencies in assessment of large wood, for example in the counting of logs that are wetted only at moderate or higher flows. I suspect that restoration projects may put in a higher percent of these than nature does (e.g., "top" logs on a jam added to improve stability, also logs placed into seasonal side channels).

So, I do not challenge the conclusion of ongoing declines in complexity, but more detail is needed to prove the case. In addition, it would be really useful for restoration efforts to partition this result among the various factors that can cause decreases in complexity and in channel context. How much results from reduction in number of beaver ponds? From loss of wood? How much is in streams at 0–1.5% gradient? Versus 2–4% gradient? How much is in second versus third versus fourth order streams? Knowing these things could really help us prioritize restoration. I doubt that the sample sizes available are sufficient for a lot of this partitioning, but it would be a productive direction for future research.

[**Response:** These are certainly good questions that may bear more investigation. This may be a task for the recovery planning process.]

P. 75. Forest and Agriculture Conversion. It is worthwhile also describing ongoing conversion trends. We are seeing three in the MidCoast, and I suspect they are playing out elsewhere (and maybe some others). First is a trend of conversion of agricultural land to forest land. As agricultural market structure becomes more difficult and owners get older, many are giving up on grazing and are planting trees. This has a variety of implications for coho habitat (jurisdiction of Forest Practices Act rather than SB1010 for example). Another trend is sale of “serious” agricultural land to hobby farmers who do not have the same income expectations from farming. This trend has led to more “ignorant” behavior such as gross overstocking with horses, but the hobby farmers tend to be receptive to advice on best practices. In addition, the last cattle dairy in Lincoln County has converted to raising beef cattle. The third ongoing trend is residential development on the banks of larger streams. Currently this is more of a problem the larger the river is and the better-developed the recreational fishery. A legal gap in riparian protection allows this trend to result in major riparian abuse.

[**Response:** We added several points to the Land management—forest and agriculture conversion subsection to capture the essence of these comments.]

Pp. 79–82. Estuary loss-gain. The caption to Figure 24 needs to explain the difference between the two histograms. I suspect one uses numbers from Good (2000) and the other from Adamus et al. (2005), but that is not explained and which is which is not identified.

[**Response:** This caption has been revised for clarity.]

Pp. 82–84. Loss/gain of Freshwater Wetlands. Another data source is potentially available for assessing freshwater wetland loss away from estuaries. If you look at soil surveys (complete for Lincoln County, not sure how complete in the rest of the ESU, but coverage should be extensive), hydric soils are good indicators of current + former wetland conditions. These are soils that form under wetland conditions, but are recognizable for quite a while (at least multiple decades) after loss of wetland hydrology. So if a hydric soils map layer were constructed from the overall soils maps, then checked for current conditions, either on aerial imagery or on the ground, a pretty good estimate of wetland trends could be obtained. I strongly suspect that such an analysis would show considerable “inadvertent” wetland loss as well as deliberate draining. The inadvertent loss would result from stream down-cutting following large wood removal or depletion. The result would be disconnection and eventual drying of lateral wetland areas. Such a map would also be very useful in identifying locations for projects to reestablish floodplain connection.

[**Response:** The Oregon Natural Heritage Information Center and The Wetlands Conservancy are engaged in a project of this kind; results are available at http://oregonexplorer.info/wetlands. This type of analysis is far beyond the scope of the BRT assessment. However, this may be a task for the recovery planning process now that LIDAR remote sensing has been flown for much of this area.]

P. 84. Land Management–Mining. I am surprised the Coquille is not mentioned in this discussion.

[**Response:** The Coquille River has been added to the discussion of mining.]

P. 84–85. Water Quality Degradation. Very recent data collection projects in the MidCoast area have identified more pervasive coliform bacteria and dissolved oxygen problems than previously recognized. The latter in particular seems to involve important coho rearing habitat, and seems connected to both agriculture and riparian domination by reed canary grass. The data are very new and preliminary, but this should be watched as a potential additional factor limiting coho recovery.

P. 90. Restoration Projects. This analysis has the potential to provide useful guidance to local groups performing restoration, but some logical lapses affect the conclusions drawn here. The detail provided, both in the text and in Appendix F, is insufficient to fully evaluate the methods or make good use of the results at the local level. Hopefully a more detailed presentation of these results will be made available to us.

Some concerns do arise from comparison of the SHAPE metrics to the identified limiting factors. Looking at Figure F1, it appears that many of the units with lower scores (darker shading) are units with limiting factors less amenable to locally developed projects. The lakes units, for example, list exotic fish as their primary limiting factor, and management of exotic fish is more of an ODFW responsibility than something local groups can or should tackle. Similarly, the hatchery impacts and water quantity issues in the Umpqua may be less amenable to local action. Certainly elsewhere local groups have invested in irrigation efficiency projects, but my sense is that some of the water quantity issues in the Umpqua are related to the operations of industrial-scale dams and reservoirs, and you should not be expecting local groups to be reporting projects to the database that address these.

So in general, what I take from this analysis is that local projects are lining up best with the limiting factors where the limiting factors can be best addressed by site-specific projects, and are not lining up so well where the limiting factors are more matters of policy (fisheries management, FERC relicensing, etc.).

[**Response:** This analysis is no longer part of the BRT document; the discussion of restoration activities is now found in the protective efforts section in the Federal Register notice. However, this type of analysis would be appropriate for a recovery planning effort.]

Overall Comments:

1. The status review has omitted a factor that may have major implications for reduced productivity of coho habitat. This is habitat loss/degradation through the action of exotic organisms. Several of these are known or can be expected to directly affect coho habitat. One category is exotic vegetation that impedes or prevents development of healthy woody riparian vegetation. Currently, particular problem species are reed canary grass, the giant knotweeds (3 species [*Polygonum* spp.]) and Himalayan blackberries (*Rubus discolor*). All these can out-compete seedlings of native riparian trees and shrubs, and maintain a low-height riparian condition that does not provide adequate shade and obviously does not contribute large wood.

The knotweeds and blackberry also appear to be poorer than native woody vegetation at bank stabilization, and may contribute to increased fine sediment input. Reed canary grass forms a thick thatch that is very effective at bank stabilization, and is also very effective at fine sediment trapping. Marshy riparian areas colonized by reed canary grass may tend to become disconnected through aggradation both from sediment trapping and from accumulation of organic remains. Reed canary grass also inhibits colonization by beavers and seems to be associated with dissolved oxygen problems.

A second category is colonization by exotic freshwater mollusks. One, New Zealand mud snail (*Potomopygyrus antipodarum*), is already present in Devils Lake and Tillamook, and perhaps elsewhere. It has the potential to displace other periphyton grazers that are important food sources for juvenile Coho. Zebra (*Dreissena polymorpha*) and quagga (*Dreissena bugensis*) mussels are also major threats, and would utterly change the ecology of any streams they colonize.

[**Response:** In response to these and other reviewers' comments, the BRT discussed this issue more fully; expanded discussion and literature citation is included in the revised document in the Ecosystem impacts of nonindigenous species subsection, Nonindigenous plant species subsection, and Nonindigenous fish subsection.]

2. I will not argue with the overall conclusion that OC coho are at moderate risk of declining to endangered status, hence deserve continued listing as threatened, even though I have concerns about several of the analyses (described above). In summary, I think the analysis of hatchery influence/ESU membership is inadequate, passage issues are being treated more effectively than recognized, the road density concern needs to be developed more as a disturbance analysis, and the habitat complexity analyses are provocative but need to be presented more fully for review and analysis. On the other hand, the implications of beaver pond declines are not fully acknowledged, more informative analyses could be done of freshwater wetland loss, water quality degradation needs also to consider dissolved oxygen and perhaps other parameters, and current and potential effects of exotic organisms need to be considered.

3. When I compare the status of OC coho to several other local stocks, I see a likelihood of comparable risk. These include OC winter steelhead, coastal spring Chinook salmon, and Siletz summer steelhead. Some of these may be protected from ESA consideration by the ESU delimitation process; if so, that process may be flawed. I am not suggesting that these all need listed, but it would be good to have a clear rationale for differentiating their status from that of coho.

Comments of Douglas County, Oregon

Comments on the National Marine Fisheries Service's listing endangered and threatened species: Completion of a review of the status of the Oregon Coast evolutionarily significant unit of coho salmon; Proposal to promulgate rule classifying species as threatened

Federal Register May 26, 2010 (Vol. 75, No. 101)

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Ronald Yockim (BA, MA, JD)

Douglas County appreciates this opportunity to provide the most recent information on the Oregon coast coho salmon (*Oncorhynchus kisutch*) with the Umpqua River basin and to provide observations to the National Marine Fisheries Service (NMFS) related to the "Proposal to Promulgate Rule Classifying Species as Threatened" as discussed in the Federal Register notice of May 26, 2010 and the draft report from the Biological Review Team (BRT) entitled "Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon."

This information is being provided to NMFS in response to its proposed rule to continue its classification of the Oregon Coast Coho Salmon Evolutionarily Significant Unit (ESU) as threatened and in response to the proposal to designate the Cow Creek (ODFW stock number 37) coho hatchery program as a threatened species within this ESU. We believe the information we are submitting is of particular importance to the NMFS status review, given that the Umpqua coho populations have been cited in past reviews as a focal point of concern for both the effects of habitat degradation and impacts from a high percentage of hatchery fish in the run. Again in the current BRT review, a concern is highlighted as to a purported decline in habitat condition in the Umpqua Basin.

Douglas County has previously compiled and submitted quantitative information on the four wild coho populations in the Umpqua Basin (See Douglas County 1994, Douglas County 2002, Cramer et al. 2004, Douglas County 2009, Douglas County 2010). The material being provided herein builds on these former submissions with specific focus on the long-term abundance of wild coho in the North Umpqua. This information relies heavily on readily available information maintained by the Oregon Department of Fish and Wildlife (ODFW), including ODFW's Winchester Dam counts of coho that date back to 1946. While the information being submitted focuses on the Umpqua River, the information is also relevant to the BRT's scientific conclusions as to status review for the entire Oregon Coast Coho Salmon ESU (Stout et al. 2010).

As discussed in more detail below, during this 64-year period of records at Winchester Dam, none of the four independent populations within the Umpqua have gone extinct, notwithstanding periods of very low abundance. During the past six decades the Umpqua River basin stratum and overall ESU population numbers have varied considerably. It is of more than passing scientific interest that notwithstanding this high degree of variability, the population has rebounded during recent years to be at or above the highest levels ever observed during this 64-

year period and most significantly the populations are above the historical levels observed in the 1940s[1] (Douglas County 2010).

