Chapter 7: The Aquatic Conservation Strategy of the Northwest Forest Plan—A Review of the Relevant Science After 22 Years

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Introduction

The Aquatic Conservation Strategy (ACS) is a regional strategy applied to aquatic and riparian ecosystems across the area covered by the Northwest Forest Plan (NWFP),

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encompassing broad landscapes of public lands administered by the U.S. Department of Agriculture Forest Service and the U.S. Department of the Interior Bureau of Land Management (BLM) (USDA and USDI 1994a). The ACS was developed during the analysis (FEMAT 1993) that led to the NWFP, but its foundation was a refinement of earlier strategies: the Scientific Panel on Late-Successional Forest Ecosystems (“The Gang of Four”) (Johnson et al. 1991), PacFish (USDA and USDI 1994b), and the Scientific Assessment Team (Thomas et al. 1993). The ACS used an ecosystem approach to management of riparian and aquatic habitats (Everest and Reeves 2007) and was designed to: 1) protect watersheds that currently had good-quality habitat and strong fish populations; and 2) halt further declines in watershed condition and restore ecological processes that create and maintain favorable conditions in aquatic ecosystems in currently degraded ecosystems (FEMAT 1993). The long-term goal (100+ years) was to develop a network of functioning watersheds that support populations of fish and other aquatic- and riparian-dependent organisms across the NWFP area (USDA and USDI 1994a). The ACS is based on preserving and restoring key ecological processes, including the natural disturbance regimes (USDA and USDI 1994a) that create and maintain habitat for native aquatic- and riparian-dependent organisms, and recognizes that periodic natural disturbances may be required to sustain ecological productivity. As a result, the ACS does not expect that all watersheds will be in favorable condition (highly productive for aquatic organisms) at any point in time, nor does it expect that any particular watershed will remain in a certain condition through time. If the ACS and the NWFP are effective, the proportion of watersheds in better condition (for native organisms) is expected to remain the same or increase over time (Reeves et al. 2004).
To “meet” and “not prevent attainment” were important phrases in the NWFP guidance for the ACS, along with the wording “maintain and restore” relative to ACS objectives (USDA and USDI 1994a). Baseline conditions against which to assess maintenance and restoration conditions were to be developed through watershed analysis, and improvements of biological and physical processes were to be evaluated relative to their range of natural variability (USDA and USDI 1994a). ACS objectives addressed: 1) diversity and complexity of watershed features; 2) spatial and temporal connectivity within and between watersheds; 3) physical integrity; 4) water quality; 5) sediment input, storage, and transport; 6) instream flows (e.g., both peak and low flows); 7) floodplain inundation; 8) riparian plant species composition and structural diversity; and 9) habitat to support well-distributed populations of native plant, invertebrate, and vertebrate aquatic- and riparian-dependent species.

The ACS sets out five components to meet its goals:

- **Riparian reserves**: Riparian reserves are specifically designated portions of the watershed most tightly coupled with streams and rivers, which provide the ecological functions and processes necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time, and habitat connectivity within and between watersheds. The reserve boundaries were considered interim until a watershed analysis was completed, at which time they could be modified based on suggestions made in the watershed analysis.

- **Key watersheds**: Key watersheds are intended to serve as refugia for aquatic organisms, particularly in the short term for at-risk fish populations, and have the greatest potential for restoration, or to provide sources of high-quality water. Tier 1 key watersheds currently have good populations or habitat, a high restoration potential, or both. Tier 2 key
watersheds provide sources of high-quality water.

- **Watershed analysis**: Watershed analysis is an analytical process that characterizes the features and processes of watersheds and identifies potential actions for addressing problems and concerns, along with possible management options. It assembles information necessary to determine the ecological characteristics and behavior of the watershed, and to contribute to the development of options to guide management in the watershed, including adjusting riparian reserve boundaries.

- **Watershed restoration**: Watershed restoration is designed to recover degraded habitat. Restoration activities focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic- and riparian-dependent organisms.

- **Standard and guidelines**: These directives impose specific requirements (standards) or recommended approaches (guidelines) for management activities in riparian reserves and key watersheds.

It is important to note a key philosophical shift in the development of the ACS and NWFP. The ACS, along with PACFISH (USDA and USDI 1994b) and the riparian component of the Tongass Land Management Plan (USDA 1997), made two substantive changes in how riparian management was formulated (Everest and Reeves 2007). First, they addressed riparian management at the watershed scale with specific emphasis on maintaining ecological functions over the long term. Second, they rejected the previous philosophy of trying to define and achieve the absolute minimum set of practices that would meet stated riparian management goals and the concept that goals could be met by implementing yet another set of best-management practices.
The new management philosophy under the NWFP represented a paradigm shift in how managers viewed resource coordination. In previous riparian rule sets, riparian and aquatic technical specialists shouldered the “burden of proof” to demonstrate resource damage from forestry activities, and the need for more comprehensive forest-practices rules to meet riparian-management goals. Under the NWFP, the precautionary principle was invoked—the burden of proof shifted (Thomas et al. 2006, USDA and USDA 1994a). Forest managers who wanted to alter the comprehensive default prescriptions for riparian management under the NWFP in order to pursue other management goals were required to demonstrate through watershed analysis that changes would not compromise established riparian-management goals.

This chapter focuses on the scientific literature related to the management and conservation of aquatic ecosystems, particularly as it has developed since the 10-year review (Reeves 2006), with particular emphasis on the area of the Northwest Forest Plan. We provide an update on the status of species listed under the Endangered Species Act (ESA) and the components of the ACS: aquatic and riparian monitoring, riparian reserves, key watersheds, watershed analysis, and watershed restoration. We also assess the implications for the potential evolution of the ACS in the next round of forest plans. In addition, we consider a review of particular issues raised by managers: 1) management and size of riparian reserves; 2) effects of roads; and 3) potential effects of climate change on aquatic ecosystems and associated fish.

Reeves (2006) provided a thorough review of the literature in the first 10 years of the ACS, and readers should refer to that publication for a review of the relevant science during that time.
Guiding Questions

In this chapter, we use the following guiding questions in our update and assessment:

1) What is the scientific basis of trends observed in the ACS monitoring program, and what are the limitations, uncertainties, and research needs related to monitoring?

2) What is known about the variation in characteristics of unmanaged streams and riparian ecosystems in relation to stream networks across the NWFP area?

3) What has been learned about the effects of riparian vegetation on stream habitat and environment?

4) What effects have human activities had on stream and riparian ecosystems?

5) What is the scientific basis for restoration management in riparian reserves, and how does restoration relate to the ecological goals of the ACS?

6) What is the capacity of federal lands in the NWFP area to contribute water for a suite of economic, recreational, and ecological uses?

7) What are the potential effects of climate change on aquatic ecosystems in the NWFP area?
Key Findings

Status of Species and Population Units Listed under the Endangered Species Act on Federal Lands in the Northwest Forest Plan Area

In 1993, only the Sacramento winter Chinook salmon (*Oncorhynchus tshawytscha*), the shorthorn sucker (*Chasmistes brevirostris*), and the Lost River sucker (*Deltistes luxatus*, native to the Klamath and Sacramento River systems) were listed as threatened or endangered under the Endangered Species Act in the area covered by the Northwest Forest Plan. Within a few years of the development of the ACS, 21 evolutionarily significant units of Pacific salmon and 3 distinct population segments of bull trout (*Salvelinus confluentus*) have been listed under the ESA (table 1). There have been three additions since the 10-year review (Reeves 2006): the Oregon Coast coho salmon evolutionarily significant unit (*O. kisutch*), and 2 other fish species, the Oregon chub (*Oregonichthys crameri*) and the Pacific eulachon (*Thaleichthys pacificus*). No population units of Pacific salmon or bull trout have warranted delisting since the ACS was developed.² However, the Oregon chub was delisted in 2015 (Federal Register 2015), becoming the first fish to be delisted because of increases in numbers. Habitat on the Willamette National Forest contributed to the recovery.

The developers of the ACS anticipated the ESA listing of distinct population segments of various species of Pacific salmon, evolutionarily significant units, and other fish species. It was

not expected to prevent the listing of any species or distinct population segment, because other
factors outside the responsibility and control of federal land managers contribute to the decline
and recovery of fish populations and will strongly influence their recovery. These factors
include:

- Degradation and loss of freshwater and estuarine habitats on non-federal lands
- Excessive harvest in commercial and recreational fisheries
- Migratory impediments, such as dams
- Loss of genetic integrity from the effects of hatchery practices and introductions combined
  with undesirable interactions (e.g., competition and predation) involving hatchery and
  naturally produced fish (NRC 1996)

Thus the ACS was an attempt to develop a strategy to guide management of aquatic ecosystems
on federal lands in the NWFP area that would meet potential ESA requirements. The ACS was
expected to make significant contributions to the recovery of the ESA-listed fish by increasing
the quantity and quality of freshwater habitat for Pacific salmon and protecting and enhancing
habitats of other species (FEMAT 1993).

The potential of federal lands to contribute to the recovery of listed fish, particularly
Pacific salmon, in many parts of the NWFP area is likely more limited than was assumed when
the ACS was developed. The primary reason for this difference is that federal lands (figs. 1 and
2) have a limited capacity to provide high-quality habitat for some of the listed fish in many
situations. Federally managed lands are generally located in the middle to upper portions of
watersheds, which tend to have steeper gradients and more confined valleys and floodplains,
making them inherently less productive for some fish (Burnett et al. 2007, Lunetta et al. 1997,
Reeves et al. 2016a). Federal lands may, however, be important sources of wood, sediment (Reeves et al. 2016a), and water (Brown and Froemke 2010, 2012) for nonfederal lands, and will be important for the potential recovery of most populations. However, their contribution to recovery may in many cases be insufficient without parallel contributions from nonfederal landownerships downstream.

The numbers of Pacific salmon and other anadromous fish returning to freshwater in the NWFP area is strongly influenced by ocean conditions, which are highly variable over time. Favorable conditions (cold water) tend to occur in the negative phase of the Pacific Decadal Oscillation (PDO) and the La Niña phase of the El Niño-Southern Oscillation (ENSO), when fish growth is strong and survival is high, resulting in strong returns of adults to freshwater (Mantua et al. 1997). Survival is low and numbers decline during warmer periods, the positive phase of the PDO and the El Niño phase of the ENSO. Winters are cold and wet in the negative PDO–La Niña phase, which also creates more favorable conditions in freshwater (Mantua et al. 1997). A positive PDO–El Niño produces dry, warm winters, reducing streamflows and increasing water temperatures. The last period of high productivity was from the late 1940s to 1977 (Mantua et al. 1997), with brief periods of favorable conditions in 1982–1983 and 1997–1998 (N. Mantua, NMFS, pers. comm.).

In the past 10 years, the numbers of Pacific salmon returning to freshwater have been relatively high because of more favorable ocean conditions (Bond et al. 2015). However, beginning in 2013, abnormally warm conditions in the Pacific Ocean (“the Blob”) developed because of lower than normal heat loss from the ocean to the atmosphere combined with a relatively weak mixing of the upper ocean layer due to an usually high sea pressure level (Bond
et al. 2015). Initial effects were most notable in the North Pacific Ocean off Alaska. Ocean
conditions changed noticeably along the NWFP area in 2014 as a result, and fish returns are
expected to decline over the next few years.

We are unable to separate the influence of ocean conditions over the last 10 years from
the influence of changes in the condition of freshwater ecosystems on federal lands that may
have occurred under the NWFP and ACS. The actual contribution of freshwater habitats to the
persistence and recovery of anadromous salmon and trout will be relatively more important when
ocean conditions move into a less-productive phase (Lawson 1993). Improvements in the
quantity and quality of freshwater habitat resulting from the ACS could result in relatively
greater numbers of fish entering the ocean, thus increasing the likelihood of persistence of many
populations during periods of low ocean productivity. However, as noted previously, the
contribution of federal lands may be more limited than expected because their potential to
provide high-quality habitat is less than originally recognized when the ACS was developed.

The status of other aquatic-riparian species in the NWFP area is not as well monitored as
that of Pacific salmon. The Oregon spotted frog (Rana pretiosa) was listed as threatened under
the ESA in 2014. It is a pond-breeding amphibian now restricted to isolated populations that
overlap the NWFP area in western Washington and Oregon.³ Five other aquatic-riparian
amphibian and reptile species are petitioned for ESA listing and are under status review: 1)
Columbia torrent salamander (Rhyacotriton kezeri); 2) Cascade torrent salamander (R.
cascadae); 3) Cascades frog (Rana cascadae); 4) Oregon slender salamander (Batrachoseps

³ http://www.fs.fed.us/r6/sfnpw/issssp/agency-policy/
wrighti); and 5) western pond turtle (Actinemys marmorata). Torrent salamanders are headwater forest species, occurring predominantly in and along the banks of small streams, with significant portions of their ranges on non-federal lands. Nevertheless, federal riparian reserves contribute habitat for localized populations of Columbia torrent salamanders and more extensive areas for Cascade torrent salamanders. The Oregon slender salamander has strong associations with down wood on the forest floor in riparian and upland forests, and has associations with older forest conditions (Clayton and Olson 2007). Cascades frogs are pond-breeders at higher elevations in the Cascade Range, where they may be affected by multiple stressors (Pope et al. 2014). Similarly, multiple threats appear to affect western pond turtles, which may occur in stream and pond systems in the NWFP area (Rosenberg et al. 2009).

Monitoring—Aquatic and Riparian Effectiveness Monitoring Program

The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) was developed to implement the ACS monitoring strategy (Reeves et al. 2004). Monitoring was to assess trends in ACS goals to restore and maintain ecological processes that create and maintain aquatic ecosystems for a suite of aquatic and riparian-dependent organisms, including fish that would potentially be listed under the Endangered Species Act, and for clean water and other ecological services (USDA and USDI 1994a). The program’s intent has been to characterize the condition of watersheds by assessing in-channel, riparian and upslope conditions. AREMP monitoring procedures were captured by Hohler et al. (2001) five years after ACS conception:
“Condition classes will be assigned by using a decision-support framework to evaluate and work with various biotic, physical, and chemical indicators. Ultimately, developing predictive models based on an understanding of the key upslope indicators associated with riparian and channel conditions will make this process more efficient. The approach rests heavily on Forest Plan related research—especially on watershed analysis, CLAMS —and disturbance-process research to design a cost-effective, integrated strategy.”

Hence, the ACS monitoring strategy postulated that management under the NWFP would be considered effective if the distribution of watersheds in better condition improved over time (Reeves et al. 2004). However, the process of first selecting indicator components and then constructing predictive models of watershed condition using multivariate indicators is not straightforward.

Hohler et al. (2001) allude to adaptive management of this process to make it more efficient—empirical analysis procedures for data elements were assumed to be developed and refined as new science became available and as data were collected as part of the monitoring program (Hohler et al. 2001, Reeves et al. 2004). Knowledge gaps in understanding aquatic/riparian systems have been recognized, such as the lack of the basic foundational characterization of the frequency distribution of expected or reference watershed conditions (Reeves et al. 2004). In addition, there are a variety of approaches possible both to identify the most relevant indicator components, and then to combine them for watershed-scale analyses. As predicted, refinements and improvements have occurred over the 20-year timeline of AREMP implementation, as a rich database of components has been assembled, new multi-scale analytical approaches have been developed, and we have a more comprehensive understanding
of aquatic-riparian conditions. Nevertheless, the discipline of landscape ecology as applied to
watersheds has not yet fully matured, and there remains considerable potential advancement for
watershed condition assessments. In that light, we reconsider the prophetic question by Hohler et
al. (2001), “can we design a more efficient process?” Here, we capture the main themes of the
current aquatic-riparian effectiveness monitoring program, and suggest areas for further research
and development. As with all evolving science, new approaches and data should continue to be
considered in all future aquatic assessments and research.

Initially, the AREMP used a conceptual decision-support system, where fuzzy-logic model
parameters and the relation between them and habitat conditions were defined using the
collective professional judgment of interdisciplinary teams, as well as the literature and
thresholds developed by existing state-level monitoring programs. Data to inform many indicator
components of this decision support model (DSM) were initially sparse, so relations between
variables and watershed condition were developed using professional opinion (Reeves et al.
2004). More recently, the AREMP revised a number of indicator components within both the
upslope and riparian geographic information system (GIS) subsets of the DSM, and modified the
in-channel analysis using empirically derived data collected as part of the monitoring program.
Four modeling approaches were evaluated, (S.A. Miller, pers. comm.), and a nearest-neighbor
approach that used thresholds derived from “least human disturbed” condition (Stoddard et. al.
2006) was used as the basis for the in-channel stream evaluation (Bates Prins and Smith 2007,
Miller et al. 2016a).
From these recent analyses that address the entire data set from AREMP inception through the 20-year post-NWFP implementation, results suggest that the overall distribution of watershed conditions has remained relatively stable (Miller et al. 2016a). Yet on a per-watershed scale, changes were apparent. Areas that were most heavily managed (e.g., intensive timber production and high road densities) before the NWFP showed the largest improvements in overall condition, primarily as a result of increased size of trees in riparian areas and reductions in roads through decommissioning. In contrast, areas that experienced wildfire showed the largest declines. Results from the upslope–riparian watershed scores in the Miller et al. (2016a) assessment supported results described in previous evaluations (Gallo et al. 2005, Lanigan et al. 2012, Reeves et al. 2006).

Using recent data, trends in in-channel stream attributes can now be evaluated since watersheds have been re-sampled given the multi-year resampling time interval imposed by the study design. As of 2016, the stream-sampling component of the AREMP evaluated just under half the re-sampled watersheds (based on the sample design, the first full resample will be completed in 2017). Although improving trends were not expected in the short term (FEMAT 1993), Miller et al. (2016a) reported that trends in mean seven-day maximum average temperatures decreased with time (2002-2013). Similarly, macroinvertebrate scores increased over the same time period. Both of these patterns, reduced thermal maxima and altered macroinvertebrate composition, suggest that aquatic conditions improved within the NWFP area, supporting ACS goals. However, AREMP data have yet not shown clear trends in the overall in-channel physical habitat conditions, which may mean that such processes are particularly variable or require a longer time to recover.
The indicator components for aquatic-riparian assessments are a critical element of watershed condition assessments and trends analyses. Different results may arise depending on the empirical or modeled data used as inputs to indicator evaluations, and given that we have imperfect knowledge of indicators, it is prudent to be cautious and acknowledge both uncertainty and alternative strategies for progress in this dynamic science-management interface.

Furthermore, indicator components should correspond to the ecological and biological objectives of the aquatic assessment, as they will subsequently frame both the rationale for the analytical approach and the interpretation of findings. Specifically, the development of aquatic monitoring programs requires a clear articulation of which biota and associated functional characteristics of habitats and ecosystems are being considered and how they are likely to be altered as a result of the actions of interest (Palmer et al. 2005, Pont et al. 2006). The failure to identify the nature of the unimpaired community in each ecological stage can make it difficult to assess the ecological consequences of detected changes (Frissell et al. 2001) and can affect the credibility of the scientists or agencies involved (Wohl 2016). As a coarse-filter approach to conservation, the panel of scientists and managers who initially framed the ACS assumed that indicator components could be used, and further, that favorable conditions for fish (e.g., streams with cold water and structural heterogeneity provided by physical habitat components such as down wood and coarse substrates), in particular, would provide suitable conditions for other aquatic and riparian organisms.

In a more recent conceptual framework, the fish could be termed the “final ecosystem services” that rely upon “intermediate ecosystem services” for their sustainability (Penaluna et al. 2016). However, the exact nature of the relationships of aquatic and riparian organisms to
forest and stream conditions is likely to vary across taxonomic groups, hence it is questionable to assume that what is good for one group is good for another (Cushman 2008, 2010). In addition, federal land managers have focused on managing freshwater habitats for particular sensitive species, with other native species as a secondary concern. In this light, previous assessments of the ACS (Gallo et al. 2005, Lanigan et al. 2012, Reeves 2006, Reeves et al. 2006) focused on watershed condition with an emphasis on components of particular relevance to native salmonids, some of which are deemed sensitive or are listed as federally threatened or endangered. Since federal lands in the NWFP area include approximately 30 percent of anadromous fish-bearing streams, the contribution of aquatic-riparian conditions in the about 70 percent of the AREMP-sampled sites without resident fish were reconsidered in the recent analyses, with consequences for indicator-component weighting, sample size, and results (Miller et al. 2016a). Yet, it is fully recognized that entire watersheds contribute to downstream fish habitat (Fausch et al. 2002).

The most recent in-channel stream assessment of the ACS (Miller et al. 2016a) moved the emphasis away from fish on the basis that “Little attention was given to the ecological thresholds of other organisms” (Miller et al. 2016a). This approach moved away from native salmonids focus and instead evaluated whether physical habitat conditions differ from what would be expected in areas of “least human disturbance”, as other monitoring efforts have done (Stoddard et al. 2006, Bates Prins and Smith 2007, Collier 2009, Al-Chokhachy et al. 2010). Thus, the approach does not specifically provide an assessment of habitat conditions for aquatic organisms. To be clear, the various approaches are not necessarily “wrong” or “right” in any sense, but it is apparent that a more in-depth understanding of alternative approaches to assigning
indicator components is an important consideration relative to what the indicators mean in the context of maintaining and restoring watershed conditions. This issue is an emerging research need in the NWFP area and in aquatic conservation in general.

Once aquatic-riparian indicators are chosen, watershed condition is assessed by integrating them into a model. Diverse analytical approaches are now available for aquatic assessments, and the aggregation of within-watershed components to assess watershed condition can be performed in various ways. Gallo et al. (2005) and Lanigan et al. (2012) used a decision-support framework, as proposed by Reeves et al. (2004), which integrated the various indicator components in a hierarchical manner to determine the condition of a given watershed as it pertains to the habitat quality for salmonids (fig. 3). Hohler et al. (2001) projected a transition in use of models, for example from using a decision-support model to a multivariate statistical model; such a transition may be timely now, given development of numerous new approaches for data-rich analyses. In the recent analysis by Miller et al. (2016a), they used a multivariate statistical model to integrate environmental variability directly into the assessment, and then considered temperature and macroinvertebrate indicator components separately from an aggregated physical-habitat indictor. Their aggregation of indicator components was empirical and not necessarily based on hypothesized ecological relationships (e.g., relative weights of different components or non-linear relationships), as might be done with a fuzzy-logic model used by Gallo et al. (2006) and Lanigan et al. (2012). However, some different consequences of fuzzy logic versus the current analytical approach are evident. For example, in the analysis by Miller et al. (2016a), three within-watershed metrics corresponded to trends in watershed condition: 1) upslope-riparian condition; 2) aquatic macroinvertebrate conditions (e.g.,
invertebrates moved higher in watersheds with improved conditions); and water temperatures
e.g., declined in improved watersheds). These measures were integrated into a comprehensive
metric in previous efforts (Gallo et al. 2005, Lanigan et al. 2012). Hence, in both cases, a
watershed may score high for one metric but low for another. The DSM models considered this
in the overall assessment, whereas Miller et al. (2016a) considered the parameters separately,
which does not reflect a single overall condition score, instead serving as multiple lines of
reporting to understand individual components. Both approaches allow an understanding of how
each component affects the overall score, yet they may yield different results and have different
ecological or biological interpretations in a conceptual framework. The issue of how best or
whether it is advisable to integrate metrics merits further investigation and reconsideration of the
objectives and uses of watershed-scale analyses.

The building blocks of models for landscape and watershed assessments need careful
conceptual consideration along several dimensions to achieve accurate representation of the
components they aggregate. For example, water-quality and stream-temperature issues are rising
to the forefront for coldwater-adapted Northwest stream species, and a recent focus has been on
stream-temperature modeling (e.g., Isaak et al. 2011). The miles of streams in the NWFP area
classified as impaired (303(d)-listed) under the Clean Water Act have likely increased in the past
10 (B. Staab, US Forest Service, pers. comm.). The recent AREMP analysis (Miller et al. 2016a)
showed a change in the distribution of water temperature toward lower temperatures along with
increases in watersheds with higher O/E scores for macroinvertebrates across the NWFP area.
Though water temperatures declined in many watersheds, an estimated 55 percent still exceed
state water-quality standards, so a decreasing temperature trend or potentially increasing O/E
trend, does not necessarily indicate the conditions within a watershed are more favorable ecologically.

In addition to having a clear rationale and a tie to the ecosystem services that an index is intended to assess (e.g., fish or other organisms, or water quality), an accounting of naturally occurring variation of the index in conditions is needed (Hughes et al. 1986, NRC 2000, Reynoldson et al. 1997). This is accomplished by including a reference population that spans the full range of conditions that an ecosystem may experience (Lisle et al. 2007, NRC 2000, Stoddard 2007). Aquatic systems are inherently multistate, subject to a variety of natural disturbances that can alter habitat conditions, biota, and processes (Penaluna et al. 2016).

A number of factors need to be better understood to improve the usefulness of the AREMP assessments. These include (1) uncertainty regarding the ecological significance of the metrics and trends of aquatic ecosystem conditions reported in all of the AREMP assessments (Gallo et al. 2005, Lanigan et al. 2012, Miller et al. 2016); (2) the role of natural variation in conditions resulting from natural disturbances and how this variation should be incorporated into large-scale condition assessments; (3) the choice of “reference condition” in systems that are naturally variable in space and time; and (4) novel ecosystems resulting from climate change, invasive species, and other anthropogenic effects.

Natural variability—

Natural variability in space and time is difficult to incorporate into traditional concepts of a “reference condition”. Generally, these assessments attempt to evaluate the degree of anthropomorphically driven change within a system; therefore, reference sites should represent the “least-disturbed conditions” (see Paulsen et al. 2008). These reference conditions may
consider local invariant environmental gradients—for streams, this might include factors such as slope, elevation, and stream size (Cao and Hawkins 2011, Montgomery 1999). However, these factors generally account for a relatively small percentage of the inherent variation (Mazor et al. 2016). Much of the local variation is driven by disturbance processes and post-disturbance recovery—and by definition cannot represent the “least-disturbed conditions”. To be effective, then, reference conditions must account for the full range of naturally occurring variation (Hughes et al. 1986, NRC 2000, Reynoldson et al. 1997).

Because the range of natural variability occurring in a system over large spatial and long temporal scales occurs across a multi-dimensional continuum, it can be difficult to incorporate into assessments. One potentially useful approach is to identify a specific ecological state in that continuum (see Wondzell et al. 2007, 2012). Then aquatic systems can be viewed as multistate systems resulting from a variety of natural disturbances as well as exogenous or anthropogenic processes that can alter habitat conditions, biota, and processes (Penaluna et al. 2016). Having the full range of potential variation classified into discrete states gives a way to begin enumerating the ways in which variability is arrayed over large spatial scales and how it changes over long temporal scales.

