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NORTHWEST FOREST PLAN
THE FIRST 25 YEARS (1994–2018)

Northwest Forest Plan—The First 25 Years (1994–2018): Watershed Condition Status and Trends

Jason Dunham, Christine Hirsch, Sean Gordon, Rebecca Flitcroft, Nathan Chelgren, Marcía Snyder, David Hockman-Wert, Gordon Reeves, Heidi Andersen, Scott Anderson, William Battaglin, Tom Black, Jason Brown, Shannon Claeson, Lauren Hay, Emily Heaston, Charles Luce, Nathan Nelson, Colin Penn, and Mark Raggon

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Cover photo: Garden Gulch-South Fork Salmon River in the Klamath National Forest. Courtesy of Brian Knees.

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Abstract

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This report describes status and trends in watershed condition across the Northwest Forest Plan (NWFP) area over the first 25 years since its inception in 1994. The program charged with this task is the Aquatic and Riparian Effectiveness Monitoring Program (AREMP), which has assembled information from field data collection, spatial datasets, and a host of landscape models to evaluate the status and trends in aquatic resources in streams and watersheds. Field data included hydrologic measurements (stream wetted widths and temperatures), geomorphic responses (instream wood and sediment), and biological responses (macroinvertebrates and aquatic organism passage). Novel statistical models were used to estimate trends in these measured responses. A suite of complementary modeled results was also employed to describe hydrometeorological drivers (e.g., drought indices and stream discharge), forest cover (upslope and riparian vegetation), and geomorphic conditions (e.g., road-related estimates of chronic and shallow landslide sediment delivery risk). Collectively, information on these responses allowed us to rigorously evaluate instream responses and hypothesize watershed drivers of those responses across the NWFP area and over time. The majority of responses we observed indicated widespread and incremental improvements from active management of forests, forest roads, and road-stream crossings as envisioned by the aquatic conservation strategy of the NWFP. Additionally, many of the responses we observed were consistent with those expected under the influences of changing climates in the Pacific Northwest. Ultimately, the long-term, broad-scale information provided by AREMP is a critical foundation for evaluating the effectiveness of federal land management and the effects of changing climates on water resources that sustain the Pacific Northwest's human and natural landscapes.

Keywords: Effectiveness monitoring, status and trend monitoring, aquatic ecosystems, riparian ecosystems, watersheds, Northwest Forest Plan, aquatic conservation strategy, Pacific Northwest.

Executive Summary

The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) was established under the Northwest Forest Plan (NWFP) to track the status and trends in stream and watershed conditions on federal lands in Washington, Oregon, and California within the 24.5-million-ac (~99,000-km²) NWFP area. Management is conducted to meet the objectives of the NWFP aquatic conservation strategy and the 2016 USDI Bureau of Land Management Western Oregon Resource Management Plans within this footprint (referred to as the AREMP area). Assessments of upslope and riparian conditions were based on mapped data (e.g., road density, vegetation) for the 1,972 watersheds with >5 percent federal ownership at two time periods: pre-NWFP and latest available data. Stream conditions were assessed with instream data (e.g., substrate, wetted width, macroinvertebrates) collected from multiple sites within 219 randomly selected watersheds with ≥25 percent federal ownership in repeating 8-year rotations beginning in 2002.

We incorporated several new methods and metrics to provide multiple lines of evidence to evaluate watershed status and trends and to understand the influences of environmental variability. These included significant new climate-related assessments of streamflow and drought to provide context for the assessment of management-related attributes. We developed improved assessments of roads to evaluate their potential for chronic and episodic shallow landslide-related sediment delivery, and we used site-specific culvert data for fish passage assessment. We used a hierarchical modeling approach to provide better estimates of instream condition trends within watersheds. For the first time, we analyzed trends at individual sites because at least two rounds of repeat sampling have been completed by AREMP field monitoring crews. To evaluate collective patterns across multiple responses, we performed an empirical multivariate analysis of watershed condition. As in past reports, patterns of stream and watershed responses are summarized by aquatic provinces and land use designations specified in the NWFP, including riparian management areas (RMAs),[1] land use allocations (LUAs), and key watershed classifications.

The combined influences of multiple processes have driven instream and watershed conditions within the AREMP area over the past 25 years. Change operating at scales ranging from local (e.g., restoration efforts such as improvements in road networks and road-stream crossings), to regional (e.g., land use regulations such as establishment of riparian management zones and changes to forest harvest practices), and to global (e.g., climate change) intersect to influence stream and upslope conditions. Some responses (e.g., temperature) can change relatively quickly (e.g., within hours), whereas other responses such as recruitment of large wood can take decades to centuries to produce observable changes. Because the responses we addressed are influenced by all processes listed here (e.g., local restoration, land use, and climate change), as well as associated disturbance processes (e.g., wildfires, floods, landslides, and other episodic events), it is difficult to unequivocally attribute observed changes to any single change agent, and in some cases

[1] Note that RMAs were originally called “riparian reserves” in the NWFP. We have chosen to refer to them as “riparian management areas” to emphasize that they allow for management activities designed to benefit aquatic and riparian-dependent resources (USDA FS 2018).

lagged effects or responses to contemporary change are likely. With these qualifications in mind, we summarize our findings in terms of climate-related responses, forest cover and stream temperature, forest conditions and instream large wood, roads, landslide risk, instream fine sediment, biotic responses, and a multivariate summary of overall watershed conditions as indicated by measured responses of stream channels combined with riparian and upslope conditions determined from spatial datasets.

Climate and Surface Water Availability

Streamflows are critical to aquatic ecological processes and have become more of a concern given the influences of a changing climate. Drought indicators, modeled annual discharge, and instream measurements of wetted width largely support climate change as a major driver of hydrological conditions across the region since NWFP implementation. Streams measured by our surveys exhibited widespread declines in wetted width across aquatic provinces sampled. Trends in drought indices and modeled annual discharge were spatially variable and often exhibited no significant change over time within individual watersheds. However, when significant trends were observed, they were most often indicative of greater prevalence of drought and declining streamflow, particularly in the southern end of the AREMP area in Oregon and California. Collectively these changes are consistent with those expected under influences of warming climates as well as droughts. Although the three lines of evidence evaluated here support variable declines in water availability across the region, it is important to note that time series analyzed here are variable in length, as well as timing (start and end), which can influence observable trends or patterns of change. Responses of instream and watershed processes to climatic changes are often superimposed on the potential influences of changes in land use and local restoration actions on the ground.

Canopy Cover and Stream Temperature

Stream temperature influences many aquatic ecological processes, most notably survival and growth of coldwater biota such as salmon and trout, which are a major management focus in the NWFP. Riparian canopy cover is the driver of stream temperature most influenced by forest management and disturbance. Over the whole AREMP area, there was little change in mean canopy cover in RMAs between 1993 (70 percent) and 2017 (72 percent). However, changes in individual subwatersheds ranged from -40 to +39 percent, where large losses in canopy cover were mostly associated with wildfire and large gains attributed to recovery from disturbance or to expansions of forest conditions into areas that were not recently forested. Overall, 509 subwatersheds experienced a >5-percent gain in canopy cover, whereas 203 experienced a >5-percent loss. Stream temperatures monitored by AREMP were relatively cool overall and within ranges reported to be suitable for supporting coldwater fishes such as salmon and trout. As expected, warmer stream temperatures, including those exceeding thresholds for coldwater fishes, were more likely to occur in lower elevation or more southerly located aquatic provinces (Franciscan, Klamath-Siskiyou, Washington-Oregon Coast Range). Annual variability in stream temperatures appeared to roughly track variability in air temperatures, although more detailed analysis is needed to confidently attribute specific processes to observed patterns of stream temperature.

Forest Conditions and Instream Large Wood

Forests contribute large wood to streams, which in turn influences numerous processes, ranging from biogeochemical cycles to creating habitats for fish, amphibians, and other species. With satellite-derived vegetation data covering the AREMP area, we used two proxy measures of standing vegetation that indicate potential availability of large wood to streams: the percentage of the RMA meeting an old-growth structure index at 80 years (OGSI 80) and the number of large trees per hectare (≥50 cm diameter at breast height [d.b.h.]) near fish-bearing streams. Across the AREMP area, an additional 4 percent of the forests within RMAs met OGSI 80 criteria in 2017 compared to 1993 (a net gain from 57 to 61 percent). Similarly, we found an overall 4-percent increase in mean large trees per hectare. At the province level, gains for both indicators were noticeably higher in the Washington-Oregon Coast Range province than for others (+15 percent for OGSI 80, +16 percent for large trees). Trends in the density of instream wood indicate spatially variable patterns for smaller size classes of large wood and consistent loss of the largest size wood in streams across the AREMP area. This may be expected as historical (pre-NWFP) forest harvest practices (loss of available trees and active removals from stream channels) across much of the AREMP area likely reduced availability of large wood that can be recruited and retained in streams. Because the dynamics of large wood occur on very long (hundreds of years) timescales, we did not expect to see dramatic changes across the AREMP area, with the potential exception of localized, episodic disturbance (e.g., wildfire, landslides, or debris flows) or targeted wood placement to restore instream wood. As available trees in the forests grow larger over time, an increasing fraction of them are expected to be available for recruitment to instream wood.

Forest Roads and Instream Fine Sediment

The presence of some fine sediment along the channel bed is normal and benefits some species, such as native lamprey, but excess fine sediment deposition can be detrimental, such as when it reduces salmon egg-to-fry survival by clogging spawning gravels. Given that road networks are often identified as a major source of management-related sediment inputs to streams, we modeled their likely contributions to chronic (runoff) and shallow landslide (episodic) sediment delivery. Road decommissioning across the AREMP area has reduced the estimated stream-connected road length on federal lands by 1608 km (a 6.6-percent reduction). Application of a GIS-based tool for relative estimation of potential chronic sediment delivery by roads to streams indicated a 4-percent decrease in mean subwatershed road sediment delivery across the AREMP area. In addition to chronic sediment delivery, we considered potential for episodic delivery of sediment by roads related to the relative likelihood of mass failures via shallow landslides. We observed an 11-percent decrease in the relative probability of sediment delivery from mass failures across the AREMP area, which is likely attributable to focused road decommissioning on high-risk locations. The instream data show declines in fine sediment across habitats in 90 percent of sampled subwatersheds and in 60 percent of sampled subwatersheds for fines in pool-tail crests. As with other responses, we observed considerable spatial variability, likely due in part to the unequal distribution of higher risk roads at the onset of the NWFP and corresponding opportunities to address them, as well as recent events such as wildfire or other disturbances. Overall, these data indicate that road and vegetation management appear to be having the desired effects of decreasing instream fine sediment on streambeds. These responses were also likely attributable to other key drivers such as changing climate, associated changes in streamflow regimes, and the local geomorphic setting.

Biotic Responses

Although the conservation of fish species is a major emphasis of the NWFP, monitoring fish populations at this scale is infeasible, and not all streams monitored by AREMP support fish. Instead, AREMP monitors aquatic macroinvertebrates (aquatic insects, mollusks, and other taxa), which are a common measure of stream water quality and ecological integrity. Both macroinvertebrate indicators as well as ecological connectivity in relation to culverts at road-stream crossings indicated improvements. Based on our assembly of multiple and overlapping sources of information on stream culverts on federal land, we were able to develop the most comprehensive assessment to date across the AREMP area. Evaluation of culverts at road-stream crossings indicated a number of crossings that we assumed had been replaced to allow for fish passage since NWFP implementation. Of the 3,193 stream culverts that have been surveyed for their potential for fish passage across the AREMP area, 773 (24 percent) are passable and 2,420 are barriers (76 percent). Barrier culverts are currently impeding access to 5500 km (10 percent) of potential fish habitat. In addition, we found 539 culverts in the databases without passage status, and 1,843 road-stream crossings not in existing databases. Overall, data on macroinvertebrate assemblages based on a reference condition approach indicate they are trending to be more similar to locations where human disturbances across the AREMP area are relatively minimal. Since the last reporting cycle, amphibian and fish species presence sampling was reintroduced with environmental DNA methods led by the USDA Forest Service Pacific Northwest Research Station, although the current incorporation of this information is limited to a case study demonstrating future utility.

Watershed Condition

Watershed condition is a collective property, reflecting the numerous interacting processes that drive outcomes, such as those tracked by AREMP. To assess how collective properties of watersheds vary across the NWFP area and have changed over time, we applied a multivariate statistical approach to integrate instream and upslope responses and summarize temporal patterns in overall watershed conditions. Temporal changes in the collective condition of watersheds were not identified across the entire NWFP area, but they were identified in the Washington-Oregon Coast Range where change was associated with gains in vegetation-related attributes and reduction in roads over time. Additionally, a slight difference over time was found in non-key watersheds where improvements in vegetation characteristics were evident, consistent with their harvested condition prior to NWFP implementation. The watershed condition analysis also showed that road density was an important metric in distinguishing among LUAs. The analysis showed that road reduction was linked to a decline in impacts for matrix and late-successional reserve (LSR) lands. Although NWFP area-wide trends were not detected, the observed responses of individual provinces, key and non-key watersheds, and LUA groupings indicate how specific factors are potentially changing. Furthermore, conditions across LUAs are generally becoming more similar to the least modified LUA (congressional reserves) over time, indicating potential recovery of lands historically subjected to more intensive forestry practices.

Land Use Allocations and Key Watersheds

Land management prescriptions across the AREMP area are not homogeneous; they are partitioned into categories, including LUAs (comprised of congressionally reserved lands, riparian management areas, LSRs, and matrix lands), as well as key watersheds, which are identified across LUAs and were selected for their value as high-quality habitat for salmonids, high water quality, and restoration potential. Congressionally reserved lands are reserved by the U.S. Congress and include wilderness areas, wild and scenic rivers, and national parks and monuments. LSRs are lands reserved for the protection, restoration, and maintenance of late-successional, old-growth forest ecosystems and habitat for associated species. Matrix lands are where most timber harvest and silvicultural activities are expected to occur. Given their likely differences, we assessed stream and watershed condition within those categories.