[**Response:** This comment encompasses two basic points that appear throughout the Douglas County comments: that the North Umpqua population (and by extension the whole ESU) has rebounded from low levels in the past, and that this rebound suggests that the ESU is unlikely to be at risk of extinction in the future. In both its 2010 preliminary report and 2011 final report, the BRT extensively discussed the recent abundance trends of the populations, strata, and ESU as whole, including the fact that ESU-wide spawning abundance in recent years has been higher than any time since the 1950s. Indeed, the BRT noted that the relatively high abundance of the ESU was an aspect of ESU status that has clearly improved since the first status review in the 1990s, and the current level of abundance was not, on its own, a significant risk factor to the ESU.

Response continued: However, the BRT also reviewed historical estimates of the ESU abundance, and noted that even with the recent upswing in spawning abundance the ESU is still only at approximately 10% of its historical abundance. In addition, although spawning abundance is relatively high, total recruitment remains below even the levels seen as recently as the 1970s, indicating that total (preharvest) abundance of the ESU has not in fact rebounded even to the levels observed just a few decades ago. All of these factors were carefully considered by the BRT in its assessment, and are discussed at length in the final BRT report.]

In addition to this quantitative data derived from actual population counts, Douglas County is also submitting comments expressing its continuing scientific concerns relative to the BRT's model assumptions. As we have addressed in prior comments (Cramer et al. 2004), the hypotheses that drive the model and assumptions therein have not been tested with quantified data to determine truth values. We continue to have these concerns over the lack of scientific validation of the model as well as new concerns that have arisen as a result of the additional changes in the model that have not been tested. These untested assumptions and new assumptions into the model have inserted an extremely high bias towards extinction.

Notable among these revisions is the BRT's decision to use peak counts of coho spawners for the metric of spawner density in determining the status of the populations based on critical abundance levels. As discussed in more detail below, this criterion is very sensitive to changes of values within the BRT's Decision Support System (DSS), as well as very sensitive to subjective interpretations as to the biological condition at the population, biogeographic, stratum, and ESU scales (Wainwright et al. 2008, Stout et al. 2010). Given this sensitivity, we are surprised that the BRT did not provide any documentation as to its scientific testing that supports the decision to change the model input data set, particularly given that it utilized the omitted data set in its prior assessments of coho populations in Oregon.

[**Response:** The analysis found in Stout et al. (2010) was not a revision to the DSS because the TRT clearly intended that peak counts be used in this metric: "Metric: This is measured by the average peak spawner density (peak adults per mile of occupied spawning habitat) in the lowest

[1] The 1946 time period wherein the Winchester Dam counts were initiated serves as an appropriate benchmark to compare the most recent 10-year period given that 1946–1947 was the end of a prior warm regime and the commencement of the 1947–1976 cool regime experienced under the Pacific Decadal Oscillation.

3 of the last 12 years." (Wainwright et al. 2008, p. 38). As was discussed in the preliminary BRT report (Stout et al. 2010), Wainwright et al. (2008) inadvertently used AUC data instead of peak counts data in their initial application of the DSS, and this error was discovered when rerunning the DSS for the BRT analysis. For a more detailed discussion, please refer to Wainwright et al. (2008) on pages 38 and 39.

Response continued: Sharr et al. 2000 was a PFMC analysis, not a BRT assessment. The depensation criterion of four fish per mile (fpm) was determined by the TRT based on a variety of modeling and analysis, considering a variety of spawner density metrics. Barrowman et al. (2003) provided support for this criterion from analysis of productivity trends in many Canadian and U.S. river systems. Neville et al. (2006) and Isaak et al. (2007) provided evidence of the fine scale at which spawning salmon interact, supporting the use of peak count data for this metric.]

In addition to our concerns as to the failure of the BRT to adequately scientifically test its hypotheses prior to redefining the model metric relative to the spawner density data input, we are further concerned that this change resulted in a secondary error by failing to account for 25% of the actual peak numbers of coho on the spawning grounds. As a result of the model change, along with this undercounting, the BRT has dramatically biased the results of the model.

[**Response:** The data set used by the BRT was provided by ODFW and was corrected for observer bias.]

Further, we note that notwithstanding the pessimistic outputs of the model, and despite the harsh ocean and freshwater conditions over the last several decades, the most recent population counts for the Umpqua River actually demonstrates a strong resiliency—a resiliency not acknowledged in the current BRT decision. Unfortunately, rather than rely on actual quantitative data, the BRT instead predicts a high risk of extinction based on pessimistic assumptions regarding depensatory mortality and future threats (Stout et al. 2010). As the data presented herein illustrates, the North Umpqua independent coho population has withstood all of the BRT's identified past and current major threats to the viability of the population.[2]

[**Response:** It is not entirely clear which models and methodologies are being questioned here. However, the main points made in these comments seem to be that because the North Umpqua population has not in fact gone extinct, despite numerous risk factors, the results of the DSS and the assessment by the BRT that the population is at risk of extinction in the future must therefore be in error. In fact, the BRT carefully considered the information discussed by the commentator, including the past variability in ESU abundance (including the North Umpqua) and threats such as the high fraction of hatchery fish and loss of high quality habitat. As the commentator points out, the ESU has demonstrated wide swings in abundance in the past in response to varying climate and other factors. As is discussed extensively in both the preliminary and final BRT reports, these wide swings in abundance, combined with ongoing threats such as climate change and habitat degradation, are one of the reasons the BRT was concerned about the viability of the ESU despite the recent upswing in abundance.

[2] It is questionable whether NMFS closely examined the most recent data set relative to environmental risks. For example, it describes the threats relating to sand and gravel operations on the Umpqua River but ignores the information provided by the State of Oregon that these same operations have ceased and are no longer permitted.

Response continued: The commentator also appears to be concerned that the negative persistence and sustainability values of the DSS for the North Umpqua population are not consistent with the relatively high current abundance of the population. However, until very recently (2008) the spawning composition of this population consisted largely of hatchery-origin fish. Consistent with both established NMFS policy (NMFS 2005b) and well-established science (e.g., McElhany et al. 2000, Nickelson 2003, Wainwright et al. 2008, Buhle et al. 2009), the DSS treats high proportions of hatchery fish in a population as a risk factor to the productivity and diversity of the population. As the BRT discussed in its reports (Stout et al. 2010 and this technical memorandum), the recent reductions in hatchery production in the North Umpqua do improve the status of the ESU and this improvement was considered by the BRT in its final assessment. If data on returns from spawners in 2008 and subsequent years with low hatchery impacts indicate that the North Umpqua population is replacing itself without continual input of hatchery fish, the DSS persistence and sustainability scores will increase to reflect the population's improved status.]

This population status is not only very well documented, it is also one that is notable for its sustainability and persistence over the past six decades despite high harvest levels of wild fish, very high percentage of hatchery fish returns mixed with wild spawners, various habitat impacts related to stream complexity on both public and private lands, low critical abundance levels, low marine survival rates, wide variations in stream flows, and other threats.

Notwithstanding that the BRT models produced results indicating that the North Umpqua population should currently be near or at a high risk of extinction, this conclusion is definitely in error, given that the actual production of wild coho currently, as well as in recent years, is near or above the 60-year average. Given the readily available data set for the coho within the Umpqua River system, one would expect that the BRT would have relied on this data set and at a minimum used it to validate the predictive certainty of the model. Rather than validate the model with existing quantitative data, the BRT relied on a model for which it acknowledged had a high uncertainty in its ability to predict a population's persistence and sustainability over the next 100 years. As discussed below, we are concerned that NMFS is unscientifically placing the focus on qualitative and untested assumptions in a model rather than unbiased quantitative population data sets.

Based on the following discussion of the scientific data set, Douglas County is of the opinion that the Oregon Coast coho is best described as an opportunistic species that is not at risk of extinction. The information herein not only validates this opinion, it also provides the proper historical context to demonstrate that the current management policies and conservation measures are and have been effective.

Not only are we concerned that the current BRT assessment does not reflect the true viability risk, as evidenced by the quantitative data that is available for the independent populations, we are also concerned that the BRT has adopted a new and untested qualitative prediction of climatic conditions for the next 100 years that also has a significantly high uncertainty of accuracy. Unfortunately, as with the other models the BRT did not test these predictive climatic models utilizing the long-term data sets that were available. In this case historic climatic records illustrate the coho evolved under a high range of climatic fluctuations, fluctuations which can be expected to occur in the future as well.

[**Response:** The BRT addressed the risks related to climate change using the best available scientific information, including a detailed review of available published, peer-reviewed literature relating to recent and future climate change in the Pacific Northwest and the likely effects of such change on OCCS. The BRT is aware of past and likely future trends and fluctuations in the local climate, and took those trends and fluctuations into account in the analysis. The BRT agrees that there is a great deal of uncertainty surrounding the effects of future climate on OCCS and took that uncertainty into account as a contributing risk factor. Much of the climate analysis does rely on predictive climate models that have been tested against long-term climate data. The BRT did not have the resources to conduct its own assessment of the accuracy of these models, instead relying on a large body of peer-reviewed scientific literature that has reported such assessments.]

We caution NMFS that the BRT report has selectively picked methodologies that support continued listing of the ESU. Further, we caution that notwithstanding these methodology concerns being raised in the past, the BRT continues to incorporate unscientific and untested methodologies into its assessment. Careful scrutiny and testing of the scientific validation of methodologies is warranted prior to any further reliance thereon.[3] This is particularly relevant, given the extensive quantitative data set that is available that does not support the model results.

[**Response:** The BRT utilized the best available scientific information, including information submitted by the commentator. The overall methodology for conducting the status review was the same as NMFS has used for many past salmon status reviews and as such it has received extensive scientific review. The BRT also utilized specific methods and analyses developed by the Oregon/Northern California Coast TRT. The TRT consisted of a range of experts from NMFS, ODFW, USFWS, and independent consultants; the tools and methods it developed reflect that expertise.]

I. Viability of Naturally Spawned Coho Populations

The four independent coho populations in the Umpqua River Basin (Lower Umpqua, Middle Umpqua, South Umpqua, and North Umpqua), have a total of 1,624 miles of spawning habitat. By way of comparison, out of the total number of spawning miles within the Oregon Coast Coho Salmon Evolutionarily Significant Unit, the Umpqua River Basin represents 30% of the available spawning miles (ODFW 2009b).

The spawning habitat within the Umpqua River Basin is comprised of 409 miles in the Lower Umpqua and Smith River (Lower Umpqua), 433 miles in the upper main stem Umpqua including the Elk and Calapooya and other tributaries (Middle Umpqua), 656 miles in the South Umpqua basin including 131 miles in Cow Creek (South Umpqua), and 126 miles in the North Umpqua (North Umpqua). The wide distribution of habitat and spawning populations within the basin serves as an effective built-in protective mechanism against any one catastrophic event resulting in the extinction of the species.

[**Response:** We agree diversity and spatial structure are important factors to consider in evaluating extinction risk, and these factors were explicitly evaluated by the BRT and discussed

[3] For example, if the models are correct, then there should be no set of circumstances which would have produced coho runs in the most recent years that exceed the historic averages.

in their report. In addition, the DSS developed by the TRT uses exactly this type of information in its diversity/spatial structure criteria.]