Not having the range of natural variation represented in the reference neighborhoods will influence the assessment of managed conditions (NRC 2000). This conceptual issue appears to arise in the analysis approach used for the recent aquatic assessment (Miller et al. 2016a), and deserves further consideration. This issue can be illustrated using the down wood data from Reeves et al. (1995), who examined three watersheds in the Oregon Coast Range that had recovered for differing lengths of time since the last large wildfire (see details in next section of
this chapter). If only the values from the watersheds that were at both an intermediate time-point and the longest time-point from disturbance were included as being in the population of reference conditions, wood values would range from 12–24 pieces of wood per 100 m of stream. If a value (e.g., number of pieces of wood/100 m) was below the 5\textsuperscript{th} percentile (fewer pieces than expected based on its associated neighborhood) it would receive a low score, and the score would increase linearly as the number of wood pieces approached the 95\textsuperscript{th} percentile (Miller et al. 2016a). Thus, a value of 13 pieces of wood/100 m would get a low score, whereas 22 would get a high score. However, if the most recently disturbed system were included in the pool of reference conditions, the lower bound would be near 6 pieces of wood/100 m, and the score for 13 pieces of wood/100 m would be much higher. It is clear that the assessment of conditions reported by Miller et al. (2016a) depends on the range of conditions found in its environmentally matched reference neighborhood, and that an incomplete representation of the range of natural variation in a reference neighborhood could strongly influence the assessment findings at the site scale.

However, it is important to note that based on the AREMP sample design, the inference is made at the watershed scale (Stevens and Olsen 2008), where multiple sites within a watershed were compared to multiple reference neighborhoods, and it is the overall distribution of scores that is used to describe conditions across the NWFP. Because this relative method of assessment rates a system relative to a particular baseline, a change in the baseline can result in different assessments for the same system that may be more congruent with assessing dynamic systems, given that the baseline is not static.
The concept of the reference condition remains important in land management—not because it is a goal of management agencies to restore systems to some previous reference condition, but rather because knowledge of the historical range of variability can help inform choices about desired future conditions and thereby help determine management and restoration goals. Thus, departure from the reference condition can provide a relative measure to evaluate habitat conditions for managers who seek to maintain or restore ecosystem and species diversity (Nanoka et al. 2007, Safford et al. 2012). The multi-state conceptual approach clearly shows that “reference conditions” are, in fact, a distribution of states within a suitably large stream network rather than a single preferred or optimal state. As such, “departure from the reference condition” is no longer a reach-scale question, but rather, a determination of whether the number of reaches (or length of stream) in each state within the network is similar to the number of reaches (or length of stream) that would be expected under a relatively natural disturbance regime. To answer this question, it is important to consider the entire range of conditions that an ecosystem can experience (Lisle 2007, NRC 2000, Stoddard et al. 2006); however, this proves problematic. It may well be impossible to find a reference system that contains the full range of ecological conditions that an unmanaged ecosystem would experience, owing to the extent and magnitude of anthropogenic effects (Miller et al. 2016b, NRC 2000, Stoddard et al. 2006). This may especially be the case in regions in the NWFP area where fire suppression has altered forest and riparian plant composition and structure (chap. 3 this volume), where invasive species are now a dominant component of communities, or where pervasive “press” disturbances such as grazing have influenced the entire landscape so that current conditions are now considered the expected norm. Thus, even areas that may appear to lack any sign of current or historical active, human-
caused disturbances can no longer be considered pristine. We suggest that further exploration regarding the definition of reference conditions and the potential consequences for the results and their application is needed, including consideration of how to use conceptual models such as those developed by Wondzell et al. (2007, 2012) to compliment reference conditions based on least-disturbed sites.

Monitoring programs have generally assumed that there is a given condition or limited set of conditions that supports aquatic organisms, primarily fish and macroinvertebrates (e.g., Karr and Chu 1998). For example, the initial curves of the response of fish to various environmental factors developed by Gallo et al. (2005) suggested that there was a relatively small range for given environmental variables that were most suitable for fish. However, recent studies of the response of aquatic organisms have demonstrated that native salmonids (Howell 2006, Rieman et al. 2010, Sestrich et al. 2011) and aquatic invertebrates (Minshall 1989) are capable of adapting to and being productive in a wide range of conditions, including those following major disturbances such as wildfire. Flitcroft et al. (2016) found that although conditions for one life-history stage of salmonids may be unfavorable, other life-history stages find alternative conditions suitable and populations may respond positively. Native salmonids may also change life-history tactics, such as by lowering the age or size at maturity (Rosenberger 2015). It is important for monitoring programs to incorporate this new perspective into the development of benchmarks and interpretation of results to better reflect the response of aquatic organisms to management and natural disturbances.

There is a growing concern about the extent to which ecosystems in the NWFP area, and elsewhere, have been affected by climate change and altered disturbance regimes, such as fire
suppression (Hessburg et al. 2005, Luce et al. 2012; also see chap. 3 this volume). We are likely seeing, or will soon see, the development of ecosystems that are different from the present and at least the near past (Hobbs et al. 2009, Luce 2012) (fig. 4). The conditions that result from these altered ecosystem trajectories could be very different from those that would be found in unaffected systems, and they may not necessarily meet social or legal expectations (Luce et al. 2012). Using benchmarks based on our understanding of aquatic ecosystems today may also affect assessments of ecological consequences from natural and anthropogenic factors. The potential implications of these changes merit a primary research focus, but in the meantime, it will be important that monitoring programs, whether they use reference conditions (e.g., Miller et al. 2016a), decision-support models (e.g., Reeves et al. 2004), or other approaches, recognize and acknowledge these potential concerns in the process of analysis and the interpretation and application of results.

The ACS monitoring program is critical to evaluating how well the goals of the NWFP are being met and what changes in management may be needed to address emerging issues and new science. AREMP, along with the broader monitoring community, faces several challenges now and in the future. There are a number of important questions and considerations regarding approaches to monitoring, including how best to integrate the effects of wildfire, and fire suppression, and climate change in assessments, as well as other emerging issues such as the effects of roads in sediment delivery to streams that would benefit from more research and development. In addition, condition assessments are often used as a way to describe species-level habitat and it is rare that monitoring programs are able to concurrently collect species-level data.
Managers continually look to incorporate watershed-level condition assessments (i.e., for individual forest-planning units or watersheds) into their decision-making and planning process, and they would benefit from multi-scale assessments that incorporate the landscape-to-regional context. The toolbox of assessment techniques to help managers in this regard is rapidly developing at this time. Simultaneously, although some of the objectives of the ACS were initially somewhat vague, as our knowledge of aquatic-riparian systems in the NWFP area increases more explicit outcomes of the ACS can be considered. Therefore, it would be prudent to convene a panel of researchers and practitioners, possibly as part of a watershed assessment center, that could review the current assessment concepts, investigate ways to understand emerging issues, and develop an integrated watershed-assessment research and management team, as has been developed for other components of the NWFP (e.g., northern spotted owl, *Strix occidentalis caurina*). Such a team could aid in assessing individual watersheds on an as-needed basis for federal land managers and in an all-lands context (e.g., for context-dependent restoration goals or understanding climate change projections), as well as providing new information to apply to refinements of the regional AREMP protocols and analysis techniques. Exploring innovative ways to make informed assessments for aquatic and riparian condition is critical to successful resource management. The role of AREMP can evolve in this regard, as it continues to work with the broader monitoring community, researchers, and partners to further our understanding of aquatic systems of the ACS.

**Components of Aquatic Conservation Strategy**
Riparian reserves—

Riparian reserves were intended to define and delineate the outer boundaries of the riparian ecosystem and to encompass the portions of a watershed most tightly coupled with streams and rivers (FEMAT 1993). These areas were assumed to provide the ecological functions and processes necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time. This includes dispersal corridors for a variety of terrestrial and riparian-dependent organisms, and connectivity of streams within watersheds (FEMAT 1993). The Forest Ecosystem Management Assessment Team (FEMAT) (1993) developed three management scenarios for riparian reserves, each requiring a reserve width on fish-bearing streams of two times the height of a site-potential tree (minimum of 300 ft [91.4 m]), which is defined as a tree that has attained the average maximum height possible given the conditions where it occurs. Management scenarios for non-fish-bearing streams varied from a width equal to one-sixth of a site-potential tree-height (minimum of 25 ft [7.6 m]) to one-half of a site-potential tree-height to one site-potential tree-height (FEMAT 1993). One scenario was integrated into each of the 10 landscape alternatives developed and evaluated by the FEMAT (1993) scientists.

The Secretaries of Interior and Agriculture selected FEMAT’s Option 9 as their preferred option, which required a riparian reserve network that was two site-potential tree-heights wide on fish-bearing streams and one-half of a site-potential tree-height on most non-fish-bearing streams. Interim boundaries of the riparian reserves were extended to a full site-potential tree-height on all non-fish-bearing streams between the draft and final environmental impact statements (USDA and USDI 1994a) to increase the likelihood of success of the ACS, and to provide additional protections from timber management and road building for nonfish organisms.
that use the area in or near streams as habitat or migratory corridors (USDA and USDI 1994a).

On some fish-bearing streams, one site-potential tree-height from the edge of a stream may not encompass the entire floodplain, which can be an important source of large wood (Latterell and Naiman 2007). In such cases, it is critical to recognize and protect the entire floodplain. This was accomplished in the ACS by requiring the boundary of the riparian reserve to extend to the edge of the 100-year floodplain (USDA and USDI 1994a). These boundaries were considered interim until a watershed analysis, which could adjust the size of the riparian reserve, was completed (USDA and USDI 1994a).

Depending on the degree of dissection of the forested landscape by streams, riparian reserves along both perennial and intermittent streams may occupy between 40 and 90 percent of the landscape (FEMAT 1993, Hohler et al. 2001). Interim riparian reserves of this magnitude, coupled with key watersheds and late-successional reserves, have provided a connected watershed-level reserve system for terrestrial, riparian, and aquatic ecosystems (Everest and Reeves 2007). However, the area of the forested landscape contained in the riparian reserves has fueled a controversy regarding riparian protection, resulting in new research to evaluate prescribed widths of riparian management areas and a re-examination of existing scientific literature on the subject (Everest and Reeves 2007). The following summarizes some of the recent key literature relating to the functions and size of riparian reserves.

**Ecological functions**—

The scientific basis for delineation of interim riparian reserves in the NWFP was derived from two sets of curves showing the relationship between various ecological functions provided by riparian zones and distance from the channel (figs. 5 and 6). These curves were developed by
FEMAT scientists based on the scientific literature that was available at the time, and on professional judgment where sources of information were incomplete (see table 2 for original sources). The original relationships (FEMAT 1993) that were incorporated into the NWFP (USDA and USDI 1994a) suggest that most ecological functions could be maintained by reserves equal to or less than the distance of one site-potential tree-height. The functions include beneficial effects of root strength for bank stability, litterfall, shading to moderate water temperatures, and delivery of coarse wood to streams (fig. 5). In addition, the majority of moderating effects on sediment delivery to streams from overland erosion associated with upland activities generally occur within a distance of one site-potential tree-height (Castelle et al. 1994, Naylor et al. 2012). The FEMAT scientists also provided a margin for error allowing for incomplete science, unknown cumulative effects, or strategic uncertainty in defining interim riparian reserves prior to watershed analysis. Everest and Reeves (2007) concluded that science published since original development of the FEMAT curves has generally supported the original assumptions and judgments.

Recent studies of wood recruitment suggest that changes in some of the ecological function curves may be supported. The graph of the relationship between the cumulative effectiveness of an ecological process and the distance for wood recruitment from the immediately adjacent riparian area in fish-bearing streams, developed in FEMAT (1993), suggested that about 60 percent of wood recruitment from the immediate riparian area along fish-bearing streams occurs within one-half of a tree-height (fig. 5a). This graph was based on a limited number of studies (McDade et al. 1990, Van Sickle and Gregory 1990) and the professional judgment of scientists involved with FEMAT. More recent studies on the sources of
wood (Gregory et al. 2003, Spies et al. 2013, Welty et al. 2002) find that, at least in the Cascade Range of western Oregon and Washington, about 95 percent of the total instream wood inputs from the adjacent riparian area along fish-bearing streams came from distances that ranged between 82 to 148 ft (25 to 45 m) from the stream (fig. 5b). The shape of this curve differs from the FEMAT curve (fig. 5a), which showed that 95 percent of the wood-recruitment function of the same streams occurs within a distance equal to about 0.95 of the height of a site-potential tree.

A primary purpose for the extension of the boundary of the riparian reserve of the Northwest Forest Plan from one site-potential tree-height to two on fish-bearing streams was to protect and enhance the microclimate of the riparian ecosystem within the first tree-height (USDA and USDI 1994a). At the time the ACS was developed, the only research on the effects of clearcutting on microclimatic conditions in adjacent forests came from upland forests on level terrain (Chen 1991). Those studies found that the influence of recent clearcuts (10 to 15 years old) extended from tens of yards (e.g., soil moisture and radiation) to hundreds of yards (e.g., wind velocity) into adjacent unharvested stands. Based on the initial work of Chen (1991), FEMAT (1993) hypothesized that a second tree-height could provide a considerable safety margin to riparian area reserves in terms of relative humidity and other microclimatic effects in the riparian reserve along fish-bearing streams (FEMAT 1993) (fig. 6a).

Since the ACS and associated ecological function curves were originally formulated, a number of research efforts have examined the effects of forest management on microclimate in riparian areas. The vast majority of this work has focused on air temperature and relative humidity in small, headwater streams; few studies were conducted along larger streams (see
review by Moore et al. 2005; Olson et al. 2007, 2014). The magnitude of harvest-related changes in microclimate in riparian areas is usually inversely related to the width of the riparian buffer and the type and extent of management activities on the outer (upslope) edge. Some studies failed to show any edge effect between clearcuts and riparian buffers composed of intact mature forest (Anderson et al. 2007, Rykken et al. 2007). Other studies have shown that edge effects ranged from a distance of 98.5 ft (30 m) (Anderson et al. 2007, Rykken et al. 2007) to 148 ft (45 m) (Brosofske et al. 1997) from the stream. At the other extreme, Ledwith, (1996) found that above-stream temperature decreased and relative humidity increased as buffer widths increased up to 164 ft (150 m). Rykken et al. (2007) attributed the lack of an edge effect to a “stream effect”, described by Moore et al. (2005), who noted that the stream can act as a heat sink and source of water vapor during the day, thus keeping near-stream microclimates cooler and more humid than areas further from the stream. Rykken et al. (2007) suggested that this stream effect might counteract harvest edge effects on microclimate, thereby reducing the distance that harvest effects penetrate into riparian zones (Rykken et al. 2007) relative to the distances measured in upland forest edges (e.g., from those projected by Chen et al. [1993] in uplands). Moore et al. (2005) also suggested that cool, moist air might be carried by down-valley breezes, contributing to this stream effect.

The FEMAT microclimate curves were based on upland studies of forest edge effects and thus they do not necessarily apply to riparian areas with a strong stream effect, protected topographic positions, and retention of some canopy in the adjacent managed stand. Reeves et al. (2016) suggest that a one tree-height buffer on fish-bearing streams (fig. 6b) would reduce most potential effects on microclimate and water temperature in near-stream environments from
timber harvest in areas on the edge of the riparian reserve, particularly when some trees are retained in the harvest unit. In general, most studies show that microclimatic changes in temperature and relative humidity seldom extend further than one site-potential tree-height from the clearcut edge into an intact riparian buffer composed of mature forest (see review by Moore et al. 2005 and references cited therein). However, the large range of effects measured in different studies demonstrates that substantial uncertainties remain. These uncertainties have important implications when considering changes in the width of the NWFP riparian reserves.

Increased stream temperature following forest harvest is one of the most frequently mentioned management concerns, and one that retention of riparian buffers is clearly designed to mitigate. Generally, the smaller the riparian area and the more extensive the activities, the greater the effect on stream temperature. Clearcut logging without riparian buffers usually leads to large, post-harvest increases in stream temperature, and the width of the riparian buffer needed to limit, or even eliminate, temperature increases remains uncertain (see reviews by Moore et al. 2005 and Leidenbach et al. 2013). Given these uncertainties, management prescriptions that reduce the width of the riparian reserve or allow some tree harvest within the reserve remain controversial.

The NWFP area encompasses a wide range of bioclimatic conditions, across its latitudinal range, west to east with distance from the ocean and rain shadow effects of mountain ranges, and with elevation. Given this variation, we describe only a few broad general patterns in riparian vegetation here. In subsequent sections, we contrast these patterns with present-day patterns in previously logged areas, especially where logging pre-dated the establishment of current forest practices rules and allowed harvest right up to stream banks. Although these general trends are important considerations, we also emphasize that more detailed local
knowledge will be critical for determining appropriate management goals and planning specific actions.

A number of studies have shown that both riparian and upland vegetation along headwater streams in moist or wet forest types are typically dominated by conifers (Nierenberg and Hibbs 2000, Pabst and Spies 1999, Sheridan and Spies 2005; fig. 7a). However, conifer density can be lower in riparian zones (Sheridan and Spies 2005), hardwoods are uncommon in both riparian and upslope areas, and there are no clear differences in shrubs between the two zones (Sheridan and Spies 2005). Many mosses and liverworts are also found at the wetted edges of small streams or on wood and rock in and along the channels (Hylander et al. 2002). In these wet forest types, tree canopies are often dense, limiting sunlight and therefore primary productivity. As a result, headwater streams depend on allochthonous (coming from outside the stream) inputs of litter and terrestrial invertebrates from the riparian forests as the primary energy source for aquatic and riparian organisms (Gomi et al. 2002, Leroy and Marks 2006, Richardson et al. 2005, Wallace et al. 1997, Wipfli 1997). Further, allochthonous litter inputs often come from areas very close to the stream. For example, Bilby and Heffner (2016) found that 95 percent of litter inputs came from within 18 ft [5.6 m] of the channel. Allochthonous material is exported to downstream areas as dissolved organic carbon (DOC), coarse (>1-mm, >0.04 in) particulate organic matter (CPOM), and to a much lesser extent, as fine particulate organic matter (FPOM) (Gomi et al. 2002, Richardson et al. 2005) and contributes to the productivity of fish-bearing streams.

Forests in riparian zones and adjacent uplands become increasingly different as the size of streams increases (fig. 7b). In the middle portions of stream networks, the riparian forest is
more diverse than along headwater streams (Acker et al. 2003, Johnson et al. 2000, Sarr and
Hibbs 2007). Riparian forests along many mid-sized streams still remain dominated by conifers,
but they are often mixed with deciduous trees such as alder (*Alnus* spp.), willow (*Salix* spp.), and
cottonwood (*Populus* spp.). Big-leaf maple (*Acer macrophyllum*) and California black oak
(*Quercus kelloggii*) can be common in riparian zones in the southern portion of the NWFP area.

Some studies in mid-sized streams have shown that conifer density nearest the stream can
be lower due to reduced survival from disturbances such as flooding (Hibbs and Giordano 1996,
Pabst and Spies 1999). Also, the availability of growing sites might be limited; conifers
preferentially establish on “microtopographic ridges” created by old tree-falls and behind wood
jams (Fetherston 2005, Fetherston et al. 1995). As a result, tree density near the stream can be
about half of that of upland stands (Acker et al. 2003, Rot et al. 2000, Wimberly and Spies
2001). Other studies suggest that there is little difference in tree density between upland and
riparian stands (Pollock et al. 2012). All these studies, however, did find that the basal area of
conifers in streamside stands is greater than in stands further from the channel or in adjacent
uplands. Streamside trees can be among the largest in a watershed (Poage and Tappeiner 2002),
and thus can be the source of the largest down trees (conifers) found in the channel (i.e., the key
pieces) (Rot et al. 2000), which are generally recruited to the channel by undercutting at high

Hardwoods are most abundant in the area closest to the channel of streams in the middle
portion of the stream network (Pabst and Spies 1999, Wimberly and Spies 2001), particularly in
unconstrained reaches (Acker et al. 2003), and decrease in density moving away from the
channel. This mix of hardwoods and conifers is important ecologically (Sponseller and Benfield
2001, Sponseller et al. 2001) and is frequently maintained by periodic flooding (Sarr and Hibbs 2007). The vegetative diversity provides diverse habitat for a suite of terrestrial and riparian organisms; hardwoods are especially important for riparian mollusks (Foster and Zieglerum 2013) and neotropical migrant and resident bird species (Pearson and Manuwal 2001). Riparian areas dominated by hardwoods, particularly nitrogen-fixing red alder (Alnus rubra), have the potential to increase primary and secondary productivity and invertebrate diversity in adjacent streams (Piccolo and Wipfli 2002, Wipfli and Musslewhite 2004). Hardwood riparian vegetation also influences the structure and composition of the associated aquatic invertebrates through top-down diversity effects on consumers (Srivastava et al. 2009) and bottom-up effects on the dominant litter traits (Cornwell et al. 2008, Kominoski and Pringle 2009, Kominoski et al. 2011, Schindler and Gessner 2009, Swan et al. 2009). Additionally, watersheds with mixed hardwood-coniferous riparian vegetation in the Oregon Coast Range received nearly 30 percent greater influx of terrestrial invertebrate biomass than streams with conifer-dominated riparian areas (Romero et al. 2005).

Although these broad general patterns hold across much of the NWFP area, we do not mean to give the impression that the riparian forests and their adjacent uplands were uniformly forested. Rather, these forests were a complex shifting mosaic of vegetation patches, presenting a landscape with high spatial variability and temporal dynamics. Wildfire was the primary factor driving forest dynamics across the Oregon Coast Range (Wimberly et al. 2000), although wind throw, insects, and disease can also be important. As a consequence, upland forests, even when assessed at large spatial scales, showed substantial variation in the area and ages of forests. For example, over long time periods, the proportion of upland forest in old-growth condition, when
summed over areas greater than 4.9 million ac (2 million ha), ranged from 25 percent to 75 percent; at the scale of late-successional reserves as specified in the NWFP (~40,000 ha), the amounts of old growth could range from 0 percent to 100 percent (Wimberley et al. 2000). Thus, riparian forest conditions, when assessed over broad landscapes, showed a distribution of conditions that resulted from the combined influences of natural disturbance and plant succession.

Wondzell et al. (2012) found similar patterns for riparian forest in the Oregon Coast Range. In addition to the factors described above for upland forests, riparian forests are also influenced by fluvial and geomorphic processes such as floods, debris flows, and bank erosion. State and transition simulations of the natural disturbance regime showed that 51 percent of the riparian network was in mature forest (stand age of 66 to 200+ yr). The simulations also showed that the long-term average forest composition was highly variable. Only 2 percent of the riparian network was in a non-forested condition, 28 percent was alder dominated, 40 percent was in mixed alder-conifer stands, and only 29 percent was in conifer-dominated stands (see table 5 in Wondzell et al. 2012). The specific results cited above pertain only to the central Oregon Coast Range, but it is clear that no single condition—defined by stand composition, structure, and tree age—can represent the full distribution of naturally occurring conditions over large areas. Rather, riparian forest conditions, when assessed over broad landscapes, showed a distribution of conditions that resulted from the combined influences of natural disturbances and plant succession.

Human impacts and restoration—
Riparian forests throughout much of the NWFP area have been changed by the land-use activities that have taken place over the last century. As a consequence, the present-day forests may frequently differ in structure and composition from the pre-settlement forests that preceded them (McIntyre et al. 2015, Naiman et al. 2000, Swanson et al. 2011). This is particularly evident in an estimated 30–35 percent of the riparian ecosystems in the NWFP area (T. Spies, pers. comm.), where riparian forested areas were harvested extensively, often to the edge of the stream, prior to the advent of current policies (Everest and Reeves 2007). In many cases, the riparian zones were subsequently planted with the most commercially valuable conifers, primarily Douglas-fir (*Pseudotsuga menziesii*), resulting in the development of dense, relatively uniform conifer stands and a decrease in hardwoods. In other cases, conifers were not successfully reestablished in logged riparian zones that are now dominated by alder with dense salmonberry (*Rubus spectabilis*) understory, as observed at the reach scale by Hibbs and Giordano (1996), and also noted in watershed-scale simulations (Wondzell et al., 2012), where under historic conditions 28 percent of the stream network was in alder-dominated riparian forests; today, alders dominate riparian zones over 40 percent of the stream network. Similarly, the removal of large conifers along rivers in the coast redwood (*Sequoia sempervirens*) range of Northern California has been associated with increased dominance by red alder (Madej et al. 2006). Clearly, the direct impacts of logging on the structure and composition of present-day riparian forest can be varied, but overall, have dramatically changed the distribution of conditions relative to those under natural disturbance regimes.

Indirect effects of logging have also modified riparian forests. For example, rates of landslides and debris flows have increased in heavily roaded and logged watersheds (Goetz et al.}
2015, Guthrie 2002, Jakob 2000), and have led to systematic changes in riparian vegetation. Debris-flow tracks are frequently scoured free of large wood and subsequently recolonized by red alder (Russell 2009, Villarian 2009), with large wood deposited in runout zones. Further, the frequency of debris flows and landslides has contributed additional sediment to stream channels, driving more severe floods, with the combined effect of increasing the width of stream channels (Lyons and Beschta 1983). Exposed gravel bars within these channels are most often colonized by hardwoods, leading to substantial changes along the stream corridor.

Fire suppression in forests with high- or moderate-frequency fire regimes has likely altered the structure and composition of riparian vegetation in ways similar to those described for upslope forested areas—namely an increase in the density of shade-tolerant conifers and a reduction in hardwoods (chap. 3 this volume). A shift in fire regimes toward less frequent and more severe wildfires is likely to cause significant changes in wood recruitment (Benda and Sias 2003); greater tree mortality and formation of debris jams in severely burned areas may shift wood dynamics toward infrequent, larger jams in a “feast or famine” cycle rather than more frequent and distributed influence of individual fallen trees. Less frequent, more extreme fires could lead to formation of debris jams that would eventually degrade without large trees available to replace them. A more frequent disturbance regime that maintains steadier wood inputs might help to better sustain associated habitat features over time.

**Restoration challenges**—

The changes to riparian forests, described above, create substantial challenges for restoration. For example, thinning of dense riparian Douglas-fir stands could open stands, allowing increased hardwood presence and thereby increasing the diversity of riparian vegetation, while also
promoting growth of the remaining trees to decrease the time needed to grow trees large enough
to act as key structural elements in the stream channel. Although such restoration treatments
might speed the restoration of many ecological functions (USDA and USDI 1994a), they also
present potential risks. The trade-offs between potential gains and potential risks appear to have
limited restoration activities, particularly within the first site-potential tree-height of streams.

Reeves (2006) estimated that 48,000 acres (19 400 ha) in the matrix of the Northwest
Forest Plan area was logged in the first 10 years of the plan. Between 2010 and 2015, an
additional estimated 38,719 acres (15 669 ha) in Region 6 had management activity (table 3).
FEMAT (1993) estimated that there were 2.2 million acres (890 000 ha) of riparian reserves
outside of other reserves and congressionally withdrawn areas in the NWFP area. Because this is
not the total area of riparian reserves, it is not possible estimate the fraction of the riparian
reserve in the NWFP area that has undergone restoration. However, it is clear that the area that
has been treated to restore ecological diversity represents a relatively small proportion of the
 riparian reserves in total and of the amount that has been altered by past activities.

Primary reasons for the limited amount of restoration activity are various and probably
include: (1) differing perspectives about the characterization of reference conditions,
conservation, and management; (2) concerns about the potential effects of mechanical treatments
on stream temperature and wood recruitment; and (3) concerns about rare and little-known
organisms that made managers reluctant to alter default prescriptions (Reeves 2006). We explore
the potential challenges associated with these restoration activities below.