Land use allocations—

Differences were commonly observed in initial conditions and rates of change in instream, riparian, and upslope responses among LUAs. Matrix lands are characterized by having smaller substrate sizes, lower wetted widths (an indicator of diminished flows), and the least amount of large wood in the streams. They showed the most gain in all vegetation attributes assessed, as well as in macroinvertebrate composition. LSR streams are characterized by a higher percentage of fines (similar to matrix lands) and higher amounts of the largest sizes of instream wood. LSR watersheds had moderate gains in all vegetation and higher gains in road-related attributes. LSR attributes showed the slowest rate of loss of proportional wetted widths. Congressional reserves are characterized by higher wetted widths, low instream fine sediment, abundant wood in smaller size categories, and relatively lower aquatic macroinvertebrate scores. The largest changes in road and vegetation metrics occurred in the LSR and matrix lands; congressional reserves showed little change over the NWFP period.

Key versus non-key watersheds—

Key watersheds were often in better condition at the initiation of the NWFP than non-key watersheds. Key watersheds had smaller proportions of fine sediments, higher instream wood densities, higher macroinvertebrate scores, and greater values in all vegetation attributes examined than non-key watersheds in both the starting and ending points summarized in this report. Key watersheds showed slower declines in wetted widths, higher loss of the largest wood, and slower improvement in macroinvertebrate composition. Non-key watersheds exhibited higher gains in all vegetation attributes. Key watersheds showed the greatest gains in road-related attributes with a 12-percent reduction in total road length compared to a 5-percent reduction in non-key watersheds. This led to greater decreases in modeled chronic and shallow landslide sediment delivery in key watersheds.

Contents

Chapter 1: Introduction

Background

The Northwest Forest Plan (NWFP) amends 19 USDA Forest Service forest plans and 7 USDI Bureau of Land Management (BLM) resource management plans across Washington, Oregon, and California (fig. 1.1). The NWFP area (also referred to as the AREMP area), covers approximately 99,000 km² (24.5 million ac), encompassing the range of the northern spotted owl (*Strix occidentalis caurina*) and eight aquatic provinces used to assess watershed condition. The NWFP provides a host of new standards and guidelines for how federal lands are to be managed within its boundaries (USDA FS and USDI BLM 1994b).

The critical role of federal forests to support diverse aquatic ecosystems was a core consideration in the design of the NWFP. Significant ecosystem services are provided by forests, including native fishes and their habitat (Penaluna et al. 2017), water for ecosystems and downstream users (Kampf et al. 2021, Luce et al. 2017), and other ecosystem services (Martin-Ortega et al. 2015). In recognition of these values, the NWFP includes an aquatic conservation strategy (ACS). Briefly, the ACS recognizes that aquatic restoration depends on conservation and management of entire watersheds and is intended over the short term (10–20 years) to stop declines in watershed conditions, and over longer time horizons (>100 years) to produce conditions that support a host of desired functions (Reeves et al. 2006). Important components of the standards and guidelines in the NWFP include land use allocations (LUAs) intended to support the ACS (box 1.1), late-successional reserve and watershed assessments, a survey-and-manage program, an interagency executive organization, social and economic mitigation initiatives, and monitoring and adaptive management (USDA FS and USDI BLM 1994b). Given these management directives, the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) was tasked with evaluating whether the NWFP ACS is achieving the goal of maintaining and restoring watershed conditions (Mulder et al. 1999, Reeves et al. 2004).

◀ Stream sampled by the Aquatic and Riparian Effectiveness Monitoring Program. Photo courtesy of Alanna Wong.

Several notable management directives were issued in the years following NWFP implementation (fig. 1.2): Many distinct population segments of five species of salmonid fishes (salmon and trout) were listed under the Endangered Species Act. These listings and the adoption of the pursuant recovery plans have strengthened protection of aquatic resources. The Roadless Rule (36 CFR Part 294) and Travel Management Rule (36 CFR 212) have restricted additional road building and encouraged removal of legacy roads. In 2016, the BLM revised its Western Oregon Resource Management Plans (USDI BLM 2016a, 2016b), working closely with regulators and stakeholders to revise their definitions of riparian management areas and key watersheds. This report uses the original NWFP definitions, however, as they were in place for most of the time period reported here and cover a majority of the NWFP area. The revised plans continue to rely on the ongoing monitoring conducted under the NWFP. All national forests and California BLM districts are still operating under NWFP direction. In addition to these changes, the Forest Service has developed and implemented a more formalized framework for watershed restoration (USDA FS 2011a, 2011b). Findings in this report should prove complementary to this framework within the NWFP area.

Unified management across multiple jurisdictions of federal ownership within the area of the NWFP offers a unique opportunity to evaluate landscape-scale effects of management intended to enhance watersheds and aquatic ecosystems. Federal administrative units within this area (i.e., national forests, BLM resource areas) have taken a diverse set of management actions designed to enhance aquatic ecosystems, including upgrading roads or replacing road-stream crossings (e.g., culverts), road decommissioning, instream and valley bottom enhancement, and riparian treatments intended to grow larger trees that will ultimately support aquatic habitat over the long term. Some of these management actions may have an immediate effect on local aquatic habitats, while others

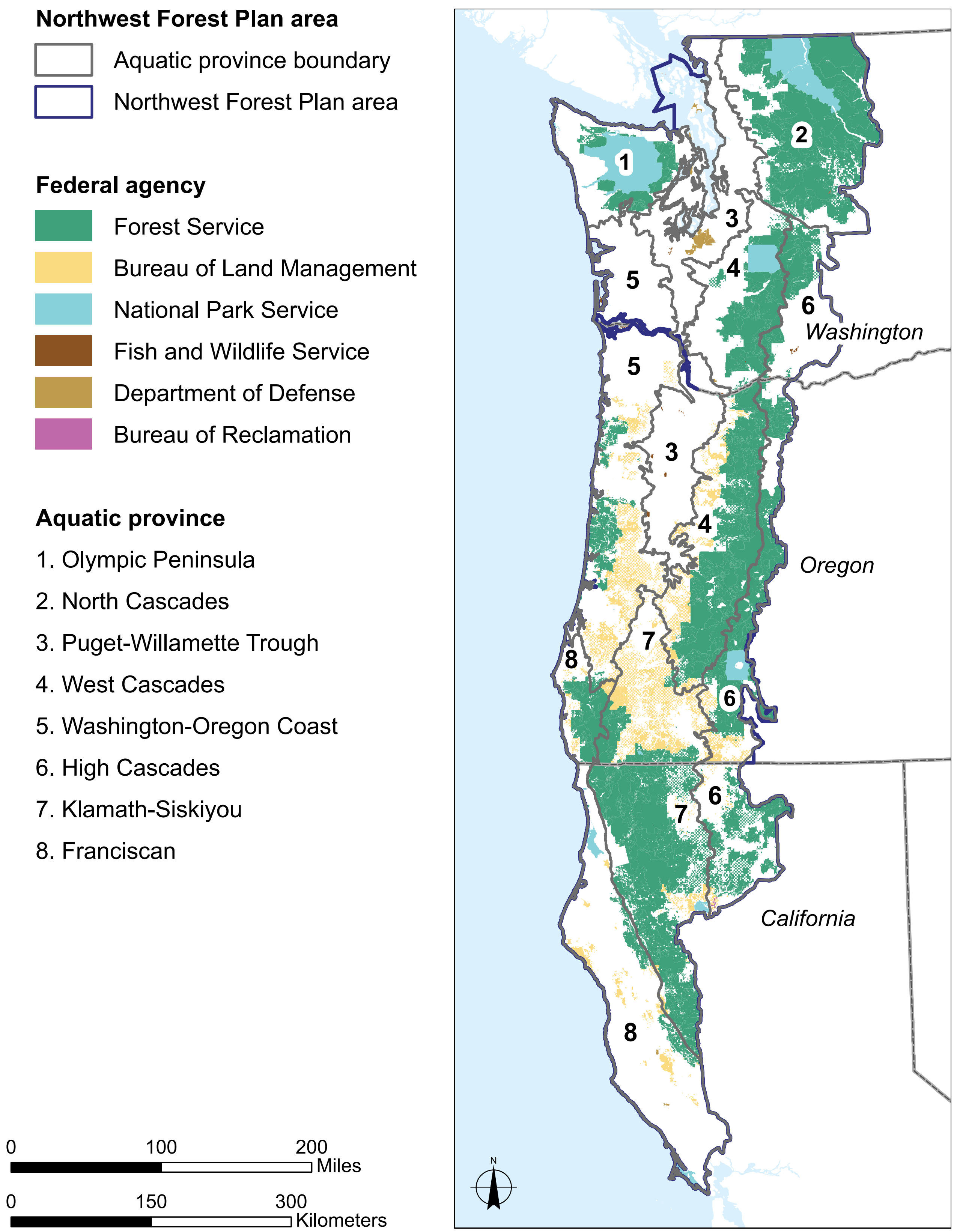

Figure 1.1—Map of the Northwest Forest Plan area showing federal lands by agency and eight aquatic provinces.

Box 1.1

Objectives of the Aquatic Conservation Strategy of the Northwest Forest Plan

"Federal lands within the range of the northern spotted owl will be managed to:

1. Maintain and restore the distribution, diversity, and complexity of watershed and landscape-scale features to ensure protection of the aquatic systems to which species, populations and communities are uniquely adapted.
2. Maintain and restore spatial and temporal connectivity within and between watersheds. Lateral, longitudinal, and drainage network connections include floodplains, wetlands, upslope areas, headwater tributaries, and intact refugia. These network connections must provide chemically and physically unobstructed routes to areas critical for fulfilling life history requirements of aquatic and riparian-dependent species.
3. Maintain and restore the physical integrity of the aquatic system, including shorelines, banks, and bottom configurations.
4. Maintain and restore water quality necessary to support healthy riparian, aquatic, and wetland ecosystems. Water quality must remain within the range that maintains the biological, physical, and chemical integrity of the system and benefits survival, growth, reproduction, and migration of individuals composing aquatic and riparian communities.
5. Maintain and restore the sediment regime under which aquatic ecosystems evolved. Elements of the sediment regime include the timing, volume, rate, and character of sediment input, storage, and transport.
6. Maintain and restore instream flows sufficient to create and sustain riparian, aquatic, and wetland habitats and to retain patterns of sediment, nutrient, and wood routing. The timing, magnitude, duration, and spatial distribution of peak, high, and low flows must be protected.
7. Maintain and restore the timing, variability, and duration of floodplain inundation and water table elevation in meadows and wetlands.
8. Maintain and restore the species composition and structural diversity of plant communities in riparian areas and wetlands to provide adequate summer and winter thermal regulation, nutrient filtering, appropriate rates of surface erosion, bank erosion, and channel migration and to supply amounts and distributions of coarse woody debris sufficient to sustain physical complexity and stability.
9. Maintain and restore habitat to support well-distributed populations of native plant, invertebrate, and vertebrate riparian-dependent species."

Source: USDA FS and USDI BLM (1994a).

are likely to take decades to result in substantive changes in aquatic conditions that are detectable through monitoring.

Evaluations of the effectiveness of the NWFP may be strongly conditioned on historical legacies and climate-related changes since the plan was implemented. Prior to implementation of the NWFP, a host of legacies from human modifications of the landscape that range from historical forest harvest practices to fire suppression (Cissel et al. 1994, Thomas et al. 2006) are still evident today in forests, watersheds, and the stream networks that drain them (Wohl 2019). For example, legacy impacts of historical log drives and splash damming are still readily observable in many streams and could take decades or longer to reverse, even though these practices are no longer implemented (Miller 2010). On a more contemporary scale, since NWFP implementation, changes in climate, wildfires, invasive species, and the social landscape have exerted influences that extend far beyond the control of federal land managers (Dalton et al. 2013, Spies et al. 2019). Accordingly, consideration of the potential influences of changing climatic conditions, for instance, and their influences on aquatic conditions is increasingly warranted.

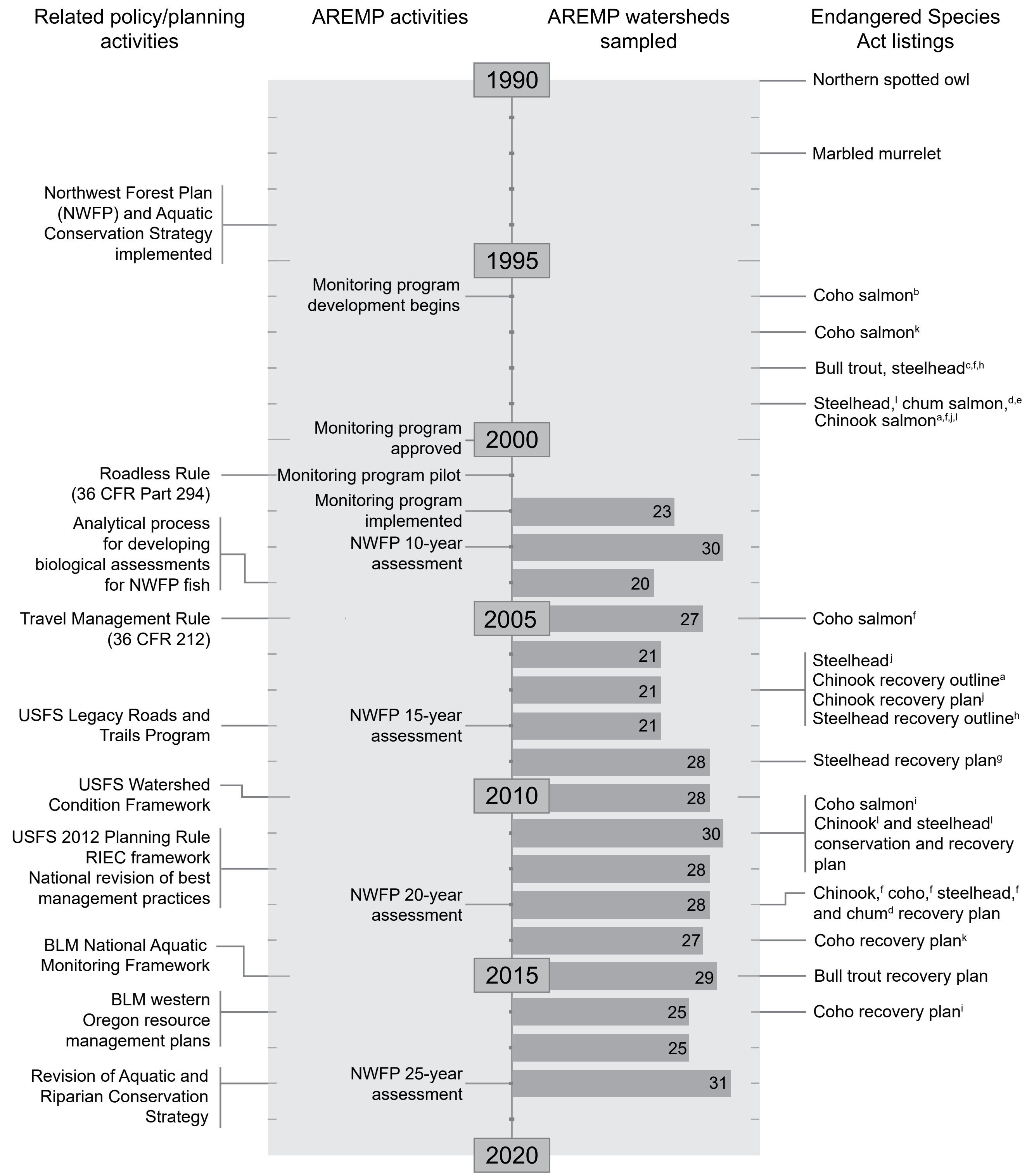

Figure 1.2—Timeline of Northwest Forest Plan aquatic monitoring program development and implementation, related policy and planning activities, and Endangered Species Act listings. AREMP = Aquatic and Riparian Effectiveness Monitoring Program, BLM = USDI Bureau of Land Management, USFS = USDA Forest Service. RIEC = Regional Interagency Executive Committee.