As discussed in more detail below, based on spawner abundance, density, and distribution within the North Umpqua independent population, the Umpqua Basin stratum, and the Oregon Coast ESU, the Umpqua River populations represent viable populations that are illustrative of coho populations throughout the ESU.

A. Wild coho population viability in the North Umpqua is supported by Winchester Dam counts since 1946

As the BRT correctly noted, the natural spawning (wild) abundance and total (preharvest) adult abundance have increased markedly over the past decade due to a combination of improved ocean conditions, lower harvest rates, and reduced hatchery production. The BRT, however, remained concerned that most of the increase in abundance to the populations could be attributed mostly to increased ocean survival rates, and to a lesser extent to hatchery and harvest recovery actions (Stout et al. 2010).

Since the 10-year window examined by the BRT provides only a very limited data set for describing events within an environment wherein decadal fluctuations of climate occur, we have compiled the ODFW data for the period of 1946 through 2009 as a more representative time frame to illustrate the spawning trends for the North Umpqua wild coho (Table 1).

[**Response:** The BRT report and analysis included the longest time series available. In the current revision we have added a reanalysis of ESU abundance from the 1880s to the present. This analysis suggests a 10-fold decrease in abundance over that time period. As detailed elsewhere, spawning trends in the North Umpqua by themselves are not sufficient to represent the entire ESU.]

We examined the extent of the increase in the wild coho population's production by calculating the estimated total population for the North Umpqua and for the entire Umpqua Basin using the spawner abundance estimates plus the fraction of the population captured in ocean fisheries as estimated by ODFW (ODFW 2010b). This catch plus escapement of adults is the metric commonly applied by fisheries managers to estimate recruitment of adult coho from each brood.

By documenting the population's and stratum's annual production over a longer time frame than the BRT utilized, we found the production numbers strongly demonstrate that despite the combined threats—including naturally fluctuating marine conditions, high risk levels of hatchery fish, and harvest rates that occurred over the past 64 years—the ongoing abundance and resiliency of wild coho over this span of time is indicative of a viable population. When one tests the BRT models against this historical data, it is readily apparent the assumptions in the model are biasing the results toward a high degree of risk.

It is of more than passing interest to note that these same model assumptions were applied in the earlier BRT reviews which also resulted in predictions of a high risk of extinction; yet, subsequent to these earlier reviews, the actual populations have consistently been among the highest on record. For example, the 2009 total population (preharvest) count of more than

Table 1. Total wild coho counts into North Umpqua, ocean harvest impact rates, and estimated total population, 1946–2009.

Return year	Wild coho Winchester count[a]	Harvest impact rate[b]	Wild coho total adult recruits[c]
2009	8,233	0.19	10,164
2008	4,027	0.04	4,195
2007	1,798	0.28	2,497
2006	3,338	0.09	3,668
2005	2,388	0.12	2,714
2004	4,025	0.22	5,160
2003	3,363	0.23	4,368
2002	4,353	0.14	5,062
2001	3,069	0.16	3,654
2000	2,449	0.13	2,815
1999	1,506	0.10	1,673
1998	1,065	0.06	1,133
1997	909	0.12	1,033
1996	1,329	0.14	1,545
1995	1,570	0.22	2,013
1994	1,162	0.02	1,186
1993	1,012	0.42	1,745
1992	1,949	0.51	3,978
1991	1,823	0.45	3,315
1990	414	0.69	1,335
1989	1,798	0.57	4,181
1988	795	0.57	1,849
1987	1,063	0.60	2,658
1986	1,000	0.34	1,515
1985	1,317	0.38	2,124
1984	10	0.27	14
1983	10	0.75	40
1982	1,175	0.58	2,798
1981	1,491	0.78	6,777
1980	335	0.68	1,047
1979	465	0.73	1,722
1978	394	0.79	1,876
1977	578	0.85	3,853
1976	262	0.87	2,015
1975	529	0.76	2,204
1973	568	0.78	2,582
1972	407	0.80	2,035
1971	638	0.77	2,774
1970	204	0.61	523
1969	563	0.67	1,706
1968	1,647	0.73	6,100

Table 1 continued. Total wild coho counts into North Umpqua, ocean harvest impact rates, and estimated total population, 1946–2009.

Return year	Wild coho Winchester count[a]	Harvest impact rate[b]	Wild coho total adult recruits[c]
1967	1,295	0.70	4,317
1966	917	0.60	2,293
1965	2,262	0.65	6,463
1964	1,166	0.67	3,533
1963	1,227	0.73	4,544
1962	548	0.57	1,274
1961	531	0.59	1,295
1960	346	0.57	805
1959	818	0.55	1,818
1958	573	0.72	2,046
1957	1,063	0.65	3,037
1956	2,760	0.66	8,118
1955	2,697	0.64	7,492
1954	389	0.64	1,081
1953	2,356	0.69	7,600
1952	3,066	0.45	5,575
1951	2,259	0.63	6,105
1950	1,375	0.62	3,618
1949	1,412	0.63	3,816
1948	790	0.63	2,135
1947	1,038	0.63	2,805
1946	1,438	0.63	3,886

[a] Jack counts are included in total. Many hatchery fish regenerated their fin marks in 1982, resulting in inaccurate (high) wild fish estimates. We caution reliance on the 1983 and 1984 populations in that the estimating techniques utilized do not accurately predict counts this low. All counts since 1991 reflect actual counts utilizing a video camera. Source of information is from ODFW records for the Winchester Dam on North Umpqua.
[b] Estimated harvest impacts to naturally produced coho in the Oregon Production Index Area. From Table III-2 in the 2009 Preseason Report I, Pacific Fishery Management Council, Portland, OR.
[c] Recruits = Winchester Dam Count / (1 – harvest rate).

10,000 wild coho is the highest on record since 1946. This compares to the record low numbers in 1983 and 1984, during which time periods less than 40 wild coho were estimated in the entire North Umpqua population.[4]

We selected the North Umpqua population as our reference for comparison based on: 1) the wild coho data for that basin are from Winchester Dam counts that are the actual fish counts of the total number of wild fish that crossed the dam since 1991 and the verified total counts

[4] It is of more than passing interest that the population did not go extinct after reaching these low levels and in fact greatly expanded from less than 40 fish to more than 10,164 roughly 25 years later. This data indicates that neither the depensation nor the extinction levels are reached at the 40 fish over the dam level. Notably even at this low number, the coho still demonstrated a strong resiliency and ability to rebound relatively quickly.

since 1946, and 2) in prior and current analyses by NMFS this independent population was considered to be at the higher risk of extinction (Stout et al. 2010).

The following analysis also takes into consideration the annual variation in ocean harvest rates and reflects the current management policies and practices. It provides the best available scientific data set for comparing accurate population numbers over the long term (60 years) (Figure 1).

The annual fish population abundance trend over the last six decades is graphically displayed in Figure 1. It is important to note that this Winchester Dam data represent the total estimates of actual wild coho for the North Umpqua independent population. The North Umpqua population abundance records are not based on expansions or assumptions of fish population status using freshwater habitat condition indices (i.e., stream complexity) or ocean condition indices; rather these counts are the product of a very consistent monitoring method of returning coho at the same site (Winchester Dam) and are verified as being very accurate population counts dating back to 1946 (ODFW unpublished files).

[**Response:** The commentator seems to have misunderstood what data were used by the BRT. The BRT did in fact use the full record of Winchester Dam Counts as provided by ODFW.]

By way of comparison, the indices included in the BRT risk assessment have a higher statistical uncertainty than the actual counts derived from the Winchester Dam.

Further uncertainty around the BRT risk assessment is created by the BRT's DSS model reliance on the untested assumptions that there are valid and sensitive (certainty) relationships for determining actual fish population productivity and sustainability.

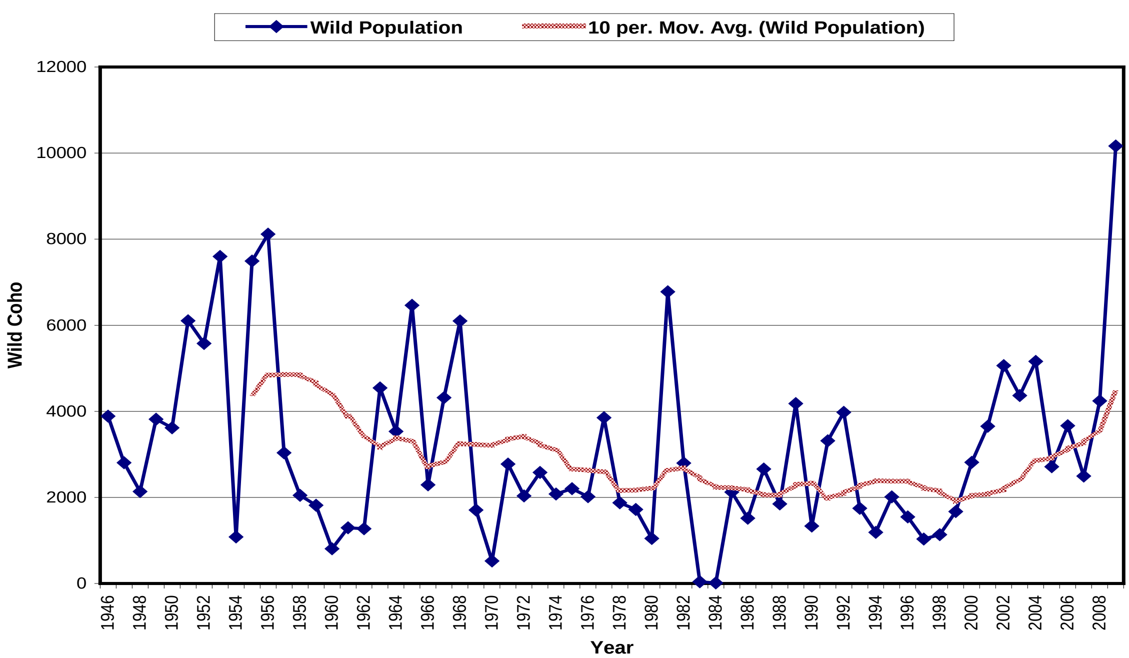

Figure 1. North Umpqua wild coho population, 1946–2009.

While the BRT recognized the limitations in its qualitative assumptions (Stout et al. 2010), it did not take the necessary steps to scientifically validate the original or revised assumptions. Further, it is unfortunate that the BRT erroneously assumed that the spawning surveys represented the total counts of fish at a 1-mile habitat reach, population, stratum, or ESU level (1,000s of miles) when it changed the spawner density metric. It simply overlooked that these counts are not 100% estimates of every fish. While we agree that the spawner density metric is an appropriate metric, we point out that the failure to utilize the best available science relative to the spawner density data and verified population production that is available on the North Umpqua River system resulted in unscientific assumptions in the metric.