Reference condition versus reference distribution—Restoration activities necessarily
require a “target”, a condition toward which the restoration activity is intended to push a system.
One way to select a target for restoration goals is to identify a minimally disturbed condition and use it as a reference to which the current condition can be compared. The minimally disturbed condition is commonly called the reference condition. Although intellectually appealing, the selection of a reference condition is not simple. For example, Pollock et al. (2012) set very stringent requirement on stand attributes that would be acceptable as a reference condition: choosing completely undisturbed, single-storied, conifer-dominated stands ranging in age from 80 to 200 years, discarding any stands that showed evidence of recent severe disturbance—including disturbances such as wildfire, insects, and disease, because these disturbances may themselves have been changed by anthropogenic fire suppression or climate change.

Pollock et al.’s (2012) study illustrates one of the problems inherent in finding undisturbed reference conditions—they are rare. Ideally, reference conditions would be identified in stands with similar potential vegetation and in relatively close proximity, or at least within the same ecoregion, so that the reference provides an appropriate comparison for similar forest stands (NRC 2000). Because of their stringent conditions, Pollock et al. (2012) were only able to identify a small number of reference stands that were widely scattered, spanning a broad latitudinal and climatic range. Further, they lumped together both upland stands and riparian stands, and the only riparian reference stands were located in the western Washington Cascade Range. Thus, Pollock et al.’s (2012) reference condition may not be appropriate to apply to riparian stands outside of western Washington or stands of other ages. However, they did focus their analyses on tree-based stand metrics (e.g., stand density and tree size) that might be somewhat similar across the region,
Miller et al. (2016b) used a different approach to identify reference conditions. First, they used monitoring plot data collected across a large number of agencies working in the NWFP area to assemble over 5,000 riparian plots that could potentially serve as reference conditions. Second, they eliminated plots that were likely to be anthropogenically disturbed by using external GIS databases of roads, mines, and other land uses in the watershed within which each plot was located. Finally, they grouped reference plots by ecoregion to help insure that the reference plots would be an appropriate comparison for other watersheds in that ecoregion. Miller et al. (2016b) do not define a specific reference condition. Rather, they define a range of values for any given metric describing the “reference condition”, a range determined by the conditions currently present within relatively little-disturbed watersheds. Like Pollock et al. (2012), Miller et al.’s (2016b) assessment was limited by the availability of little-disturbed reference plots, and no reference plots could be found in several highly disturbed ecoregions—namely the Willamette Valley and the Puget Sound Lowlands.

Part of the debate about restoration needs for riparian areas may derive from a lack of understanding of riparian reference conditions (as a goal for restoration), how they vary with scale, and across watersheds and the NWFP region. Although many studies (e.g., Acker et al. 2003, Hibbs and Sarr 2007, Pabst and Spies 1999), as mentioned previously, have found that riparian vegetation and upland vegetation frequently differ in structure, composition, and dynamics depending on stream size, some have argued that the differences between riparian and upland vegetation are small and that upland conditions can be used as a reference target for riparian management. For example, Pollock et al. (2012) argued that “Both forest types [upslope and riparian] are generally similar, but riparian stands have more live tree wood volumes and
basal areas, suggesting they may be growing on sites that are more productive”. They contended that restoration in riparian zones across western Washington and Oregon should primarily promote conifer-dominated stands with densities and sizes of live and dead trees that are similar to or exceed those of uplands. To understand this perspective, it is important to understand the design of the study (Schuett-Hames et al. 2005) that formed the basis of the reference conditions used by Pollock et al. (2012). Schuett-Hames et al. (2005) targeted mature conifer-dominated riparian stands in the western Cascades of Washington, and the study “was not designed or intended to sample from all stand types, stream types…or age categories from riparian vegetation.” Nor did it sample any riparian forests in Oregon, and all but one of the riparian forests came from the western Washington Cascades. Thus, it does not appear that the results from Pollock et al. (2014) can be extrapolated to the full range of unmanaged riparian vegetation conditions that exist, or to riparian vegetation outside of western Washington. Others (Gregory 1997, Pabst and Spies 1999, Welty et al. 2002, Wimberly and Spies 2001) have found that the type and magnitude of differences in features between upslope and riparian forests can be large, suggesting that upslope vegetation should not be assumed to be a reference for designing and assessing managed strategies for riparian vegetation. The differences of opinion regarding the conservation and restoration goals and approaches for riparian areas have made it difficult for managers and regulators to design and implement management actions in riparian reserves. Wondzell et al. (2012) used “state and transition” models to explore the range of ecological states of the riparian network of a large river network in the central Oregon Coast Range. They first used a GIS tool to divide the stream network and its valley floor into discrete reaches. Each reach was classified into a potential geomorphic and vegetation type and a state-
and-transition model was developed for each potential type that included all possible states that could result from succession, natural disturbance, and land-use activities. Transitions among states were based on likely reoccurrence probabilities of natural disturbances and the frequency at which land-use practices occur. Stand development through time was simulated as a deterministic process based on underlying growth rates of trees and development of other vegetation types. The model frameworks were based on similar models developed to simulate upland forest dynamics in the region. However, the upland models were modified to better represent expected differences from riparian stands and to include disturbances that were unique to riparian areas.

Wondzell et al. (2012) clearly stated that their simulation results “should be interpreted as hypotheses of likely outcomes”. Development of their model state-classes was limited by the lack of examples of undisturbed state-classes, forcing them to rely on expert judgement rather than empirical data. Further, validation data were not readily available. They did use a combination of field sampling and lidar-based analyses of stand structure and composition to generate current conditions, and the model simulations of the current condition seemed reasonable. However, those comparisons do not constitute robust validations of the models. Despite these limitations, the models can be used to “hindcast” expected historical distribution of riparian forest conditions.

We suggest that each of the approaches examined above—that of Pollock et al. (2012), Miller et al. (2016b), and Wondzell et al. (2012)—offers important ideas that would contribute to building a “reference condition” based strategy to evaluate current conditions, to evaluate likely outcomes of planned management activities, and to help evaluate the trade-offs between
potential risks and benefits of any planned management strategy. For example, a modeling
approach like the state-and-transition models of Wondzell et al. (2012) could be used to generate
an expected historical distribution of states for the riparian vegetation within a stream network.
The approach of Miller et al. (2016b) could be used to identify relatively little-disturbed
watersheds within the ecoregion and the monitoring plots located within those watersheds. The
individual plots could then be “cross-walked” back to specific states in the state-and-transition
model, and those states could be attributed with values for various metrics, much as was done by
Pollock et al. (2012). Because anthropogenically disturbed states are also included in the state-
and-transition models, something similar could be done to attribute these states with empirical
data. The models could then be used to hind-cast the historic distribution of state classes, and
descriptive metrics from the empirical data could then be linked to the historic distribution. This
result could be compared to the current condition. Also, the models could be used in forward
simulations, incorporating different land-use choices, to project how the distribution of
conditions might be expected to change over time in response to various management strategies.

Riparian thinning and stream temperature—Because the current distribution of
conditions of riparian forests in many stream networks is far different from the historic
distributions, there is substantial interest in active restoration treatments—especially thinning
dense conifer plantations (Reeves et al. 2016) or logging hardwood-dominated stands and
replanting to convert them to conifer dominance (Cristea and Janisch 2007). Although these
treatments are not inconsistent with the ACS, which generally allowed thinning for ecological
objectives in the area beyond 120–150 ft (36.6–45.7 m) to a distance of one site-potential tree-
height, they could potentially exceed the 0.3°C “non-degradation standard” for water-quality
impacts from logging. The 0.3°C standard is critically important, limiting potential cumulative effects from multiple actions, none of which individually might be sufficient to impair water quality. Alternatively, restoration treatments might speed the attainment of desired future conditions. These decisions pose critical management challenges. Clearly, there are risks from any active restoration treatment, but choosing not to act also poses risks, not only by increasing the time needed to attain a desired future condition, but also leaving the riparian zone at greater risk of uncharacteristic disturbance—for example, dense conifer stands might be more prone to wildfire.

Reach-scale studies clearly demonstrate that solar radiation is the primary factor affecting stream water temperatures during summer (Leinenbach et al. 2013). Thus, the likely effect of forest harvest on stream temperatures will be a function of the amount of shade lost. The largest effects will generally be seen with clearcut logging right to the stream banks, whereas retention of forested buffers tends to reduce these effects, as does thinning rather than clearcutting outside the buffer. The actual magnitude of stream-temperature increases can be highly variable, in part determined by factors such as discharge, water depth, width, and flow velocity, hyporheic exchange, and groundwater inflows (Janisch et al. 2012, Johnson 2004, Moore et al. 2005). Topographic shading can also influence water temperatures, particularly in small streams flowing in narrow, steep-sided valleys, as much as or perhaps more than shade from streamside forests.

Unfortunately, relatively few studies have examined the effects of riparian thinning on stream water temperature. A few studies have examined clearcut harvesting combined with partial harvest of riparian buffers (Kreutzweiser et al. 2009, Macdonald et al. 2003, Mellina et al.
2002, Wilkersen et al. 2006). These studies, like those cited above, suggest that the effect of riparian thinning on summer stream temperatures will be positively correlated with the amount of forest canopy removed, and thus the amount of shade lost (Leinenbach et al. 2013). However, even fairly substantial thinning may not substantially change angular canopy density, so that the amount of shade lost can be much smaller than the amount of tree basal area removed (Kreutzweiser et al. 2009). Further, any shade loss and stream temperature increases from riparian thinning are likely to be short lived because riparian forest canopies can close relatively quickly (within 3 years) after thinning (Chan et al. 2006). The potential magnitude of stream temperature increases in response to riparian thinning will be highly dependent on the thinning prescription, but with light thinning these restoration treatments are unlikely to substantially increase stream temperatures.

Managers thus face the question: Are there places in the stream network where riparian thinning would help speed attainment of the reference distribution and where present day thermal regimes would suggest that small temperature increases would not have significant detrimental effects on fish (Groom et al. 2011) or other organisms of interest? This question tends to be examined at the reach scale. That is, examining the potential effects of a thinning treatment on the development of riparian forest stand attributes as done by Pollock et al. (2012), or summer maximum temperatures (Groom et al. 2011), in the treated reach. Rarely are these questions also examined in the context of the distribution of reference conditions across the larger watershed. Are the conditions of the treated reach over-represented with respect to the reference distribution, or under-represented? In the Oregon Coast Range, for example, it is clear that not all reaches would be maintained in conifer-dominated mature forest under a natural disturbance regime
(Wondzell et al. 2012). If dense, young, conifer-dominated stands are currently more abundant than expected from the reference distribution, then should some of those stands be thinned, perhaps mimicking wind-throw events that open stand canopies and allow development of multi-storied, mixed stands? And if so, how many should be treated to better change the long-term trajectory of conditions from the current distribution toward one that is closer to the reference distribution?

Riparian thinning and large, instream wood—The absence or diminished quantity of wood in streams throughout the NWFP area also is a primary concern for managers and regulators because wood is important for creating habitat and performing other ecological functions. Thinning and other active management in plantations in riparian zones can reduce the potential amount of wood that can be delivered to streams (Beechie et al. 2000, Pollock et al. 2012) and the forest floor (Pollock and Beechie 2014, Pollock et al. 2012) if the trees are removed from the site. Additionally, thinning may negatively affect habitat, at least in the short run, for some species that are favored by dense areas of conifers (See chap. 3 in this volume for more details), potentially increase water temperature (Leinenbach et al. 2013), and reduce carbon storage (D’Amore et al. 2015). However, there are also many potential benefits to thinning, including increasing structural diversity, species richness, flowering and fruiting of understory shrubs and herbs (Burton et al. 2014, Carey 2000, Hager et al. 1996, Muir et al. 2002), and faster development of mature-forest conditions including very large trees with thick limbs that may be used for nesting by marbled murrelets (*Brachyramphus marmoratus*) (Carey and Curtis 1996, Franklin et al. 2002, Tappeiner et al. 1997, chap. 5 this volume). Furthermore, variable-density thinning of the overstory in the second-growth riparian forest could accelerate recovery of old-
growth characteristics by promoting dominance of redwood in the southern portion of the NWFP area (Keyes and Teraoka 2014).

Considerable research on wood dynamics in the NWFP has been done in wet forests of Washington, California, and Oregon, but there has generally been less research in areas with drier forest types, including Northern California. Riparian areas in redwood-dominated forests are particularly distinctive due to the exceptional productivity, low mortality, and slow decay of those trees (Benda et al. 2002). Benda and Bigelow (2014) compared wood volumes across four different regions of Northern California, including the Coast Range, Klamath Mountains, Cascade Range, and Sierra Nevada, as well as variation associated with forest management. They noted that coastal streams had much higher wood volumes, which they attributed to higher forest biomass and growth rates of redwood forests, as well as slower decay of large wood pieces. They also observed that some second-growth forests along streams in that region had wood volumes comparable to those in old-growth forests, due to heavy debris remaining from tractor-era logging before the 1970s. Although the volumes were similar, streams in old-growth areas had fewer but larger logs (Benda et al. 2002). Benda and Bigelow (2014) also found that streams in the Cascades and Sierras that they characterized as more heavily managed had larger volumes of stream wood than less intensively managed areas in the same regions. They conjectured that managed forests could have higher rates of tree mortality due to stem exclusion than more mature, but not yet decadent, unmanaged forests.

A panel of scientists from the USDA Forest Service and NOAA Fisheries recently reviewed the published literature on the effects of thinning (Spies et al. 2013). The major conclusions are summarized below:
• Accurate assessment of thinning effects requires site-specific information. The effects of thinning regimes on dead wood creation and recruitment (relative to no-thinning) will depend on many factors including initial stand conditions, particularly stand density, and thinning prescription.

• Conventional thinning generally produces fewer large dead trees. Thinning with removal of trees (conventional thinning) will generally produce fewer large dead trees across a range of sizes over the several decades following thinning and the lifetime of the stand relative to equivalent stands that are not thinned.

• Thinning to develop old-growth structure is most beneficial in dense young stands less than 80 years and especially less than 50 years old.

• In terms of production of large dead trees, higher-density stands are likely to see more benefits from thinning, and lower-density stands fewer benefits.

• Conventional thinning can accelerate the development of very large-diameter trees. In stands that are conventionally thinned, the appearance of very large-diameter dead trees (greater than 40 in [102 cm]) may be accelerated by up to 20 years relative to unthinned plantations, depending on thinning intensity and initial stand conditions.

• Stands thinned with prescriptions that leave some or all of the dead wood may more rapidly produce both large-diameter dead trees in the short term and very large-diameter dead trees (especially greater than 40 in [102 cm]) in the long term, relative to unthinned stands.

• Thinning can increase the amount of pool-forming wood only when the thinned trees are larger in diameter than the average diameter of pool-forming wood (which varies with stream size).
• Effects of thinning on instream wood need to be placed in a watershed context. Assessing
  the relative effects of riparian thinning on instream wood loads at a site and over the long
  term requires an estimation of the likely wood recruitment that will occur from both the
  banks and from upstream transport, and the rate of decay and downstream transport of wood
  from the site.

• The ecological effects of thinning on instream habitat will vary depending upon location in
  the stream network. Riparian management practices can be varied to match the ecological
  functions of streams.

• Variation in thinning is essential (i.e., don’t do the same prescription everywhere).

Since Spies et al. (2013) summarized the state of the science, other studies have increased
our understanding of the effect of restoration thinning in riparian areas. Benda et al. (2015)
simulated the idea of adding wood to channels during thinning by modeling the amount of
instream wood that would result from thinning a stand from 400 trees/acre to 90 trees/acre (from
988 trees/ha to 222 trees/ha), which is considered a moderate amount of thinning, then
directionally falling or pulling over varying proportions of the trees scheduled for harvest (table
4). This was compared to the amount of wood that would be expected in the stream if the
existing stand was not thinned. Not surprisingly, the amount of wood increased above the “no
thin” level immediately after the tipping simulation in all of the wood-addition options.

However, the cumulative total amount of wood expected in the stream over 100 years relative to
the unthinned stand varied depending on the amount of wood delivered. Adding ≤10 percent of
the wood that would be removed during thinning produced less wood in the channel over time
than the unthinned option. When 15–20 percent of the volume of thinned trees from one side of
the stream was directed to the stream at each entry, the total amount of dead wood in the channel exceeded the unthinned scenario over time (table 4). Carah et al. (2014) found that adding unanchored wood into the stream was less costly than securing the wood, and improved habitat conditions for Coho salmon. Reeves et al. (2016) included wood addition (tree-tipping) as a component of options for managing the riparian reserves on O&C lands of the BLM in western Oregon to accelerate attainment of restoration objectives.

*Ecological trade-offs*—There are potential ecological consequences of limiting management only to the outer portions of the riparian reserves that have been considered in the assessment of proposed projects. The focus on single values, such as water temperature or wood recruitment, comes at the expense of recognizing the myriad ecological processes that create and maintain the freshwater habitats of Pacific salmon (Bisson et al. 1997, 2007) and the ecological context in which they evolved (Frissell et al. 1997). This is especially relevant to the goals of the ACS, which are broad and include more than aquatic conditions. In recognition of this issue, Holling and Meffe (1996) referred to a single focus as an example of a “command-and-control approach” to natural resource management. They contend that this approach often fails when it is applied to situations in which processes are complex, non-linear, and poorly understood, such as in aquatic ecosystems in the NWFP area, and that it may lead to further degradation or compromising of the ecosystems and landscapes of interest (Dale et al. 2000, Hiers et al. 2016, Rieman et al. 2006). Managing for a single purpose (e.g., maximizing dead wood) may compromise or retard other ecological functions, such as development of hardwoods and shrubs or sources of the largest trees, which are found in areas of low density near the stream (see previous discussion), and ultimately may alter the structure of the food web (Bellmore et al.
Pollock and Beechie (2014) state that “Species that utilize large-diameter live trees will benefit most from heavy thinning, whereas species that utilize large-diameter deadwood will benefit most from light or no thinning. Because far more vertebrate species utilize large deadwood rather than large live trees, allowing riparian forests to naturally develop may result in the most rapid and sustained development of structural features.” We agree that trade-offs exist and that prioritization will be needed.

The choice of priority conservation targets (e.g., dead wood, plant community diversity, large live trees, geomorphic disturbances) for riparian management is a difficult one to make, involving scientific criteria, risk assessment, and social values. Given the diversity of conditions in riparian areas at watershed and regional scales, it would make sense not to apply one-size-fits-all strategies but rather to develop priorities based on a watershed-scale view (see below). For example, Pollock and Beechie (2014) state “Management strategies that seek to create a range of large live and dead tree densities across the landscape will help to hedge against uncertain outcomes related to unanticipated disturbances, unexpected species needs, and unknown errors in model assumptions.” It will be important to consider the full suite of ecological functions across a watershed; focusing only on one condition or metric may limit recovery of riparian ecosystems in ways that prevent full achievement of the broad objectives of the ACS. Given these broad objectives, a more comprehensive watershed- and regional-scale consideration of all ecological processes, and studies to develop new and more complete approaches, may be more fruitful than focusing only on one or two metrics.

A context-dependent approach to riparian conservation and management—
A key component of the ACS is watershed analysis (FEMAT 1993), which is supposed to provide the context of a given location for adjusting the boundaries of, and allowing activities within, riparian reserves. However, the former intent of watershed analysis was never realized (Reeves et al. 2006), owing to a number of factors, including cost of analysis and the need to consider a multitude of species and their ecological requirements. Neither FEMAT (1993) nor the NWFP (USDA and USDI 1994a) provided explicit criteria for changing the boundaries or demonstrating that proposed changes would meet or not prevent attainment of ACS objectives over the long term. In addition, at the time, credible analytical tools to aid decision making were lacking (Reeves 2006), and a fixed-width approach is easy to administer and apply, and is less costly than developing site-specific recommendations (Richardson et al. 2012). As a result, adjustments have proven difficult for the agencies to make (Naylor et al. 2012, Richardson et al. 2012), and interim boundaries of the riparian reserves remained intact in the vast majority of watersheds (Baker et al. 2006).

Since the development of the ACS, there has been a call in the scientific literature to allow discretion in setting site-specific activities (Lee et al. 2004, Pollock and Beechie 2014, Richardson et al. 2012), which can be economically beneficial (Tiwari et al. 2016). Greater flexibility in the management of riparian areas would depend on the “context” of the area of interest (Kondolf et al. 2006, Montgomery 2004), and the primary management objective for the specific area (Burnett and Miller 2007). However, development of such an approach has been limited because of the reliance on “off-the-shelf” and one-size-fits-all concepts and designs, rather than on an understanding of specific features and capabilities of the location of interest (Kondolf et al. 2003, Naiman et al. 2012). A mix of approaches could be undertaken, recognizing
ecological and other goals such as timber harvest, especially if applied over larger spatial scales (Burnett and Miller 2007, Miller and Burnett 2008, Olson and Rugger 2007) and if consideration is given to the distribution of populations of concern and connectivity among them (Olson and Burnett 2009, Olson and Kluber 2014, Olson et al. 2007).

There have been a few attempts to design and implement a site-specific approach. Cissel et al. (1999) proposed a plan based on variation in the disturbance patterns (in this case, wildfire) in the target watershed, and called for harvest of some older trees and a revision of the interim riparian reserves for the Central Cascades Adaptive Management Area. Olson and Rugger (2007) proposed a two-tiered approach to riparian management to first identify reaches where sensitive species occur, then manage their critical habitat elements, hence varying riparian reserve management with species distributions. Olson and Burnett (2009) applied sensitive-species filters to criteria for designations of connectivity habitat. Inter-watershed connections provided by riparian areas are critical avenues of movement to new habitats. None of these approaches have been implemented to date.

Reeves et al. (2016a) proposed a more comprehensive context-dependent approach for management of the riparian reserves in the matrix of federal lands in western Oregon that divided the riparian reserve into inner and outer zones, with management tailored to the specific features and characteristics of individual stream reaches (Option B of Reeves et al. 2016a). The context-dependent option was informed by new research, tools, and concepts, including:

- The influence of the width of riparian area on microclimate (see earlier discussion)
- Movement of amphibians along non-fish-bearing streams (Olson and Burton 2014, Olson et al. 2007)
• The distance to, and sources of, wood for fish-bearing streams (Spies et al. 2013)

• Intrinsic potential, a concept for assessing the capability of a given set of geomorphic conditions in a stream reach to provide habitat for selected species of Pacific salmon (Burnett et al. 2007)

• NetMap (Benda et al. 2007), a geo-spatial platform for watershed analysis that can, among other things, identify the location of key ecological processes that influence aquatic and riparian ecosystems on the landscape and in the stream network

• Concepts for managing riparian ecosystems and the activities that affect them, such as ecological forestry (Franklin and Johnson 2012) and tree-tipping (Benda et al. 2015)

Under the context-dependent option, current interim riparian reserves of two site-potential tree-heights along fish-bearing streams and one site-potential tree-height along non-fish-bearing streams would be retained in late-successional reserves and other special land designations (Reeves et al. 2016a). In lands allocated as matrix under the NWFP, the area of interest for aquatic conservation (which Reeves et al. [2016a] referred to as the riparian conservation area) extended upslope from the stream for a distance equal to the height of one site-potential tree along fish-bearing and non-fish-bearing streams. The riparian conservation area was divided into an inner and an outer zone depending on “ecological context,” based on four characteristics of each stream reach—susceptibility to surface erosion, debris flows, thermal loading, and habitat potential for target fish species—to determine the width of the inner zone.

The entire riparian conservation area of the most ecologically sensitive stream reaches along fish- and non-fish-bearing streams was managed solely for ecological goals. In other fish-bearing and non-fish-bearing streams, the inner zone was 100 ft (30.5 m) and 50 ft (15.3 m) wide,
respectively (Reeves et al. 2016a). Active management was limited to stands age 80 or younger (Spies et al. 2013), and tree-tipping (Benda et al. 2015) was used throughout the riparian reserve to ensure that harvest did not negatively affect wood recruitment to the stream (table 4).

Using the matrix in BLM managed lands in western Oregon to illustrate the application, Reeves et al. (2016a) estimated an average of 46 percent of the riparian reserve in a watershed would be managed solely for the ACS goals. Also, an estimated average 36 percent would achieve ACS goals along with other potential goals, which could include timber production, and 18 percent could be returned to the terrestrial prescriptions of the NWFP (Reeves et al. 2016a).

In late-successional and other reserve allocations, which cover approximately half of the BLM lands in western Oregon, interim riparian reserves would remain unchanged. Assuming that half of the interim riparian reserves on BLM lands in western Oregon would remain unchanged, and applying their study estimates of changes in matrix, Reeves et al. (2016a) estimated that about 72 percent of the interim riparian reserves would remain solely devoted to ACS goals, and 19 percent would likely meet ACS goals and could also provide opportunity for achievement of matrix goals including limited timber production. The reduction of the width of the riparian reserve along fish-bearing streams to one tree-height would return an estimated 9 percent of interim riparian reserves to matrix on these lands.

The analysis of Reeves et al. (2016a) was not intended to provide a single recommendation for managing riparian ecosystems. The primary purpose was to re-evaluate riparian conservation strategies using the latest scientific evidence. This or other options should be viewed as working hypotheses to be tested with monitoring and adaptive-management experiments. The analysis provides an example of how a context- and landscape-dependent
approach could be designed to address multiple conservation goals of the ACS, the commodity
goals of the NWFP, and the significant challenges of climate change. Although new science
has refined our understanding of the ecological processes in riparian ecosystems, uncertainties
and information needs remain. Thus, an adaptive-management approach and further research is
critical for continual improvement and evaluation of this and other options for meeting the goals
of the ACS (Stankey et al. 2005).

Key watersheds—

Tier 1 key watersheds (a total of 141, covering 8,154,500 acres (3,300,000 ha) (fig. 8)
were intended to serve as refugia for aquatic organisms or to have high potential for restoration
(USDA and USDI 1994a). Tier 2 key watersheds provide sources of high-quality water, and
comprised 23 watersheds covering a total of about 1,112,000 acres (405,000 ha) (fig. 8). Key
watersheds are aligned as closely as possible with the late-successional reserves of the NWFP
(areas designated to protect late-successional and old-growth ecosystems) and other officially
designated reserve areas to maximize ecological efficiency (USDA and USDI 1994a), and to
minimize the amount of area in which timber harvest activities were restricted. A primary
objective for tier 1 key watersheds is to aid in the recovery of ESA-listed fishes, particularly in
the short term (FEMAT 1993). Tier 1 key watersheds currently in good condition were assumed
to serve as centers for potential recovery of depressed populations. Those with currently
degraded conditions were expected to have the greatest potential for restoration and to become
future sources of good habitat.

Ecological conditions in key watersheds have improved more than in non-key watersheds
(Gallo et al. 2005, Lanigan et al. 2012, Miller et al. 2016a). The primary reason for this
difference was that more than twice as many miles of roads were decommissioned in key
watersheds as in non-key watersheds (Gallo et al. 2005, Lanigan et al. 2012), suggesting that
land-management agencies appear to recognize key watersheds as priority areas for restoration.

Key watersheds were originally selected based on the professional judgment of the
scientists involved with the development of the ACS, in consultation with fish and aquatic
biologists and hydrologists from the national forests and BLM districts covered by the NWFP.
Also, they were tightly aligned with late-successional/old-growth reserves, based, in part, on the
assumption that streams in old-growth forests would be most favorable for fish (FEMAT 1993).
New techniques (e.g., NetMap Benda et al. [2007]) and understandings of aquatic ecosystems
now provide a different perspective on aquatic ecosystems and how they operate in space and
time.