[a] California Coastal Evolutionarily Significant Unit (ESU).
[b] Central California ESU.
[c] Central California Coast Distinct Population Segment (DPS).
[d] Columbia River ESU,
[e] Hood Canal summer-run ESU.
[f] Lower Columbia River DPS (Steelhead)/ESU.
[g] Mid Columbia River DPS.
[h] Northern California DPS.
[i] Oregon Coast ESU.
[j] Puget Sound DPS (Steelhead)/ESU.
[k] Southern Oregon/Northern California Coasts ESU.
[l] Upper Willamette River DPS (Steelhead)/ESU.

Aquatic conditions within streams in the NWFP area have been tracked by AREMP with field-based monitoring initiated in 2002 (fig. 1.2) (Gallo et al. 2005) and followed by publication of a supporting conceptual framework (Reeves et al. 2004).[1] Subwatersheds selected for field-based monitoring by AREMP had to include at least 25 percent of federal land ownership along the length of its major streams, a percentage considered to represent a significant contribution of federal lands to watershed condition (Reeves et al. 2004). Subwatersheds and sample site reaches used to characterize the subwatershed were selected following sample survey designs developed for the Environmental Protection Agency's (EPA's) National Rivers and Streams Assessment (fig. 1.3) (Gallo et al. 2005, Paulsen et al. 2008, Stevens and Olson 2004, Stevens et al. 2007). A variety of stream types are encompassed by the field-based monitoring. These range from low-order, high-gradient streams to higher order, low- and mid-gradient streams and include both fish-bearing and non-fish-bearing streams. Subwatershed areas sampled by AREMP ranged from 33.6 to 193.2 km² (see app. 1).

AREMP reporting occurred in 5-year intervals and evolved from strong reliance on standardized province-level criteria based on the best available science[2] when data were scarce (Gallo et al. 2005), to more empirically based assessments scaled to individual stream geomorphic attributes as monitoring data became available (Miller et al. 2017). At the same time, analytical tools and predictive models have advanced considerably and provide new opportunities for quantifying key drivers and watershed conditions (Reeves et al. 2018). Accordingly, this report provides an updated evaluation of instream and riparian conditions in AREMP watersheds based on newly available data (up to 2018), analytical tools, and predictive models that capture key drivers of watershed conditions. This report focuses on riparian management areas (RMAs) (see footnote 1), key watersheds, and LUAs (fig. 1.3). RMAs were designed to enhance habitat for riparian-dependent species through water quality protection, connectivity for dispersal of terrestrial species, and many other services provided by riparian zones (Naiman et al. 2010). This report incorporates a more refined RMA delineation process as compared to previous reports. We moved from a medium- (1:100k) to a high- (1:24k) resolution stream layer, and from a uniform width (300 ft) to variable buffer width (100 ft to two site-potential tree heights) by water feature type, as specified in the ACS. However, it was not feasible to model some ACS criteria at this scale, including 100-year floodplains, unstable areas, and inner gorges. This may result in some changes in riparian summary statistics compared with prior reports. Likewise, we have used exact key watershed delineations rather than approximating to subwatersheds. Key watersheds were defined under the ACS and are areas intended to "serve as refuge for aquatic organisms, particularly in the short term for at-risk fish populations, to have the greatest potential for restoration, or to provide sources of high-quality water" (Haynes et al. 2006). LUAs were linked with different management prescriptions from the most protected, congressional reserves, to the most flexible, matrix. This report simplifies the LUAs recognized by the NWFP into categories that account for the major differences in land management: congressional reserves, late-successional reserves, and matrix lands (for an explanation of all LUA categories, see app. 1: table A1.1). Congressionally reserved areas are designated by the U.S. Congress and encompass wilderness areas, wild and scenic rivers, and national parks and monuments. Late-successional reserves are managed for the protection and restoration of late-successional, old-growth forest ecosystems and habitat for associated species. Matrix lands are outside of other reserved categories where most timber harvest and silvicultural activities are expected to occur.

Objectives

This report evaluates status and trends in the condition of individual instream and upslope metrics and compares multiple variables assessing watershed conditions during the 25 years of the NWFP spanning 1994–2018 (table 1.1). The overall objective is to quantitatively track key elements of the NWFP ACS. Monitoring conducted by AREMP was originally intended to address the nine objectives identified by the original ACS (box 1.1), with effectiveness

[1] Note that in the Puget-Willamette Trough aquatic province, no subwatersheds were sampled as it represents only <1 percent of the NWFP area.

[2] Land management decisions are complex. However, the USDA Forest Service uses the best available science and other tools, such as environmental analysis and extensive public engagement, to help in its decisionmaking.

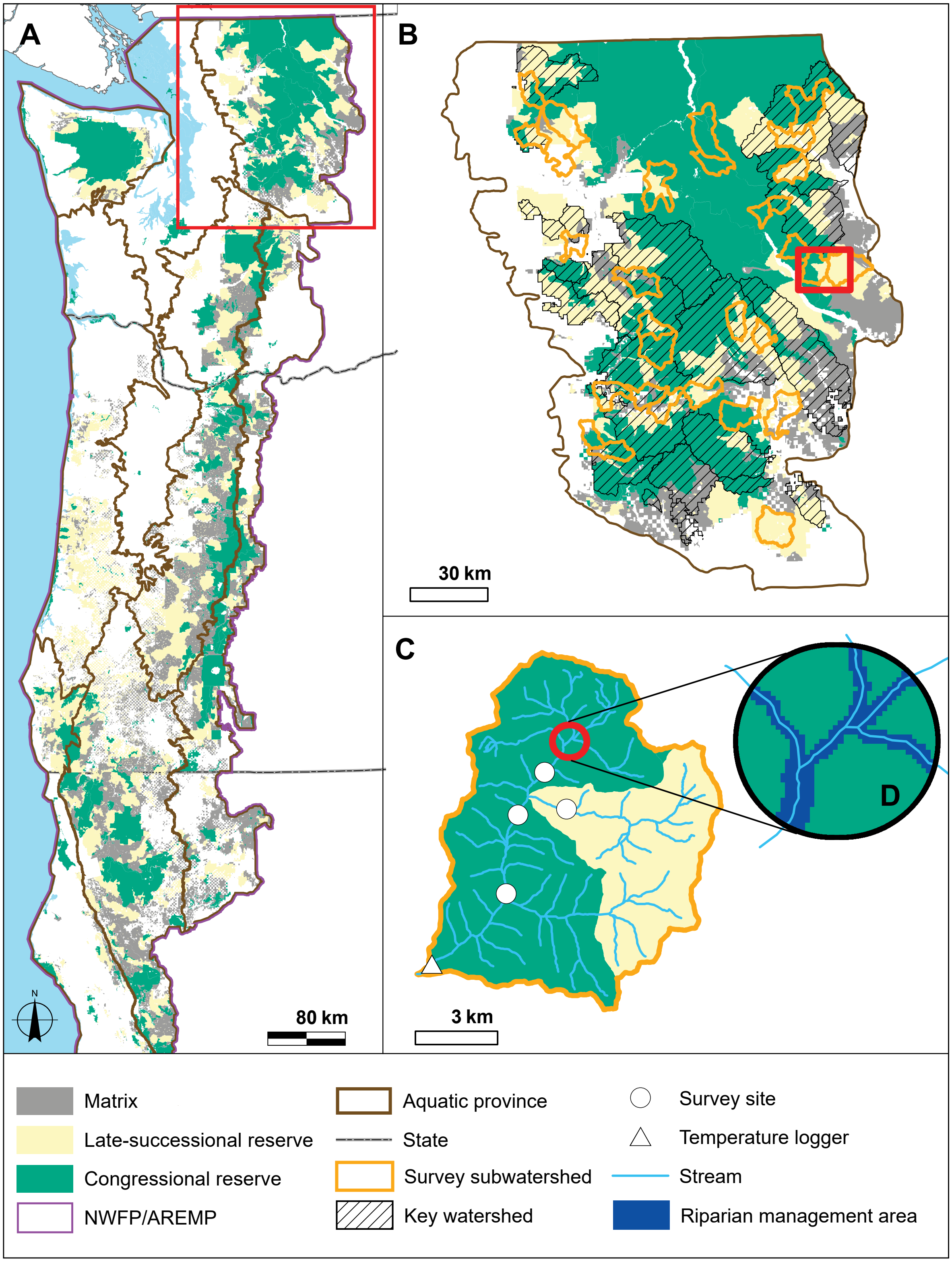

Figure 1.3—Map of the Northwest Forest Plan (NWFP) and Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area showing (A) federal lands by aquatic province; (B) North Cascade province with land use allocation, key watersheds, and AREMP-surveyed subwatersheds; (C) an AREMP-surveyed subwatershed with sampled survey sites and temperature logger site; and (D) riparian management areas.

Table 1.1—Sample size of individual instream and upslope response variables evaluated in this 25-year report

Response type	Sample size	Responses
Instream[a]	Surveyed watersheds (n = 219 subwatersheds)	Wetted width Water temperature Instream wood Transect sediment fines Pool tail sediment fines Particle size distribution Macroinvertebrate composition
Upslope[b]	Full AREMP area (n = 1,972 subwatersheds)	Canopy cover in riparian management area Old-growth structure index in riparian management area Streamside large-tree density Chronic sediment from roads Roads and shallow landslide risk Habitat connectivity
	Surveyed watersheds (n = 219 subwatersheds)	Drought index Annual discharge

[a] Instream responses were assessed with data collected from 2002 through 2018.

[b] Upslope responses covered broader time frames than instream responses, generally spanning at least 1993–2018 (see main text for details).

of the NWFP evaluated in terms of instream, riparian, and upslope watershed conditions and is continued under the Western Oregon Resource Management Plans. Three types of monitoring— implementation, effectiveness, and validation—are used to evaluate watershed restoration (Kershner 1997). Simply defined, implementation monitoring refers to how management actions are tracked (e.g., did they occur, and how were they implemented?), whereas effectiveness monitoring refers to whether or not a given action or series of actions led to attainment of objectives, and validation monitoring refers to the process of evaluating the validity of assumptions behind effectiveness monitoring. Although AREMP is focused on effectiveness monitoring, there are elements in this report that address each of these.

A challenge associated with addressing ACS objectives is the lack of specific quantifiable outcomes, which can make it difficult for identifying whether desired outcomes have been attained. ACS objectives were written to address natural processes and variability. By contrast, more simply quantified responses, such as temperature criteria (Poole et al. 2004) or reference conditions (Kershner 1997, Stoddard et al. 2006), are widely applied but do not capture the full complexity of processes envisioned by the ACS (Bisson et al. 2009, Reeves et al. 2018). This report attempts to bridge those perspectives by coupling rigorous quantitative modeling of instream responses with detailed process narratives and summaries of upslope/riparian conditions that describe how specific processes interact to produce instream responses. Instead of using a standardized scoring scale as in previous reports, which involves some type of reference condition, whenever possible we simply report on changes in the indicators directly (e.g., change in riparian canopy cover), which is intended to be more transparent for readers.

Results from statistical analyses (box 1.2) of spatial and temporal variation in measured instream responses are coupled with the most relevant watershed drivers (e.g., climate, forest cover, roads), based on hypothesized process linkages described in a series of process narratives (app. 5). Within the up-front summary we provide brief interpretations as to what patterns in the results could mean relative to the process narratives and other hypotheses. Given the scope and complexity of responses considered herein, we reserve more detailed interpretations for anticipated future publication of results.

Findings in this report are organized to provide an up-front summary of the most salient results for the overall area, by province, between key and non-key watersheds, and by LUA. Additional information is provided in the

Box 1.2

Guide to Interpretation of the Statistics Used in This Report

We employed a diversity of approaches to analyze the many facets of watershed condition in this report. Most upslope responses (outside of measurements in stream channels) are based on existing spatial data, including a number of model-based data products. Spatial data used are available for all subwatersheds across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area, and thus are not a sample but a full census. Accordingly, we report these without inferential statistics (i.e., sampling error). Although spatial data can provide continuous coverage of the AREMP area, spatial data often have unknown levels of uncertainty that can propagate in modeling processes that combine multiple datasets. Therefore, we report mean values directly for each indicator and display the variability in subwatershed-level results by using a minimum change threshold based on examination of the data and known uncertainty where available. Only one of our upslope indicators (canopy cover) has been assessed for error rates near the scales of our summaries, with a resulting root mean square error of 3–4 percent (depending on scale). Readers may keep this margin of error in mind for the reported status and trend numbers. In appendix 3, we provide brief descriptions of the indicators and links to further information. We present results in several formats addressing management decisions (AREMP area, subwatersheds, aquatic province, land use allocation, and key and non-key watersheds; fig. 1.3) to allow readers to see this variability and assess patterns for their own interpretations.