We also note that viability models for predicting fisheries' responses are in relatively early stages of development, and as a result, the level of uncertainty in the projections remains high, especially when projecting out over very long time periods (i.e., 100 years) (IMST 1999).

[**Response:** The BRT does not disagree that there is significant uncertainty in long-term projections. This is why the BRT considered many aspects of Oregon Coast coho salmon ecology in assessing status and used a variety of information (population viability modeling, the TRT's DSS, habitat assessments, climate assessments, assessment of other threats) in conducting its assessment. The BRT also was careful to characterize the degree of certainty of its conclusions and this was extensively discussed in both its preliminary and final reports.]

While the North Umpqua population production (preharvest estimates) over the last six decades covers a period wherein there has been large variation of marine survival rates, it is notable that this independent population has increased significantly in recent years (10-year moving average trend line in Figure 1), and in fact the population is at or above the highest abundance levels documented over the 64-year period of data.

This scientifically derived quantification of the population's status alone strongly supports our assessment that the population is sustainable and persistent and has been so over a long-term span—a span that included all of the threats that the BRT used in its viability assessment for the independent populations and the ESU.

The observations of actual fish returns over the long term not only provides an indicator of the actual health status of a population, it also provides a critical monitoring tool that can be used to verify the scientific confidence level to be attributed to the qualitative model. As the Independent Multidisciplinary Science Team (IMST) cautioned, if model performance is of questionable scientific validity or reliability as is the case herein, then key indicators (e.g., observations of actual fish returns) should be used to supplement model estimates or replace the models (IMST 1999).

Another example wherein data was readily available to allow reasonable predictions yet was only selectively utilized is illustrated by the marine survival rates. While Douglas County agrees that marine survival rates are an important risk factor to include in this assessment and are especially critical during periods of low productively, we continue to have concerns over the manner in which this metric was applied in the model. As we expressed in prior comments relative to shifts in ocean conditions (Douglas County 1994), the BRT has made several key assumptions about future marine conditions that are not consistent with the known variability in ocean conditions. This is a particularly glaring omission, given that the BRT adopted these assumptions without taking the necessary steps to validate these assumptions against historic data on marine conditions. Key information was available on which the BRT could have assessed the marine conditions in both intra-annual and inter-decadal time frames. The recurring pattern of climatic variability is now well documented and accepted. As illustrated in Figure 2, this pattern provides evidence of “reversals” of prevailing polarity and has distinct patterns of variation at intra-annual to inter-decadal time scales (Mantua et al. 1997).

Research in the 1990s demonstrated that marine ecosystem productivity is strongly correlated with the climatic regime changes brought about by the Pacific Decadal Oscillation (PDO), especially in the Northeast Pacific. This quantitative data supports the conclusion that in the future it is very likely that the PDO will continue to change polarity every few decades, as it has over the past century, and with it the abundance of salmon species (Mantua et al. 1997).

The warm regime of the 1900 to 1946 time period is represented by the above [zero] line … areas. Similarly, the cool regime of the 1946 to 1977 period is represented by the … [areas below the zero line]. This shift in regimes is also depicted on the follow National Fisheries Science Center illustration.

The PDO index (Figure 3), illustrates a series of shifts from warm regimes … [bars above the zero line] of the 1925 to 1946 time period and the 1977 to 1998 time period back to cold regimes … [bars below the zero line] of the 1947 to 1976 time periods. Mantua related these regimes to adult salmon catches in the Northeast Pacific and demonstrated that these two events were correlated with the PDO. He found that major changes in the phase of the PDO have resulted in corresponding shifts in ocean biological productivity; warm eras have seen enhanced coastal ocean productivity of salmon in Alaska with corresponding inhibited productivity off the West Coast of the contiguous United States. Similarly, cold eras have resulted in relatively high salmon production in California, Oregon, and Washington and low salmon production in Alaska.

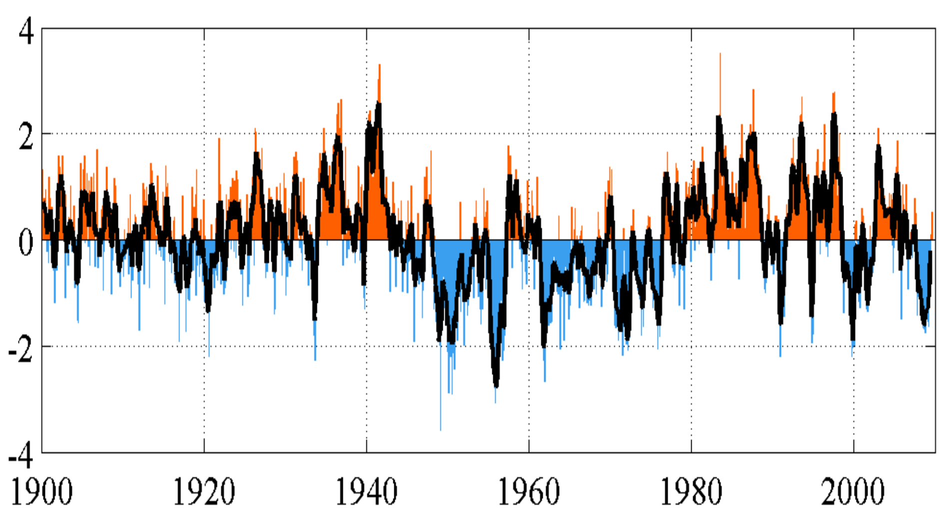

Figure 2. Monthly values for the PDO index: 1990–September 2009. Source: Mantua 2000, updated to 2009 with online monthly values.

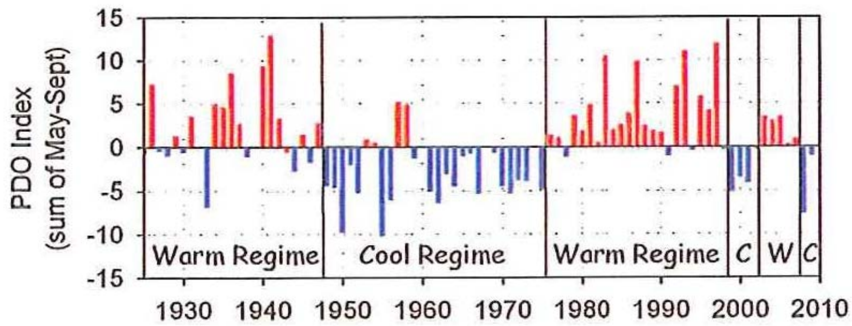

Figure 3. Pacific Decadal Oscillation warm-cold regimes.

The recent 1998 flip in the PDO is widely considered responsible for the current cooling ocean environment and upsurge in salmon runs.[5]

This shift in fisheries responses between the Alaska fishing zones and the Oregon-Washington regions is evidenced by the following excerpts from fishing reports of the time:

Pacific Fisherman Sept. 1915

> "Never before have the Bristol Bay [Alaska] salmon packers returned to port after the season's operations so early."
>
> "The spring [Chinook salmon] fishing season on the Columbia River [Washington and Oregon] closed at noon on August 25 and provided to be one of the best for some years."

Pacific Fisherman 1939

> "The Bristol Bay [Alaska] Red [sockeye salmon] run was regarded as the greatest in history."
>
> "The [May, June and July Chinook] catch this year is one of the lowest in the history of the Columbia [Washington and Oregon]."

National Fisherman 1972

> "Bristol Bay [Alaska] salmon run a disaster."
>
> "Gillnetters in the Lower Columbia [Washington and Oregon] received an unexpected bonus when the largest run of spring Chinook since counting began in 1938 entered the river."

Pacific Fishing 1995

> "Alaska set a new record for its salmon harvest in 1994, breaking the record set the year before."

[5] While there is reliable hindsight on which to assess climatic events, climate is highly variable and attempts to predict future events is fraught with high degree of error—particularly the further into the future one attempts to predict. The PDO is very valuable, however, in demonstrating the wide variations in climate experienced as well as to place into context the responses we experience today.

“Columbia [Washington and Oregon] spring Chinook fishery shut down; west coast troll coho fishing banned.”

These early reports illustrate the temporal variations in fisheries productivity that track the robust, recurring pattern of ocean-atmospheric variability of the PDO. Similar to the historic reports above, the coho populations from the Winchester Dam counts (Table 1) also dramatically illustrate this same phenomenon.

By overlaying the PDO cycles (Figure 3) on top of the graph depicting the 10-year moving average (Figure 1 and Figure 6), a similar variation in productivity is observed (Figure 4). Note the productivity is depicted as a 10-year average which blurs the regime changes but nonetheless depicts the overall shift between cold and warm regimes on the 30–50 year cycles.

When one looks further back into the historic record, one finds similar climatic shifts dating back to 1735 in Oregon. The USGS study of tree rings covering the period of 1734 through 1930 described similar oscillations as measured by the percentage deviation in normal precipitation in the Harney Basin of eastern Oregon [Figure 5].

The USGS report depicts the percentage deviation from normal precipitation as being a positive above the zero axis and a negative deviation below the zero axis. The positive deviations correspond to the cool regimes while the negative deviations correspond to warm regimes described by Mantua et al.

While the BRT acknowledged its uncertainty in its predictive model for quantifying risks from marine conditions for the foreseeable future, given global climatic change information, it

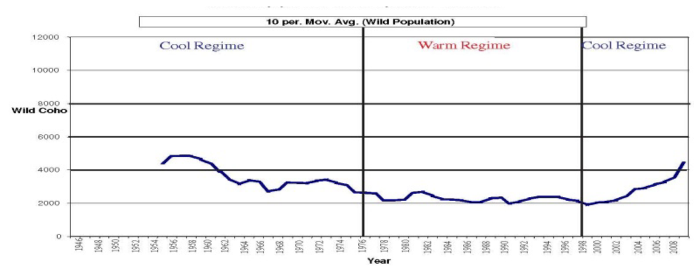

Figure 4. [North Umpqua wild coho population 1946–2009.]

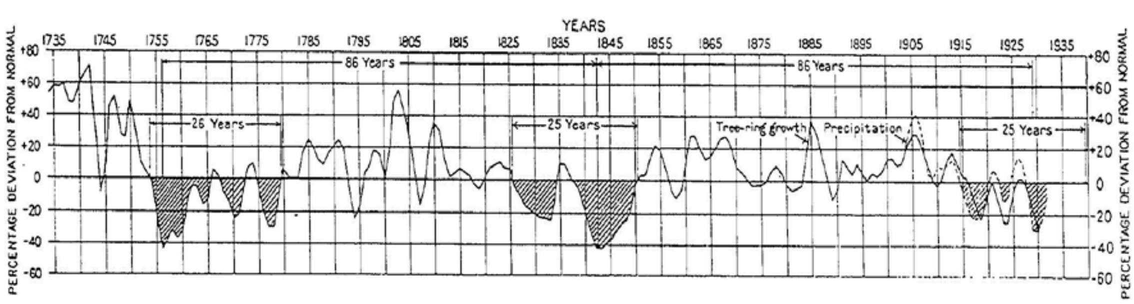

Figure 5. Geology and groundwater resources of the Harney Basin, Oregon, Geological Survey Water Supply Paper 841, U.S. Department of the Interior, 1939, p. 13.

nonetheless adopted a pessimistic strategy in the face of this uncertainty (Stout et al. 2010). It did not, however, test its predictive model against the available historic records. This is a particularly glaring omission, given the BRT's acknowledgment that it is relatively new to developing models for global warming and its failure to examine the coho in the context of the historic PDO events.