New concepts such as intrinsic potential of fish habitat (Burnett et al. 2007), projections
of climate change, and new questions as to whether stream conditions in old-growth forests are
actually most favorable for native salmonids (Bisson et al. 2009, Reeves and Bisson 2009,
Reeves et al. 1995) are pivotal concepts that reframe our understanding of aquatic ecology and
ecosystems. No formal evaluation of the potential effectiveness of the network of key watersheds
was conducted during development of the NWFP, or has been since it was implemented. Fish
populations in need of attention are clearly identified now, and it would be useful to see whether
the current system is beneficial to those fish in terms of the overall distribution and the suitability
of individual watersheds. Additionally, the distribution of other sensitive aquatic-riparian species
(e.g., ESA-listed or petitioned herpetofauna) could contribute to this assessment.

Watershed analysis—
Watershed analysis was designed to provide the context for management activities in a particular sixth-field watershed, and the basis for developing project-specific proposals and determining restoration needs. It was envisioned as an analytical and not a decision-making process, involving individuals from a variety of scientific disciplines (USDA and USDI 1994a). Management agencies were expected to complete a watershed analysis before activities (other than minor ones) were initiated in key watersheds or riparian reserves (USDA and USDI 1994a). The version of watershed analysis advocated in the NWFP differs from previous versions (e.g., Washington Forest Practices Board 1993) and involves multiple disciplines and issues other than those that are specifically aquatic.

Baker et al. (2006) estimate that about 500 watershed analyses existed by 2003, but their quality and effectiveness vary widely. No formal assessment of watershed analyses has been completed at this time, so it would be prudent to conduct a comprehensive review and evaluation of watershed analyses, and consider incorporating new analytical tools such as NetMap (Benda et al. 2007) to help improve the process and reduce costs while increasing the usefulness of the product. The watershed analysis process could also be reexamined so that it is conducted more efficiently and considers the appropriate spatial scales, including the watershed of interest and its context within the larger basin. The latter could be particularly relevant for effective planning at a landscape scale.

New Perspectives on Conservation of Riparian and Aquatic Ecosystems
The Aquatic Conservation Strategy of the Northwest Forest Plan was premised on the view that aquatic ecosystems were dynamic in space and time, exhibiting a range of potential conditions, similar to the terrestrial systems in which they are embedded. (FEMAT 1993). Aquatic ecosystems in Pacific Northwest forests are multifaceted and complex, and can be conceptualized as a set of explicit ecological states (Penaluna et al. 2016, Reeves et al. 1995, Rieman et al. 2015). Each state has particular abiotic and biotic conditions, functions, and processes at any given time. The number and variety of ecological states in a domain (i.e., the range of conditions or range of natural variability for an ecosystem) is in constant flux in response to changes in local conditions, stochastic processes, legacies of past disturbance, and time since past disturbance (Beechie et al. 2010; Benda et al. 1998; Liss et al. 2006; Miller et al. 2003; Reeves et al. 1995; Resh et al. 1988; Rieman et al. 2006, 2015; Wondzell et al. 2007). Examples of the variation that aquatic ecosystems can experience through time are shown for the central Oregon Coast Range (Reeves et al. 1995) (fig. 9a and table 5), and eastern Oregon (Wondzell et al. 2007) (fig. 9b). Larger streams and rivers in the lower portion of the network are less variable through time; those in the upper and middle portions are more dynamic (Naiman et al. 1992). Because of the variation in the size and asynchronous nature of disturbance events (Allen et al. 1997, Malard et al. 2000, Schindler et al. 2010, Wiens 2002), conditions will vary over time among watersheds, resulting in a mosaic of biophysical conditions across the landscape. Unmanaged and minimally disturbed aquatic systems may actually exhibit a wider range of conditions than more heavily managed systems (Lisle 2002, Lisle et al. 2007).

A contrasting view holds that aquatic ecosystems tend to be in an equilibrium or steady state, and when disturbed they are expected to return to predisturbance conditions relatively...
quickly (Resh et al. 1988, Swanson et al. 1988). Biological (Vannote et al. 1980) and physical conditions (Rosgen 1994) are presumed to be relatively constant through time and to be “good” (barring human interference) in all systems at the same time. Conditions in aquatic systems with little or no human influence and natural disturbance, particularly those associated with late-successional and old-growth forest, are assumed to have the most favorable conditions for fish (Fox and Bolton 2007, Murphy and Koski 1989, Pollock and Beechie 2014, Pollock et al. 2012) and other aquatic organisms, and are most frequently used as references against which the condition of managed streams (e.g., Index of Biotic Integrity, Karr and Chu 1999) and effects of management actions can be assessed. Systems experiencing disturbances, such as wildfire or floods, are frequently immediately “restored” by attempting to reduce or eliminate erosional processes. For example, the US Forest Service attempted to reduce erosion in headwater streams following a wildfire in Colorado (Chin et al. 2015). However, reaches lower in the stream network downcut, creating other concerns. Although this static ecosystem view is being questioned in the general ecological literature (Hiers et al. 2016, Jackson et al. 2009, Montgomery 1999, White and Jentsch 2001), it is still being used to guide management and assess effects on aquatic ecosystems, and persists in environmental laws and policies developed in the 1970s, such as the Clean Water Act (Craig 2010).

Resilience is the capacity of an ecosystem to absorb change and remain within the ecosystem state and domain in the face of natural disturbances and human stressors (Desjardins et al. 2015). As ecosystems undergo larger shifts from human stressors, the ecosystem can be redefined, with a completely different set of characteristics and a compromised or altered range of conditions (Bisson et al. 2009, Reeves et al. 1995). Some ecosystem components may persist
through this transition, whereas others may be new, leading to a different set of novel states that may result in the loss of selected ecosystem services and conditions for at least some native species (Penaluna et al. 2016).

The physical aspect of these dynamics is understood conceptually (see review in Buffington 2012), but few mechanistic models currently exist to help understand the potential effects of management on dynamic ecosystems (but see Wondzell et al. 2007). As a result, consideration of dynamics remains largely conceptual, and holistic models of basin function (i.e., watershed analyses) are generally lacking, limiting the development of process-based applications of river management and restoration (Beechie et al. 2010). Also, there is also a tendency to focus on mean or median conditions rather than temporal variability, often dismissing this as “noise” and losing sight of the considerable inherent variability that characterizes riparian and aquatic ecosystems (Fausch et al. 2002, Montgomery 1999) and is ecologically critical (Hiers et al. 2016). Accounting for this variability and for nonstationarity of fluvial processes is central to assessing potential effects of climate change on riverine ecosystems (Buffington 2012, Miller et al. 2003, Montgomery 1999).

Consideration of large spatial and temporal scales is critical to the development of nuanced management and conservation strategies for ecosystems (Dale et al. 2000, Holling and Meffe 1996), including a range of conditions for aquatic ecosystems (Fausch et al. 2002, IMST 1999, Liss et al. 2006, NRC 1996). This shift requires moving from the current focus on relatively small spatial scales, with little or no consideration to the relevance of time, to a focus that considers large spatial scales, specifically ecosystems and landscapes, over relatively long time periods (10s to 100s of years) (Bisson et al. 2009, Naiman and Latterell 2005, Poff et al. 1997,
Reeves et al. 1995). An example of the importance of relations between scales can be seen in the “portfolio effect” of the behavior of populations of sockeye salmon (*Oncorhynchus nerka*) in Bristol Bay, Alaska identified by Schindler et al. (2010). This study found large variation in the number of fish in any local population over time. However, the variation among the various populations was asynchronous—not all were high or low at the same time. As a result, the total number of fish was relatively constant at the landscape scale, a pattern similar to the amount of old-growth historically found in the Oregon Coast Range (Wimberly et al. 2000).

The NWFP, and now the new Forest Service planning rule (USDA 2012), require managers to consider large spatial scales in designing, implementing, and evaluating management actions. This can be daunting given the lack of scientifically sound examples of how to operate at large temporal and spatial scales (North and Keeton 2008, Reeves and Duncan 2009, Thompson et al. 2009) and the lack of adequate tools and guidance. Shifting the management focus to the landscape level and longer time intervals may require recognition of the principles of hierarchy theory and the relation among levels of organization to increase the potential for success of future riparian policies and practices (Fausch et al. 2002).

Regulators may recognize the need to apply policies and regulations across broad areas, but may be constrained by the regulatory framework with which they are operating, and generally default to single standards that are applied across broad areas (e.g., NMFS’s matrix of pathway and indicators, NMFS 1999). This premise is incorrect, however (Allen and Starr 1982; O’Neill et al. 1986, 1989); instead, it is important to recognize that a multiwatershed landscape operates differently through time than does a single watershed, and that smaller spatial scales tend to be more variable over time than larger scales (Benda et al. 1998, Wimberly et al. 2000). Increasing
levels of aggregation, especially as spatial scales increase, may obscure important system processes (Clark and Avery 1975) and may result in unrealistic expectations for ecosystems and contribute to the contention that often surrounds large-scale management proposals (Allan and Curtis 2005, O’Neill 1986, Shindler et al. 2002). Also, the failure to recognize the different levels of ecological organization and the potential response of each to component parts of disturbance and management may incur unintended economic and social costs, such as repeated investment in ineffective restoration and management strategies (Caraher et al. 1999, Dale et al. 2000).

The emerging consideration of ecosystem dynamics and large spatial and temporal scales has implications for approaches to restoration of aquatic-riparian ecosystems. Many restoration efforts have focused mainly on improving habitat attributes, primarily wood placement, to a lesser degree on shade improvement for water temperature. These efforts too often aim to bring “stability” to degraded systems and are viewed as the final phase of restoration (see Palmer et al. 2014). The dynamic approach, not yet broadly practiced, focuses on restoring ecological processes (Beechie et al. 2009, 2010; Bernhardt and Palmer 2011), including periodic inputs or reoccurrences of these important habitat attributes. This requires a shift from reliance on only a set conditions (e.g., number of pieces of wood per unit length) or channel classification (e.g., Rosgen 1994) to a quantitative approach based on ecological processes, theory, empirical field methods, and limited modeling (Kline and Cahoon 2010, Wohl et al. 2005).

Some researchers have pointed out that although restoration of ecological processes, such as flow, water temperature, habitat complexity, and connectivity, is a critical consideration in restoring many streams, it may not be sufficient for degraded channels, and can even worsen the
ecological condition of the stream (Louhi et al. 2011, Tullos et al. 2009). For example, in restoring floodplain overflow potential, if riparian trees are removed from a previously closed-canopy stream, the underlying energy regime may change from allochthonous resources to one driven by primary production. This may shift the stream further away from the desired ecological state and often toward algae-dominated streambeds and higher temperatures (Sudduth et al. 2011). Similarly, if the hydrologic regime is restored but there is no nearby source of invertebrate colonists, then the instream communities will remain altered (Sundermann et al. 2011). Finally, an over-reliance on an in-channel focus (i.e., small scale) may not address the stressor(s) that most limit recovery of the aquatic ecosystem; quite often this factor is water quality, and thus ecological recovery will not occur until the stressor is addressed (Beechie et al. 2010, Kail et al. 2012, Selvakumar et al. 2010). Examples of process-focused restoration are presented below in the section on climate change.

In addition to considering spatial complexity, temporal dynamics are particularly important to understand because many key ecological processes such as canopy closure, tree fall, and fuel loading are related to the age of trees in riparian areas as well as time since disturbance. Temporal dynamics can be examined using models, but long-term studies and monitoring are needed to understand how systems respond over time (Hassan et al. 2005). One strategy that may be appropriate is to design monitoring to focus more on changes following major disturbances rather than focusing simply on short-term trends.

The other challenge posed by a dynamic perspective of aquatic ecosystems is the consideration of large spatial scales. Restoration efforts are generally performed at small spatial scales, with only a relatively small percentage of any watershed receiving activity (Ogston et al.)
2014, Roni et al. (2010). Roni et al. (2010) estimated that a minimum of 20 percent the habitat of
a given species in a watershed should be restored to detect a 25 percent increase in smolt (salmon
or steelhead) numbers, the minimum detection level for most monitoring programs. They found
that floodplain restoration yielded greater increases than in-channel restoration. However,
because of the large variability in numbers for in most populations (Bisson et al. 1997, Schindler
et al. 2010), Roni et al. (2010) suggested that 100 percent of the habitat should be restored to be
assured of having a significant ecological impact.

Non-fish-bearing streams—

The ecological importance of headwater streams, which generally make up 70 percent or more of
the stream network (Downing et al. 2012, Gomi et al. 2002), was not well known or understood
at the time of the development of the ACS, but it is now well established in the scientific
literature (Leigh et al. 2016, Richardson and Danehy 2007). Headwaters are sources of sediment
(Benda and Dunne 1997a, 1997b; May and Lee 2004; Zimmerman and Church 2001; see review
by McDonald and Coe 2007) and wood (Bigelow et al. 2007; May and Gresswell 2003, 2004;
Reeves et al. 2003) for fish-bearing streams, provide habitat (Kelsey and West 1998, Olson et al.
2007, chap. 6, this volume) and movement corridors (Olson and Burnett 2009, Olson and Kluber
2014) for several species of native amphibians and macroinvertebrates (Alexander et al. 2011,
Meyer et al. 2007), including recently discovered species (Dieterich and Anderson 2000), and
may be important sources of food for fish (Kiffney et al. 2000, Wipfli and Gregovich 2002; see
reviews by McDonald and Coe 2007 and Clarke et al. 2008). Wood jams in small streams are
important sites of carbon storage (Beckman and Wohl 2014), and these streams export large
amounts of carbon; one-third is emitted to the atmosphere and the remainder transported
downstream (Argerich et al. 2016).

Tributary junctions of headwater streams and larger channels are important nodes for regulating material flows (Benda et al. 2004, Gomi et al. 2002, Montgomery et al. 2003) and cold water (Ebersole et al. 2015) in a watershed, and are the locations where site-scale effects from management activities are often observed (Richardson and Béraud 2014). These locations have unique hydrologic, geomorphic, and biological attributes and vary in the types and amount of materials delivered to the channel, making them sites of high biodiversity (Benda et al. 2004, Danehy et al. 2012)

Headwater streams are among the most dynamic portions of aquatic ecosystems (Benda et al. 2005, Hassan et al. 2005, McDonald and Coe 2007, Naiman et al. 1992). Headwater habitats may range from simple to complex, depending on the amount of time since disturbance (such as landslides and debris flows). Following evacuation by a debris flow, headwater depressions and channels fill with material from the surrounding hillslopes, including large wood that falls into these channels, forming obstructions behind which sediments and wood accumulate (Benda and Cundy 1990, May and Gresswell 2004), and then empty again with the next landslide or debris flow (fig. 10). As a result, headwater streams are likely to exhibit a range of conditions across the landscape at any point in time.

This cycle of filling and emptying results in a punctuated movement of sediment and wood to larger, fish-bearing streams (Benda et al. 1998, Naiman et al. 1992), which contributes to the long-term productivity of many aquatic ecosystems (Benda et al. 2003, Hogan et al. 1998, Reeves et al. 1995). A common consequence of past clearcutting is an absence of down wood to replenish the refilling process. This lack of wood may result in a chronic movement of sediment
to larger channels, which could lead to both non-fish-bearing and fish-bearing channels
developing characteristics different from those that occurred before forest management. Such
conditions could be outside the range of watershed conditions to which native biota are adapted
(Beschta et al. 2004), limiting the effectiveness of conservation and recovery programs.

Wood enters streams via chronic and episodic processes (Bisson et al. 1987). Chronic
processes, such as tree mortality and bank undercutting (Bilby and Bisson 1998, Murphy and
Koski 1989), generally introduce single trees or relatively small numbers of trees at frequent
intervals. Wood from headwater streams, which originates from within 131 ft (40 m) of the
channel (May and Gresswell 2003), is delivered to fish-bearing streams by large, infrequent
events, such as wind throw (Harmon et al. 1986), wildfire (Agee 1993), severe floods, landslides,
and debris flows (Benda et al. 2003; May and Gresswell 2003, 2004; Reeves et al. 2003).

Geomorphic features of a watershed influence the potential contribution of upslope wood
sources; steeper, more highly dissected watersheds will likely have a greater proportion of wood
coming from upslope sources than will watersheds with lower stream densities and gradients.
Also, there is wide variation in the potential of headwater streams to deliver sediment and wood
to fish-bearing streams, depending on channel steepness and angle of entry along the run-out
track, among other factors (Brayshaw and Hassan 2009; Benda and Dunne 1997a, 1997b;
Burnett and Miller 2007; May 2007). Culverts and other stream crossings can also impede wood
movement from smaller to larger streams (Trombulak and Frissell 2000).

The presence of large wood from headwater streams influences the behavior of landslides
and debris flows and the response of the channel to such events. Large wood in debris flows and
landslides influences the run-out length of these disturbance events (Lancaster et al. 2003).
Debris flows without large wood move faster and for longer distances than those with wood, and they are less likely to stop high in the stream network. A debris flow without wood is likely to be a concentrated slurry of sediments of various sizes that can move at relatively high speeds over long distances, scouring substrate and wood from the affected channels. These types of debris flows are more likely to negatively affect fish-bearing channels, as compared to the potentially favorable effects that result from the presence of wood. Woodless debris flows can further delay or impede the development of favorable conditions for fish and other aquatic organisms. In contrast, those containing wood can help store sediments (Bunn and Montgomery 2004) and build terraces that can persist for extended time-periods (Lancaster and Casebeer 2007, May and Lee 2004).

Intermittent streams, which can make up half of the stream network (Datry et al. 2015), connected to larger fish-bearing streams can provide important seasonal habitats for rearing and spawning by fish (Boughton et al. 2009; Wigington et al. 2006). In the Oregon Coast Range, growth and survival of juvenile coho salmon was higher in intermittent streams than the perennial mainstem (Ebersole et al. 2006, 2009; Hance 2016). Identification, protection, and restoration of these streams is important to the success of conservation efforts for native fish across the NWFP area.

A rich non-fish vertebrate community dominated by invertebrates inhabits headwater streams throughout the NWFP area. For example, Olson and Weaver (2007) reported about 450 fish of 3 species and 180 amphibians of 12 species in 106 stream reaches in 12 study sites in western Oregon. Torrent salamanders dominate intermittent streams (Olson and Burton 2014, Olson and Weaver 2007), and appear to be sensitive to thinning in narrow riparian management
areas; NWFP riparian reserves appear to be benefiting retention of this aquatic-dependent community in abundant small streams in the region. Nevertheless, two torrent salamander species are currently petitioned for listing under the ESA; both have ranges that include significant tracts of nonfederal lands.

Continuing and Emerging Issues of Concern

Water—

Federal lands are important sources of freshwater for human consumption, recreation, agriculture, and environmental needs in the United States, something that was not well recognized at the time the NWFP and ACS was developed. These lands produce 24.2 percent of the water nationally, 18 percent and 1.5 percent from Forest Service and BLM lands, respectively (Brown et al. 2008). In the West, federal lands contribute 66 percent of the mean annual water supply, 51 percent of which comes from Forest Service lands and 5.4 percent from BLM lands (Brown et al. 2008). Management strives to maintain the quality and quantity of this water.

The contribution of water from federal lands in the NWFP is also important; however, exact estimates are not available currently and were beyond the scope of this review. At the state level, the majority of water in the three NWFP states (California, Oregon, and Washington)

4 This includes the states of Arizona, California, Colorado, Idaho, Montana, New Mexico, Nevada, Oregon, Utah, Washington, and Wyoming
originates from federal lands (table 6), with bulk coming from Forest Service lands. Within the
NFWP area, the amount of water produced on federal lands varies among national forests and
watersheds. Some forests, such as the Willamette and Deschutes National Forests, make
relatively large contributions, 40 percent or more, to rivers in the watersheds where they are
found (fig. 11a and b, respectively). Conversely, contributions from other forests are smaller
(less than 20 percent) (fig. 11c) but nonetheless important. See the section on climate change for
potential future issues pertaining to water availability and temperatures.

Roads—
Roads provide necessary motorized access for forest management, recreation, and other
beneficial purposes (Gucinski et al. 2001), but they can also have detrimental effects on the
landscape. The focus of the NWFP and ACS has been to address the negative effects of roads on
aquatic ecosystems through a broad program of road decommissioning and upgrading, including
remediation of chronic sedimentation and barriers to aquatic organism movement. Several
syntheses describe the types, causes, and effects of road networks and streams, and meta-
analyses concerning the ecological effects of roads and the delivery of sediment from low-
standard mountain roads have been published (Croke and Hairsine 2006, Forman and Alexander
1998, Jones et al. 2000, Trombulak and Frissell 2000). Reducing the effect of roads and
associated infrastructure remains a challenge for federal management agencies, and others.

The vulnerability of roads to hydrologic changes and the associated effects on aquatic
and riparian ecosystems differ based on topography, geology, slope stability, design, location,
and use. Roads can affect streams directly by:

2) Imposing barriers to the migration of aquatic organisms, including access to floodplains and off-channel habitats (Clarkin et al. 2005, Gibson et al. 2005, Sagar 2004, Trombulak and Frissell 2001)

3) Increasing stream temperatures (Wenger et al. 2011)

4) Causing changes in channel morphology (Hassan et al. 2005)

5) Introducing exotic species (McKinney 2003)

6) Increasing harvest and poaching pressure (Lee et al. 1997, Trombulak and Frissell 2000)

7) Changing hillslope hydrology and resulting peak flows (Jones and Grant 2001)

Roads penetrating remote and otherwise-intact forested landscapes can have particularly significant effects on aquatic ecosystems (Forman et al. 2003, Havlick 2002, Trombulak and Frissell 2001). The ecological consequences of these effects are shown in table 7.

The effects of roads vary widely depending on local features (Al-Chokhachy et al. 2016). Recently developed techniques, such as the Geomorphic Roads Analysis and Inventory Package (GRAIP) (Black et al. 2012) can be employed to identify priority locations of sources of sediment, including culvert failures, landslides, and gullies. A modified version of this technique has been incorporated into NetMap (Benda et al. 2007) to reduce the amount of field time needed to assess roads. Evaluating the effectiveness of new analytical approaches and the potential consequences of not considering other effects, or the limited length of roads treated, could be a research priority.
A significant number of watershed-improvement actions implemented under the NWFP and other large-scale forest-planning efforts involve decommissioning roads that have a high probability of contributing to landslides, and that are not regarded as essential to meeting local forest objectives, as well as removing road-related impediments to upstream and downstream movements of aquatic organisms (Switalski et al. 2004). The watershed analysis component of the ACS identified forest roads where: 1) drainage systems hastened runoff from storms and promoted sedimentation of streams; 2) unstable fill materials concentrated water and increased the risk of landslides; and 3) the roadbed encroached on riparian zones (Kershner 1997). Since NWFP implementation, the Aquatic and Riparian Effectiveness Monitoring Program (S. Miller, pers. comm.) estimated that 6.7 percent of the road network has been removed or closed (5,390 out of 80,750 miles total [8674 of 129 954 km]) in the NWFP area. Additionally, 10 percent of the impassable road crossings (209 of 2114) have been made passable on Forest Service lands in Region 6 since the inception of the NWFP (J. Capurso, pers. comm.).

Though restoration of fish passage is often listed as a top priority for stream restoration in the Pacific Northwest (GAO 2002, Roni et al. 2002), recent work has contributed much to our understanding of just how complex this issue is in practice (McKay et al. 2016). Advances have been made in culvert inventory and assessment (Clarkin et. al 2005), ecosystem-based restoration approaches (USDA 2008), and effectiveness monitoring (Heredia et al. 2016, Hoffmann et al. 2012). Until recently, however, the ecological benefit of these efforts has been difficult to quantify beyond the level of individual projects. A new study evaluating the effectiveness of passage restoration at the level of an entire forest (Siuslaw National Forest, Chelgren and Dunham 2015) found that individual culvert replacements successfully increased the probability
of upstream access for all fishes in the study area. Results of this work also showed that the net
benefit of culvert replacements was fairly modest across the extent of the forest, however, when
expressed in terms of gains in kilometers of stream occupied or increases in fish numbers
resulting from restoration. The authors hastened to add that some limitations of the study design
could have influenced these findings (Chelgren and Dunham 2015), but results of this study
nonetheless point to the value of programmatic (vs. project-only) evaluations of culvert
restoration. This echoes more general recommendations for following the cycle of adaptive
management on National Forests (Marcot et al. 2012) and the recommended scales for managing
watersheds (Fausch et al. 2002, Neeson et al. 2015).

Much of the guidance for fish-passage restoration on Federal lands in the Pacific
Northwest was issued by an assessment in 2002 (GAO 2002). This assessment highlighted the
need for larger scale assessments, as noted above, as well as the economic challenges associated
with passage restoration, which are only beginning to be addressed. For example, a follow-up to
the Chelgren and Dunham (2015) study by Reagan (2015) evaluated costs and benefits of
remaining culvert replacement opportunities on the Siuslaw National Forest in relation to
multiple objectives, including benefits to fish (estimated from Chelgren and Dunham 2015),
maintenance of transportation networks, and the probability of culvert failures based on culvert
sizing and influences of floods and major erosional events in streams. The Reagan (2015)
analysis explicitly quantified economic costs and benefits of restoration in relation to these
objectives and their relative assumed values. This work (along with others in the region, e.g.,
Pilson 2012) has demonstrated the value of a proactive, economic analysis of multiple objectives
to identify priorities for restoration investments in a programmatic context. These new tools, if
applied, can more completely address standing recommendations to land-management agencies in the Pacific Northwest (e.g., GAO 2002) to more efficiently invest limited resources to benefit fisheries and other management objectives through culvert replacements.

Because road access management must take into account social, economic, and environmental objectives (Daigle 2010), the decision-making process for dealing with roads is complex. A decision matrix for identifying actions is shown in figure 12. In many cases, limited funds or socioeconomic issues may preclude closing or removing roads identified as high priority for treatment on the basis of their effects on riparian ecosystems. Also, a road network may be needed to effectively implement landscape-scale restoration projects that might involve widespread thinning and prescribed fire (Franklin and Johnson 2012), and for fire management, fuel reduction, and fire control. The vast road system on private and state lands that abut federal lands also needs to be considered in road assessments. Studies in Redwood National Park suggest that removal of logging roads can yield carbon storage dividends, in particular by preventing soil erosion (van Mantgem et al. 2013). Understanding how to balance fire management, recreation, and other needs against potential negative aspects of roads will require a concerted cooperative effort of managers and physical, biological, and social scientists. (See section below on climate change for additional discussion of roads.)

Climate Change

Since 1994, our knowledge of climate change in the NWFP area has greatly improved, just as dealing with climate change has become an important aspect of environmental planning in the Forest Service and BLM. Many advances have come from models that forecast trends in
temperature, precipitation, and snowpack, and associations of these trends with the habitat conditions for various species. Although there is general agreement about the direction of trends in many meteorological parameters, the rates and amounts of change at specific locations in the NWFP area vary among models (Climate Impacts Group 2009; also see chap. 2 this volume). Further, other climate-related changes such as increases in forest insect and disease outbreaks and uncharacteristically severe wildfires may accentuate the undesirable effects of meteorological and hydrological trends, resulting in threats to both terrestrial and aquatic ecosystem integrity. Although developing proactive measures that would ameliorate undesirable effects of climate change on forest resources was not a centerpiece of the NWFP, one of the Plan’s main objectives was to restore forest ecosystems that were resilient in the face of natural and anthropogenic disturbances. The question is, how well does the NWFP address climate-related threats to native fishes and other aquatic biota as they are currently perceived? (See chap. 2 this volume for further details.)