For instream attributes, status and trends were estimated using hierarchical Bayesian models. Similar to the upslope responses, we present results in several groupings: overall AREMP area, subwatershed, aquatic province, land use allocation, and key and non-key watersheds. Generally, we summarize trends based on rate of change per decade. To summarize responses at finer scales, we report the percentage of subwatersheds with an increasing or decreasing trend. We use credibility intervals to describe the uncertainty and the level of confidence in the estimated trends similar to the confidence intervals used in frequentist statistics. We report significant differences in trends and conditions for the responses with strong probabilistic support (probability >95 percent). For differences in trends and conditions with less probabilistic support, we characterize them as moderate (90–95 percent), or weak (85–90 percent). For other responses with trends or other patterns summarized in this report (e.g., drought indices or stream discharge), we used a variety of conventional (frequentist) statistical methods for inference, which we describe within each section.

appendices, including detailed descriptions of the AREMP area (fig. 1.1); field methods; field data summaries by national forest, BLM district, and national park; and analytical methods used to statistically analyze data. The front-facing material is organized into the following sections:

- Climate and surface water availability (changes in climate, modeled streamflow, and measured sizes of streams at summer low flows)
- Forest canopy cover and stream temperature
- Forest condition and instream large wood
- Roads, landslide risk, and instream fine sediment
- Biotic responses (culverts and connectivity for fish, patterns of variability in stream macroinvertebrate assemblages)
- Multivariate assessment of integrated watershed condition

Chapter 2: Climate and Surface Water Availability

Although the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) was not initially designed to evaluate the effects of climate, climate is an overarching driver of nearly every process influencing watershed conditions (Furniss et al. 2010, Spies et al. 2019). A large body of evidence indicates that climate change is fundamentally altering the Pacific Northwest (e.g., Abatzoglou et al. 2014, Dalton et al. 2013, Reidmiller et al. 2018, Wuebbles et al. 2017). Documented climate-related changes include warming water temperatures (Arismendi et al. 2012; Isaak et al. 2016, 2018), stream drying (Jaeger et al. 2019), earlier runoff and reduced summer streamflows (Luce and Holden 2009, Sawaske and Freyberg 2014), and potential for increasing winter floods, particularly in watersheds where warming has led to winter rains replacing snowfall (Safeeq et al. 2016, Wenger et al. 2010). Changes include not only increases (e.g., temperature) or decreases (e.g., low flows in streams), but also regional changes in the spatial synchrony and year-to-year variability in climate responses, making climate-related events such as drought more synchronized and spatially contiguous across watersheds in the region (Black et al. 2018, Overpeck and Udall 2020). We considered climate change in the context of changes as an indicator of drought, modeled stream discharge, and measured wetted widths of streams in sampled watersheds. These lines of evidence provide a means of associating meteorological changes in climate and drought conditions to modeled and measured hydrological conditions (van Loon 2015). Modeled outputs indicating changes in climate and stream discharge were adapted to provide context for understanding watershed condition status and trends within the AREMP area relative to potential influences of changing climatic conditions. Trends in these outputs were estimated using Kendall's tau for statistical significance and Sen's slope estimator (trend and slope, respectively) (Helsel and Hirsch 2002).

◄ Aquatic and Riparian Effectiveness Monitoring Program survey crew members at North Fork Dillon Creek in Klamath National Forest. Photo courtesy of Leah Diggins.

Climate: Drought Assessment

The broadest indicator of climatic changes we considered was the Standardized Precipitation-Evapotranspiration Index (SPEI) (Vicinte-Serrano et al. 2010), a widely used indicator of relative drought (app. 3). We computed a monthly scaled SPEI across sampled subwatersheds (12-digit hydrologic unit code[1] [HUC]). We summarized yearly SPEI values across the normally driest months of the year: July, August, and September. We considered annual summer variability in SPEI for three time periods: 1980–2018, 1980–1993, and 1994–2018. The first time period represents the entire range of available information, whereas the second and third time periods represent years preceding and following implementation of the NWFP, respectively.

Overall and by Province

Overall—

Across the AREMP area, variable expression of drought using SPEI is evident in the available data (fig. 2.1). The SPEI units can be roughly interpreted as follows: non-drought conditions correspond to values >-0.5; mild drought conditions correspond to values from -1 to -0.5; moderate drought corresponds to values from -1.5 to -1; severe drought corresponds to values from -2 to -1.5; and extreme drought corresponds to values <-2 (McKee et al. 1993, Paulo et al. 2012). Generally, in any given year, a few subwatersheds meet a severe drought classification, with most subwatersheds, over time, in the non-drought condition classification.

Overall, across the full available time series we considered (1980–2018), many AREMP subwatersheds trended toward lower values of summer (July, August, September) SPEI and thus greater drought severity (fig. 2.2). Within the narrower time periods preceding (1980–1993) and following (1994–2018) implementation of the NWFP, trends toward lower values of SPEI are most obvious within the southern extent of the AREMP area. Across the NWFP time period, however, the trend

[1] For more information, see the USGS Water Resources of the United States webpage: https://water.usgs.gov/GIS/huc.html.

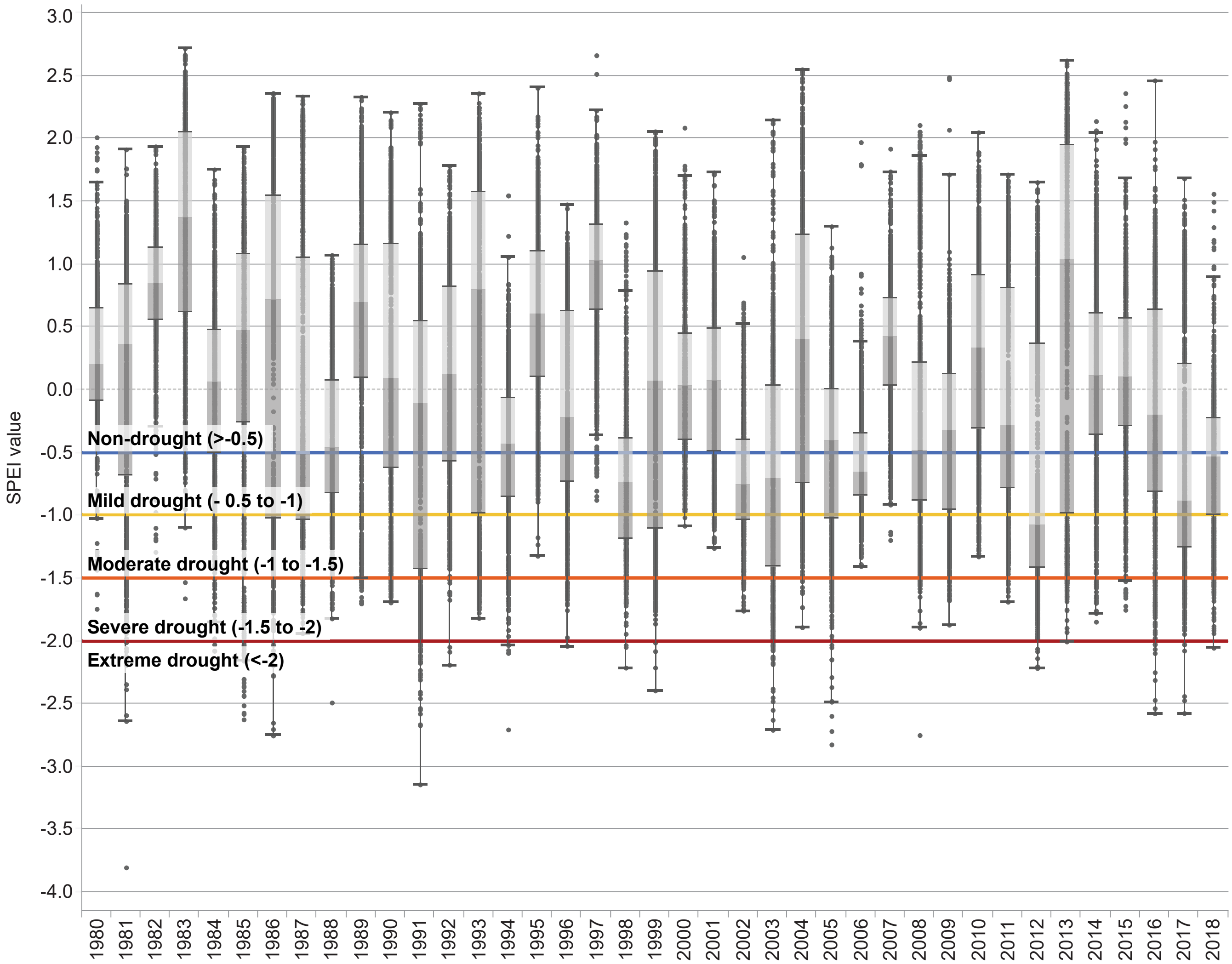

Figure 2.1—SPEI (Standardized Precipitation-Evapotranspiration Index) values from 1980 through 2018 across the Aquatic and Riparian Effectiveness Monitoring Program area. Horizontal reference lines indicate values that identify established levels of drought severity for SPEI: non-drought (>-0.5), mild drought (-0.5 to -1), moderate drought (-1 to -1.5), severe drought (-1.5 to -2), and extreme drought (<-2).

appears to indicate drier conditions in many locations over time. More than half of the subwatersheds were in drought conditions 7 of the 25 years since NWFP implementation (1994), compared to 1 of the 13 preceding years. In 2012, more than half of the subwatersheds were estimated to be in moderate or greater drought condition. Notable exceptions occur throughout the AREMP area, highlighting the importance of considering both local and regional drought variability (Kovach et al. 2019).

Province—

Summer drought conditions (July, August, September) in all provinces appear to be becoming more common than wetter conditions in each of the time series assessed based on the slope values of the trend lines, except the Franciscan province in the most recent time period (fig. 2.3). However, some subwatersheds in the Klamath-Siskiyou showed a trend toward wetter summer SPEI conditions in the 1980–2018 time series and in the shorter 1980–1993 time series. Since the start of the NWFP, some subwatersheds in the northern part of the Franciscan province showed significant drought recovery. Overall, significant trends toward drier

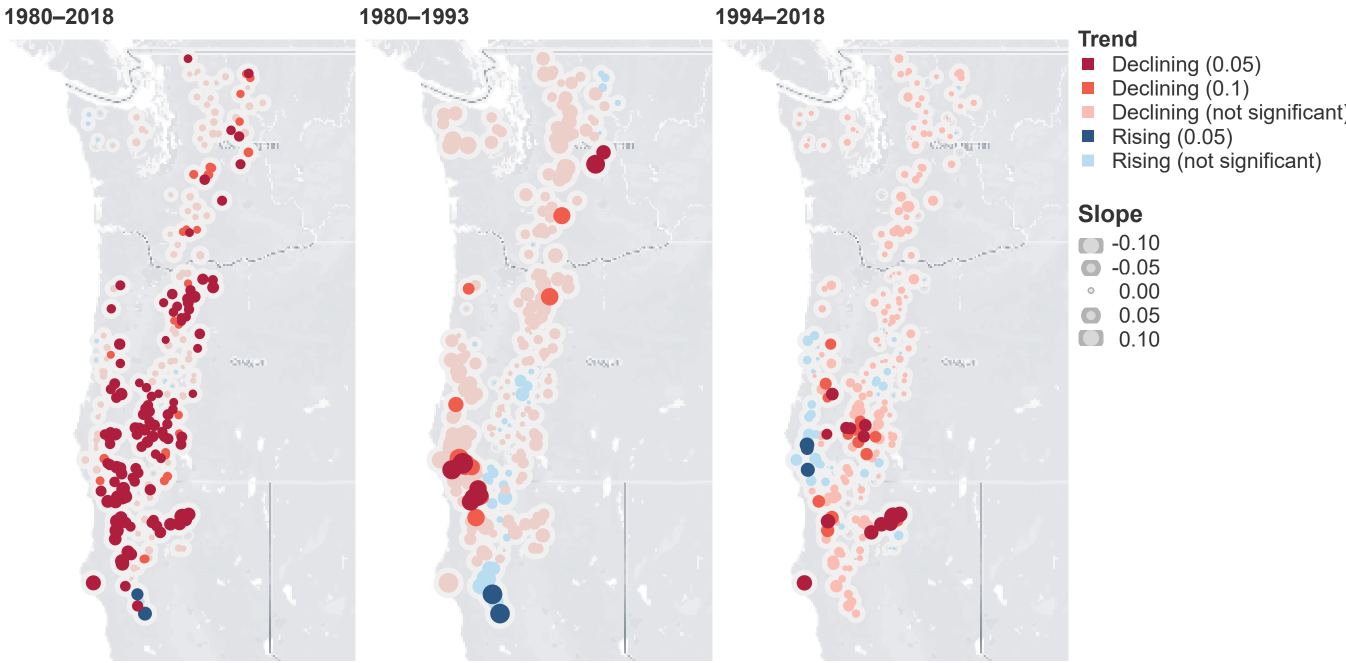

Figure 2.2—Map of trends in SPEI (Standardized Precipitation-Evapotranspiration Index) values for each Aquatic and Riparian Effectiveness Monitoring Program subwatershed during summer (July, August, September) for 1980–2018, 1980–1993, and 1994–2018. Trend significance values of SPEI slopes are displayed with color intensity; slope is displayed with point size.