[**Response:** Douglas County does not specify what particular "key assumptions about future marine conditions" they question, so it is difficult to respond to this comment. However, any assumptions made by the BRT are consistent with the scientific literature regarding marine survival of coho salmon. The BRT agrees that fluctuations in marine conditions (including the PDO and other factors) strongly affect survival of OCCS, and has accounted for such fluctuations in its analyses.]

In examining the 64-year record of North Umpqua coho population production in the context of the PDO, one finds a more reliable tool for quantifying risk of extinction from climatic events. These quantitative data sets provide long-term historical evidence that the coho salmon species (independent populations and ESU-level) has evolved under many diverse and extreme habitat conditions, within both the marine and freshwater environments. These climatic extremes have resulted in the coho evolving with a genetic "natural climate insurance" that has been formed through many years of climatic and environmental conditions and cycles. This ability to survive within these climatic extremes is clearly demonstrated in the populations that are present today within the ESU.

Given this historical record, one would anticipate a qualitative prediction for the coho ESU over the next 100 years that is more likely optimistic, or at a minimum similar to the past century rather than the pessimistic approach of the BRT. One must assume that there will continue to be variability in season-to-season and year-to-year climatic rhythms, that there will be continued environmental uncertainty similar to that under which the species evolved, and that the range of variability in the future will be similar to that encountered in the past. Speculation as to future climate change may help planners prepare for all conceivable threats, however, this speculation does not satisfy the requirement that NMFS use best available scientific data in its determinations. When one examines the coho history in the climatic context, the pessimistic prediction of marine survival of coho in the foreseeable future is simply not warranted by the historical record.

The tie between salmon productivity and the PDO not only illustrates the widespread variation on which the coho have evolved and survived, it also demonstrates the error in not examining global warming predictions in the context of the greater historic record. Unfortunately, the BRT took a prospective model without validating the projected environment in the context of available historic records. Inclusion of speculative futures in the model enables NMFS to direct the model outcomes and therefore bias the results.

When one examines the variability in coho populations for the period of 1946 to the present in the context of the historical records of the PDO, one clearly becomes cognizant of the resilience of this species in the face of a highly variable climatic record. Further, one also readily becomes aware that the fluctuations experienced since the 1990s are not out of the realm of conditions experienced in the past—nor are they out of the realm to be expected to continue into the future.

B. Relationship of North Umpqua wild coho population compared to other independent populations in the Umpqua stratum and the OCCS ESU

In examining the coho spawner abundance estimates for the 1990–2009 time period (ODFW 2010b), one finds a similar annual variation in the abundance of wild coho returns irrespective of whether it is the North Umpqua or the entire Umpqua River basin. This variation allows us to examine the known population of the North Umpqua and make predictions for the entire basin. For example, North Umpqua spawner returns ranged from about 700 fish in 1997 to 9,600 fish in 2009, with a 20-year average of 2,700 spawners and a median of 2,200 returns (Table 2). By way of comparison, the Umpqua Basin total spawner returns ranged from 3,400 in 1997 to 65,000 in 2009, with a 20-year average of 21,100 spawners and a median of 16,300 returns.[6]

One can readily assess the trend in spawner abundance for the North Umpqua, then extrapolate these numbers to the four independent Umpqua Basin populations (Umpqua stratum) to arrive at an accurate indication of annual production of wild coho for the entire stratum as well as the ESU.

[**Response:** The BRT is required to utilize the best available scientific information in conducting its assessment. To focus exclusively on dam counts for one population as an index of abundance for the entire ESU would be to ignore a vast amount of useful information collected from elsewhere in the ESU. In addition, the importance of spatial structure and diversity to ESU viability is well documented in the scientific literature (e.g., see review by McElhany et al. 2000). Evaluating an entire ESU from dam counts for a single population would ignore important differences among populations. It would essentially restrict our analysis to a small amount of information, ignoring the vast majority of the information available to NMFS. It also does not take into account that the habitat in the North Umpqua population is not typical of the rest of the ESU, nor does it reflect the diversity of other habitats found in the ESU.]

In addition to the coho salmon population estimates set forth above, we were able to determine estimates of the annual wild coho spawner abundance for each of the four independent populations in the Umpqua (Table 3). The abundance values in Table 3 represent estimates for three of the populations and actual fish counts for the North Umpqua during the period of 1991 to the present.

While there has been considerable variation in spawner returns over the past 20 years, in most cases the annual returns for the four independent populations track with each other, therefore indicating that the marine survival and ocean harvest rates were uniform influences on each population for that annual cohort.

[6] While ESU harvest impacts have varied significantly within these time periods, for this analysis we assumed that the individual harvest rate for the four populations was relatively equal. This assumption is appropriate given that most of the wild coho from the Umpqua system are harvested in the ocean fisheries and estuary where the four independent wild coho populations are mixed in the ocean, resulting in the harvest is spread across all four populations (Melcher 2005, ODFW 2009b).

Table 2. Coho population estimates for North Umpqua River and Umpqua River basin, based on adult spawner abundance and ocean harvest rates for 1990–2009. Total population estimate calculated from spawner abundance divided by (1 − ocean harvest rate %).

Return year	OCN ocean harvest (%)[a]	Spawner abundance	Total population estimate[b]	Spawner abundance	Total population estimate[b]
		North Umpqua		Umpqua Basin	
2009[c]	19	7,813	9,646	52,685	65,043
2008	4	3,438	3,581	32,306	33,652
2007	28	1,410	1,958	11,783	16,365
2006	9	3,062	3,365	18,154	19,950
2005	12	2,113	2,401	42,676	48,496
2004	22	3,705	4,750	31,346	40,187
2003	23	3,005	3,903	29,607	38,451
2002	14	3,780	4,395	37,591	43,710
2001	16	2,951	3,513	35,702	42,502
2000	13	1,838	2,113	12,233	14,061
1999	10	1,186	1,318	7,685	8,539
1998	6	727	773	9,153	9,737
1997	12	727	826	2,960	3,364
1996	14	1,075	1,250	10,824	12,586
1995	22	1,460	1,872	12,809	16,421
1994	2	851	868	5,336	5,445
1993	42	933	1,609	10,224	17,627
1992	51	1,607	3,280	3,759	7,671
1991	45	1,273	2,315	4,873	8,860
1990	69	376	1,213	4,113	13,268
Average 1990–2009	22	2,167	2,747	18,791	21,100
Median 1990–2009	15	1,533	2,214	12,008	16,325

[a] Updated harvest rate impact rates estimates and total population numbers for previous years. ODFW and PFMC data.
[b] Total estimated number of naturally produced coho prior to harvest.
[c] Preliminary estimated coho spawner abundance and harvest impact rate for 2009.

Based on these spawner abundance estimates, the four independent populations in the Umpqua demonstrate an increasing trend in production during the most recent years. Notably, the data also illustrate that in 11 of the last 12 years the total spawner abundance has been in excess of the 20-year median—a result that tracks well with the projected PDO influences discussed earlier.

When one compares the coho spawner abundance in the Umpqua River basin with the entire ESU, one finds that on average, the Umpqua Basin represents about 18% of the total wild coho spawner returns to the ESU (Table 4). For example, in 16 of the last 20 years (1990–2009) the percent returns for the Umpqua stratum represented 13% to 21% of the entire ESU. In most years the Umpqua River spawner abundance is a very good indicator of annual production and trends for the entire ESU.

Table 3. Annual wild coho spawner abundance estimates in Umpqua River basin, 1990–2009.

Return year	Lower Umpqua	Middle Umpqua	South Umpqua	North Umpqua	Total Umpqua
2009*	16,670	13,346	14,856	7,813	52,685
2008	12,267	4,594	12,007	3,438	32,306
2007	4,237	1,587	4,549	1,410	11,783
2006	7,994	4,852	2,246	3,062	18,154
2005	18,591	7,608	14,364	2,113	42,676
2004	8,046	7,911	11,684	3,705	31,346
2003	12,760	10,220	3,622	3,005	29,607
2002	14,492	10,904	8,415	3,780	37,591
2001	8,850	10,758	13,143	2,951	35,702
2000	3,696	4,638	2,061	1,838	12,233
1999	2,323	1,723	2,453	1,186	7,685
1998	5,118	823	2,485	727	9,153
1997	935	593	705	727	2,960
1996	4,904	2,048	2,797	1,075	10,824
1995	6,803	2,667	1,879	1,460	12,809
1994	1,689	1,948	848	851	5,336
1993	4,804	1,431	3,076	933	10,244
1992	1,769	192	201	1,607	3,769
1991	1,316	NA	2,284	1,273	4,873
1990	589	640	2,508	376	4,113
1990–2009 average	6,893	4,657	5,309	2,167	18,792
1990–2009 median	5,041	2,667	2,653	1,534	12,008

*Preliminary estimated coho spawner abundance. North Umpqua includes Winchester Dam counts (ODFW 2010a).

Comparing the annual spawner returns of the North Umpqua with the returns for the total ESU (Table 3), we found a close relationship which strongly supports the use of the North Umpqua data as a good index of the spawners within the total ESU (ODFW 2010b) (Figure 6).

The North Umpqua spawner abundance demonstrates that the four independent Umpqua River populations, as well as the ESU, continue to be viable over a long time span. In fact the data set illustrates that they are not only viable, but have significantly increased in recent years to a point wherein they are now near or above the historic levels that have occurred over the past 60 years (Figure 6).

II. Model Results Do Not Reflect Abundance or Persistence of Populations in the ESU

A. Spawner density levels

Recent years of high spawner abundance have validated Douglas County's previous comments that the NMFS models unduly overestimated the risk of extinction. As we discussed in earlier submissions, a continuing error in the BRT's model is the use of spawner density as one of the critical abundance criterion utilized to determine the persistence truth value for a population.

Table 4. Annual estimates of wild coho spawner abundance in Umpqua River basin compared to Oregon Coast coho ESU, 1990–2009.