In this section we focus on a review of recent advances in our knowledge of the likely effects of climate change on native fishes of the NWFP area. We examine climate change effects on fish lifecycles, with a principal focus on anadromous salmonids, a group of species that has received the most scientific attention as well as significant conservation effort (table 8) (additional discussion in the appendix of this chapter). Watershed improvements undertaken through the NWFP are related to potential climate effects on fish lifecycles, and to the capacity of populations to adapt and persist through time. Finally, we discuss the role that federally managed forests in the NWFP area play in conserving native fishes in a changing climate, when viewed in a broader matrix of different landownerships and other landscape-scale uses.
Climate change in the NWFP area—

Projected changes in climate are usually derived from models based on historical data coupled with scenarios incorporating alternative assumptions about future greenhouse gas emissions. These assumptions range from high global rates of economic development and human population growth to conservative industrial and population growth rates and widespread implementation of “clean” technologies. Model outcomes are often displayed as incremental changes in an environmental parameter of interest such as air temperature, sea level, or precipitation over a fixed time period. Projected changes in climate under different scenarios are plotted to provide a range of outcomes at a given point in time, with scenarios incorporating intermediate assumptions about future greenhouse gas emissions generally believed to represent the most realistic expectations.

Air and water temperatures—

Virtually all climate models forecast a gradual rise in air temperature over the next century. Recent changes in climate appear to be happening more rapidly than in at least the past 1,000 years (IPCC 2007), and have included a global average warming of 1.4°F (0.8°C) during the past 120 years. According to the Intergovernmental Panel on Climate Change (IPCC) (2007), most general circulation models suggest 2 to 7 times more warming will occur in the next 50 to 100 years, with projected increases in mean global surface temperatures by the year 2100 ranging from 2.7°F to 3.6°F (1.5 to 2.0°C) relative to a 1850–1900 time frame, depending on CO₂ emission scenarios (IPCC 2014). The 2014 IPCC Synthesis Report (Summary for Policy Makers: 10) states:
“It is virtually certain that there will be more frequent hot and fewer cold
temperature extremes over most land areas on daily and seasonal timescales, as
global mean surface temperature increases. It is very likely that heat waves will
occur with a higher frequency and longer duration. Occasional cold winter
extremes will continue to occur.” [emphasis theirs]

The finding that climate change will include both gradual long-term temperature trends as well
as increases in the frequency and duration of extreme events has important implications for
aquatic ecosystems in the NWFP area.

Air temperature changes in forests of the NWFP area are predicted to be generally
consistent with global climate models, although somewhat more variable, with forecast increases
ranging from 1 to 6.3°F (0.5 to 3.5°C) depending on the greenhouse gas emission scenario used
in the model and on forest location (Latta et al. 2010). Overall, these authors noted that relative
temperature increases were more apparent at higher elevations than at lower elevations, and that
proximity to the Pacific Ocean moderated the rate of change. Mote and Salathé (2010) examined
a broad suite of IPCC climate models and found that, by the 2080s, average air temperatures in
the Pacific Northwest were predicted to increase 2.9°F (1.6°C) under the coolest scenario and by
10.3°F (5.7°C) under the warmest scenario. In most models the greatest absolute temperature
increases were projected for summer months, although warming was also forecast in other
seasons. Sea-surface temperatures showed less warming over the same period than those
modeled over land.

Similar to air temperatures, water temperatures are expected to rise in much of the NWFP
area as a result of climate change (Isaak et al. 2011 [NorWeST]). Modeled water temperatures
were primarily developed from models of the relation between air and water temperatures, and are projected to be stressful to lethal for many native salmonids (e.g., Isaak et al. 2012, Wade et al. 2013) (see later discussion for more details). More recent projections suggest that the extent of temperature change may not be as great as originally projected, particularly at higher elevations (Isaak et al. 2016). However, other researchers (Arismendi et al. 2014) have questioned the ability to project future water temperature from past relations between air and water temperatures. In addition, Arismendi et al. (2013a) found that recent trends in water temperature have been more variable than those reported by Isaak et al. (2012)—using empirical records, they found that water temperatures increased in some systems and decreased in others. Also, Leach et al. (2016) found variability in water temperature in a headwater stream of the Oregon Coast Range that was not captured by the NorWeST model (Isaak et al. 2010), but noted that the model was not scaled to track such small-scale effects. Although there is some uncertainty about the extent of temperature changes that streams in the NWFP area will experience, it is clear that dealing with water temperatures will be a major challenge for managers.

Potential patterns of changes in water temperature are highly variable when examined at the local scale (Leach et al. 2016, Reeves et al 2016b, Turshwell et al. 2016) (fig. 13). This variability is a result of local conditions such as stream orientation, topographic shading, and elevation, and strongly influences physical and biological attributes and resultant ecosystem integrity (Gomi et al. 2002, Thorpe et al. 2006). Understanding this variability will be crucial to developing effective restoration and mitigation programs. Watershed analysis tools, such as NetMap (Benda et al. 2007), can help identify areas that can provide thermal refugia and areas
where riparian restoration efforts (fig. 14) could help reduce water temperatures to levels less
stressful or even optimal for native fish, despite climate change (Lawrence et al. 2014, Ruesh et
al. 2012)

Hydrology—

Predicted future changes in streamflow on national forests in the PNW are fundamentally tied to
changes in the region’s climate. Detailed modeling using a variety of climate models and
assumptions predicts that the Pacific Northwest will warm considerably (anywhere from <2°C to
>6°C average annual temperature) by the year 2100 (Salathé et al. 2007). However, predicted
changes in annual precipitation are much less certain, and most models project that future
precipitation will remain approximately the same as it has been for the past 50 years (Salathé et
al. 2007). Most predictions of changing streamflows for the PNW therefore focus primarily on
the effects of changes in temperature. Seasonal changes in precipitation are showing up in the
data (i.e., Safeeq et al. 2013), but are difficult to resolve regionally and consequently are not as
well understood.

A key factor affecting both high and low streamflows in the future will be the fate of
snow and the seasonal snowpack. Snowpack dynamics are important in understanding
streamflow regimes because snow represents a dominant form of storage on the landscape. When
precipitation falls as snow it is not available for runoff or groundwater recharge until it melts.
Similarly, the rate and timing of snowmelt are first-order controls on both peak and low
streamflows, as discussed below.

A particularly crucial dimension of snowpack dynamics is the geographical location of
the rain-snow transition on the landscape. This transition is controlled by elevation and
determines how much of the winter precipitation falls as rain versus snow. Although often
visualized as a fixed elevation, this transition is better seen as a stacked sequence of elevationally
controlled zones or ranges with imprecise and regionally varying boundaries (Nolin and Daly
2006, Klos et al. 2014). In general, for any area there is an elevation below which virtually all
winter precipitation falls as rain and above which it falls as snow. Elevations in between are
defined as the transitional snow zone (TSZ) that receives both rain and snow; the latter typically
melts out fairly rapidly.

The widely varying elevational gradients in the PNW in general, and on national forest
land in particular, therefore impose considerable variability in the response of the landscape to
changing climate. Depending on the proportion of the landscape that occupies each of these
zones, a warming climate, hence a rising snow line, may transition the landscape from a zone
dominated by seasonal snow accumulation and melt (snow zone) to one that receives a mixture
of rain and snow (and rain-on-snow)—the TSZ. Or it may push the landscape out of the TSZ and
into the rain zone (Klos et al. 2014, Luce et al. 2014)

The effects of a changing climate are already apparent in the snow data for the Pacific
Northwest. As winter and spring temperatures have climbed over the past 50–70 years, spring
snowpacks have been smaller (Mote 2003, Hamlet et al. 2005, Mote et al. 2005) and have melted
out earlier (Stewart et al. 2005, Hamlet et al. 2007). Moreover, the aforementioned zonal changes
are already occurring in some landscapes as snow zones transition to the TSZ, and the TSZ
becomes rain-dominated (Tohver et al. 2014). These trends are expected to continue across much
of the region (Luce et al. 2014).
However, snowpack dynamics alone do not determine what future streamflow regimes will look like on national forests in the area of the Northwest Forest Plan. Recent work has shown that another first-order control is the landscape-scale drainage efficiency: the inherent, geologically-mediated efficiency of landscapes in converting recharge (precipitation) into discharge (Tague and Grant 2009; Safeeq et al. 2013, 2014). In essence, the drainage efficiency determines how quickly precipitation, either as rain or snowmelt, becomes streamflow. Although drainage efficiency is “hard-wired” into the landscape on millennial timescales, and thus is not changing with climate, it mediates the climate-influenced signals and therefore has to be considered in predicting future streamflow regimes. This is particularly true for low-flow regimes, but influences peak flows as well. Basically, the drainage efficiency of a landscape is determined by the rate at which water moves through the subsurface. In steep landscapes with shallow soils, water moves laterally through the subsurface rapidly through both saturated and unsaturated pathways, drainage efficiency is high, and streams respond rapidly to recharge events. In flatter landscapes with deep, permeable, porous, or fractured bedrock, water moves slowly as deep groundwater, drainage efficiency is low, and streams respond slowly to recharge events but may have sustained high base flows.

**Effects of climate change on peak flows**—Here we broadly consider how both climate and drainage efficiency can shape predictions of future streamflows on national forest lands. We distinguish between effects on peak and low flows, as the mechanisms of streamflow generation are different in each case. Finally, we discuss how these broad predictions can be refined for individual forests, which is beyond the scope of the current analysis.
There are several hydrologic mechanisms by which climate could increase peak flows in rivers and thus their propensity to flood. More intense or frequent rainstorms are one mechanism, and some research has suggested that a warming atmosphere may result in a more northerly storm track for the North Pacific, potentially resulting in more intense precipitation (Salathé 2006). However, these results have large uncertainties and are not well represented in most global circulation models. A somewhat better understood mechanism is the shifting potential for rain-on-snow events in the Pacific Northwest as the climate warms. Rain-on-snow (ROS) events are known to be a potent flood-producing mechanism in steep mountain landscapes in the Pacific Northwest (Harr 1981, Marks et al. 1998, McCabe et al. 2007). In general, landscape susceptibility to ROS events is determined by climate and topography; the effects of climate warming on ROS are similarly influenced by the same controls; and climate warming may increase, decrease, or not affect the risk, depending on whether snowpacks are cold or warm (i.e., near the freezing point). As summarized by Hamlet and Lettenmaier (2007):

“Cold river basins where snow processes dominate the annual hydrologic cycle (\(<-6^\circ\text{C} \text{ average in midwinter}\)) typically show reductions in flood risk due to overall reductions in spring snowpack. Relatively warm rain-dominant basins (\(>5^\circ\text{C} \text{ average in midwinter}\)) show little systematic change. Intermediate or transient basins show a wide range of effects depending on competing factors such as the relative role of antecedent snow and contributing basin area during storms that cause flooding. Warmer transient basins along the coast in Washington, Oregon, and California, in particular, tend to show increased flood risk.”
A more recent analysis looked at a range of factors influencing peak flows, including ROS in Oregon and Washington, and developed a model of sensitivity to peak-flow increases based on perturbing the temperature in the model using warming scenarios from 2020–2080 and the A1B emissions scenario (Safeeq et al. 2015). They concluded that corresponding changes in snowpack dynamics may result in large (>30 to 40 percent) increases in peak flows, primarily in the Cascade Range and Olympic Mountains. The North Cascades, in particular, were most vulnerable (fig. 15). Lower-elevation areas were less likely to be affected but were still vulnerable to larger floods generated from upstream reaches in vulnerable landscapes. The analysis yielded regional sensitivity maps for Oregon and Washington that can be used to characterize the risk on individual national forests and landscapes. These watersheds are also likely more susceptible to warming (Arismendi et al. 2013a, 2013b; Poole and Berman 2001; van Vliet et al. 2011). Streams at higher elevations should retain flows; with stable, cooler water temperatures, they will be critical cool-water refugia for native fish (Isaak et al. 2012, 2015; Luce et al. 2014; Wenger et al. 2011).

Similar refugia can be projected for amphibian species as well (Shoo et al. 2011), with myriad ecological consequences. For lentic-breeding amphibians in the NWFP area, higher-elevation-adapted Cascades frogs (*Rana cascadae*) may be faced with shifts in their breeding habitat conditions. In addition, they may encounter novel interactions with species associated with warmer, lower-elevation habitats, such as native northern red-legged frogs (*Rana aurora*), which may spread to higher elevations with altered climate. The low- to mid-elevation-adapted red-legged frogs may in turn encounter invasive American bullfrogs (*Lithobates catesbeianus*), which now occur at warm, low elevations. Similarly, as mountain stream flows are projected to
change, torrent salamanders (*Rhyacotriton* spp.), associated with intermittent streams, could have a truncated active season, retreating below ground as small streams dry earlier in the season (possibly affecting survival and reproduction). They may also move downstream and be faced with new interactions with larger predatory salamanders or fish in perennial reaches. If they migrate downstream, their over-ridge dispersal to new watersheds may be affected, as distances between flowing water-bodies increase. Hence their populations could become more isolated and vulnerable to stochastic events. For terrestrial-breeding salamanders, we can project the consequence of warmer, drier conditions by examining the distribution of current species in the drier portion of the Northwest; these are species for which climate change has already occurred. Optimal habitat for the Siskiyou Mountains salamander (*Plethodon stormi*) is modeled to occur on the shaded side of mountain ridges and in cooler riparian areas in the arid southern Oregon landscape (Suzuki et al. 2008), and the black salamander (*Aneides flavipunctatus*) appears to become a riparian associate in dry portions of its range (Nauman and Olson 2004). Hence for cool, moisture-dependent species, riparian areas and north-facing slopes with hill shading may become more important with projected changes in climate. Alternatively, as for torrent salamanders, their activity pattern may be altered, with reduced surface activities during dry times and possible consequences for survival. Range shifts for temperature- and moisture-dependent species have also been projected for pathogens of aquatic organisms, such as the amphibian chytrid fungus (*Batrachochytrium dendrobatidis*), which is predicted to increase in occurrence probability in the NWFP area with climate change (Xie et al. 2016).

**Effects of climate change on low flows**—Snowpack dynamics and drainage efficiency combine to determine the sensitivity of individual landscapes to a warming climate (Tague and
Grant 2009, Safeeq et al. 2013). There has been a general trend over the past 50 years for less
snow in winter and earlier snowmelt, which reduces spring, early summer, and late summer
flows in the western United States (Leppi et al. 2011, Safeeq et al. 2013), with the lowest
percentile flows decreasing across the PNW (Luce and Holden 2009). Hydrologic models such
as the Variable Infiltration Capacity (VIC) model, coupled with down-scaled climate simulations
have been used to generate predictions of future low flows across much of the Pacific Northwest
(i.e., Hamlet et al. 2010).

However, snowpack changes are not the only factor determining future low flows. Other
recent work has shown that the drainage efficiency (slow versus fast) mediates the signal from
climate-induced changes in snowpack and snowmelt. Employing a simple exponential decay
model to describe the recession behavior of streams coupled with a regional-scale estimation of
variations in aquifer drainage characteristics, Safeeq et al. (2014) developed a sensitivity map for
changes in summer streamflow across Oregon and Washington. As with the VIC products and
peak-flow maps previously described, these maps provide water and landscape managers with a
spatially explicit representation of where future changes in low flow are likely to be most
pronounced. For example, these maps show that areas drained by young volcanic rocks with
depth, slow groundwater systems, such as the High Cascades, may be particularly vulnerable to
declines in summer streamflow, whereas areas with shallow sub-surface aquifers and limited
potential to store water are less sensitive to changes in low flows.

Assessing climate change impacts on streamflow at the scale of individual national
forests—The discussion above highlights how existing tools and models can be used to give
technically sound predictions about the magnitude and timing of streamflow changes in specific
landscapes. Although not a trivial exercise, any national forest can use the spatially-explicit models already developed to make first-order forecasts for changes in streamflow regimes. The products to date cover most but not all forests in the area of the Northwest Forest Plan. Extending results to these unmapped forests (mostly in Northern California) would require some extrapolation, but is well within the scope of the existing data. Tools and approaches such the concept of “hydrologic landscapes” can expedite this (Winter 2001, Wigington et al. 2013, Patil et al. 2014).

Furthermore, there are several examples to date of individual or groups of national forests and other federal lands that have coordinated efforts to develop detailed assessments of likely hydrologic changes that can serve as models for other Forests and regions. Specific examples include the Olympic National Forest (Halofsky et al. 2011), the Quinault Indian Nation on the Olympic Peninsula (Reeves et al. 2016b), the Blue Mountain Adaptation Partnership (BMAP) (Halofsky and Peterson in press 2016), and the upcoming report from the Southern Central Oregon Adaptation Partnership (SCOAP).
Extreme events—

Increased frequency of extended, severe droughts and intense winter storm events (IPCC 2014) will also affect aquatic ecosystems and fish populations in forested landscapes (Ward et al. 2015). The pattern of changes will vary widely within and among watersheds (fig. 16) depending on local features, which makes it difficult to generalize the effects. However, changes in the seasonal timing of annual hydrographs and more frequent extremely low or high flows are likely to affect native fish populations to some degree. Changes in flows that lead to earlier spring runoff and prolonged periods of summer low flows could have important implications for the habitat of (ISAB 2007) and food chains encompassing fish and other aquatic organisms (Power et al. 2008, Wooton et al. 1996) (see discussion later in this section for more details).

Amphibians that inhabit ephemeral ponds and streams are likely to be especially vulnerable to drought and general climate change effects (Blaustein and Olson 1991, Shoo et al. 2011).

Ocean conditions—

Over the last several decades, the importance of the marine environment to fish that spend part or all of their lives at sea has been recognized as a major factor regulating population abundance. Climate-related changes in the ocean potentially important to native fishes in the NWFP area include acidification (Orr et al. 2005), increased sea-surface temperatures (IPCC 2007), changes in wind and current patterns (Rykaczewski and Dunne 2010), and sea-level rise (IPCC 2007). Absorption of anthropogenic CO₂ by the upper ocean decreases pH and carbonate ion concentrations (Orr et al. 2005), increasing acidity and inhibiting the ability of planktonic organisms to form calcium carbonate (CaCO₃), a key component of the exoskeleton. Many of these organisms form the base of the food chain that supports anadromous fishes during the
marine phase of their lifecycles. The subarctic Pacific Ocean naturally has higher carbon
concentrations than most other ocean basins, and the effects of acidification are expected to
occur sooner and be more pronounced there (Cooley et al. 2012).

Rising sea-surface temperatures may reduce the amount of preferred thermal habitat for
anadromous salmonids in the ocean and potentially limit their marine distribution (Aziz et al.
2011, Welch et al. 1995). As areas with suitable temperatures decrease or shift northward,
Pacific salmon could become concentrated in smaller foraging zones, resulting in increased
competition for limited food resources (Grebmeier et al. 2006, Mantua et al. 2009, Johnson and
Schindler 2009, Welch et al. 1995). Salmon may be able to partially compensate for these
changes by using cooler subsurface waters; however, deeper water may provide reduced food
resources, increased competition with other marine species, or greater exposure to predation

Other potentially important climate-related changes in the marine environment include
sea-level rise (IPCC 2007) and altered patterns of coastal upwelling (Wang et al. 2015). The
consequences of sea-level rise for nearshore fishes are uncertain and will be strongly influenced
by local topography; new habitat could be created in some areas but lost in others. Saltwater
inundation may affect species that sometimes spawn immediately above tidewater (e.g., pink and
chum salmon, *O. gorbuscha* and *O. keta*). Both positive and negative effects on estuaries are also
possible as new land is inundated, and the degree to which estuarine productivity is altered will
be influenced by the extent of human development. Where development of estuary and coastal
shorelines is extensive, sea-level rise will likely result in more seawalls, channelization, and
other measures to prevent flooding during storm surges (Neumann et al. 2015).
Changes in the patterns of coastal upwelling in the NWFP area could have very significant effects on anadromous fishes as well as other animals that depend on marine food webs. Wind-driven ocean currents regulate the strength of coastal upwelling along the Pacific Coast, where nutrients from deep ocean waters fuel plankton blooms that are critical to marine food webs that support salmon (Francis and Sibley 1991). Long-term shifts in the timing and intensity of coastal currents and upwelling have accompanied climate change in the eastern Pacific Ocean, with winter and spring storm tracks gradually shifting northward (Salathé 2006) and upwelling along the coast in the NWFP area becoming more erratic and unpredictable (Bylhouwer et al. 2013). Anadromous salmonids are particularly vulnerable to changes in upwelling because survival of fish in the first few weeks after entering the ocean depends heavily on their ability to feed and grow large enough to avoid predation (Beamish and Boullion 1993, Pearcy 1992, Walters et al. 1978).

The occurrence of interdecadal shifts in sea-surface temperatures and related weather patterns (Pacific Decadal Oscillation—PDO) from cool/wet to warm/dry conditions (Mantua et al. 1997) further complicates the productivity of marine environments along fish migration routes, with more favorable ocean conditions occurring when the NWFP area is in a cool/wet phase than in a warm/dry phase. Wang et al. (2015) used a suite of climate models to forecast upwelling over the next century and found that by the year 2100 coastal upwelling will likely start earlier, end later, and be more intense in the northern latitudes (British Columbia, Canada and southeast Alaska) than in southern latitudes (Northern California and Oregon). Wang et al. (2015) also noted that an intensification of upwelling could actually promote plankton productivity, but in extreme cases could also result in large swaths of anoxic conditions.
developing over broad areas, leading to massive die-offs of marine life where such conditions
develop. Taken together, the new information on climate-related PDO cycles and trends in
upwelling patterns suggest that the marine environment along the Pacific Coast is becoming
more variable spatially and temporally, with Northern California and Oregon being somewhat
more likely to exhibit unpredictable ocean conditions than in more northerly latitudes. For
migratory organisms such as anadromous salmonids whose lifecycles are adapted to being in the
right place at the right time for feeding and reproduction, introducing more variability into the
part of their lifecycle where most growth occurs is likely to add to population destabilization.

Climate effects on fish lifecycles—

Although the extent to which a particular fish population in the NWFP area will be affected by
climate change depends to a large degree on changes that occur at the local level, climate-related
effects, both favorable and unfavorable, can accumulate across multiple life-history stages.
Restricting an understanding of climate influences to a single life-history stage may well
underestimate the total effect on the population. Further, because of the wide geographic ranges
of many native fishes and the heterogeneity of aquatic environments in which they reside,
climate effects may be expressed differently across the range of a given species. Locally adapted
life histories vary over broad landscapes, and different species; even stocks of the same species
can exhibit dissimilar responses to similar climate trends (Schindler et al. 2010). A number of
papers have investigated the potential effects of climate change on Pacific salmon, but these have
primarily been overviews (e.g., Bryant 2009, ISAB 2007) or results of modeled effects on a
given life-history stage (e.g., Crozier and Zabel 2006, Rand et al. 2006) and its associated habitat
(e.g., O’Neal 2002). A comprehensive review of the effects of climate change on native fishes in
the NWFP area across their ranges, including effects accumulated across multiple life-history stages, is lacking.

Understanding the potential consequences of altered future conditions, particularly where the perceived effects may not be lethal, requires consideration of the effects at each life-history stage (Fleming et al. 1997, ISAB 2007, Jonsson and Jonsson 2009). Changes at one life stage can cascade throughout the remaining stages, significantly altering population response. Focusing on anadromous Pacific salmonids, it is possible to examine the overall impacts of climate change by identifying effects at each life-history stage and discussing how those impacts might be propagated through succeeding stages. These effects and potential management options are listed in table 9. It is also possible to identify attributes of Pacific salmon lifecycles that promote their adaptive capacity to climate change along with options for managers and decision makers to protect and enhance those attributes in order to mitigate potential climate change effects in the NWFP area.

**Other climate-related factors**—

Climate warming will lead to an increase in the frequency and severity of wildfires (IPCC 2007; chap. 2 this volume). Wildfire trends in the NWFP area will be complex because the area includes a wide range of forest types, elevations, weather regimes, and forest-management histories (Hessburg and Agee 2003); hence risks of damage to native fish habitats are likely to be highly variable across the region. In addition to altering wildfire frequency and intensity, climate change will also influence outbreaks of insects and forest diseases (Spies et al. 2010) in some cases, leading to alterations of forest stands that affect aquatic habitats. Wildfires, insects, and forest diseases should not be viewed strictly as threats to native fishes, however—they may also
provide benefits. They can create openings and patches along water bodies that result in more complex stream channels and greater biodiversity (Flitcroft et al. 2016, Reeves et al. 1995, Rieman et al. 2006). In addition, the erosional processes that accompany these disturbances are important for recruiting wood and coarse sediment that form essential habitats for many aquatic organisms (Benda et al. 2004). Thus, actions that seek to control erosion and other ecological processes that occur following wildfire may have long-term and unintended negative consequences for aquatic ecosystems (Chin et al. 2016).

The effects of climate change on aquatic ecosystems in the NWFP area expressed through wildfires, insects, and diseases will be complex and difficult to predict, but it will be important to examine the current responses to wildfire and consider making potential changes to allow fire to be more ecologically beneficial. Climate change will likely influence the expansion of nonindigenous plant and animal species in the NWFP area, while at the same time either reducing or even extirpating native species (Dale et al. 2001). Nonindigenous species include undesirable invasives, species undergoing expansion of their native ranges, and nonnative species deliberately introduced for commercial, recreational, or cultural reasons. They can occur in both terrestrial (riparian) and aquatic ecosystems. Nonindigenous species are not always harmful to native fishes or their habitats, but in some instances they can: 1) compete with, prey upon, hybridize with, or infect native species with novel pathogens; 2) greatly alter the structure of food webs; or 3) cause habitat changes that reduce the productivity of desirable aquatic organisms.

Sanderson et al. (2009) provide a useful summary of underappreciated threats to salmon posed by nonindigenous vertebrates, invertebrates, and plants. They conclude that the threats
posed by nonindigenous species may equal or outweigh threats posed by traditionally perceived causes of decline—habitat alteration, harvest, hatcheries, and hydroelectric production. Many of the nonindigenous fishes known to harm native fishes of the NWFP area are warmwater fish species deliberately introduced from eastern North America. In some river basins these forms have largely displaced native fishes from dominant roles in the aquatic food webs of low-elevation, low-gradient rivers (ISAB 2012). Continued warming will favor the expansion of warm-adapted species in western North America (Rahel et al. 2008), and shrinking headwater flows resulting from longer, drier summers (Moore et al. 2007) could force cool-adapted native species lower in drainage systems where there will be greater opportunity for unwanted interactions with established populations of introduced game fishes.