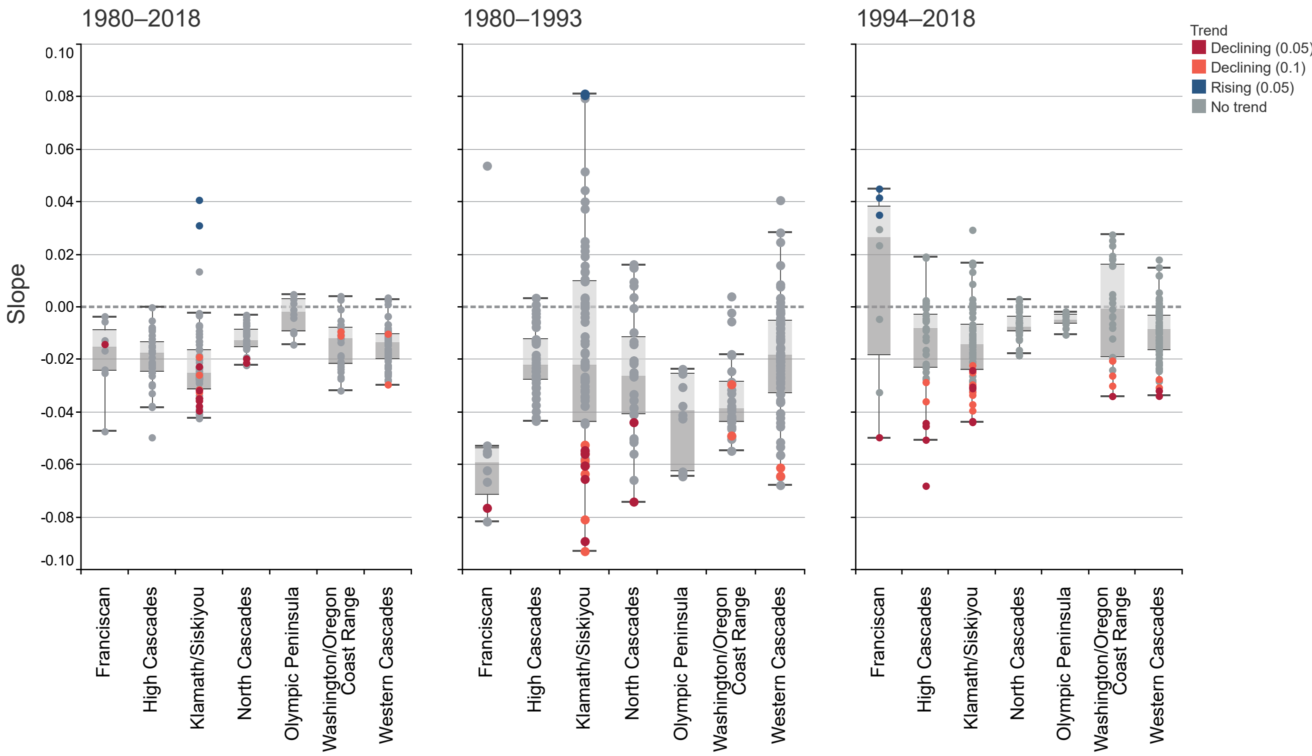

Figure 2.3—Annual trends in SPEI (Standardized Precipitation Evapotranspiration Index) values by aquatic province for each AREMP subwatershed during summer (July, August, September) for 1980–2018, 1980–1993, and 1994–2018. Trend significance values of SPEI slopes are displayed (p-values) with color and intensity.

conditions since the start of the NWFP occurred in the southern provinces. No sites with significant trends were observed in the Olympic and North Cascades provinces during the most recent (1994–2018) time period .

Climate: Annual Discharge

To consider how past meteorological conditions translate into available water, we considered annual discharge. Discharge was modeled using the National Hydrologic Model and the Precipitation-Runoff Modeling System (NHM-PRMS) (Regan et al. 2018) for each subwatershed selected for field-based monitoring in the AREMP (see "Discharge" in app. 3 for methods). Unlike the SPEI, the NHM-PRMS is a physically based model specifically designed to simulate stream discharge. Like SPEI, the NHM-PRMS integrates multiple pathways of influence for temperature, precipitation, and other factors influencing water availability to streams. We considered discharge over three time periods: 1982–2016, 1982–1993, and 1994–2016 (2016 is the most recent year with available modeled data). We report discharge in terms of specific discharge (mm/day), which is the product of volumetric discharge standardized by catchment area. For this report, we considered one descriptor of flow: annual specific discharge. Other components of the flow regime are potentially important for a host of processes (e.g., timing, magnitude, duration, frequency, and predictability) (Poff et al. 1997). Our intent here was to provide an overall assessment of water availability on an annual time step across all subwatersheds in the AREMP area, and for that, annual specific discharge is a useful summary.

Overall and by Province

Overall—

Annual specific discharge estimated by the NHM-PRMS indicates variability across the AREMP area with both increasing and declining specific discharge for the full time series (1982–2016). A greater prevalence of declining discharges was observed in the southern extent of the AREMP area (fig. 2.4). Numerous AREMP subwatersheds exhibit no trends in discharges for the full time series. For the time period preceding the NWFP (1982–1993), discharges in AREMP subwatersheds exhibited more obvious declines, whereas such declines were less pronounced for the time period following implementation of the NWFP (1994–2016). For the full time series, trends in most subwatersheds monitored by AREMP were not significant, but when they were, declines were most evident.

Province—

NHM-PRMS estimates of mean annual discharge exhibited spatial variability across the aquatic provinces of the AREMP area. The North Cascades and Olympic Peninsula provinces show the highest mean annual discharge values and also the largest range in values across subwatersheds within a province. The Klamath-Siskiyou and High Cascades showed the lowest mean annual discharge values and the smallest range in values from 1982 to 2016 (fig. 2.5). Trends in annual specific discharge at the province scale parallel overall patterns indicating that trends in discharge, when observed, generally decline. The North Cascades showed some sites with potential increases in annual discharge (fig. 2.6). Sites with the largest potential declines in annual discharge were observed in the Olympic Peninsula and Klamath-Siskiyou aquatic provinces.

Surface Water Availability: Wetted Width

The AREMP monitoring plan (Reeves et al. 2004) did not specify measurements of stream discharge, so we relied on a closely related indicator that is logically linked to modeled discharges and SPEI: wetted widths of streams. We considered trends in wetted widths expressed as a proportion of subwatershed-specific average bankfull width from 2002 to 2018 (for methods, see app. 2). The subwatershed-specific bankfull width benchmark was used to provide a constant frame of reference for comparisons across sites and to account for a statistical requirement of normality. Changes in relative wetted width indicate only changes in wetted width because we used average bankfull width for the benchmark. Another indicator of climate influence is stream temperature (Arismendi et al. 2012; Isaak et al. 2016, 2018), which is addressed in the following section (for methods, see app. 2). We also considered annual changes in discharges (see previous section). Wetted width by administrative unit is presented in appendix 6.

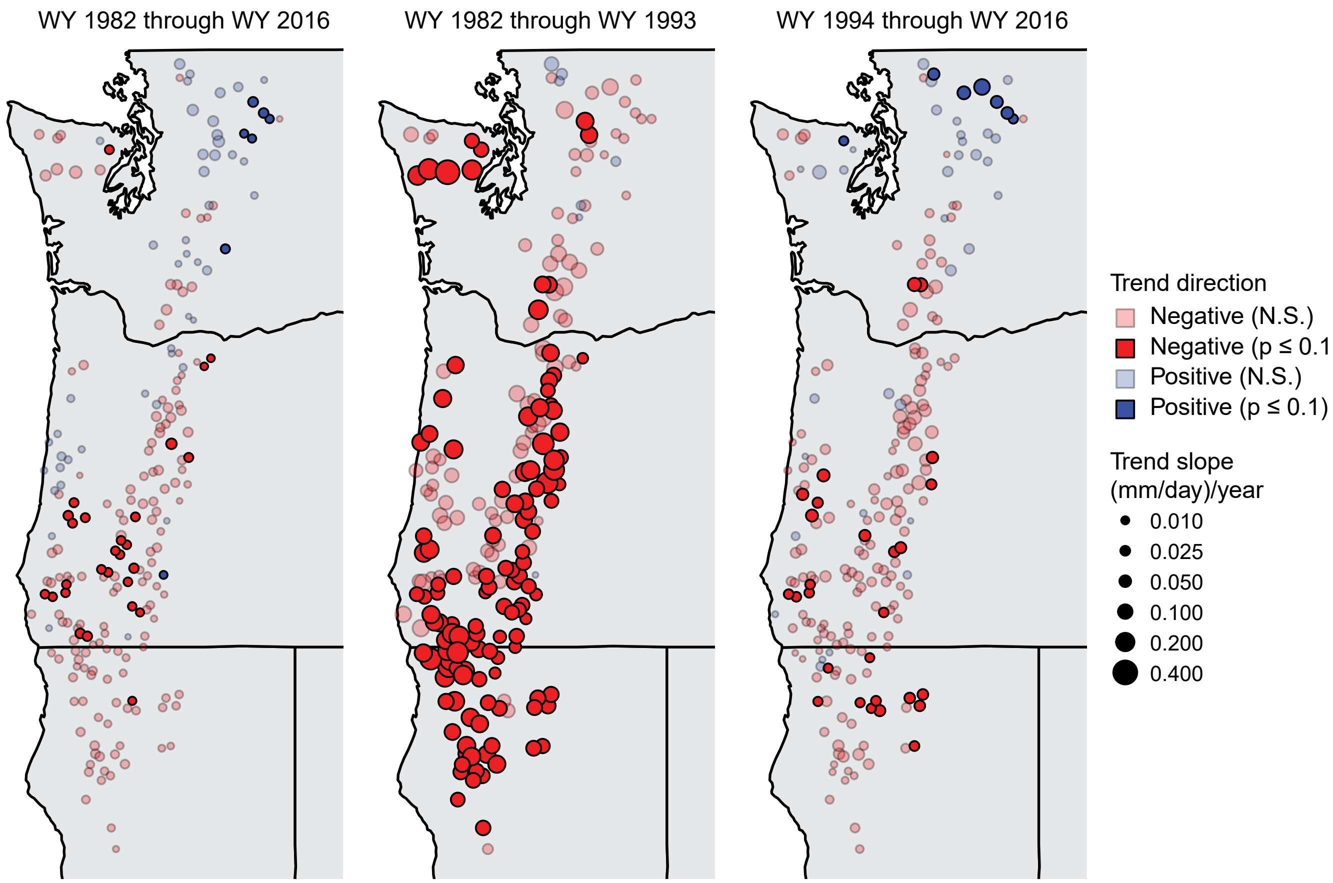

Figure 2.4—Map of trends in specific discharge ([mm/day]/year) values for each Aquatic and Riparian Effectiveness Monitoring Program (AREMP) subwatershed for 1982–2016, 1982–1993, and 1994–2016 water years (WYs). Trend significance values of slopes are displayed with color intensity; slope is displayed with point size.

Overall and by Province

Within the time period of instream measurements recorded by the AREMP (2002–2018), we found declining trends in wetted width of streams were evident overall (grand trend) and across all aquatic provinces within the AREMP area (fig. 2.7). The overall linear trend in wetted width was -9.3 percent of the long-term average bankfull width benchmark of the site per decade (95-percent credibility interval from -15.0 to -3.9 percent). Across the AREMP extent, 82 percent of subwatershed-level trends were negative (95-percent credibility intervals from 63 to 94 percent) for wetted width. Declines in instream wetted widths were most pronounced in the High Cascades, Klamath-Siskiyou, and Olympic Peninsula aquatic provinces. The least pronounced decline in wetted width was found in the Franciscan aquatic province.

Key Watersheds and Land Use Allocations

Key watersheds—

Average wetted widths were higher in key watersheds, and there was moderate support for flatter average slope in declining trends across years in key watersheds compared to non-key watersheds (fig. 2.8).

Land use allocations—

Matrix lands had lower wetted widths on average than LSR and congressional reserve lands. LSR lands were distinguished from matrix and congressional reserve lands by having flatter average slope in their declining trends from 2002 to 2018.

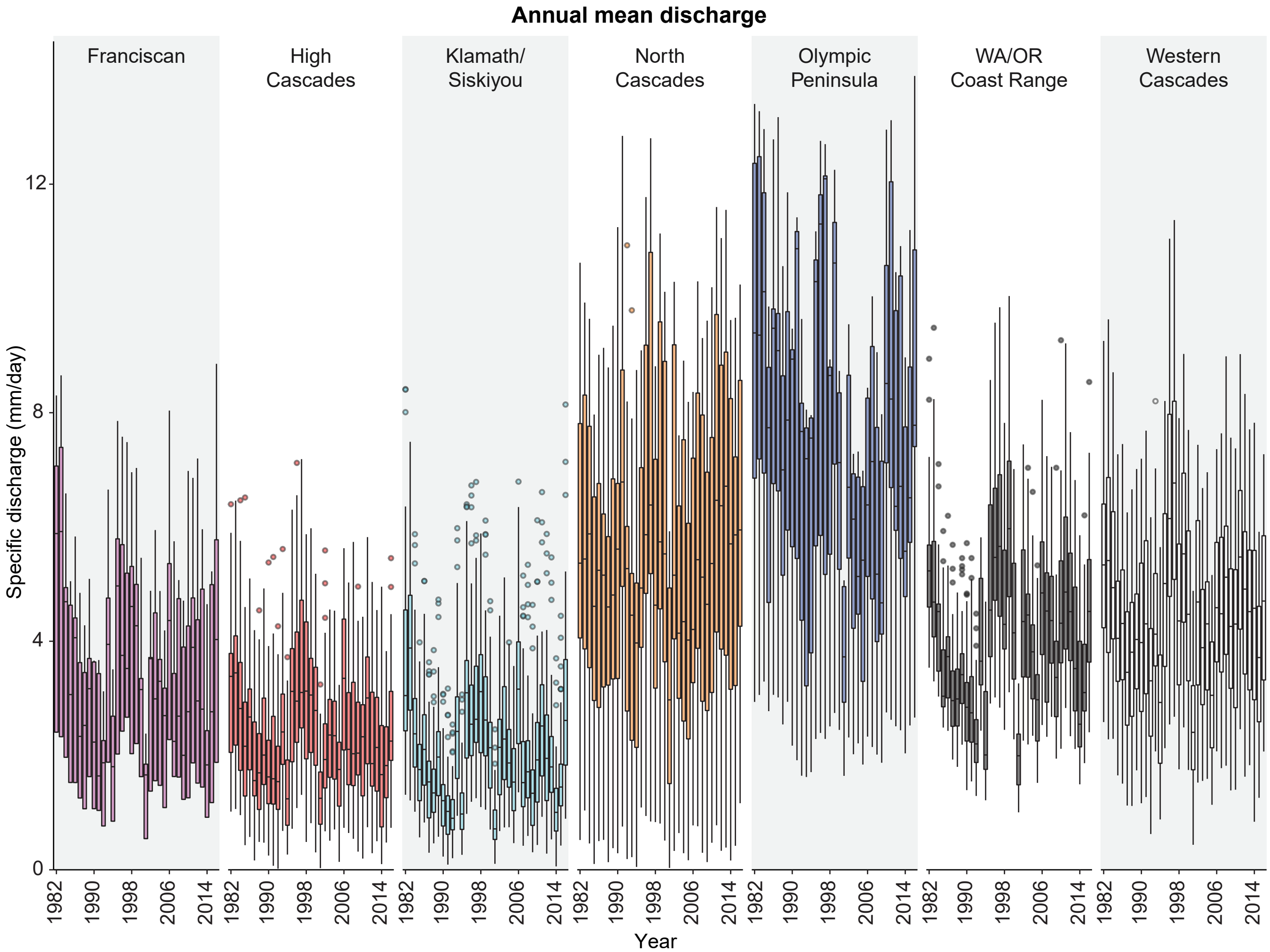

Figure 2.5—Annual mean specific discharge ([mm/day]/year) for each Aquatic and Riparian Effectiveness Monitoring Program subwatershed across aquatic provinces, 1982–2016. Colors display aquatic provinces. Discharge is modeled from the National Hydrologic Model and Precipitation-Runoff Modeling System and summarized by water year. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated.