Return year	Umpqua Basin	Oregon ESU	% of ESU
2009*	52,685	230,458	23%
2008	32,306	165,324	20%
2007	11,783	66,169	18%
2006	18,154	128,838	14%
2005	42,676	154,131	28%
2004	31,346	175,380	18%
2003	29,607	228,681	13%
2002	37,591	260,570	14%
2001	35,702	172,617	21%
2000	12,233	70,110	17%
1999	7,685	49,128	16%
1998	9,153	32,004	29%
1997	2,960	23,398	13%
1996	10,824	74,021	15%
1995	12,809	53,979	24%
1994	5,336	44,169	12%
1993	10,244	55,344	19%
1992	3,759	42,197	9%
1991	4,873	37,602	13%
1990	4,113	21,279	19%
1990–2009 average	18,792	104,270	18%

*Preliminary estimate of coho spawner abundance (ODFW 2010a).

A demonstration of the lack of scientific reliability of the model's reliance on this element is illustrated by the wild coho spawners in both the North Umpqua as well as the overall Umpqua stratum. ODFW surveys demonstrate that wild coho spawners have increased significantly over the past 10 years in relation to the total fish returns within the spawning areas since 1990 (Table 5).[7]

To calculate the spawner density, we used the available data relative to total spawner returns divided by the number of spawner miles[8] over the selected time period. Given the long history of spawner returns on the North Umpqua, we selected the wild coho spawner return data as the data set that represents the most accurate record of spawner density in either the Umpqua stratum or the larger ESU (Figure 7).

[7] Significantly, during this period, there has not been a large decrease in the available spawning miles. While restoration projects have definitely increased fish passage and access to spawning stream miles in the basin, the number of these additional miles in relation to the overall total miles represents a proportionately small number of miles. Given that the increase in access is limited in comparison to the overall miles, the observed increase in spawner density is most directly related to increased spawner abundance levels basin wide, not a change in the number of spawner miles.

[8] In the North Umpqua, there are currently approximately 126 miles of coho spawning habitat.

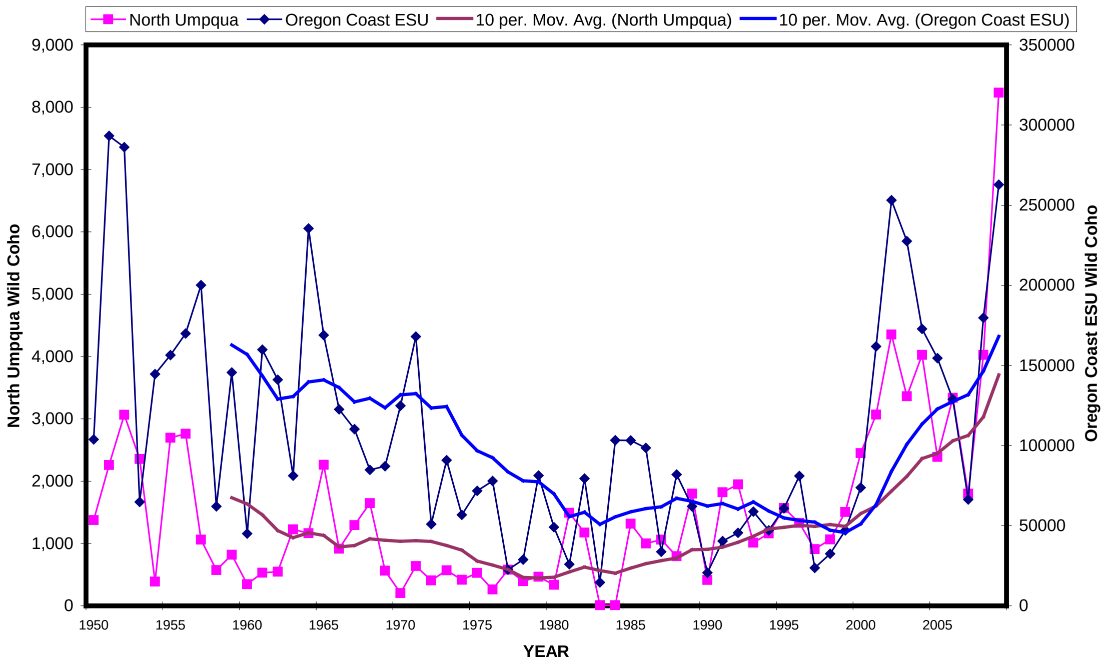

Figure 6. Numbers of wild coho returns to the North Umpqua and Oregon Coast ESU, 1950–2009. [Top line without symbols is Oregon Coast ESU; bottom line without symbols is North Umpqua.]

In Table 6, we summarize the wild coho returns in the North Umpqua segregated by decades in order to provide the history of spawner abundance since records were first maintained at Winchester Dam (1946). This data set includes wild spawner numbers and total spawner numbers (wild and hatchery).

We divided the 10-year spawner average by the number of miles (126) in the basin to provide a data set for spawner density (Table 6). We utilized six decades since this closely reflects the three cycles of three generations of coho (9 years) typically utilized to assess spawner density.

In earlier BRT reports, as well as in other publications (i.e., Sharr et al. 2000 and Wainwright et al. 2008), the spawner density of four fish per mile has been utilized as the baseline in viability assessments as representing the minimum threshold wherein depensation problems based on low spawner abundance in the population can be avoided. Consistent with this approach, the current DSS model also includes the criteria of the average of the lowest 3 years of spawner density over the past 12 years to address the persistence prediction.

Based on our analysis, the spawner density for the North Umpqua independent population over the 10-year period from 1970 to 1979 averaged 4 wild fish per mile.[9] While at this density one would expect the population to face depensation problems, in fact, following this 1970–1979 time period we experienced two decades (1980s and 1990s) wherein the spawner density of wild and total coho was well above the depensation threshold level—in large part due

[9] In our analysis, we did not count hatchery fish in determining the spawner density or critical abundance.

Table 5. Annual wild coho spawners per mile in Umpqua River basin, 1990–2009.

Return year	Lower Umpqua	Middle Umpqua	South Umpqua	North Umpqua	Total Umpqua
2009*	41	31	23	62	32
2008	30	11	18	27	20
2007	10	4	7	11	7
2006	20	11	3	24	11
2005	45	18	22	17	26
2004	20	18	18	29	19
2003	31	24	6	24	18
2002	35	25	13	30	23
2001	22	25	20	23	22
2000	9	11	3	15	8
1999	6	4	4	9	5
1998	13	2	4	6	6
1997	2	1	1	6	2
1996	12	5	4	9	7
1995	17	6	3	12	8
1994	4	4	1	7	3
1993	12	3	5	7	6
1992	4	1	1	13	2
1991	3	NA	3	10	3
1990	1	1	4	3	3
1990-2009 average	17	11	8	17	12
1990-2009 median	12	6	4	12	7

* Preliminary estimated coho spawner abundance, (ODFW 2010a).

to the high abundance of hatchery fish returns to the Umpqua Basin (Figure 8 and Figure 9) until the elimination of the hatchery program on all but Cow Creek (located on the South Umpqua). This rebound in spawner density over the depensation level of four fish per mile, illustrates that the North Umpqua population does not have a long-term persistence problem that can be related to low spawner densities.

[**Response:** The spawner density levels cited were greatly influenced by hatchery returns, which make it impossible to assess the response of the wild population to low abundance events. Even if that were not the case, it would be inaccurate to conclude that because a population recovered from low abundance in the past that it will recover in the future under different conditions. Depensation includes random effects that do not necessarily happen every time low abundance occurs and recovery from one low abundance event does not necessarily mean that the population will recover from the next low abundance event. That is why the TRT included a criterion that assesses low spawner density, as it presents a risk to the recovery of the population.]

Notwithstanding the available data set illustrating a sustainable population, the BRT concluded that there was a high uncertainty of sustainability based on its interpretation of the modeled DSS scores.

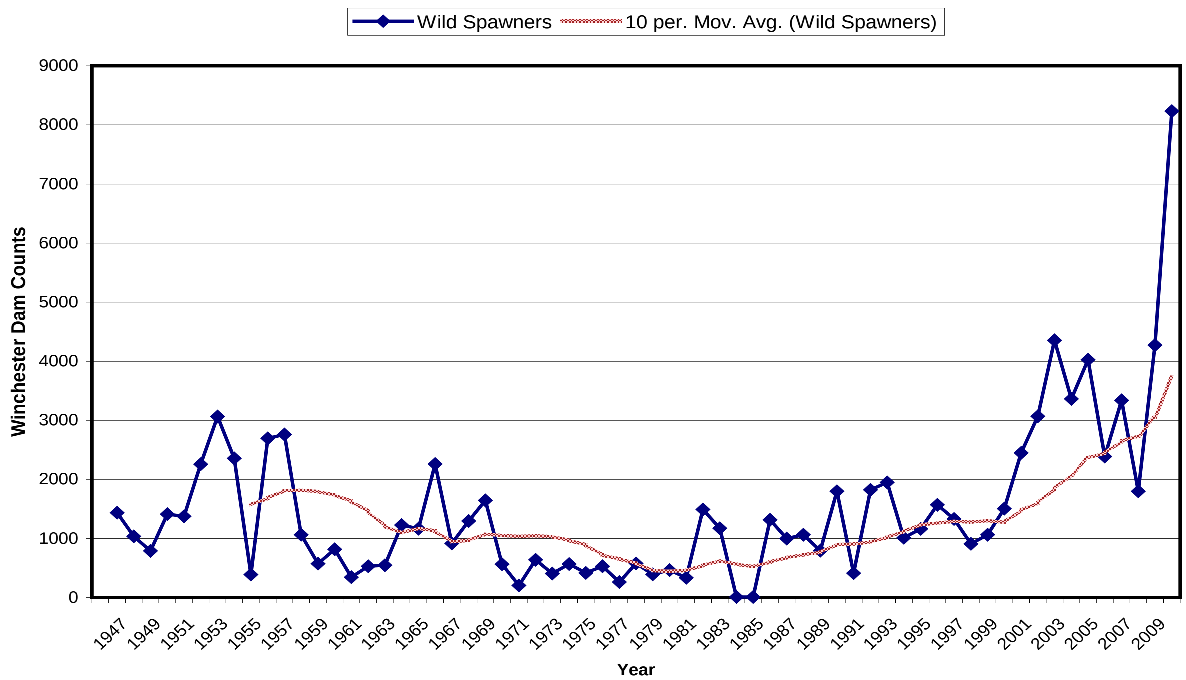

Figure 7. Wild coho returns to the North Umpqua, 1946–2009.

Table 6. Average and median numbers and spawner density of wild and total coho counted at Winchester Dam, North Umpqua, 1946–2009, by decade. Source: ODFW unpublished data.

	Average wild count	**Median wild count**	**Average wild density**	**Median wild density**	**Average total density[a]**	**Median total density[a]**
1946–1949	1,170	1,225	9	10	9	10
1950–1959	1,736	1,817	14	14	14	14
1960–1969	1,050	1,042	8	8	8	8
1970–1979	446	441	4	4	4	4
1980–1989	899	1,032	7	8	29	23
1990–1999	1,274	1,246	10	10	41	41
2000–2009	3,709	3,351	29	27	99	106
64-year average	1,487	1,164	12	9	31	31

[a] Total wild and hatchery returns divided by 126 spawner miles.