Restoration and Response to Climate Change under the ACS

Watershed improvements implemented in the Northwest Forest Plan— An important goal of the Northwest Forest Plan was to create a managed federal forest landscape in which natural ecological processes sustained healthy populations of native fish and wildlife (USDA and USDI 1994a). Architects of the Aquatic Conservation Strategy of the NWFP recognized that federally managed forests might anchor the recovery of imperiled native fishes, but because of their location relative to state and private forests as well as other types of land use (which tended to be located in lowland areas), they could not ensure that appropriate conservation measures would be applied throughout the full suite of freshwater environments to which many native species, particularly anadromous salmonids, were exposed (Sedell et al. 1997). Nevertheless, many of the aquatic conservation actions that emerged from the NWFP
were considered at the time to provide more protection to aquatic and riparian habitats than had ever before been implemented on multiple-use forests in the Pacific Northwest (NRC 1996). The region’s national parks and designated wilderness areas were also considered to possess high-quality habitats where natural ecosystem processes could operate. However, the scarcity and location of such areas (Reeves et al. 2016a, Sedell et al. 1994) were generally believed to be too inadequate to prevent species or their distinct population segments (evolutionarily significant units) from becoming imperiled, or to hasten recovery. Given the impacts of climate change on fish lifecycles as discussed above, how the NWFP as it is currently being implemented addresses these potential impacts merits priority for future research.

**Restoration of mid and late-seral forest stands**—Concurrent with the restoration of mid- and late-seral stands in the NWFP area will likely see a reduction in large openings caused by regeneration harvests (clearcuts) and by wildfire as result of continuing fire suppression (chap. 3 this volume). As forest stands grow older in the seasonally transient snow (“rain-on-snow”) zone, snowfall interception by branches will diminish the accumulation of ground-level snow and prolong melting and runoff processes during subsequent rain events (Harr 1986). Peak flows were found to increase by as much as 20 percent in small watersheds and 30 to 100 percent in larger basins over a 50-year period in the western Cascades of Oregon in response to road building and clearcutting (Jones and Grant 2001). However, a recent synthesis of peak runoff studies in western Oregon and Washington (Grant et al. 2008) concluded that the incremental contribution of clearcutting to peak flows in the transient snow zone was minor relative to other types of human disturbance, and would likely be confined to stream reaches possessing 2 percent gradients with sand and gravel substrates. In
areas where climate change results in an expansion of the transient snow zone, restoration of late-seral stands is likely to reduce the frequency and possibly duration of flows that are capable of mobilizing substrates of some fish-bearing streams, which could benefit survival of developing fish eggs and alevins as well as the abundance of amphibians and benthic macroinvertebrates.

One climate change trend with important implications for native fishes is the lengthening of low-flow periods during the warm season; aquatic organisms in watersheds with reduced snowpack will be especially affected by lower summer flows. Although not thoroughly investigated, the capture of fog by tree branches can result in “fog drip” that contributes to runoff during times when rainfall is scarce (Harr 1982). Whether climate change will alter the frequency of foggy days in the NWFP area is poorly understood, but there is preliminary evidence that the intensification of wind-driven upwelling in the California current as a result of increased CO2 could lead to intensified fog and increased moisture flux along the Pacific Northwest coast during the upwelling season (Snyder et al. 2003). However, Johnstone and Dawson (2016) reported that fog frequency along the northern California coast declined by 33 percent in the 20th century. Nonetheless, restoration of late-seral stands will result in taller trees with larger limbs, which could capture more moisture and deliver some of it to streams during a season when water is in short supply.

The progressive impoverishment of large wood in Pacific Northwest streams, particularly large-diameter habitat-forming tree trunks and rootwads, has long been recognized (Bisson et al. 1987, Sedell and Swanson 1984). Climate change is expected change the frequency and severity of fires and the incidence of forest pathogen outbreaks in many parts of the NWFP area (see
chap. 3), However, the ensuing recruitment of large wood to streams, a key component of fish
habitat, may be limited if landslide-prone headwalls that normally deliver this material to
channels during and following natural disturbances no longer contain trees of the size needed to
form and maintain structural fish habitats. The importance of wood recruited to streams from
unstable hillslopes is often underappreciated. For example, Reeves et al. (2003) found that 65
percent of the large wood pieces and 47 percent of the large wood volume in an Oregon coastal
stream originated from upslope sources. Meeting the challenge of climate change would include
inventorying and mapping unstable headwall areas, protecting them from forestry-related
disturbance, permitting natural wood delivery processes to occur, and allowing late-seral stands
to develop on them where appropriate (Cissel et al. 1999).

Road decommissioning and fish passage barrier removal—

Reducing the hydrological and biological effects of forest roads in the NWFP area will likely
improve watershed resilience to the adverse effects of climate change on aquatic ecosystems.
Road cuts are known to be a major contributor to accelerated runoff during storms by
intercepting subsurface flow and capturing it in ditches, which rapidly deliver water and fine
sediment to streams (Wemple and Jones 2003). As the intensity of storms increase with gradual
warming and, in some parts of the NWFP area, with greater precipitation, the risk of streambed-
mobilizing runoff events will rise as well. Reducing the exacerbating effects of road drainage
networks on peak flow in watersheds where roads have been decommissioned could lessen the
potential for severe storms to scour eggs and alevins in stream gravels and likewise reduce the
intrusion of harmful fine sediment into spawning substrates. In addition, eliminating road-related
initiation points for landslides through road decommissioning will help return the frequency of
mass wasting in watersheds to more natural levels.

Road corridors can serve as important invasion routes for nonnative species, especially
nonnative plants (Heckman 1999), and climate change is likely to favor continued expansion of
nuisance and harmful exotic herbaceous species in watersheds (Dale et al. 2001). The effect of
invasive plants on riparian ecosystems in federally managed forests has received relatively little
study, but some plants (e.g., Asian knotweed, Polygonum spp.) are capable of displacing native
vegetation (Urgenson et al. 2009) and disrupting the transfer of organic material from streamside
vegetation to stream channels. Invasive plant control programs are costly, and even in riparian
zones where treatments have been applied the long-term reestablishment of native plants has
been difficult to achieve (Claeson and Bisson 2013). Therefore, reducing road densities in
NWFP watersheds should help forestall the movement of unwanted nonnative plants into
sensitive riparian areas and protect the integrity of native plant assemblages.

Floodplain protection—
One of the key tenets of the Aquatic Conservation Strategy was that connections between
streams and rivers and their associated floodplain and wetland habitats should be protected and,
if necessary, restored (Reeves et al. 2006). In valleys where rivers are unconstrained and riparian
forests are well developed, off-channel habitats such as braided streams, oxbow lakes, springs,
and other floodplain features provide important seasonal rearing habitats for a wide variety of
aquatic and terrestrial species. Floodplains are considered to be among the most biophysically
complex and diverse systems on earth (Bayley 1995). Flood pulses that redistribute sediment and
organic matter create a dynamic mosaic of physical habitat features (Junk et al. 1998, Stanford et
al. 2005) on floodplains, which support diverse and productive biological communities. In forested regions of the Pacific Northwest, flood-induced channel migration creates a variety of aquatic habitat patches that vary in age and connectivity with the main channel, from connected side channels that reside within the active flood zone to disconnected side channels that become connected only during larger flood events.

Flood-induced erosion and deposition of substrate also create dynamic and heterogeneous plant communities. Early-successional species such as alder, willow, and cottonwood are generally found on newly deposited sediments, whereas mixed-species (deciduous and conifer) mature forests and old-growth conifer forests are found on older and more stable floodplain surfaces (Naiman et al. 2010). This spatial heterogeneity can also create highly complex and spatially structured food webs (Bellmore et al. 2013), which may be important for mediating the strength of predator-prey interactions and promoting biodiversity and resilience (Bellmore et al. 2015)

In the context of large-scale environmental stressors such as climate change, intact floodplains may be hubs of ecological resilience. The biological and physical diversity found across floodplains may promote ecological resilience in river networks via at least two pathways. First, enhanced species diversity in floodplains may provide functional redundancy within species guilds, whereby individual species extirpations may not significantly reduce ecological function (e.g., primary/secondary production, nutrient cycling) until some critical threshold is exceeded (Walker et al. 1992). Second, the physical heterogeneity or spatial complexity found across floodplains may provide critical refugia for individual species (Boughton and Pike 2013). For example, groundwater upwelling in floodplain springbrooks can provide cold-water thermal
refugia when main-channel waters exceed thermal optiums for a particular species (Ebersole et al. 2003, Torgersen et al. 1999). Unfortunately, however, many river-floodplain systems have been severely altered by human disturbance, which has constrained the physical processes that create and maintain habitat heterogeneity in floodplains (Tockner and Stanford 2002), and the associated resilience these habitats may provide. Although active restoration efforts are frequently targeted at recreating specific floodplain habitats (e.g., side channels), the reestablishment of natural channel-forming processes (Beechie et al. 2013), such as the “natural flow regime” (Poff et al. 1997), may be most successful at restoring the biophysical complexity of floodplains over the long term and help negate potential effects of climate change.

Winners and losers—
Climate change is projected to lead to changes in the distribution and abundance of native fishes and a host of other aquatic-riparian organisms in the NWFP area. Some species will be adversely affected by climate-mediated shifts in environmental conditions; others may actually benefit from the changes. Whether conditions for each species will become more or less favorable for a particular species depends on physiological requirements, life-history and migratory patterns, habitat preferences, and shifts in aquatic community composition. Fishes that prefer warm water and benefit from alterations in aquatic food webs and hydrologic regimes that accompany climate change will likely increase in abundance and expand their ranges. Other native fishes that prefer cool water will likely suffer losses from recently established predators and competitors; elements of their habitats that are needed at different points in their lifecycles will likely decrease in abundance, and their ranges will either contract or shift northward. Population fragmentation in cool-water fishes is also likely to increase as favorable thermal conditions
retreat to higher elevations, and smaller populations may suffer reduced genetic variability that threatens long-term survival (Kovach et al. 2015).

For anadromous species, survival and growth at sea will depend on how climate change alters upwelling patterns, plankton blooms, forage fish populations, predator abundance, and other potentially limiting variables. In the fisheries management community there is no clear consensus on whether freshwater or marine environments are “more important” to regulating the abundance of Pacific salmon, but it has become apparent that both ecosystems can exert a strong influence on run size, and that there are many uncertainties about how these two ecosystems interact to govern population viability and resilience.

In the NWFP area, climate change will lead to freshwater alterations that will be more or less favorable for some fish species relative to others. In figure 17 we list life-history strategies of fish that could increase vulnerability to the types of habitat change discussed earlier in this chapter; these include inflexible habitat specialization, extended freshwater rearing (1 year or more), low movement and spawning stray rates, potential for extended exposure to high water temperatures in their preferred habitats, and fall spawning placing them at risk of exposure to flow extremes. We also list life-history and habitat requirements that are likely to fare better in future climates. These include being able to use many different habitat types (habitat generalist); an abbreviated period in freshwater prior to seaward migration; high movement and spawning stray rates; either brief exposure to high water temperatures or a tolerance of prolonged elevated temperatures; and spring spawning that occurs after peak winter flows. Fishes are then arrayed along a risk scale ranging from those we believe to be less vulnerable to harm from climate change to those that may be moderately vulnerable, and finally to those that may be at high risk
of long-term harm. No species possesses all life-history attributes that are well adapted to
thrive under predicted climate regimes, just as no species possesses only attributes that are ill
adapted to all projected future conditions. However, based on what is known about climate-
related trends in freshwater habitats and on detailed knowledge of the life-history requirements
of native Pacific Northwest fishes, we suggest that there will be winners and losers among fish
assemblages. To some extent, the NWFP addresses many of the habitat changes likely to be
associated with climate-related alterations in federally managed forests, but some changes (e.g.,
trends in marine conditions) will not be materially affected by NWFP implementation.

The geographical distribution of native fishes and the variation in their life histories,
combined with the wide range of effects of climate change on freshwater environments, make it
difficult to predict which species will benefit most from NWFP aquatic habitat protections. In
figure 18 we divided the NWFP area into four zones: eastern, western, northern, and southern.
The western zone includes watersheds draining coastal mountain ranges, whereas the eastern
zone includes central lowlands of the NWFP area (Puget Sound, Willamette Valley, and
California’s Central Valley) and western drainages of the Cascade Range and Siskiyou
Mountains. The northern zone includes all river systems north of the Columbia River; the
southern zone includes river systems southward to San Francisco Bay. The zones are not
mutually exclusive, because the northern and southern zones include both eastern and western
areas; however, some fishes occur primarily in coastal systems and others are found primarily in
eastern portions of the NWFP area.

Based on different types of improvements to aquatic habitats from implementation of the
NWFP that mitigate harmful effects of climate change as discussed above, figure 17 lists native
salmonids that are likely to benefit in some way from NWFP protections. A few of the fishes
e.g., Chinook and coho salmon, steelhead [anadromous *O. mykiss*], and coastal cutthroat trout
(*O. clarkii clarkii*) are found throughout the NWFP area and therefore occur on each list; others
(e.g., westslope cutthroat trout, *O. clarkii lewisi*) are limited to relatively small regions of the
NWFP area. Figure 17 does not include nonnative species or nonsalmonids. In general,
nonsalmonids (e.g., native minnows and suckers) are likely to benefit from climate warming and
may or may not respond to NWFP aquatic habitat improvements such as fish-passage barrier
removal. Nonnative salmonids (e.g., introduced chars—brook and lake trout, *Salvelinus*
*fontinalis* and *S. namaycush*) will probably be adversely affected by climate change, but also
may or may not benefit from NWFP actions. Other introduced species, especially warmwater
fishes (e.g., sunfishes and basses, Centrarchidae and *Micropterus* spp.) will likely become more
abundant, and may increase the risk of predation, competition, and exotic disease exposure to
native fishes. However, restoration of riparian habitats may reduce water temperatures and
restrict expansion of these fish (Lawrence et al. 2014).

Implementation of the NWFP represented a significant improvement in the protection
and management of freshwater habitats in federal forests of the Pacific Northwest. Although not
directed at mitigating the negative effects of climate change on native aquatic organisms at its
outset, the protections provided under the NWFP will provide benefits to populations of native
cold-water fishes throughout their lifecycles and will help maintain the diverse mosaic of habitat
types on the landscape that is essential for population resilience (Bisson et al. 2009, Beechie et
al. 2013). However, although many aquatic and riparian habitats in federal forests are likely to
retain favorable conditions for aquatic-riparian biota or to slowly improve as watershed
restoration actions are undertaken, it is important to recognize that federally managed forests are usually embedded in a landscape that includes many different types of land uses, and that the standard of environmental protection under other management regimes is quite different from ACS-based standards and guidelines of the NWFP (Reeves et al. 2016a). Climate-related changes in aquatic and riparian habitats on nonfederal lands may be much less favorable for native aquatic organisms and more favorable for a variety of nonnative species.

As the biological communities of whole river systems are transformed under a changing climate, there will be a continuing need to monitor the role that federal forests play in conserving native aquatic organisms in the NWFP area. It will be critical for planners to identify vulnerabilities to climate change as well as incorporate approaches that allow management adjustments as the effects of climate change become apparent (Joyce et al. 2009). Because of the nature of environmental variability, the inevitability of novelty and surprise, and the range of management objectives and situations across the NWFP area, no single approach will fit all situations. A range of management options could include practices focused on mitigating or negating the effects of climate change by building resistance and resilience into current ecosystems, and on managing for change by enabling ecosystems and associated biota to adapt to climate change (Joyce et al. 2009, Perry et al. 2015). Better and more widespread implementation of already-known practices that reduce the effects of existing stressors represents an important “no-regrets” strategy (Joyce et al. 2009). These management opportunities will require consideration of the Forest Service’s adaptive capacity, including availability of personnel with the expertise to conduct required technical analyses, and being
able to work cooperatively with the public and other federal agencies to develop and implement the resulting management strategies.

The marine environment is likely to be a major challenge for Pacific salmon in the NWFP area. The predicted effects of climate change on the oceans, including acidification and increased temperatures, and their potential ecological consequences, reduced survival and size of returning adult fish, were described earlier. Pacific salmon have survived climate shifts in the past (Waples et al. 2009) and likely have the ability to persist in many areas of their current range even in more pessimistic climate change scenarios. Salmonid populations exhibit large genetic and phenotypic diversity relative to many other bony fishes (Crozier et al. 2008, Schindler et al. 2010, Waples 1991) and can adapt to changing conditions rapidly (Healey and Prince 1995, Quinn et al. 2001). This genetic and phenotypic diversity has allowed for persistence in highly dynamic and ecologically diverse environments in the past (Greene et al. 2009, Moore et al. 2014, Waples et al. 2009) and will be a key to future survival (Mangel 1994). (However, we note that Gienapp et al. [2008] cautioned that our knowledge about the role of genetic variation and the ability of natural populations to respond adaptively to current and future environmental change is limited, and that assuming that adaptation will or can happen is risky because of the uncertain rate and extent of climate change, effects of invasive species, and altered ecological processes.) The challenge to managers will be to conserve natural environmental complexity in space and time so it can provide the physical templates for maintaining genotypic and phenotypic diversity in populations that are currently strong, or to restore environmental complexity where it is currently compromised.
Research Needs, Uncertainties, information Gaps, and Limitations

The scientific basis of the Aquatic Conservation Strategy of the NWFP is still sound and is supported by new science produced since its inception by FEMAT in 1993. However, we have learned much about relationships of riparian vegetation to stream habitats and environments that have refined and modified some hypotheses that were used to develop the ACS in the early 1990s. Among the issues advanced by research findings in the past 20 years are:

- The ecological, physical, and biological importance of headwater and intermittent streams
- The contribution of periodic disturbances to the resilience and productivity of aquatic ecosystems
- The inherent variation of aquatic ecosystems in space and time
- A better understanding of the variation in where key ecological processes occur within the stream network and the development of new tools to identify these locations
- An understanding of the variation in the capacity of aquatic ecosystems to provide habitat for various fish species
- Awareness of climate change and its potential effects

Emerging science suggests that the absence of disturbance and management in terrestrial ecosystems, primarily fire, may be affecting vegetation, and combined with climate change, is likely altering these ecosystems (chap. 3, this volume). The same trends are likely occurring in riparian and aquatic ecosystems in a manner that is not fully understood at present; this could be a useful subject for research conducted in an adaptive-management context to provide information to managers, regulators, and policy makers in a timely manner.
Climate change is expected to affect aquatic and riparian ecosystems throughout the NWFP area, though with much uncertainty. Effects will likely vary widely within and among watersheds, and new approaches are needed to identify this variation and help craft strategies and programs for mitigation and adaptation. Having the capacity to do the needed analysis will also be critical for the involved agencies to successfully meet this challenge in a timely and effective manner, particularly in an era when budgets and personnel for federal land-management agencies are declining (see chap. 8, this volume). Thus, development of cost-effective and scientifically sound analysis procedures performed with close collaboration between research and management is the key to addressing this need.

There are continuing challenges about how to consider the effect of climate change on aquatic biota. Much of the focus has been on individual species. Research that focuses on understanding potential effects over the life-history of species and how effects may cascade through life-history stages as well as consideration of community level effects is critical. Understanding the effects on water quantity and quality are also important, particularly across spatial scales within watersheds, among watersheds, and across seasons and years. It is likely that aquatic and associated terrestrial ecosystems will change in uncertain ways under a changing climate, and that this change will vary widely across the NWFP area. Considerable research is needed on the dynamics of aquatic ecosystems in space and time and how biota respond and are adapted to changes.

The contribution of federal lands to the conservation and recovery of ESA-listed fish continues to be important. However, federal lands alone are likely to be insufficient in geographic scope to reach the comprehensive goals of the NWFP relative to recovery of listed
fish, particularly many Evolutionarily Significant Units of Pacific salmon, as originally expected by FEMAT (1993) and the record of decision (USDA and USDI 1994a). Although the geomorphic setting of streams on federal lands maybe as capable of providing sufficient favorable habitat, particularly for salmon, as originally expected, streams on state and private lands may have a much greater potential to provide habitat in many watersheds. Thus, it will be important to work closely with adjoining landowners and other interested parties to develop more comprehensive efforts across species ranges. The development of incentive programs is likely to be important to develop partnerships for fish-habitat management across landownerships. Developing an understanding of the variation in the capacity of watersheds to provide favorable conditions for fish and other aquatic biota could be critical to the success of such programs.

The ACS monitoring program is critical to evaluating how well the goals of the NWFP are being met and what changes in management may be needed to address emerging issues and new science. There are a number of important questions and considerations regarding approaches to monitoring aquatic ecosystems that would benefit from more research and development. It would be prudent to convene a panel of researchers and practitioners to review the current assessment concepts and protocols for bio-assessments and develop a complementary research program, as has been done for other components of the NWFP (e.g., northern spotted owl). Understanding the appropriateness and consequences of using different approaches is particularly critical owing to the growing concern about climate change, fire suppression, and other emerging issues. The monitoring community faces these and other challenges now and into
the future. Exploring innovative ways to make informed assessments of aquatic and riparian condition is critical to successful resource management.

Monitoring the effectiveness of the ACS will continue to be important. Some meaningful uncertainties remain regarding the aquatic-riparian monitoring approaches, especially relative to whether they are capable of capturing the effects of the ACS on a wide range of ecological processes and species of aquatic and riparian ecosystems. Research is needed to test the ecological validity of individual metrics and different ways of combining metrics to represent different components of complex and diverse aquatic and riparian ecosystems and communities. It would, therefore, be prudent to compare alternative approaches in the face of the new understanding about the behavior of aquatic and riparian ecosystems in time and space and the yet-to-be-understood effects of disturbances or lack of disturbance, climate change, and novel ecosystems. A related research need is to better understand the relationship of the productivity of aquatic biota, which include organisms other than salmonids, in the context of different upland vegetation and in-channel successional stages or restoration treatments. This type of information can feed into watershed assessments to better ensure that the effects of the ACS are captured more comprehensively relative to the biota that are a key ecosystem service of the aquatic-riparian ecosystems. In particular, we lack information about the amount, pattern, and type of restoration activities that have occurred in upland and riparian forests. Implementation monitoring has not been adequate to enable a sufficient understanding of the consequences of restoration actions (or lack of actions), especially relative to how they may have altered aquatic ecosystems in space and time.
Roads and their effects will continue to be a major issue in the NWFP area. Both research on road effects and the continued development of analysis tools such as GRAIP (Black et al. 2012) are important. The same is true for effects of culverts on ecological processes and the movement of aquatic biota. These are current priorities given the uncertainties of climate change. Also, understanding how to balance fire management, recreation, and other needs against potential negative aspects of roads will require a concerted cooperative effort of managers and physical, biological, and social scientists.

A key uncertainty that has emerged from our analysis is how to understand and assess the effects of “no action” management options and trade-offs managing for one factor (e.g., water temperature or wood recruitment) on other ecological processes or attributes. The assumption has been that focusing on one concern would not influence other processes or attributes, and that taking no action was synonymous with having no effect. However, these assumptions are questionable and deserve increased consideration and focus by researchers.

Several other topics relating to the components of the ACS merit further research. Watershed analysis could be reexamined so that it is conducted more efficiently and considers the appropriate spatial scales, including a smaller watershed of interest and its context within a larger basin. The larger-scale context is particularly relevant for effective landscape scale planning. In addition, no formal evaluation of the potential effectiveness of the network of key watersheds was conducted during development of the NWFP, nor has such an evaluation been attempted since it was implemented. New concepts, tools, and emerging understandings about aquatic ecosystems are now available to better assess and increase the potential effectiveness of key watersheds.
Conclusions and Management Considerations

The goal of the Aquatic Conservation Strategy (ACS) of the Northwest Forest Plan (NWFP) was to maintain and restore aquatic-riparian ecosystems on federal lands within the range of the northern spotted owl. A review of monitoring efforts and the pertinent scientific literature suggests that: 1) aquatic ecosystems in the NWFP area are likely improving as expected, albeit slowly; 2) the fundamental tenets and framework of the ACS are sound, and we are gaining more explicit understanding of several components over time; and 3) opportunities exist for implementing parts of the ACS differently while continuing to achieve its goals. The third finding is particularly applicable to the riparian reserve component of the ACS, where more active management may help to address potential concerns about the effects of the lack of natural disturbance (primarily wildfire), and climate change. Our understanding of the effects of these factors on aquatic ecosystems is incomplete (though evolving) at this time, but because there could be significant implications for the productivity of these systems, they will continue to be a major focus of research. In the meantime, to address disturbance-related concerns, land managers would benefit from a comparison of alternative approaches in the context of their site-to-watershed situation, inclusive of how those aquatic ecosystems behave in space and time. A dynamic watershed perspective would foster the development of new management approaches.

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hydrologic variability threatens depleted anadromous fish populations. Global Change


Appendix A: Influence of Climate Change on Life-Stages of Pacific Salmon

Adults

The species of anadromous Pacific salmonids found in the NWFP area and their freshwater and marine residence times are shown in table 8. The freshwater environment is used for both growth and reproduction; the marine environment is used for growth and the initiation of sexual maturity. Depending on species, fish may spend from 1 to 5 or more years in the eastern Pacific Ocean before returning to freshwater to spawn. An exception is coastal cutthroat trout, which generally make limited forays into nearshore areas and typically do not range more than 65 miles (100 km) from natal rivers (Trotter 1989). Owing to marine heterogeneity, the disparate migration patterns of various stocks, and the widely varying amount of time spent at sea, the influences of climate change on survival and growth of different populations of salmon in the ocean will vary.

Although the specific effects of climate change on the marine survival and growth of salmon will depend on the location of their natal rivers and their movements at sea, some trends seem to be common to populations along the Pacific Coast. Possibly as a result of decreasing pH and increasing temperature, salmon are becoming smaller and sometimes younger upon return to freshwater, and exhibit reduced marine survival rates. The size of returning adults of most Pacific salmon species has generally trended downward over the last 3 decades of the 20th century (Bigler et al. 1996), although there have been multiyear periods when both sizes and abundances have increased (Helle et al. 2007). Some populations of sockeye salmon in Bristol
Bay, Alaska have returned to spawn at a younger age in the second half of the 20th century. (Hodgeson et al. 2006, Robards and Quinn 2002).

A decline in the size of the largest fish caught in a Chinook salmon fishing derby in Juneau, Alaska, has occurred over the past 69 years (fig. A1). Although the origin of these salmon is not known with certainty, it is possible that some originated from rivers in the NWFP area, as migratory routes for some Pacific Northwest Chinook salmon stocks include southeast Alaska. There are many possible explanations for the observed declines in fish size, but several previously published examinations of adult salmon sizes from either commercial fishing records (Helle et al. 2007) or Alaskan fishing derbies (Fagan 1988) attributed at least some of the decline in size to increased competition due to large numbers of hatchery-produced salmon (Bigler et al. 1996, Francis and Hare 1997). However, such relationships are not simple. Helle et al. (2007) analyzed data for different species and stocks from northern Alaska to Oregon and concluded that adult body size resulted from both density-dependent factors (competition) and density-independent factors (environmental conditions). Long-term trends in body size observed over time in a Juneau, Alaska, fishing derby are weakly correlated to the gradually warming 42-month average Pacific Decadal Oscillation (PDO) index observed prior to when the fish were caught (fig. A1), suggesting a linkage between ocean conditions and fish size that portends future size declines under a warming climate. The relationship between gradual warming and shifts in the frequency and intensity of PDO fluctuations is unclear, but significant PDO regime shifts can signal major changes in the earth’s biophysical systems (Reid et al. 2016).