Discussion

Collectively, results for a major drought indicator (SPEI), modeled discharges (from the NHM-PRMS), and observed trends in wetted widths point to variable declines in water availability across the AREMP area. Although the three lines of evidence evaluated here point to declines in water availability across the region, particularly since implementation of the NWFP, it is important to recognize that a host of factors can act to influence water availability to aquatic ecosystems (see "Stream Discharge" in app. 5), although most are influenced by climate. It is also important to recognize that processes influencing hydrology at scales too fine to track with existing spatial datasets used to model SPEI and discharges from the NHM-PRMS could play an important role in driving declines in wetted widths observed from ground-based data (Kovach et al. 2019). Furthermore, hydrologic drought can emerge via multiple and complex pathways, with processes (patterns of precipitation and temperature) that strongly vary in space and time and strongly interact with local factors (Crausbay et al. 2020, van Loon 2015).

Interpretation of climate-related trends (e.g., SPEI, discharge, or other indicators) can strongly depend on the length of records, as well as when they begin and end (Easterling and Wehner 2009). Accordingly, for all longer term and climate-related responses addressed herein, we report a range of time series to provide more robust assessments of trends relative to the NWFP time period.

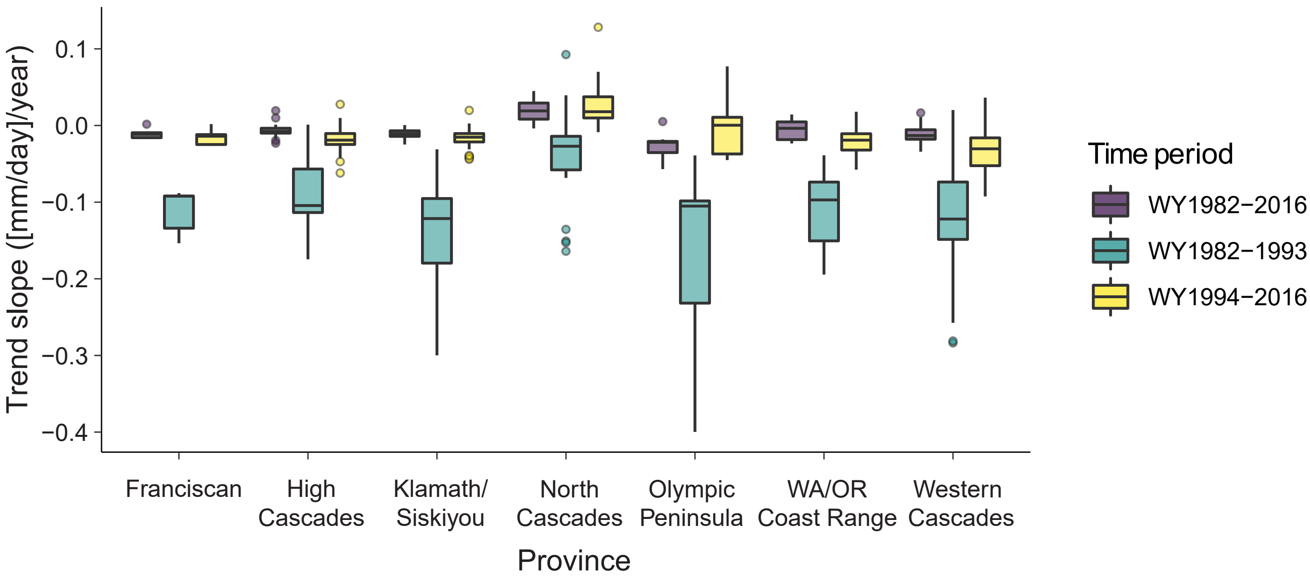

Figure 2.6—Trend slope values displayed as median total annual specific discharge ([mm/day]/year) for each Aquatic and Riparian Effectiveness Monitoring Program subwatershed for each aquatic province. Colors display time periods of record: 1982–2016, 1982–1993, and 1994–2016. Discharge is modeled from the National Hydrologic Model and Precipitation-Runoff Modeling System. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated.

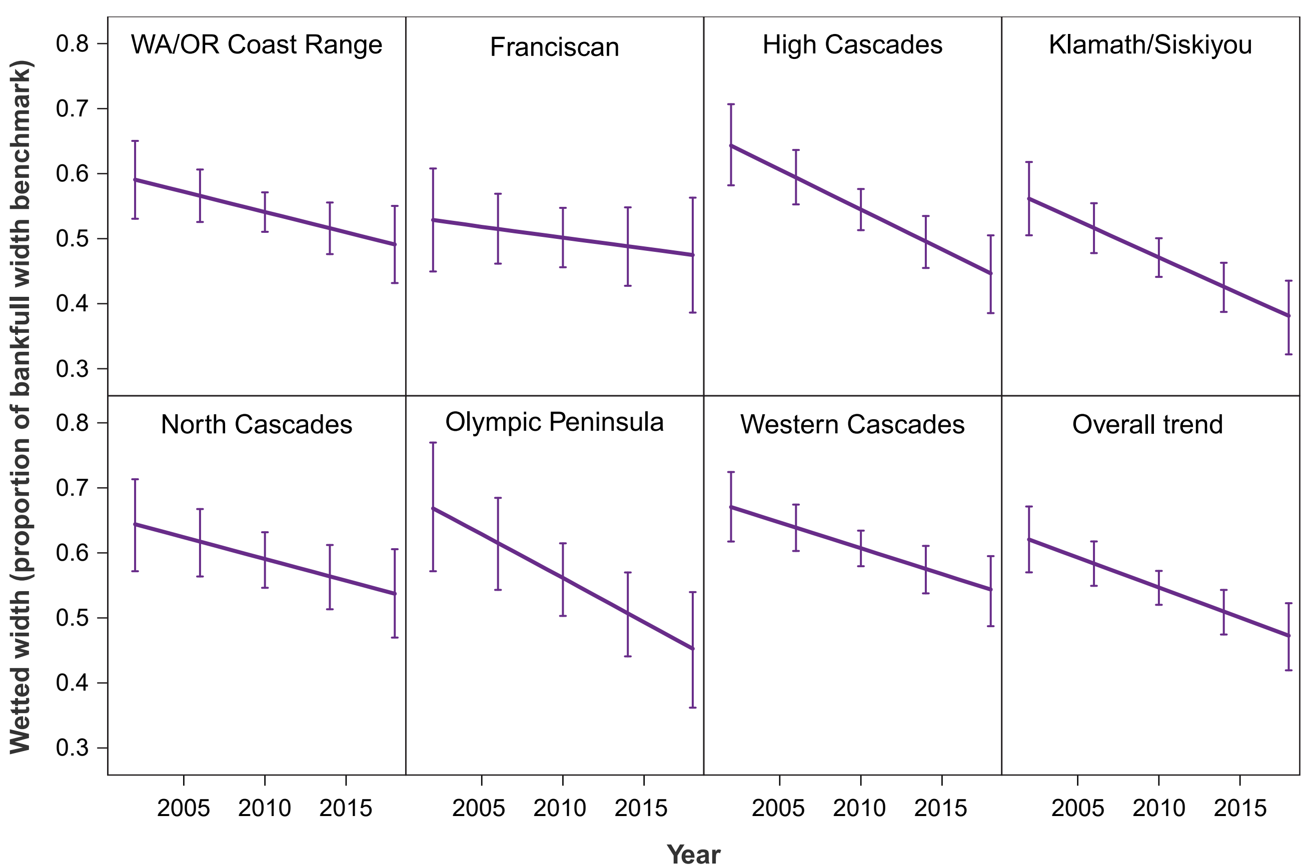

Figure 2.7—Average trends in relative wetted widths (as a proportion of bankfull width at the site) of streams monitored by the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) for 2002–2018. Solid lines represent average trends of sampled subwatersheds within aquatic provinces. Drop-lines are 95-percent credibility intervals. Results are shown by province and overall across the AREMP area.

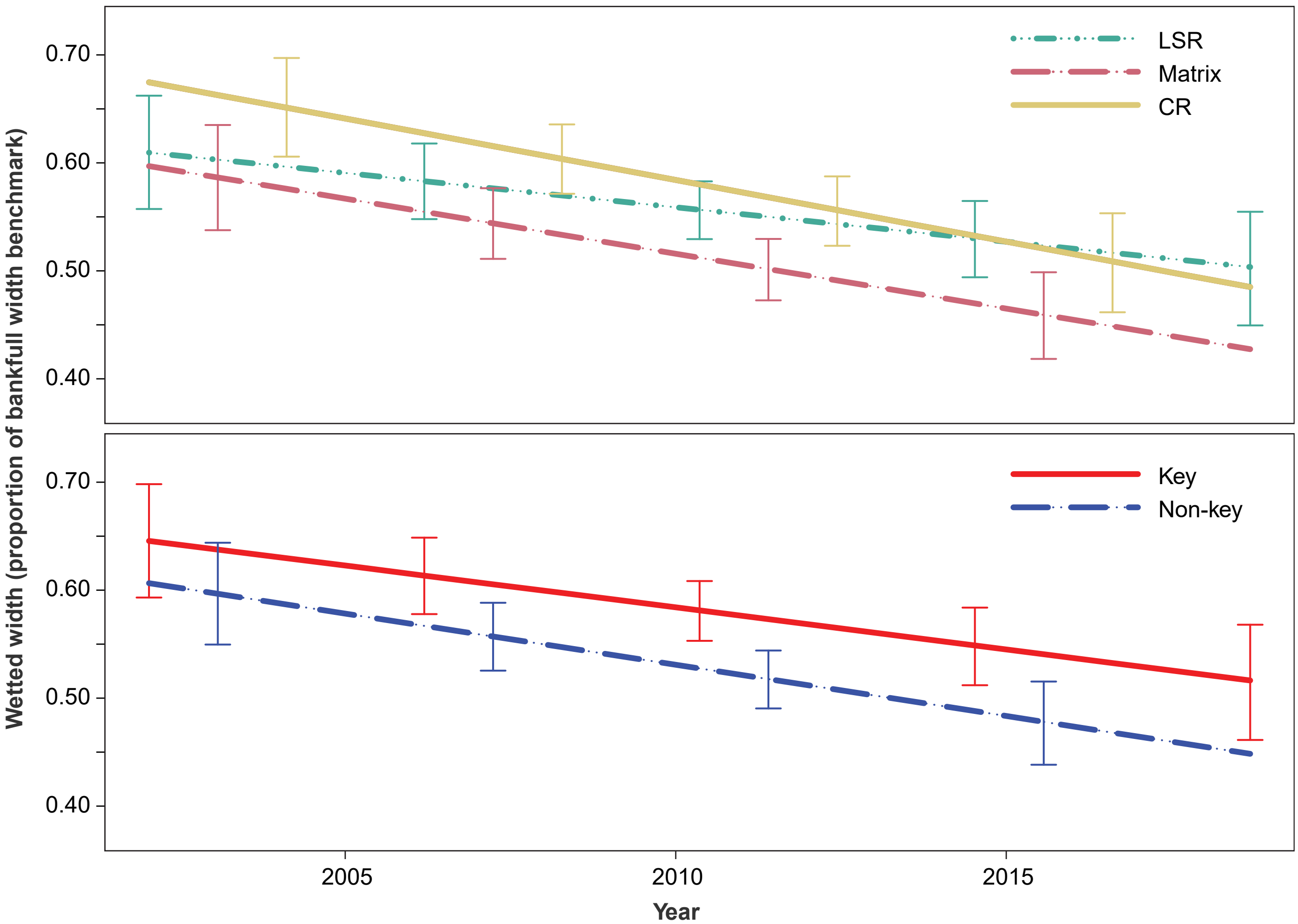

Figure 2.8—Average trends in relative wetted width (as a proportion of benchmark bankfull width at the site) by key and non-key watershed; and by land use allocation for 2002–2018. Bold lines represent average trends of sampled subwatersheds. Drop-lines are 95-percent credibility intervals. LSR = late-successional reserve, CR = congressional reserve.

Similar qualifications applied to interpretations of trends in relative wetted widths, which were only available after the NWFP was initiated (2002–2018).

A second note of caution in interpreting results for SPEI and NHM-PRMS discharges is related to how the statistical significance of trends was considered. For frequentist statistical tests, we report trends considered to be "significant" using statistical alpha levels of 0.1 and 0.05 and note that interpretation of such values has been the topic of much discussion in the literature.[2] It is also important to note that the statistical tools applied herein are designed for assessments of monotonic trends (Helsel and Hirsch 2002), and thus other forms of change may be revealed by alternative analyses. Similar reasoning applies to interpretation of Bayesian statistical analyses (Marden 2000). Whether interpreted in terms of their statistical, physical, or biological significance, however, our findings effectively describe spatial and temporal variability in climate-related conditions across the AREMP area and the fundamental flow-dependent processes that are relevant to watershed conditions (Poff et al. 1997).