B. Model results do not reflect actual production

While the BRT was concerned its population functionality criterion was strongly influenced by basin size since the four populations in the large Umpqua basin scored at the highest positive level under the model input,[10] rather than rely on this result or validate the model or otherwise investigate this result, the BRT simply eliminated this criterion from the risk assessment and chose to use other habitat-based metrics. Unfortunately, the BRT did not

[10] Which is not surprising given the actual spawning counts and total number of coho present in the basin.

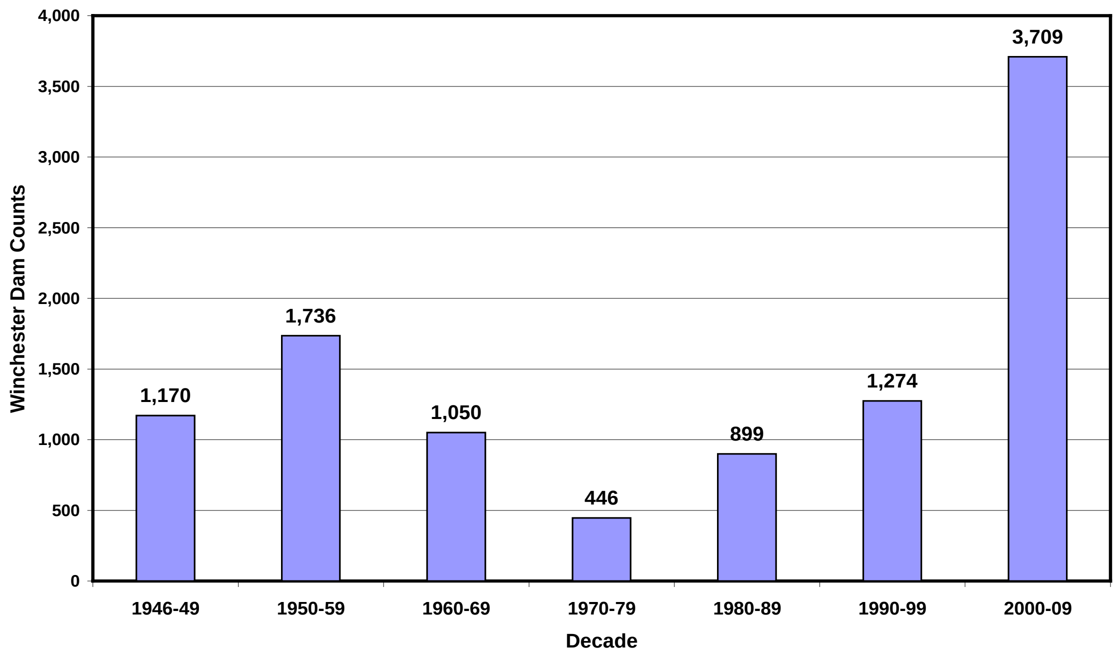

Figure 8. North Umpqua wild coho spawners average by decade, 1946–2009.

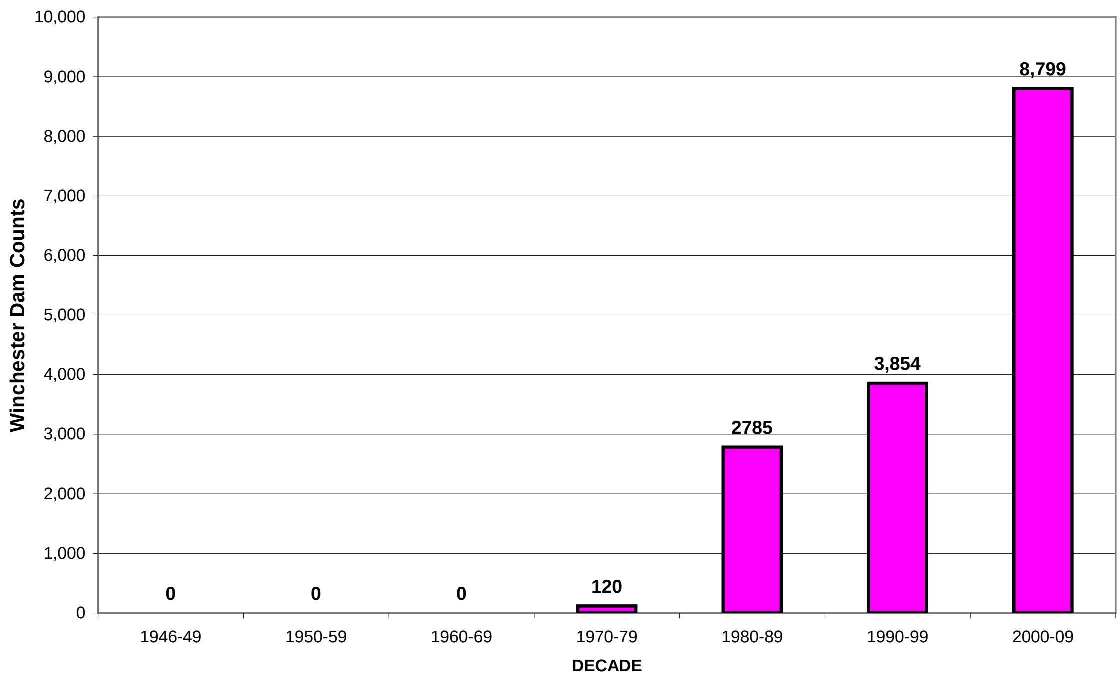

Figure 9. North Umpqua hatchery coho spawners average by decade, 1946–2009.[11]

[11] Hatchery releases on the North Umpqua were terminated in 2006.

validate the original model nor did it validate its modification of the model.[12] This was surprising, given that under the BRT assessment, the North Umpqua, as the smallest of the four subbasins within the ESU, would have been expected to have resulted in one of the most accurate functionality scores.

[**Response:** This appears to be a misunderstanding of the report. The BRT included the population functionality criterion in the DSS. We discussed the need for reconsideration of this criterion in the future. In addition, the BRT did not rely only on the DSS in our deliberations, but rather included other factors and sources of information as well.]

This failure to accept its own model results is illustrative of the bias that the BRT utilized throughout its modeling. Rather than accept the model output, the BRT simply changed the model. Changing the model resulted in DSS scores for habitat functionality that not only differed significantly from actual habitat utilization, but also produced a biased appearance that the habitat status and trend for the most part are impaired and decreasing (Stout et al. 2010).

It is the BRT's selectivity in the use of data that results in the counterintuitive DSS model results wherein the functionality of habitat within the relatively small North Umpqua basin is defined by the model as decreasing while at the same time fish productivity has actually been increasing.

We note that the DSS model scores for persistence and sustainability were –0.94 and –1.00 respectively for the North Umpqua. These negative scores are descriptive of a wild coho population that has a very high certainty of being in very poor shape and one that is in danger of extinction in the near future.

The DSS characterizes a population with these scores as one where:

- abundance is low and declining,
- distribution of juveniles and spawners is very limited,
- genetic integrity is severely compromised,
- widespread destruction/alteration that create and maintain habitat has occurred,
- heroic measures are required for successful restoration,
- no opportunities for harvest, and
- hatchery programs should be restricted to captive breeding activities only.

Under the DSS scoring, one would therefore expect the numbers and productivity of the North Umpqua wild coho population to be dismal and at the lowest levels in history. Yet in fact, the opposite is true for the actual numbers of wild coho—and the persistence of the independent population as reflected in actual population counts over the last 60 years—simply do not reflect this result. The actual quantified data set for the North Umpqua reveals wild fish production in

[12] The scientific approach would have been to truth test the model to either validate the result or delete the criterion as unscientific; in this case the BRT did neither, opting instead to simply eliminate the criterion since it did not produce the result it desired.

the North Umpqua that is neither impaired nor reflecting a declining trend. The long-term abundance data set actually demonstrates the population has a high certainty of persistence and sustainability, is in very good shape, and is at a very low risk of extinction.

The BRT's negative scores for current viability status of the North Umpqua population is not merely counterintuitive, it simply does not reflect what is truly occurring in the North Umpqua population.

The lack of correlation between the actual quantified data (population numbers and spawning counts) and the model's output reinforces our earlier concerns over the lack of reliability in the model. We continue to request that the BRT reexamine its model utilizing recognized scientific analyses and data—including truth testing. During the prior 2005 comment period, we and the ODFW raised similar concerns over the model's results differing significantly from actual data. The existing data set for the North Umpqua not only demonstrates a very high certainty of a viable population, it also provides an excellent tool for validating the model.

C. The BRT arbitrarily changed population assessment model metric for spawner density

In spite of the lack of evidence that depensation (reduced survival at low population size) actually plays a key role in the population dynamics of coho salmon, the BRT continues to include depensation in its model, even while acknowledging that the model predictions are sensitive to the assumed depensation. Rather than removing or discounting the effects of depensation from the assessment, the BRT has amplified the effect that depensation has on coho. The manner in which depensation is applied is contrary to the best available science. Cramer et al. (2005) found that evidence supplied to NMFS by ODFW for Oregon coastal streams showed that coho survival continued to increase at the lowest observed spawner densities of less than one fish/mile. Thus even the criterion of less than 3.2 spawners/mile[13] used previously by ODFW for assigning the onset of depensatory mortality was a substantially higher spawner density than for which such mortality was actually observed.

Another significant departure in the model that results in a significant bias towards listing is evidenced by the BRT election to use "peak count" data, rather than continue with the standard fisheries' approach to utilizing "area under the curve" (AUC) for assessing the viability and risk level of the ESU.

In explaining this shift, the BRT stated:

> The BRT concluded that peak abundance counts were more likely to capture the potential for depensation because the effect occurs for fish that are on the spawning grounds at the same time (that is, fish need to find mates that are on the spawning grounds at the same time they are.)

The BRT further stated:

> The Critical Abundance criterion, 'PP-3,' in Wainwright et al. (2008), was discovered to have been evaluated using the wrong data set by the TRT (Wainwright et al. 2008). It was originally calculated using area-under-the-curve

[13] The BRT compounded this error by using a spawner density of 4.0/mile as its depensation trigger point.

> (AUC) spawner data rather than peak-count data as specified in the criterion. The updated Critical Abundance values are based on peak counts. AUC counts are almost always higher than peak counts … . Peak counts are simply the highest number observed at any one time. The object of the criterion was to evaluate the likelihood of depensation due to low spawner numbers. … This effect, termed 'depensation,' is thought to become a problem at spawner densities below four fish per mile (Wainwright et al. 2008).