Decreases in adult body size resulting from changing environmental conditions in the ocean could also lead to reduced reproductive success. In Pacific salmon, both the number of
eggs (Hankin and McKelvey 1985, Healey and Heard 1984) and egg size (Quinn and Vøllestad 2003) are directly related to the weight of adult females. Reproductive capacity of populations could decline if females have fewer eggs (McElhany et al. 2000). Egg size, primarily related to yolk reserve, can also be an adaptation to the environment in which eggs develop. Fish that spawn in warmer areas tend to have larger eggs compared to those from cooler areas because the efficiency of yolk conversion to body tissue is reduced at higher temperatures (Fleming and Gross 1990). The survival and body mass at hatching of eggs incubating at warmer future temperatures could therefore be compromised if egg size does not increase as well.

Food webs in aquatic and riparian ecosystems are supported by the influx of marine-derived nutrients from returning adult salmonids (Bilby et al. 1996, Schindler et al. 2003). The productivity of many streams and rivers within the range of Pacific salmon is influenced by the quantity of marine-derived nutrients from salmon carcasses (Helfield and Naiman 2001, Willson et al. 2004). A reduction in the size and number of returning adult salmon could compromise the capacity of freshwater ecosystems to produce new salmon, with carryover effects on the wide variety of aquatic and terrestrial organisms that may also benefit from the consumption of eggs during the spawning period (Cederholm et al. 2001; Garner et al. 2009). The growth of juvenile salmon during the spawning season is important for their overwinter survival (Lang et al. 2006). Energy derived from eggs consumed by returning adults can also allow for longer migrations and extended spawning times (Copeland and Venditti 2009); thus fewer, smaller eggs could diminish this potential energy source.

According to climate change predictions for most rivers in the NWFP area, returning adult salmon will face warmer temperatures and lower flows if migrations take place in summer.
Some species and life-history types, such as stream-type ("spring") Chinook salmon and summer steelhead in the southern and middle portions of the Pacific Coast range of Pacific salmon, return to fresh water in spring or early summer months and hold in rivers and streams for several months before spawning. Adults feed infrequently and usually rest in large pools with cool water. Such pools are not abundant in late summer and early fall, with cool-water refuges likely to become even less available at those times as climate continues to warm. This circumstance suggests that holding and migrating adults may become increasingly stressed, which will diminish their reproductive potential and increase prespawning mortality. Beechie et al. (2006) believed that the loss of summer prespawn staging habitats in rivers entering Puget Sound, Washington, could result in the replacement of stream-type Chinook salmon by ocean-type Chinook salmon whose fall run timing avoids exposure to warm, low-flow summer conditions. For populations undertaking long upstream migrations to spawning grounds, elevated stream temperatures will incur higher metabolic costs and mortality (Rand et al. 2006), and fish that do arrive at spawning grounds may have reduced reproductive capacities (Miller et al. 2011). Warmer temperatures may also limit gonadal development; Pankhurst et al. (1996) found that female steelhead did not ovulate when temperature exceeded 70°F (21°C). The extirpation of Atlantic salmon in the southern portion of their distributional range is attributed to reproductive failure associated with elevated water temperatures in freshwater spawning areas (McCarthy and Houlihan 1997).

Elevated water temperatures during migration can have indirect effects on returning adults. Returning adults may be more vulnerable to disease and parasites if conditions are warmer in fresh water (Johnson et al. 1996, Ray et al. 2012) However, Stocking et al. (2006)
found no relation between water temperature and infection of salmonids with *Ceratomyxa shasta* in the Klamath River, California. Adults that are unable to find cool-water holding areas during migration in warmer water may be particularly vulnerable to disease because warm water will favor rapid disease transmission and virulence of warm-adapted pathogens that could lead to fish kills (Mathes et al. 2010). For example, Miller et al. (2011) presented evidence that elevated temperatures in British Columbia’s Fraser River have likely contributed to the virulence of a virus that infects adult sockeye salmon prior to entering the Fraser River, resulting in a high incidence of prespawning mortality.

Rising sea level (IPCC 2007) may affect the reproductive success of species that spawn close to tidewater, particularly some pink and chum salmon populations. For small populations that spawn in streams just above the high-tide level, elevated sea levels could reduce the available spawning habitat if suitable spawning sites upstream are inaccessible. The development and persistence of less-favorable ocean conditions could potentially influence the degree of anadromy in populations that possess both anadromous and nonanadromous (fully freshwater-resident) lifecycle options. Steelhead, the anadromous form of *O. mykiss*, persists at least in part because there is a fitness advantage associated with migrating to the ocean to feed and returning to freshwater to spawn (Quinn and Myers 2004). If this advantage is reduced or lost, residency could increase in populations, assuming that changes in the freshwater environment are suitable for the persistence of the freshwater life-history variant of rainbow trout (Benjamin et al. 2013, Rosenberger et al. 2015, Sloat and Reeves 2014). Other Pacific Coast populations of *O. mykiss* maintain primarily resident populations in locations
where the marine environment is believed to be unfavorable for survival and growth, as in
southern California (Behnke 2002).

3 Eggs and Alevins

Eggs and developing embryos will likely be affected by two different aspects of climate
change—increased temperatures during egg incubation and altered hydrographs. Under some
climate scenarios, winter temperatures are predicted to increase at faster rates than are summer
temperatures for Alaska (IPCC 2007), whereas the opposite is true for the more southerly NWFP
region (Mote and Salathé 2010).

Most research on climate effects on native fish has focused on the potential for elevated
summer temperatures (e.g., Crozier and Zabel 2006; Isaak et al. 2010). However, the effect of
elevated winter temperatures may be as, and perhaps even more, pronounced and ecologically
significant than increases in summer temperatures. Increased winter temperatures in the NWFP
area will result in more precipitation falling as rain rather than snow. Watersheds that historically
developed a seasonal snowpack will experience a trend from snow to rain, resulting in more
rapid runoff in winter and early spring when snow usually falls, and lower late-spring and early-
summer flows due to reduced snowmelt (Hamlet and Lettenmaier 2007, Hamlet et al. 2005,
Tague and Grant 2009). In Washington state’s transitional drainage systems that historically
possessed both fall/winter and spring/summer runoff peaks, the shift to a rain-dominant
hydrograph is expected to be the most dramatic. Substantial increases are anticipated in the
magnitude and frequency of extremely high-flow events in winter coupled with substantial
reductions in summer low flows (e.g., Mantua et al. 2010, Elsner et al. 2010). However, because
snowpack will be reduced, rivers with snowmelt-dominated hydrographs could likely see a
reduction in the magnitude of high flows during spring runoff. Loukas and Quick (1999) predicted that floods in the snowmelt-dominated continental portions of British Columbia will decrease in magnitude by 7 percent and in volume by 38 percent, and occur as many as 20 days earlier, as a result of the snow-to-rain transition. In coastal areas, Loukas and Quick (1999) projected that there would be little change in the timing of floods, but that on average peak-flow magnitude (+14 percent), flood volume (+94 percent), frequency (+11 percent) and duration (+44 percent) would all increase.

High-flow events will influence egg and alevin survival, depending on the depth of the redd, the size of the female, and the location of spawning in the stream network. Eggs in shallower redds will be more susceptible to being scoured than will those in deeper redds, and smaller salmon often excavate shallower redds than larger salmon (van den Berghe and Gross 1989). It has been speculated that increased peak flows during the incubation period could result in decreased survival of eggs and embryos in populations exposed to hydrologic regimes that have become more prone to gravel-mobilizing flows (Battin et al. 2007)

Potential effects of hydrographs altered by climate change are likely to vary among species and life-history forms. In most drainages of the NWFP area, scour is likely to increase the most in small streams or in confined, steep rivers, affecting fish such as bull trout that spawn in the late fall and early winter when the most severe storms tend to occur along the northwestern Pacific Coast (Isaak et al. 2012). Fish spawning in lower-gradient, unconfined areas, such as coho, Chinook, pink, and chum salmon, could be less affected. Studies that have examined potential effects of increased flows on streambed scour (Battin et al. 2007, Leppi et al. 2014, Shanley and Albert 2014) assumed a uniform relationship between flood magnitudes and the
vulnerability of salmon populations and their habitat. However, the geographic range of Pacific
salmon is characterized by exceptional topographic complexity and watershed dynamism
(Montgomery 2000), which can generate considerable diversity in watershed- and stream reach-
scale responses of habitat to flood disturbance (Buffington 2012, Montgomery and MacDonald
2002). Thus, effects of increased flows are unlikely to be similar among watersheds or even
among reaches within stream networks.

Previous research has demonstrated that stream-channel response potential varies
according to position within the dendritic structure of stream networks (Benda et al. 2004),
variation in valley and reach-scale confinement (Coulthard et al. 2005, Montgomery and
Buffington 1997), and differences among species in their use of habitats created by this
physiographical complexity (Goode et al. 2013). In terms of management, the floodplain
connectivity may ameliorate the effects of future increases in discharge on streambed dynamics.
Floodplain connectivity in unconfined reaches provides a “stress release valve” (McKean and
Tonina 2013) that limits vulnerability of salmon spawning habitat even in large floods with
return intervals of decades to centuries (Goode et al. 2013, Lapointe et al. 2000, McKean and
Tonina 2013). In this regard, maintaining or restoring connectivity between streams and adjacent
floodplains will mitigate near-term responses to increased flood magnitudes. Additionally,
maintaining or restoring channel complexity and hydraulic roughness from large wood may
further mitigate the effect of higher flows on salmon spawning habitat (Buffington 1995, Sloat et
al. in press).

The rate of development of eggs and the size of fish at emergence is related to water
temperature. Egg development depends on the accumulation of degree days (Neuheimer and
Taggart 2007). Even slight increases in temperature can accelerate rate of development and ultimately result in earlier time of emergence from the gravel (McCollough 1999) (fig. A2). Accelerated development leads to smaller individuals at emergence because metabolic costs decrease the efficiency of yolk use (Beacham and Murray 1990, Elliott and Hurley 1998). Upon emergence, smaller fish are more susceptible to displacement at higher flows. Some fish species may be more influenced by thermal shifts during incubation than others; Beacham and Murray (1990) suggested that coho salmon are adapted for low water temperatures during development and could experience lower survival under warming climate scenarios.

There are important ecological implications of climate-related changes in the time and size of fish at emergence. Earlier emergence can result in an extended growing season, a benefit that can lead to increased fitness. Holtby (1988) found that an increase of 1.3°F (0.7°C) in winter water temperatures following timber harvest in Carnation Creek on the west coast of Vancouver Island, British Columbia, resulted in coho salmon emerging six weeks earlier. Size at age increased because of the extended growing season, which resulted in more fish completing their freshwater-rearing life history in one year rather than two. Coho salmon in Carnation Creek also smolted and moved to sea about two weeks earlier following timber harvest (which raised stream temperatures); however, marine survival declined, possibly as a result of the decoupling of the timing of smolt migration and marine plankton blooms (Holtby and Scrivener 1989). Similarly, warmer winter temperatures increased the length of the growing season of recently emerged sockeye salmon in southwest Alaska. Like coho salmon in Carnation Creek, sockeye salmon grew faster and more underwent smolt transformation at age 1+ during warm periods rather than
at age 2+ in cooler periods (Schindler et al. 2005). However, age-1+ smolts were smaller than age-2+ smolts and were expected to have decreased marine survival.

Juveniles

Juvenile Pacific salmon (defined here as recently emerged fry up to, but not including, smolts) face a number of challenges from the potential effects of climate change. These challenges will include elevated temperatures and altered streamflows, both of which can affect physical and biological aspects of stream habitats. The type and extent of flow effects will vary depending on the time of emergence. For example, fish emerging in the late winter and early spring may experience high flows caused by earlier snowmelt. The consequences a changing hydrograph will depend to a large degree on the geomorphic setting in which spawning and emergence occurs. In some settings, increased flooding could improve use of floodplain habitats when fish in wide, geomorphically unconstrained channels have access to habitats where floodplain vegetation is intact and secondary channels are available.

Low-gradient streams and rivers can be important areas for post-emergent and seasonal growth (Brown and Hartman 1988, Moore and Gregory 1988, Peterson 1982a), and marginal areas with reduced water velocities provide refuge against downstream displacement. Fry that emerge at a smaller size if water temperature is warmer can potentially overcome their size disadvantage by gaining an early start on the growing season (Holtby 1988). Juvenile salmonids in rain-dominated hydrographic regimes often move into the lower reaches of the channel network or into off-channel habitats in autumn to seek refuge from unfavorable water velocities in the main channel (Ebersole et al. 2006, Everest 1975, Peterson 1982b, Solazzi et al. 2000). In high-elevation snowfall-dominated drainage systems, however, climate warming might not
significantly increase mid-winter flood flows and facilitate access to floodplain habitats if precipitation still falls as snow.

Under several climate scenarios, the onset of the low-flow period is expected to occur up to 4 to 6 weeks earlier in most areas as a result of warming (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009). An extended period of low discharge over the dry season would likely decrease the amount of habitat suitable to juvenile salmonids, and this effect could be most pronounced in small to mid-sized streams (Stewart et al. 2005), resulting in some reaches that formerly held surface flows throughout the year becoming intermittent or even drying completely. As noted by Battin et al. (2007), flow reductions in headwater areas during the dry season could force resident fishes downstream in the stream network as well as compromise their ability to cope with drought by reducing the network of connected, perennially flowing channels. Additionally, the downstream displacement of headwater-rearing fish will expose them to warmer temperatures than those to which they are adapted, and possibly to harmful biological interactions with native and nonnative species inhabiting the lower watershed.

The consequences of climate-induced changes in low flows for juvenile salmonids such as Chinook salmon and steelhead that often rear in rivers are likely similar to those in smaller streams, although the risk of river reaches becoming intermittent is lower because drainage areas are larger. Mantua et al. (2010) found widespread declines in summer discharge for many Washington state rivers under climatic warming scenarios. Likewise, Luce and Holden (2009) examined hydrographic records from drainage systems throughout the Pacific Northwest and found that summer flows in all types of hydrologic regimes have been declining, thus providing increasingly smaller rearing areas to river-dwelling species.
In addition to lower flows, elevated summer water temperatures will likely have strong ecological effects on juvenile Pacific salmon, with the direction and magnitude of influence varying geographically, by species, and by life-history type. Water temperature influences the metabolism, food consumption, and growth of an individual (Brett et al. 1969, Warren and Davis 1967, Wurtsbaugh and Davis 1977). Age and size of individuals also influence thermal effects; younger and smaller fish are most susceptible to thermal extremes (Brett 1952) and to short-term thermal variation (Elliott 1994). There is a temperature range in which an individual performs best given a certain level of food resources, and beyond that range, metabolic costs increase such that growth declines (Warren 1971). Increased temperature could potentially affect juvenile salmonids in opposing ways (Li et al. 1994). Warmer water could enhance primary and secondary aquatic production, leading to greater food availability; however, if the increased metabolic demands of higher temperatures lower food-conversion efficiency or if the organisms benefiting from higher temperatures are not preferred food items, the net effect of warming could be reduced growth (Bisson and Davis 1976). In southern portions of a species’ range, elevated temperatures could reduce the suitability of rearing areas for juveniles during the summer as temperatures exceed the point at which gains resulting from increased aquatic production are offset by physiological costs, resulting in reduced summer growth rates (Marine and Cech 2004). In contrast, growth rates of juveniles in more northern areas could increase if projected temperature changes stimulate aquatic productivity while remaining within the preferred physiological range for the species.

If the net effects of elevated temperatures resulting from climate change in southern areas reduce summer growth (Isaak et al. 2010, Royer and Minshall 1997, Scarneccchia and Bergersen
1987), juveniles will be smaller entering the winter (ISAB 2007), and overwinter survival may
decrease (Quinn and Petersen 1996). However, thermal increases may be beneficial for growth
during other seasons if abundant food is present. Sogard et al. (2010) found that juvenile
steelhead in the central coast of California attained the most growth in the spring and autumn,
and that juvenile coho salmon grew in in the winter in coastal Oregon (Ebersole et al. 2006,
2009).

Outcomes of interactions between salmonids and nonsalmonids can be influenced by
changing water temperatures. Rearing salmonids tend to out-compete nonsalmonids for food
resources and preferred feeding areas at cooler temperatures, whereas nonsalmonids have the
advantage at warmer temperatures (Petersen and Kitchell 2001, Reeves et al. 1987). The
susceptibility of juvenile salmonids to disease could also increase at warmer temperatures and
could be compounded by the presence of competitors that are less susceptible to the pathogens
infecting salmon and trout (Reeves et al. 1987). Additionally, warmer temperatures could lead to
increased predation from nonnative warmwater fish (ISAB 2007, Petersen and Kitchell 2001).
The aggregate results of these indirect effects are likely to be changes in the structure and
composition of fish communities in the affected stream systems (ISAB 2012), particularly in the
southern portions of the NWFP area where the potential for interaction with warmwater species
is greatest due to widespread introduction and proliferation of nonnative warmwater fishes.

The effects of climate change on rearing habitats for juvenile salmon at the local level
will depend, to some degree, on the geomorphic features of a particular location. Crozier and
Zabel (2006) suggested that two climate-influenced factors—stream temperature and flow—
could affect habitat in different ways: narrow, confined streams were predicted to be more
responsive to flow changes, and geomorphically unconfined streams would be more sensitive to
temperature changes. In addition, the future quantity and quality of freshwater rearing habitat of
Pacific salmon may also be influenced by predicted increases in the magnitude and frequency of
large disturbances. Climate change scenarios predict an increase in exceptional flood events
caused by transitions from snow to rain, accelerated glacial melt, wildfires, and forest pathogen
outbreaks (Dale et al. 2001, Hamlet and Lettenmaier 2007). Frequent large floods promote
landsliding and stream sedimentation in many areas (Miller et al. 2003). The effects of floods
and associated erosion events on freshwater habitat will vary depending on the geomorphic
setting, the magnitude and legacy of the event, the interval between succeeding disturbances, and
the extent to which the affected ecosystem has been altered by past human activities (Reeves et
al. 1995, Rieman et al. 2006).

Increased disturbance frequency and severity can have short-term negative consequences
for fish populations, including substrate scour and fine sediment intrusion that reduces egg and
alevin survival and macroinvertebrate abundance in confined channels, displacement of juveniles
downstream, and loss of surface flow in summer in reaches where porous material has been
deposited in the channel. However, in functionally intact systems there is a strong potential for
aquatic habitat complexity to improve with flooding because floodplain linkages can be re-
established and large wood will be recruited to the channel network (Bisson et al. 2009). Long-
term changes might be favorable to rearing salmon if the cumulative effects of climate change on
water temperature, fine sediment levels, and surface flows remain within limits tolerable to
juvenile salmon or only exceed those thresholds for a short duration.
Population productivity after large disturbances will also be enhanced by the presence of adjacent fish populations that provide sources of colonizers that help initiate recovery and that add to the phenotypic and genetic diversity of affected populations (Schtickzelle and Quinn 2007). But it is also possible that in highly altered watersheds where the cumulative harmful effects of climate change exceed environmental tolerance limits, the damage caused by large-scale disturbances is so great, and there are no nearby populations to provide new colonists, that local population extirpation will occur.

Lakes are important rearing habitats for sockeye salmon and will also be affected by climate change, although there are relatively few drainage systems in the NWFP area that support sockeye salmon runs. Potential effects will vary greatly depending on the location and features of the lake, but a primary effect will be the magnitude and seasonality of warming, with epilimnetic water and the timing of spring and autumn turnover experiencing the greatest changes (Stefen et al. 2001). Slight warming of deep lakes could lead to increased sockeye growth rates if temperatures stimulate primary and secondary production without significantly affecting the availability of cooler water during periods when the epilimnion becomes too warm for efficient metabolism. This benefit could be offset during the growing season by a reduction in the delivery of inorganic nutrients and dissolved organic carbon from terrestrial systems as a result of decreased spring and summer flows. Reduced inputs of nutrients and dissolved organic carbon from the surrounding watershed could result in diminished algal production, which would result in deeper light penetration and additional warming of the lake (Schindler et al. 1990).

The productivity of zooplankton, the principal food of juvenile sockeye salmon in lakes, will be affected by climate change, but whether or not the changes are beneficial will depend on
ambient thermal and hydrologic regimes. In Alaska, warming temperatures have resulted in
earlier ice melt, greater densities of zooplankton, and increasing sockeye growth rates (Schindler
et al. 2005). In contrast, earlier onset of spring in western Washington’s Lake Washington has
advanced lake stratification by 20 days in recent years, resulting in earlier diatom blooms and a
decline in cladocerans (*Daphnia* spp.), which are important prey species for juvenile sockeye
rearing in the lake (Winder and Schindler 2004).

Smolts

Anadromous salmonids typically undergo the smolting process and move to the ocean in spring,
although seaward migrations of some salmon stocks occur throughout the year. Water
temperature, day length, and changes in flow are the principal cues influencing the timing of
parr-smolt transformations. Environmental signals affecting smolting can be divided into
regulating and controlling factors (Byrne et al. 2004). Regulating factors act on juvenile salmon
before the migration and influence the physiological aspects of smolting. Controlling factors
operate during migration and affect the speed of downstream movement. Water temperature and
day length appear to be key regulating factors (Jonsson and Jonsson 2009). Day length is not
influenced by climate change, but increased temperature will affect the onset of smoltification.
For Pacific salmon, elevated winter temperatures can result in earlier migration times of smolts.
Chinook salmon have been observed to migrate earlier in warmer years than in cooler years
(Achord et al. 2007, Roper and Scarneccchia 1999), but Jonsson and Jonsson (2009) cite a suite of
other studies on Atlantic salmon, brown trout (*Salmo trutta*), and steelhead where water
temperatures did not affect the timing of smolt migration. Under certain conditions, elevated
temperatures may even inhibit parr-smolt transformation. Adams et al. (1973) found that
smolting in steelhead held at 59°F (15°C) or higher led to reductions of ATPase activity needed to initiate the smolt transformation process. Thus, the effect of altered temperature on timing of smolt migration remains unpredictable and likely will vary widely across populations.

To a large extent, streamflow determines the rate at which smolts move downstream (Connor et al. 2003, Smith et al. 2002). Climate model projections of stream runoff (Snover et al. 2003, Tague and Grant 2009) suggest that the onset of the low-flow period will occur 4–6 weeks earlier over much of the NWFP area in the next century. Projections of the annual cycle of elevated flows from melting snow for more northerly areas are not currently available, but we assume that they will be similar. The consequences of altered flows are likely to be population-specific, with the timing and smolt survival rates of those populations that tend to migrate later or are required to move long distances likely to be the most affected by climate change.

The survival of smolts entering the ocean depends on a number of factors (Pearcy 1992). Larger smolts tend to have higher survival rates than do smaller fish (Holtby and Scrivener 1989, Quinn and Peterson 1996, Slaney 1988), possibly because they are better able to avoid predation. The size of an individual at smolting is influenced by its size at the beginning of the previous winter. Brown and Hartman (1988) found that stream and groundwater warming caused by logging in a coastal Vancouver Island watershed resulted in increased overwinter growth of presmolt coho salmon, and Holtby and Scrivener (1989) suggested that this growth advantage led to higher smolt-to-adult return rates through improved ocean survival.

Conditions in marine nearshore areas at the time of ocean entry are known to strongly influence ocean survival (Rechisky et al. 2009). In the coastal area influenced by the California current—primarily the southern half of the distributional range of many Pacific salmon species—
potential changes in the timing and intensity of upwelling have important implications for smolts
(Barth et al. 2007). Cold, nutrient-rich waters are pushed into nearshore areas by northerly winds
in the late spring and early summer, producing favorable conditions for plankton production
(Nickelson 1986, Scheuerell and Williams 2005). Under one climate change scenario, upwelling
is projected to intensify but occur later in the summer (Snyder et al. 2003), which decouples the
timing of smolt migration relative to plankton blooms for early-entry salmon smolts.

The abundance of predators in nearshore areas can also influence marine survival of
smolts (Pearcy 1992). Coho salmon from Carnation Creek on the west coast of Vancouver
Island, British Columbia, entered the ocean about two weeks earlier as a result of increased
growth as juveniles (Holtby 1988), but survival declined compared to the timing of prelogging
smolt migration. It was believed that predation by mackerel (*Scomber japonicus*) and hake
(*Merluccius productus*) contributed to the decline, as both species moved into Barkley Sound
during periods of warm sea-surface temperatures. Elevated ocean temperatures could also result
in the expansion of subtropical predators such as the Humboldt squid (*Dosidicus gigas*) into
Pacific Northwest waters, further increasing predation pressure on salmon smolts (Christensen
and Trites 2011, ISAB 2007).