Overall, results from this assessment of three major indicators of climate- or drought-related influences on hydrologic processes support the conclusions of the NWFP science synthesis (Spies et al. 2019), which

[2] Alpha levels formally refer to statistical probability of falsely rejecting the hypothesis that statistically significant trends are in fact not present in the data. This "frequentist" approach to statistical inference has been the subject of much debate in the literature (Lukacs et al. 2007, Nicholls 2001). Note also that longer time series may have greater statistical power to detect trends, relative to shorter time series.

identifies climate change as a major driver across the region. Numerous local factors (e.g., Coble et al. 2020, Jaeger et al. 2019, Leibowitz et al. 2016) are also likely acting to influence water availability, but they are not currently tracked by regional monitoring efforts. If a better understanding of these processes is desired, additional investments in finer scale monitoring of hydrological responses and factors influencing water availability to parameterize appropriate physically based or statistical models would be necessary (Crausbay et al. 2020, Kovach et al. 2019). Given that federal lands produce much of the available surface water in the region (Luce et al. 2013), this may become an important question (e.g., Hafen et al. 2020).

Chapter 3: Forest Canopy Cover and Stream Temperature

Riparian conditions, including canopy cover, affect streams through numerous pathways (Naiman et al. 2005). In the context of land management, influences of riparian cover on exposure of stream surfaces to solar radiation and consequences for the heat budget of streams are primary concerns (see "Temperature" in app. 5). Stream temperature influences nearly every ecological process, most notably survival and growth of coldwater taxa such as salmon and trout, which are a major management focus of the Northwest Forest Plan (NWFP) (McCullough et al. 2009). The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) has collected stream temperature data since 2002, but it was not until 2011 and 2012 that enough temperature data loggers were deployed to allow for the characterization of thermal conditions among aquatic provinces. A robust trend analysis of thermal conditions is not yet feasible with existing datasets due to sample size and time limitations (e.g., Arismendi et al. 2012). Here, we report the patterns and trends of a major driver of stream temperature, forest canopy cover, as well as stream temperatures from records available across provinces within the AREMP area.

Canopy Cover

Canopy cover of live trees was assessed within the variable width riparian management areas (RMAs) as defined in the NWFP to target areas (riparian zones) most likely to directly affect stream thermal conditions. Various aspects of forest canopies have potential to influence incoming solar radiation and thermal conditions in streams (app. 5), but here we used cover of live trees as an overall surrogate for riparian shading. We used modeled forest vegetation (Davis et al. 2022) extracted from RMAs to derive canopy cover (see "Forest Characteristics" in app. 3 for methods). Values for canopy cover in RMAs were summarized at the subwatershed scale across the entire AREMP area. Analyses compared average RMA canopy cover values in 1993 and 2017. We used a minimum change threshold of 5 percent for canopy cover when reporting individual subwatershed scale changes.

◄ Lower North Fork Siletz River valley in the Bureau of Land Management Northwest Oregon District. Photo courtesy of the Aquatic and Riparian Effectiveness Monitoring Program.

Overall and by Province

Overall—

The average value for RMA canopy cover in the AREMP area changed from 70 percent in 1993 to 72 percent in 2017. Within the NWFP area, however, there was considerable variation in individual subwatersheds where canopy cover ranged from 0 to 94 percent and changes over the monitoring period ranged from -40 to +39 percent. Approximately one-quarter of the subwatersheds (509 out of 1,972) experienced an increase of ≥5 percent cover and approximately one-tenth (203) experienced a decline of ≥5 percent (fig. 3.1). Losses of RMA canopy cover in subwatersheds were often associated with large-scale disturbances such as wildfire. For example, a loss of 40 percent was identified at Lake Creek (North Cascades) that was heavily burned by the Fawn Peak fire complex in 2003. This compares with locations where large gains in RMA canopy were identified that tend to be associated with recovery from disturbance, or to expansions of forest conditions into areas that were not historically forested. An example of the latter is a gain of 39 percent on the North Beach Peninsula of the Willapa Bay National Wildlife Refuge (Washington-Oregon Coast Range) where shore pine may have expanded into historically dune-dominated environments (Kumler 1969).

Province—

Among provinces within the AREMP area, average values for RMA canopy cover were consistently highest in the Olympic Peninsula province (80 percent in 1993 to 83 percent in 2017) and lowest in the Klamath-Siskiyou (64 to 65 percent) and High Cascades provinces (65 to 67 percent) (fig. 3.2). Changes in cover were most apparent in the Puget-Willamette Trough province (+4 percent), and Western

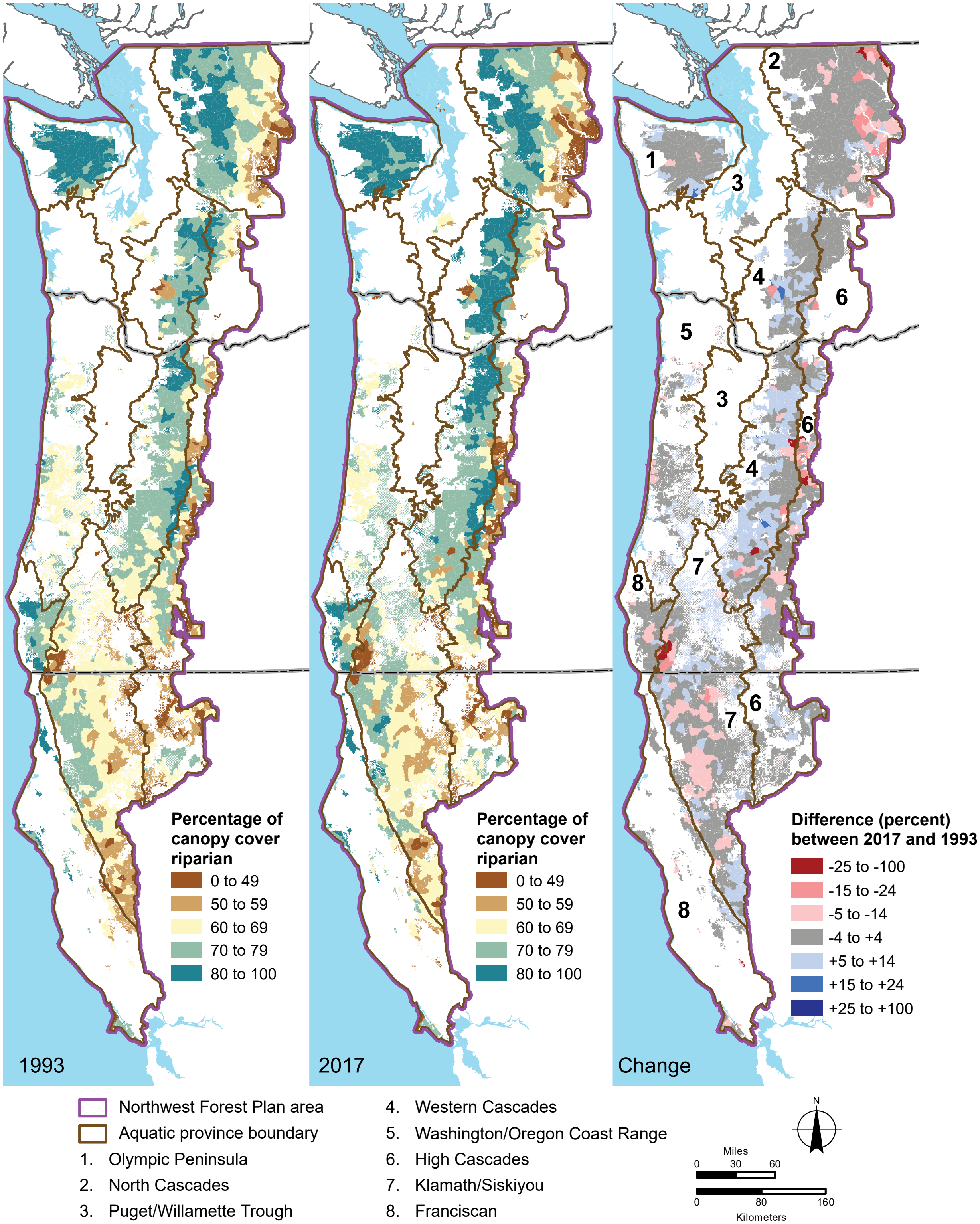

Figure 3.1—Mean percentage of riparian canopy cover by subwatershed among aquatic provinces within the Aquatic and Riparian Effectiveness Monitoring Program area for 1993, 2017, and the difference between these two time periods.

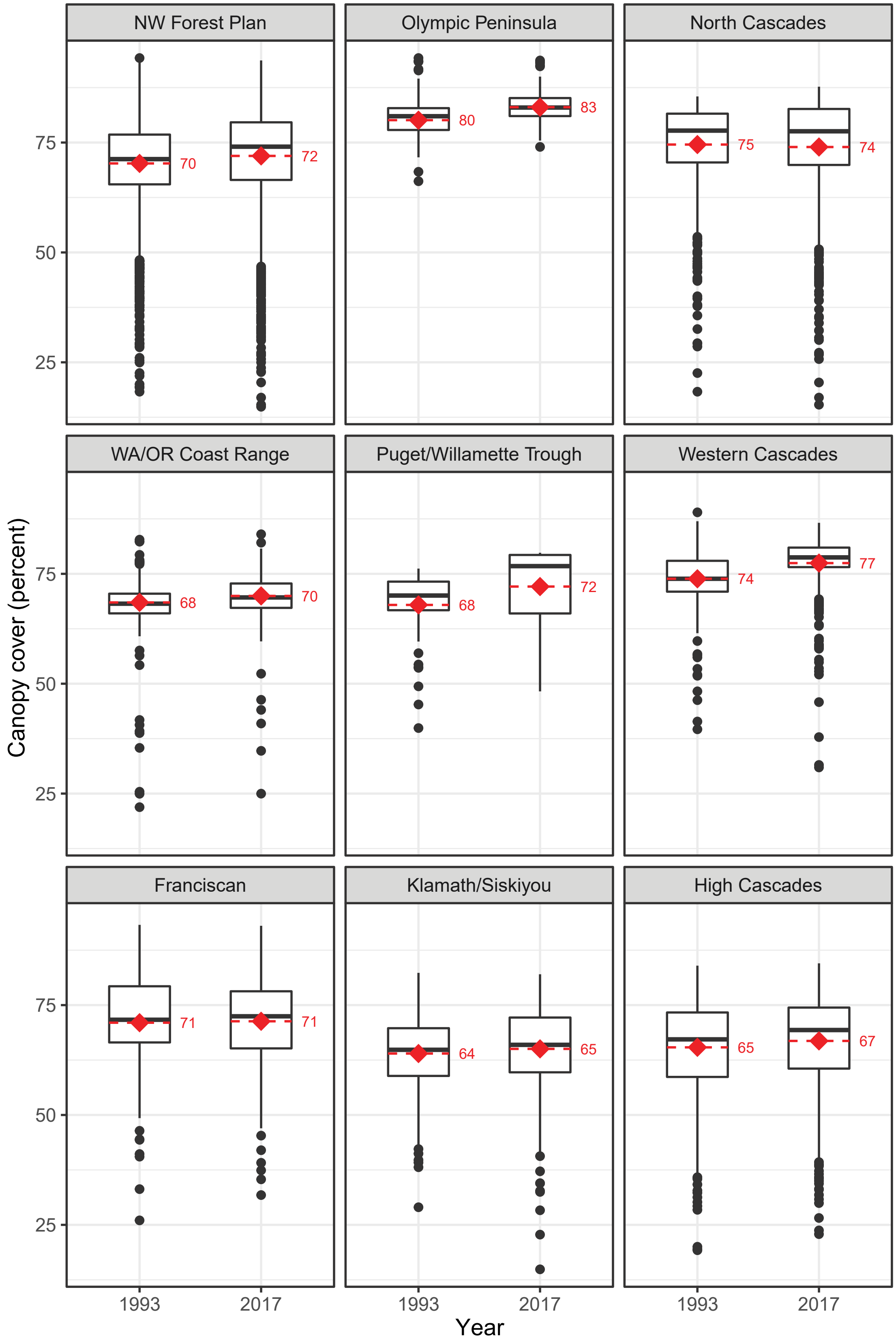

Figure 3.2—Mean percentage of riparian canopy cover by subwatershed and aquatic province within the Aquatic and Riparian Effectiveness Monitoring Program area for 1993 and 2017. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

Cascades and Olympic Peninsula provinces (+3 percent); the least in the Franciscan province (0 percent) and North Cascades province (-1 percent). All other provinces exhibited changes in the 1 to 2 percent range.

Key Watersheds and Land Use Allocations

Key watersheds—

In 1993, average RMA canopy cover was higher in key watersheds (72 percent) than non-key watersheds (69 percent) (fig. 3.3). Canopy cover increased by 1 percent in key watersheds compared to 2 percent in non-key watersheds, resulting in 2017 mean cover values of 73 percent (key) and 71 percent (non-key).

Land use allocation—

In 1993, average RMA canopy cover was higher in congressionally reserved lands (72 percent) and late-successional reserves (LSRs) (72 percent) than in matrix lands (67 percent). Canopy cover decreased by 1 percent from 1993 to 2017 in congressionally reserved lands, and increased by 2 percent in LSRs and by 4 percent in matrix, resulting in 2017 mean cover values of 71 percent in congressionally reserved and matrix lands and 74 percent in LSRs (fig. 3.4).

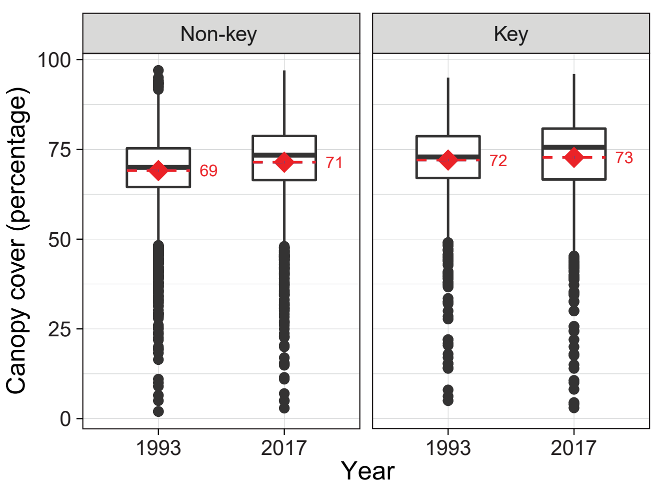

Figure 3.3—Mean percentage of riparian canopy cover by subwatershed for key and non-key watersheds within the Aquatic and Riparian Effectiveness Monitoring Program area for 1993 and 2017. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

Stream Temperature

Stream temperatures have been monitored by AREMP since 2002 in select subwatersheds. Temperature loggers were not placed at individual field sampling sites. Rather, a single temperature data logger was deployed lower in the subwatershed to capture outflow from upstream tributaries (see "Temperature" in app. 2 for field methods). Timing of deployment and recovery of temperature loggers varied depending on field crew availability and scheduling. In 2011 and 2012, a comprehensive temperature monitoring program was implemented by AREMP in Oregon and Washington to support other data collection efforts. The number of subwatersheds with temperature data available for analysis varied by year, aquatic province, key/non-key watershed, and land use allocation (LUA) across the AREMP area (table 3.1). Due to small sample sizes in some years, trend analyses were not attempted with the temperature datasets. Rather, we present patterns in broad categories such as year or aquatic province.