The BRT further acknowledged that the DSS scores resulted in a higher certainty of sustainability than persistence, which it described as an initially counterintuitive result. The BRT explained this disconnect by stating:

> Even though population persistence is included in the sustainability criteria, the structure of the DSS logic tree and mathematical form of fuzzy logic used allows such a result. The cause of the relatively higher sustainability scores stem partly from the high scores for population functionality (PF) which was originally intended as a measure of habitat availability. Most populations scored very high for this criterion. However, in reviewing the DSS, the BRT was concerned that the metric for functionality—sufficient habitat to support a minimum number of smolts required for genetic integrity—did not adequately reflect the habitat quality issues this criterion was intended to address. The values for population functionality are strongly correlated with basin size … . For example, the largest river system in the ESU, the Umpqua River, had all four populations with a functionality score of 1.0, even though there are serious concerns about habitat conditions in these populations.[14] The BRT had several new analyses available to help evaluate habitat functionality … and felt that these new analyses were more informative than the TRT's Population Functionality criterion.
>
> In addition, the data set adjustment from AUC counts to peak counts for Critical Abundance lowered the persistence score substantially. Persistence is evaluated using three factors, while sustainability uses seven (including the three persistence factors). As a result, persistence is much more sensitive to changes in a single factor than is sustainability, so this score is considerably lower than was reported in Wainwright et al. (2008) and, counterintuitively, lower than the sustainability score.

While the BRT was of the opinion that Wainright et al. (2008) used the wrong data set (AUC versus peak count), in fact, Wainright et al. followed not only the protocol generally accepted in the fisheries' profession, it was also the same protocol that was used in prior BRT assessments (e.g., Sharr et al. 2000). Both Wainright and Sharr knowingly used the AUC data set in reaching the conclusion that four fish per mile standard was the best metric for the

[14] When one examines the habitat conditions of concern, one finds that the sand and gravel mining has ceased, the instream water rights have not varied for more than 50 years and have been augmented with flows from Galesville Reservoir, stream side buffers have been implemented under the Northwest Forest Plan and the State Forest Practices Act, and other extension restoration projects have been successfully undertaken by the Partners for the Umpqua River and others. The serious concerns no longer exist or have been overstated. The sheer numbers and increasing population bring into question whether the habitat in fact justifies the "serious concerns" statement.

determination of the population density wherein there was risk of extinction as a function of depensatory factors affecting spawning success at low densities.

Rather than follow the standard AUC, the BRT elected to change from AUC to peak counts, effectively extrapolating data beyond the original experimental design under which this metric was developed. Unfortunately, the BRT does not provide any references or scientific analysis as to why the metric is still scientifically valid if peak counts is the measure as opposed to AUC. Nor does the BRT provide any justification for departing from the conclusions reached by Wainright and Sharr.

While the BRT elimination of part of the methodology relating to population functionality as described in Section B above in and of itself may not have been critical, the change away from AUC becomes critical in that it resulted in a significant bias as a result of the model continuing to utilize the less than four fish per mile as the numerical threshold for assuming there is a small-population demographic risk.

We caution that if the BRT continues to set the criteria at four fish per mile while utilizing the peak counts rather than the AUC, it needs to recognize that it is taking the original science out of context wherein the four fish per mile quantum was developed. To eliminate this bias, the BRT should reexamine the use of the same truth membership function for its critical abundance criterion (PP-3) notwithstanding its change to peak counts.

While we consider the shift to peak counts as a critical error in its own right, the bias is magnified when one recognizes that the BRT was assuming that the survey techniques utilized in developing the number of spawners per mile provided data on 100% of the population and that this population number was a reliable basis to assess depensation. Both were errors, for the survey techniques that are utilized for spawning surveys are recognized as understating the true number of fish per mile and are recognized as only a random sampling of the stream miles.[15]

The scientific literature documents the significant undercounting that results from this form of stream sampling. For example, Solazzi (1984) found that on average surveyors observed only 75.5% of adult coho, and only 49.1% of jack coho that were present in the survey sections. Solazzi further reported that surveyors observed only an average of 62.8% of coho carcasses. Solazzi observed that these percentages of coho and jacks observed did not significantly differ before or after the date of peak spawning (Solazzi 1984).

Based on Solazzi's analysis, if the survey resulted in an adult peak count of four fish per mile, then the actual number of coho spawners in that reach would be at least five adults per mile. In addition to these five adults per mile, there could also be at least one jack coho. If one is to use the spawner surveys to assess depensation, then the number needs to be corrected to add

[15] In this sampling technique, a large population may be present in the mile adjacent to the mile actually sampled. With a large unsampled neighboring population, it is scientifically impossible to accurately assess depensation for either the population sampled or the stream population as a whole.

in the additional 25% of adult spawners as well as the jacks which are not included in the spawner survey numbers reported in the ODFW data set.[16]

Our review reveals that the BRT failed to recognize the limitations in the sampling techniques that led to the area under the curve being the recognized standard as opposed to the peak count. By switching to peak counts for a depensation assessment without making the corrections improperly extrapolates data beyond the scientific design. This error can be corrected by either expanding the peak counts observed or by modifying the model's truth membership function.

As a result of these errors, the BRT's depensation risk at four fish per mile at peak times was erroneously triggered in the model for this viability assessment.

As the scientific literature illustrates, if the observed peak count is only 75% of total count, and one wants the threshold to be negative (below "0"), then the model should indicate this at three fish per mile recorded peak count, not four.

Since the AUC survey data used in previous assessments demonstrate the variability of timing of spawner return over several weeks and includes more accurate abundance data that bears on the possible problems that may be associated with depensation, we recommend that if depensation is important for the model, then at a minimum the BRT return to the AUC model rather than the peak count strategy that was adopted. Further, since the depensation model is so sensitive to low numbers, it should be adjusted to reflect actual populations rather than merely random samples which have a high risk of missing critical population blocks.

In addition, the adult peak counts do not account for the additional jack spawners or that there is some overlap in fish entering and spawning in the survey reach.

[**Response:** The BRT noted several places where the DSS could potentially be improved, and if the TRT or others update the DSS, this may be a useful comment to consider during that process. The BRT utilized the TRT product for its analysis as one important factor to consider in evaluating the viability of the ESU, but other than updating (or in some cases correcting) the input data did not elect to modify the DSS. However, the BRT extensively discussed the DSS structure and results, and considered the strengths and limitations of the DSS in its final assessment.]

Illustrating how the model is subject to bias as a result of not reflecting actual population behavior, we compared the actual counts derived from Winchester Dam with the spawner survey data. We found that when actual wild adult coho counted at Winchester Dam are compared with the spawning survey estimates, in four out of the past five years the spawning survey estimates resulted in density numbers that were well below the actual number of spawners (Table 7).

Further illustrating the care that must be used in applying spawner densities to measure depensation is demonstrated by data derived from sampling on the Smith River in the lower Umpqua Basin. The Smith River study found that there is an overall negative bias of about 27%

[16] We note that the Winchester Dam counts indicate that about 22% of the spawners in the North Umpqua are jacks (ODFW unpublished data).

Table 7. Wild coho counts at Winchester Dam compared to spawner survey estimates on the North Umpqua above Winchester Dam.

Year	Winchester Dam counts*	Survey estimates	Winchester minus survey	
			Fish	**% bias**
2009	7,724	2,310	+5,414	–70%
2008	3,438	1,500	+1,938	–56%
2007	1,410	1,081	+329	–23%
2006	3,000	2,154	+846	–28%
2005	2,113	3,692	–1,579	+75%

* Adjusted for harvest and brood stock collection.

associated with spawner survey-based estimates (Jacobs 2002). Again illustrating that the use solely of spawner surveys would most likely directly bias any predictive model based on spawner abundance numbers.

We also note that the current AUC monitoring conducted annually for spawner survey estimates has been previously recognized as having a low precision on a basin and stratum level (Jacobs and Nickelson 1998). If spawner density metrics derived from small population levels are used in the model, then the BRT model must recognize an uncertainty ranging up to ±80% of the stated value.

Jacobs and Nickelson (2008) caution that for a given level of sampling, coast-wide estimates will always be substantially more precise than estimates for individual basins, and precision is directly related to the number of surveys conducted and the size of the geographic unit where inference is being drawn.

When used for the spawner density criterion, the peak count data would be expected to result in an equal to or higher range of uncertainty whenever the number of surveys is below 50 reaches for each population (Beidler and Nickelson 1980). Beidler and Nickelson found that the "peak count" to "total" ratio ranged from 0.20 to 0.74, which they concluded resulted in peak counts being a poor indicator of abundance on a year to year basis. Based on this study, the use of peak count data simply may not be a valid assumption for annual spawner density estimates in assessing critical abundance (depensation).

It appears that the BRT's model, wherein depensation using peak counts is incorporated, overstepped the scientific certainty values associated with this data, and in addition, it erroneously did not take into consideration the factor of about 25% unobserved fish and the jacks that are actually present on the spawning grounds.

While we are concerned that the BRT model was not based on the best available scientific data and results in a significant bias relative to depensation on the ESU level, we also note that as a practical matter it should not be an issue on the North Umpqua, given that actual quantitative data from the 1946–2009 time period illustrates that contrary to the model, this population has shown resiliency even during low density levels. Most telling is the fact that this population is increasing in production following years when the model predicted significant depensation.

Similar conclusions should be reached for the entire ESU, given that the Winchester Dam data set not only provides the best science for assessing the health of the Umpqua populations, it also provides a highly accurate data set on which to validate the model assumptions. Utilizing actual data that is available for the North Umpqua wild coho population would significantly change the overall scores for persistence and sustainability for not only this population, but for many other independent populations and most likely for the entire ESU.

III. Summary

Based on our review of the available scientific information, it is clear that the North Umpqua independent population is near or above historical levels of abundance as compared to the six decades of data from the Winchester Dam counts, has been increasing in the recent decade, and is not in danger of extinction in the near future (at least for 60 more years), given the threats currently identified.

Notwithstanding this actual data and history, the current BRT's risk assessment concluded that this population has a very high certainty of being in very poor shape and in danger of extinction within the near future (within 100 years) assuming the very same threats it has encountered in the past.

Based on model results for the Umpqua Basin stratum relative to abundance and resiliency, we are concerned that at least one of the BRT model's major criterion assumptions is not valid or reliable.

Based on actual quantitative data on fish abundance and density, along with the historic climatic patterns, there clearly is not sufficient certainty that the qualitative model is the best available science on which to base a decision that this ESU is threatened in the foreseeable future.

We recognize and share the BRT's uncertainty as to the ability to predict future habitat and climatic conditions; however, given the wide variation experienced over the last 300 years, as evidenced by tree rings, we strongly recommend that the BRT look to the historical record.

The historical review shows a well-documented history and scientific record of persistence and sustainability for the North Umpqua population. This history strongly supports the findings that current hatchery releases, harvest practices, escapement, and ocean and freshwater habitat conditions are not likely to change in a manner that would further risk the viability of this wild coho population and others within the ESU.

[**Response:** The BRT did examine the historical record and recognized that there are strong climate driven fluctuations in abundance and productivity. We incorporated fluctuations in both the TRT criteria and the BRT risk assessment.]

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