Nearshore conditions in northern portions of the NWFP area will also be influenced by
climate change. In some locations, melting glaciers could increase iron levels in nearshore areas
(Westerlund and Ohman 1991). Iron levels are often considered limiting to primary production in
the North Pacific, and increased iron levels in freshwater plumes could potentially enhance
marine food webs (Rose et al. 2005) and thus improve growth and survival of young salmon. The
projected effects of climate change on the ocean ecology of Pacific salmon will therefore result
from the combined influences of several factors, notably predation, food resource abundance, and both intra- and interspecific competition.
**Table 1**—Evolutionarily significant units (ESUs) of Pacific salmon and trout (*Oncorhynchus* spp.), distinct population segments (DPSs) of bull trout (*Salvelinus confluentus*), and fish and amphibian species listed under the Endangered Species Act that occur in the area covered by the Northwest Forest Plan. Petitioned = under review for Endangered Species Act listing; T = Threatened; S = suspected occurrence

<table>
<thead>
<tr>
<th>Species</th>
<th>ESU/DPS/Species</th>
<th>National forests (NF) and Bureau of Land Management (BLM) districts where ESU, DPS or species occurs</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fish</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coho salmon</td>
<td>Lower Columbia/southwest Washington</td>
<td>Gifford Pinchot NF, Mount Hood NF Siuslaw NF, Umpqua NF, Siskiyou NF, Eugene BLM, Coos Bay BLM, Medford BLM, Roseburg BLM, Salem BLM Rogue River-Siskiyou NF, Six Rivers NF, Shasta-Trinity NF, Klamath NF, Mendocino NF, Arcata BLM, Kings Range National Conservation Area (NCA), Redding BLM, Medford BLM, Coos Bay BLM Ukiah BLM</td>
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<tr>
<td></td>
<td>Oregon coast</td>
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<td></td>
<td>Southern Oregon/northern California</td>
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<tr>
<td></td>
<td>Central California coast</td>
<td></td>
</tr>
<tr>
<td>Chinook salmon</td>
<td>Puget Sound</td>
<td>Mount Baker-Snoqualmie NF, Olympic NF, Gifford Pinchot NF Gifford Pinchot NF, Mount Hood NF, Salem BLM Okanogan NF, Wenatchee NF Mount Hood NF, Willamette NF, Eugene BLM, Salem BLM Six Rivers NF, Mendocino NF, Arcata BLM, Kings Range NCA, Ukiah BLM Mendocino BLM, Mendocino NF, Shasta-Trinity NF Shasta-Trinity NF, Mendocino BLM, Redding BLM</td>
</tr>
<tr>
<td>Species</td>
<td>ESU/DPS/Species</td>
<td>National forests (NF) and Bureau of Land Management (BLM) districts where ESU, DPS or species occurs</td>
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<tr>
<td><strong>Chum salmon</strong></td>
<td>Hood Canal summer</td>
<td>Olympic NF</td>
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<td></td>
<td>Columbia River</td>
<td>Salem BLM</td>
</tr>
<tr>
<td><strong>Steelhead</strong></td>
<td>Lower Columbia</td>
<td>Gifford Pinchot NF, Mount Hood NF, Salem BLM</td>
</tr>
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<td><strong>Steelhead</strong> (cont.)</td>
<td>Mid-Columbia</td>
<td>Gifford Pinchot NF, Mount Hood NF, Wenatchee NF</td>
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<td></td>
<td>Upper Columbia</td>
<td>Wenatchee NF, Okanogan NF</td>
</tr>
<tr>
<td></td>
<td>Upper Willamette</td>
<td>Willamette NF, Salem BLM, Eugene BLM</td>
</tr>
<tr>
<td></td>
<td>Northern California</td>
<td>Six Rivers NF, Mendocino NF, Mendocino BLM, Arcata BLM, Ukiah BLM, Kings Range NCA</td>
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<tr>
<td></td>
<td>Central California coast</td>
<td>Arcata BLM, Kings Range NCA</td>
</tr>
<tr>
<td></td>
<td>Central Valley, California</td>
<td>Shasta-Trinity NF, Mendocino NF, Winemba</td>
</tr>
<tr>
<td><strong>Bull trout</strong></td>
<td>Klamath River</td>
<td>Deschutes NF, Gifford Pinchot NF, Mount Hood NF, Wenatchee NF, Okanogan NF, Willamette NF, Eugene BLM</td>
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<tr>
<td></td>
<td>Columbia River</td>
<td>Winemba</td>
</tr>
<tr>
<td></td>
<td>Coastal-Puget Sound</td>
<td>Gifford Pinchot NF, Mount Baker-Snoqualmie NF, Olympic NF</td>
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<tr>
<td><strong>Lost River sucker</strong></td>
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<td>Winemba</td>
</tr>
<tr>
<td><strong>Shortnose sucker</strong></td>
<td></td>
<td>Winemba</td>
</tr>
<tr>
<td><strong>Pacific eulachon</strong></td>
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<td>Siuslaw NF, Six Rivers NF</td>
</tr>
<tr>
<td><strong>2. Amphibians</strong></td>
<td></td>
<td>Klamath Falls BLM, Medford BLM, Columbia River Gorge NSA (S), Gifford Pinchot NF, Deschutes NF, Fremont-Winema NF, Mount Hood NF, Willamette NF</td>
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<tr>
<td><strong>Oregon spotted frog</strong></td>
<td></td>
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<tr>
<td>Species</td>
<td>ESU/DPS/Species</td>
<td>National forests (NF) and Bureau of Land Management (BLM) districts where ESU, DPS or species occurs</td>
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<td>----------------------------------------</td>
<td>------------------------------------------------------</td>
<td>---------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Cascades frog (petitioned)</td>
<td></td>
<td>Salem BLM, Medford BLM (S), Roseburg BLM, Wenatchee NF, Gifford Pinchot NF, Mount Hood NF, Mount Baker-Snoqualmie NF, Umpqua NF, Willamette NF, Deschutes NF, Olympic NF, Rogue River-Siskiyou NF</td>
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<tr>
<td>Oregon slender salamander (petitioned)</td>
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<td>Mount Hood NF, Willamette NF, Columbia River Scenic Area</td>
</tr>
<tr>
<td>Cascade torrent salamander (petitioned)</td>
<td></td>
<td>Salem BLM, Eugene BLM, Columbia River Gorge NSA, Gifford Pinchot NF, Mount Hood NF, Willamette NF</td>
</tr>
<tr>
<td>Columbia torrent salamander (petitioned)</td>
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<td>Salem BLM (S), Siuslaw NF</td>
</tr>
<tr>
<td>Western pond turtle (petitioned)</td>
<td></td>
<td>Coos Bay BLM, Eugene BLM, Klamath Falls BLM, Medford BLM, Roseburg BLM, Salem BLM (S), Columbia River Gorge, Mount Hood NF, Rogue River-Siskiyou NF, Siuslaw NF, Umpqua NF, Willamette NF, Fremont Winema NF</td>
</tr>
</tbody>
</table>
Table 2—Literature sources used to develop the original curves of ecological functions in riparian reserves in FEMAT (1993)

<table>
<thead>
<tr>
<th>Function</th>
<th>Sources</th>
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</thead>
<tbody>
<tr>
<td>Root strength</td>
<td>Burroughs and Thomas 1977</td>
</tr>
<tr>
<td></td>
<td>Wu et al. 1986</td>
</tr>
<tr>
<td>Wood delivery</td>
<td>Beschta et al. 1987</td>
</tr>
<tr>
<td></td>
<td>Van Sickle and Gregory 1990</td>
</tr>
<tr>
<td></td>
<td>McDade et al. 1990</td>
</tr>
<tr>
<td>Litter fall</td>
<td>Professional judgment</td>
</tr>
<tr>
<td>Shading</td>
<td>Steinblums 1977</td>
</tr>
<tr>
<td></td>
<td>Beschta et al. 1987</td>
</tr>
<tr>
<td>Microclimate</td>
<td>Chen 1991</td>
</tr>
</tbody>
</table>
Table 3—The estimated area of riparian reserves in the Northwest Forest Plan area in Region 6 of the U.S. Forest Service in which management has occurred from 2010—2015.

<table>
<thead>
<tr>
<th>National forest</th>
<th>Area of riparian reserve managed (ha)</th>
<th>Ha = hectare (Source: Region 6)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Commercial</td>
<td>Noncommercial</td>
</tr>
<tr>
<td>Olympic</td>
<td>750</td>
<td>454</td>
</tr>
<tr>
<td>Gifford-Pinchot</td>
<td>1031</td>
<td>301</td>
</tr>
<tr>
<td>Umpqua</td>
<td>883</td>
<td>622</td>
</tr>
<tr>
<td>Willamette</td>
<td>2835</td>
<td>no data</td>
</tr>
<tr>
<td>Mt. Hood</td>
<td>674</td>
<td>0</td>
</tr>
<tr>
<td>Siuslaw</td>
<td>3923</td>
<td>203</td>
</tr>
<tr>
<td>Rogue River-Siskiyou</td>
<td>142</td>
<td>616</td>
</tr>
<tr>
<td>Okanogan-Wenatchee</td>
<td>331</td>
<td>2150</td>
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<tr>
<td>Deschutes</td>
<td>168</td>
<td>461</td>
</tr>
<tr>
<td>Fremont-Winema</td>
<td>0</td>
<td>0</td>
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<tr>
<td>Mt. Baker-Snoqualmie</td>
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<td>0</td>
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<tr>
<td><strong>Total</strong></td>
<td><strong>10 862</strong></td>
<td><strong>4807</strong></td>
</tr>
</tbody>
</table>
Table 4—Predicted cumulative wood storage (m³ · 100 m⁻¹) over the simulated century (includes decay), showing differences between the no-treatment alternative in comparison to various combinations of single- and double-entry thins, a 10-m buffer, and tree-tipping of between 5 percent and 20 percent (From: Benda et al. 2016)

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Change from no treatment</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Percent</td>
<td>Single-entry thin</td>
<td>Double-entry thin</td>
</tr>
<tr>
<td>No treatment (reference)</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Thin</td>
<td>-33</td>
<td>-42</td>
<td></td>
</tr>
<tr>
<td>Thin, buffer</td>
<td>-7</td>
<td>-11</td>
<td></td>
</tr>
<tr>
<td>Thin, tip 5 percent</td>
<td>-15</td>
<td>-15</td>
<td></td>
</tr>
<tr>
<td>Thin, tip 10 percent</td>
<td>-6</td>
<td>+1</td>
<td></td>
</tr>
<tr>
<td>Thin, tip 15 percent</td>
<td>+1</td>
<td>+16</td>
<td></td>
</tr>
<tr>
<td>Thin, tip 20 percent</td>
<td>+6</td>
<td>+24</td>
<td></td>
</tr>
</tbody>
</table>
Table 5—Features of streams from the Oregon Coast Range used in figure 6A (from: Reeves et al. 1995)

<table>
<thead>
<tr>
<th>Feature/Stream</th>
<th>Harvey Cr. (1)</th>
<th>Franklin Cr. (2)</th>
<th>Skate Cr. (3)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Time since disturbance</td>
<td>90–100 years</td>
<td>160–180 years</td>
<td>&gt;330 years</td>
</tr>
<tr>
<td>No. of pieces of wood/100 m</td>
<td>7.9</td>
<td>12.3</td>
<td>23.5</td>
</tr>
<tr>
<td>Mean depth of pools (m)</td>
<td>0.9</td>
<td>0.35</td>
<td>0.1</td>
</tr>
<tr>
<td>Dominant substrate</td>
<td>Gravel</td>
<td>Gravel</td>
<td>Bedrock</td>
</tr>
<tr>
<td>Percent of fish assemblage</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>juvenile coho salmon</td>
<td>98.0</td>
<td>85.0</td>
<td>100.0</td>
</tr>
<tr>
<td>Percent of fish assemblage</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>juvenile steelhead</td>
<td>1.0</td>
<td>12.5</td>
<td>0.0</td>
</tr>
<tr>
<td>Percent of fish assemblage</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>juvenile cutthroat trout</td>
<td>1.0</td>
<td>2.5</td>
<td>0.0</td>
</tr>
</tbody>
</table>
Table 6— Contribution of federal lands and agencies to the total mean annual water supply of states in the Northwest Forests Plan area. (Percent of mean annual water supply) (from: Brown et al. 2008).

<table>
<thead>
<tr>
<th>State</th>
<th>All federal lands</th>
<th>Forest Service</th>
<th>Bureau of Land Management</th>
<th>Other</th>
</tr>
</thead>
<tbody>
<tr>
<td>California</td>
<td>61.1</td>
<td>46.6</td>
<td>5.5</td>
<td>9.0</td>
</tr>
<tr>
<td>Oregon</td>
<td>55.3</td>
<td>44.0</td>
<td>9.4</td>
<td>2.0</td>
</tr>
<tr>
<td>Washington</td>
<td>60.2</td>
<td>41.5</td>
<td>0.0</td>
<td>18.7</td>
</tr>
</tbody>
</table>
Table 7—Summary of effects of road on aquatic ecosystems and associated biota (modified from: Robinson et al. 2010)

<table>
<thead>
<tr>
<th>Ecological effect</th>
<th>Habitat loss/ degradation</th>
<th>Habitat fragmentation</th>
<th>Direct mortality</th>
<th>Stress</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low population density</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Low population reproductive rates</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Area occupied restricted</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Over-harvest</td>
<td></td>
<td></td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Changes in water quality</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Changes in hydrologic functions</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Species</td>
<td>Residence time</td>
<td>Marine</td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------------</td>
<td>----------------</td>
<td>-----------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pink salmon</td>
<td>Oncorhynchus gorbusa</td>
<td>&lt;30 days</td>
<td>2 years</td>
<td></td>
</tr>
<tr>
<td>Chum salmon</td>
<td>O. keta</td>
<td>&lt;30 days</td>
<td>2–5 years</td>
<td></td>
</tr>
<tr>
<td>Sockeye salmon</td>
<td>O. nerka</td>
<td>few months–2 years</td>
<td>2–5 years</td>
<td></td>
</tr>
<tr>
<td>Coho salmon</td>
<td>O. kisutch</td>
<td>1–2 years</td>
<td>2 years</td>
<td></td>
</tr>
<tr>
<td>Chinook salmon</td>
<td>O. tshawytscha</td>
<td>few months–1 year</td>
<td>2–6 years</td>
<td></td>
</tr>
<tr>
<td>Steelhead</td>
<td>O. mykiss</td>
<td>1–3 years</td>
<td>2–4 years</td>
<td></td>
</tr>
<tr>
<td>Coastal cutthroat trout</td>
<td>O. clarkii clarkii</td>
<td>2–4 years</td>
<td>Short forays into nearshore environment</td>
<td></td>
</tr>
</tbody>
</table>
### Table 9—Potential effects of climate change on anadromous salmonids of the Pacific Northwest, by life-history stage

<table>
<thead>
<tr>
<th>Life stage and habitat</th>
<th>Potential effect of climate change</th>
<th>Ecological consequences</th>
<th>Ecological implications</th>
<th>Potential actions</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Adults</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Increased sea-surface temperatures (IPCC 2007; Aziz et al. 2011)</td>
<td>Smaller size at return</td>
<td>Reduced population reproductive capacity (Hankin and McKelvey 1985, Healey and Heard 1984)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Change in life-history expression (O. mykiss)</td>
<td>Loss of steelhead life-history (migratory) and increase in rainbow trout life-history (resident) (Benjamin et al. 2013, Quinn and Myers 2004, Rosenberger et al. 2015, Sloat and Reeves 2014)</td>
<td>Population monitoring with consideration of life-history types</td>
</tr>
<tr>
<td>Freshwater</td>
<td>Sea-level rise (IPCC 2007)</td>
<td>Increased estuary habitat</td>
<td>Increased life-history diversity (Bottom et al. 2005)</td>
<td>Population monitoring with consideration of life-history types</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loss of spawning habitat in areas close to coast</td>
<td>Reduced population productive capacity</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Increased flooding during surge events</td>
<td>Reduced egg survival</td>
<td></td>
</tr>
<tr>
<td>Life stage and habitat</td>
<td>Potential effect of climate change</td>
<td>Ecological consequences</td>
<td>Ecological implications</td>
<td>Potential actions</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------------</td>
<td>------------------------</td>
<td>------------------------</td>
<td>-------------------</td>
</tr>
<tr>
<td>Eggs and Alevins</td>
<td>Increased water temperature (Isaak et al. 2010)</td>
<td>Increased stress</td>
<td>Reduced survival to spawning grounds (Rand et al. 2011) and population reproductive capacity (Miller et al. 2011, Pankhurst et al. 1996) Increased susceptibility to disease and parasites (Johnson et al. 1996, Mathes et al. 2010, Miller et al. 2011, Ray et al. 2012)</td>
<td>Monitoring water temperatures in entire stream network to identify areas of thermal refugia</td>
</tr>
<tr>
<td>Freshwater</td>
<td>Elevated winter water temperatures</td>
<td>Increased rates of development (McCollough 1999, Neuheimer and Taggart 2007) Earlier time of emergence (Holtby 1988)</td>
<td>Smaller size at emergence (Beacham and Murray 1990, Elliot and Hurley 1998) Increased growth rates, earlier timing of smolting, and smaller size at ocean entry, but decreased marine survival (Holtby and Scrivener 1989, Schindler et al. 2005)</td>
<td>Year-round monitoring of water temperatures Increase availability of floodplain and off-channel habitats</td>
</tr>
<tr>
<td>(Freshwater)</td>
<td>Increased winter flows (Hamlet et al. 2005, Hamlet and Lettenmaier 2007, Tague and Grant 2009)</td>
<td>Increased scour of redds (Battin et al. 2007)</td>
<td>Reduced survivala (Battin et al. 2007, Leppi et al. 2014, Shanley and Albert 2014)</td>
<td>Increase connection to floodplain, remove roads and infrastructure that restrict access to floodplain, with wood placement in and near low-gradient spawning areas</td>
</tr>
<tr>
<td>Life stage and habitat</td>
<td>Potential effect of climate change</td>
<td>Ecological consequences</td>
<td>Ecological implications</td>
<td>Potential actions</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------------</td>
<td>-------------------------</td>
<td>------------------------</td>
<td>------------------</td>
</tr>
<tr>
<td>Freshwater</td>
<td>Higher spring flows (Hamlet et al. 2005, Hamlet and Lettenmaier 2007, Tague and Grant 2009)</td>
<td>Increased access to floodplain and off-channel habitats</td>
<td>Increased growth and survival if floodplains and off-channel habitats available (Brown and Hartman 1988, Moore and Gregory 1988, Peterson 1982a); decreases if not</td>
<td>Increased access to floodplain and off-channel habitats</td>
</tr>
<tr>
<td></td>
<td>Earlier onset of low flows (Hamlet et al. 2005, Hamlet and Lettenmaier 2007, Tague and Grant 2009)</td>
<td>Reduced habitat availability (Stewart et al. 2005, Battin et al. 2007, Luce and Holden 2009, Mantua et al. 2010)</td>
<td>Reduced survival (Battin et al. 2007, Mantua et al. 2010)</td>
<td>Identify areas in network that are likely to be refugia during low-flow period and improve habitat conditions, including improving riparian conditions to reduce water temperature</td>
</tr>
<tr>
<td>Life stage and habitat</td>
<td>Potential effect of climate change</td>
<td>Ecological consequences</td>
<td>Ecological implications</td>
<td>Potential actions</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------------</td>
<td>-------------------------</td>
<td>------------------------</td>
<td>-------------------</td>
</tr>
<tr>
<td>(Juveniles)</td>
<td>Increased summer water temperatures (Isaak et al. 2010)</td>
<td>Reduced growth and survival if temperature increases are beyond favorable range (Scarnecchia and Bergersen 1987, Royer and Minshall 1997, Marine and Cech 2004, Crozier and Zabel 2006, Isaak et al. 2010)</td>
<td>Smaller size and reduced survival (ISAB 2007, Quinn and Petersen 1996)</td>
<td>Improve riparian conditions</td>
</tr>
<tr>
<td></td>
<td>Increased nonsummer water temperatures</td>
<td>Increased growth if temperatures move into more favorable range</td>
<td>Altered outcomes of interactions with other species (Reeves et al. 1987; Petersen and Kitchell 2001; ISAP 2007, 2012)</td>
<td>Reduce growth and survival (ISAB 2007, Petersen and Kitchell 2001, Reeves et al. 1987)</td>
</tr>
<tr>
<td></td>
<td>Increased growth rates (Ebersole et al. 2006, Sogard et al. 2010)</td>
<td>Increased growth and survival</td>
<td>Change in structure and composition of fish communities (warm water species increase) (ISAB 2012)</td>
<td>Year-round monitoring of water temperatures</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Provide access to intermittent streams, off-channel habitats, and floodplains</td>
</tr>
<tr>
<td>Life stage and habitat</td>
<td>Potential effect of climate change</td>
<td>Ecological consequences</td>
<td>Ecological implications</td>
<td>Potential actions</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------------</td>
<td>-------------------------</td>
<td>------------------------</td>
<td>-------------------</td>
</tr>
<tr>
<td><strong>Smolts</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Warmer water temperatures (Isaak et al. 2010)</td>
<td>Smaller size at ocean entry</td>
<td>Reduced marine survival (Holtby and Scrivener 1989, Quinn and Peterson 1989, Slaney 1988)</td>
<td>Improve riparian conditions</td>
</tr>
</tbody>
</table>
Figure 1—Proportion of fish-bearing and non-fish-bearing stream length categorized as most ecologically sensitive in the six study watersheds from Reeves et al. (2016a), by ownership.
Figure 2—Distribution of ecologically important stream reaches on federal and nonfederal lands in Myrtle Creek, Oregon. (From: Reeves et al. 2016a)
Figure 3—Example of decision process for assessing the ecological condition of a watershed. (From: Lanigan et al. 2012). AVE = average of scores, MIN = minimum, D50 = median particle size, EPT = Ephemeroptera, Plecoptera, Tricoperta index. [ ] = Use the indicator if context switch is true.
Figure 4—Conceptual roles for disturbance in a changing climate. Disturbance could continue to operate much as it always has (a), with unique disturbance/recovery patterns, or it could become the catalyst that forces ecosystems to shift rapidly and via alternate and uncertain pathways (b) in response to climate. (From: Luce et al. 2012)
Figure 5—(A) Relation of distance from stream channel to cumulative effectiveness of riparian ecological functions (FEMAT 1993: V-27); (B) modified effectiveness curve for wood delivery to streams as a function of distance from the stream channel. The curve was changed based on scientific literature developed since the original curve was portrayed in FEMAT (1993). (From: Spies et al. 2013)
Figure 6—(a) Relation of distance from stream channel to cumulative effectiveness of factors influencing microclimate in riparian ecosystems (FEMAT 1993, p. V-27); (b) modified effectiveness curve for relative humidity as a function of distance from the stream channel. The curve was changed based on scientific literature developed since the original curve was portrayed in FEMAT (1993). (From: Reeves et al. 2016a)
Figure 7—Conceptual representation of vegetative conditions in headwater (a) and mid-order streams (b) in the Northwest Forest Plan area.
Figure 8—Location of key watersheds in the Northwest Forest Plan area.
Figure 9—Examples of the range of conditions that aquatic ecosystems may experience: (a) Central Oregon Coast (Reeves et al. 1995) (‘1’ is stream that is 90-100 years from the last disturbance, ‘2’ 160-180 years, and ‘3’ more than 330 years, see table 5 for specific details), and (b) eastern Oregon (Wondzell et al. 2007).
Figure 10—Conceptual illustration of the changes in channel morphology based on the time since the previous debris flow. (From: May and Gresswell 2004)
Figure 11—Contribution of selected National Forests in the Northwest Forest Plan area to the mean annual flow of adjacent watersheds. (A) Deschutes National Forest, (B) Willamette National Forest, (C) Siuslaw National Forest, and (D) Six Rivers National Forest. Source: https://wwwapps.fs.usda.gov/rmrs/projects/national-forest-contributions-streamflow

A.
Figure 12—A decision matrix for identifying potential options for managing roads (from: Robinson et al. 2010).

<table>
<thead>
<tr>
<th>Preventive management:</th>
<th>Mitigate presence effects:</th>
</tr>
</thead>
<tbody>
<tr>
<td>• perform regular maintenance</td>
<td>• maintain fish passage</td>
</tr>
<tr>
<td>• monitor for signs of presence effects</td>
<td>• prevent erosion</td>
</tr>
<tr>
<td></td>
<td>• soften edge effects</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Decommission road:</th>
<th>Decommission road:</th>
</tr>
</thead>
<tbody>
<tr>
<td>• abandon</td>
<td>• rip or obliterate</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Road importance:</th>
<th>Ecological impact</th>
</tr>
</thead>
<tbody>
<tr>
<td>low</td>
<td>low</td>
</tr>
<tr>
<td>high</td>
<td>high</td>
</tr>
</tbody>
</table>
Figure 13—Current (A) and projected (2040) (B) summer water temperatures (°C) in the study basins in the Treaty of Olympia area (From: Reeves et al. 2016b).
Figure 14—Example of identification of potential locations where riparian restoration could provide potential thermal refugia for native fish. Analysis from NetMap (Benda et al. 2007).

Areas that could benefit from increased shade
(reduced thermal energy input to channels)

Thermal energy in larger rivers cannot be significantly influenced by increasing shade, except very locally. Smaller channels in areas with no shade would benefit most, and many of these overlap with high-quality coho habitat potential (high IP scores).
Figure 15—Predicted decline in flow levels in absolute (A) and relative (B) terms over the summer. From: Safeeq et al. (2014).
Figure 16—Percentage of reduction in average summer (A) and winter (B) flow levels from current to 2040 in study basins in the Olympic Peninsula area (from Reeves et al. 2016b).
Figure 17—Life cycle and habitat preference strategies of freshwater fishes that are considered in this paper to be favored (“winning”) and disfavored (“losing”) in future climates of the NWFP area. Beneath the lists of winning and losing strategies is a grouping of fishes along a gradient of low to high risk from climate effects. These groupings, somewhat subjective, are based on current knowledge of each species’ life histories, spawning and rearing locations in watersheds, and residence time in fresh water.

<table>
<thead>
<tr>
<th>Winning Strategies</th>
<th>Losing Strategies</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat generalist</td>
<td>Habitat specialist</td>
</tr>
<tr>
<td>Shorter time in fresh water</td>
<td>Long freshwater rearing</td>
</tr>
<tr>
<td>High stray rate</td>
<td>Low stray rate</td>
</tr>
<tr>
<td>Spring spawning</td>
<td>Fall spawning</td>
</tr>
<tr>
<td>Brief exposure OR high tolerance to high temperatures</td>
<td>Extended exposure to high temperatures</td>
</tr>
<tr>
<td>Chum Salmon</td>
<td>Sockeye Salmon</td>
</tr>
<tr>
<td>Pink Salmon</td>
<td>Coho Salmon</td>
</tr>
<tr>
<td>fall Chinook Salmon</td>
<td>spring Chinook Salmon</td>
</tr>
<tr>
<td>winter steelhead</td>
<td>summer steelhead</td>
</tr>
<tr>
<td>native minnows</td>
<td>Bull Trout</td>
</tr>
<tr>
<td>native suckers</td>
<td>Mountain Whitefish</td>
</tr>
<tr>
<td>many non-natives</td>
<td></td>
</tr>
</tbody>
</table>

Lower risk → Higher risk
Figure 18—Native salmonid fishes in the NWFP area that are likely to benefit in some way from the environmental protections from the harmful effects of climate change, grouped by different geographical zones (see text).

<table>
<thead>
<tr>
<th>Western</th>
<th>Eastern</th>
<th>Northern</th>
<th>Southern</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coho Salmon</td>
<td>Sockeye Salmon</td>
<td>Coho Salmon</td>
<td>Coho Salmon</td>
</tr>
<tr>
<td>Sockeye Salmon</td>
<td>spring Chinook Salmon</td>
<td>Sockeye Salmon</td>
<td>Spring Chinook Salmon</td>
</tr>
<tr>
<td>spring Chinook Salmon</td>
<td>summer steelhead</td>
<td>fall Chinook Salmon</td>
<td>fall Chinook Salmon</td>
</tr>
<tr>
<td>fall Chinook Salmon</td>
<td>winter steelhead</td>
<td>summer steelhead</td>
<td>summer steelhead</td>
</tr>
<tr>
<td>summer steelhead</td>
<td>Coastal Cutthroat Trout</td>
<td>winter steelhead</td>
<td>Coastal Cutthroat Trout</td>
</tr>
<tr>
<td>winter steelhead</td>
<td>Westslope Cutthroat Trout</td>
<td>Chum Salmon</td>
<td>Westslope Cutthroat Trout</td>
</tr>
<tr>
<td>Coastal Cutthroat Trout</td>
<td>Bull Trout</td>
<td>Pink Salmon</td>
<td>Chum Salmon</td>
</tr>
<tr>
<td>Chum Salmon</td>
<td>Mountain Whitefish*</td>
<td>Bull Trout</td>
<td>Pink Salmon</td>
</tr>
<tr>
<td>Pink Salmon</td>
<td></td>
<td></td>
<td>Bull Trout</td>
</tr>
<tr>
<td>Bull Trout</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Mountain Whitefish, *Prosopium williamsoni*
Figure A1—Winning weights of Juneau’s Golden North Salmon Derby from 1947 through 2015 and the preceding 42-month average Pacific Decadal Oscillation (PDO) index. Positive deviations in the PDO index occur in warmer than average PDO cycles, and negative deviations indicate cooler temperatures. Also see Fagen (1988) and Reid et al. (2016).
Figure A2—Changes in time of emergence of Chinook and coho salmon as a result of a 1°C increment increase in water temperature during egg development. (From: McCollugh 1999).