The most comprehensive temperature data were available in the month of August. Therefore, we reported the 7-day average of daily maximum temperature in August statistic (see "Temperature" in app. 2 for details). This statistic is commonly used in applying water quality criteria to protect cold water to support salmonids (Falke et al. 2016, McCullough et al. 2009). Depending on temperature criteria employed, values exceeding 16–18 °C are used by different states to account for exposures considered more likely to cause sublethal stress in salmonids (Falke et al. 2016).

Overall and by Province

Overall—

Annual summaries of temperature across the sampled subwatersheds in the AREMP area indicate spatial variability over time in mean thermal conditions, but some degree of coherence among sites when considered across years. Air and water temperatures also exhibited a degree of coherence across years (fig. 3.5). Throughout the monitored time, 2011 was the coolest water year (mean = 13.75 °C), and 2009 was the warmest (mean = 17.50 °C).

Province—

When summarized by aquatic province, the Franciscan, Klamath-Siskiyou, and Washington-Oregon Coast Range

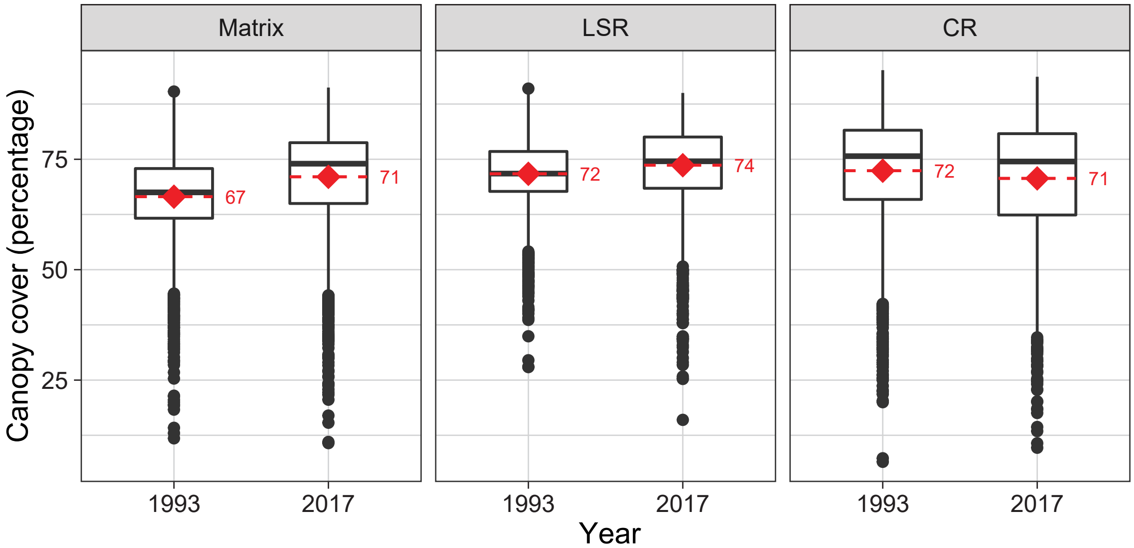

Figure 3.4—Mean percentage of riparian canopy cover by subwatershed and land use allocation for 1993 and 2017 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

Table 3.1—Number of temperature logger sites by year for aquatic province, watershed class (key/non-key), and land use allocations (LUAs)

	Year																
Province	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**
	Number of subwatersheds with temperature data available																
Franciscan	0	0	2	0	0	0	0	1	0	0	0	1	2	1	1	1	0
Klamath/ Siskiyou	3	7	4	7	7	7	6	5	5	4	12	11	7	9	10	13	8
High Cascades	0	2	5	5	4	2	1	1	2	6	14	11	7	6	7	8	1
Western Cascades	3	3	7	10	11	10	4	7	7	18	28	26	38	31	31	25	17
WA/OR Coast Range	3	5	1	1	1	6	2	5	4	5	7	8	9	9	14	8	1
North Cascades	0	2	2	2	1	2	1	3	0	7	2	6	5	12	10	6	6
Olympic Peninsula	0	0	1	2	0	2	2	0	0	0	0	2	4	3	6	5	1
Class																	
Key	5	8	9	12	13	11	5	7	9	12	22	26	30	26	28	26	13
Non-key	4	11	13	15	11	18	11	15	9	28	41	39	42	45	51	40	21
LUA																	
LSR	3	8	11	12	9	13	6	12	10	18	26	24	31	30	39	30	17
Matrix	2	5	6	10	10	10	7	6	2	11	24	23	22	26	23	21	10
CR	4	6	5	5	5	6	3	4	6	11	13	18	19	15	17	15	7

CR = congressional reserve, LSR = late-successional reserve.

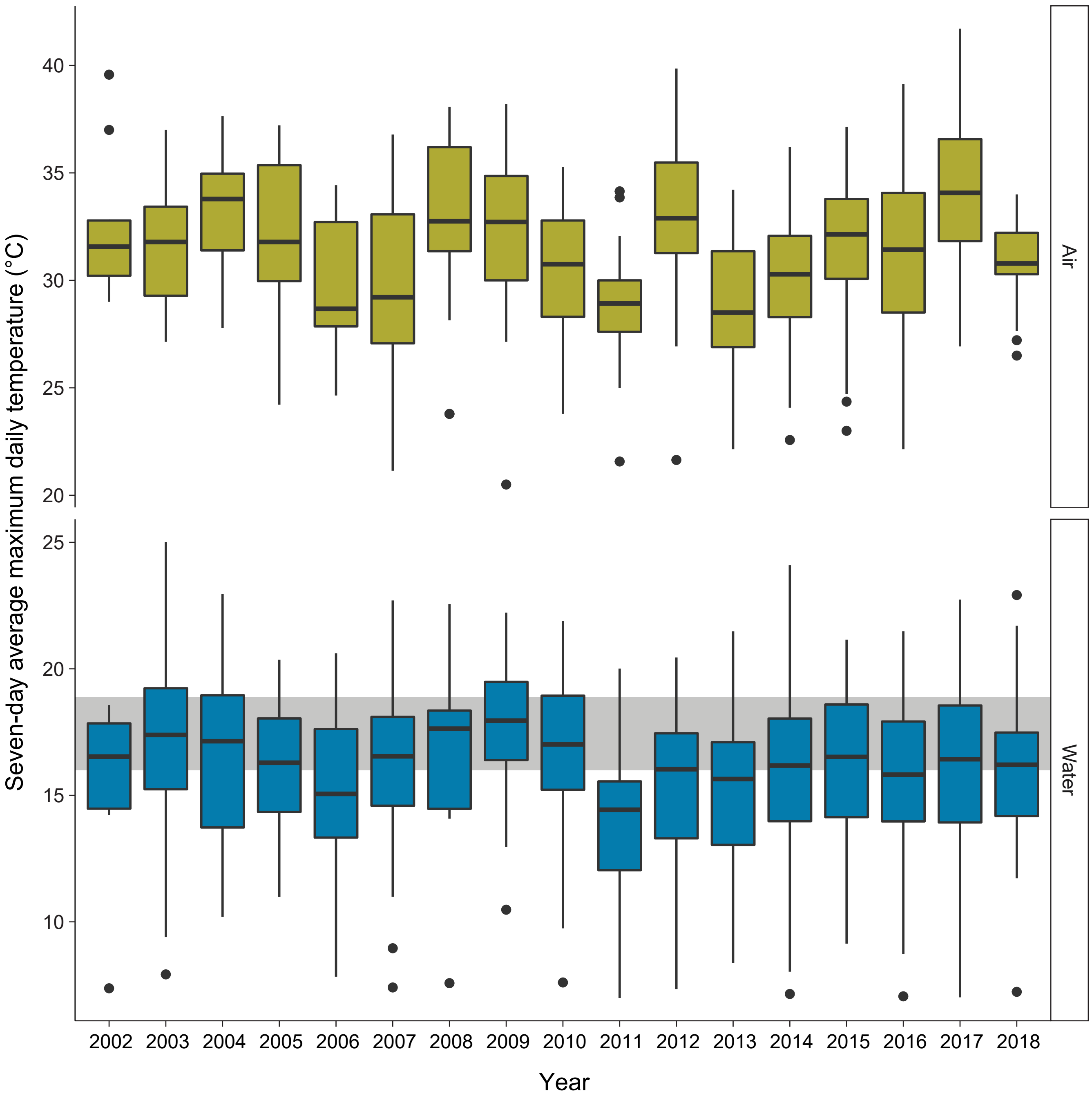

Figure 3.5—Overall distribution at paired sampling locations of air and water temperatures summarized for the month of August across subwatersheds sampled by the Aquatic and Riparian Effectiveness Monitoring Program, 2002–2018. Air temperatures are summarized from DayMet (Thornton et al. 2016). Air and water temperatures are summarized as the warmest week for average maximum daily water temperature (°C) in August for each year. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated.

tend to have the warmest August 7-day average maximum daily water temperature. These temperatures were variable, with some provinces exhibiting frequent 7-day average maximum daily water temperatures above the thresholds of 16–18 °C, and others with much cooler conditions. The High and Western Cascades provinces exhibited the most internal variability (fig. 3.6).

Key Watersheds and Land Use Allocations—

Key watersheds—

Box plots indicate similar thermal patterns between key and non-key watersheds across the entire dataset, or when key and non-key watersheds were grouped by aquatic provinces (fig. 3.7).

Land use allocations—

Subwatersheds with predominantly LSRs frequently had warmer August 7-day average maximum daily water temperatures than congressional reserve subwatersheds (fig. 3.8) within the same aquatic province.

Discussion

Estimated canopy cover within RMAs changed less than 5 percent within the AREMP area from 1993 through 2017. Other studies suggest that the rate and magnitude of forest disturbance has declined since the inception of the NWFP (Kennedy et al. 2012). This overall steady state may suggest that increased wildfire and other disturbance have offset increases from decreased forest harvest, as well as

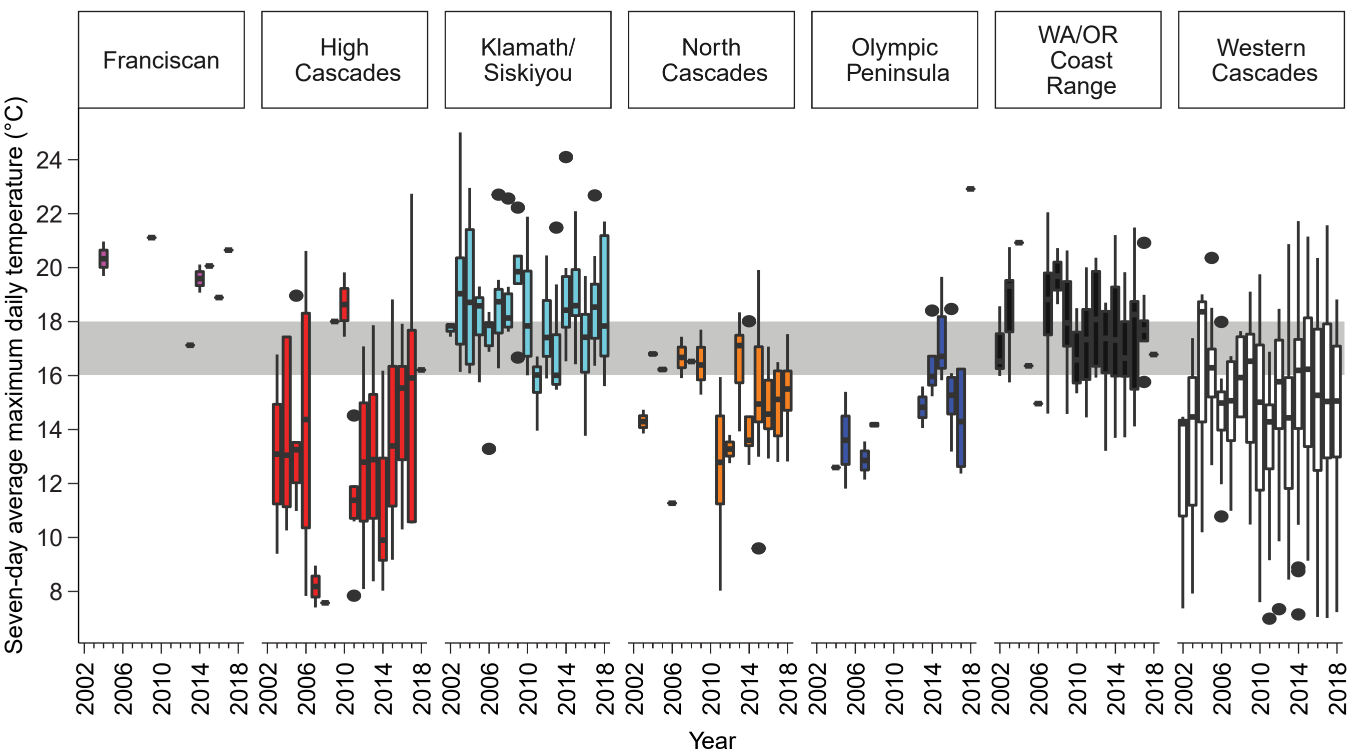

Figure 3.6—August 7-day average maximum daily water temperature (°C) by sites within provinces, 2002–2018. The gray bar highlights 16–18 °C, thermal conditions that have been associated with potentially stressful effects on coldwater fishes. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated.

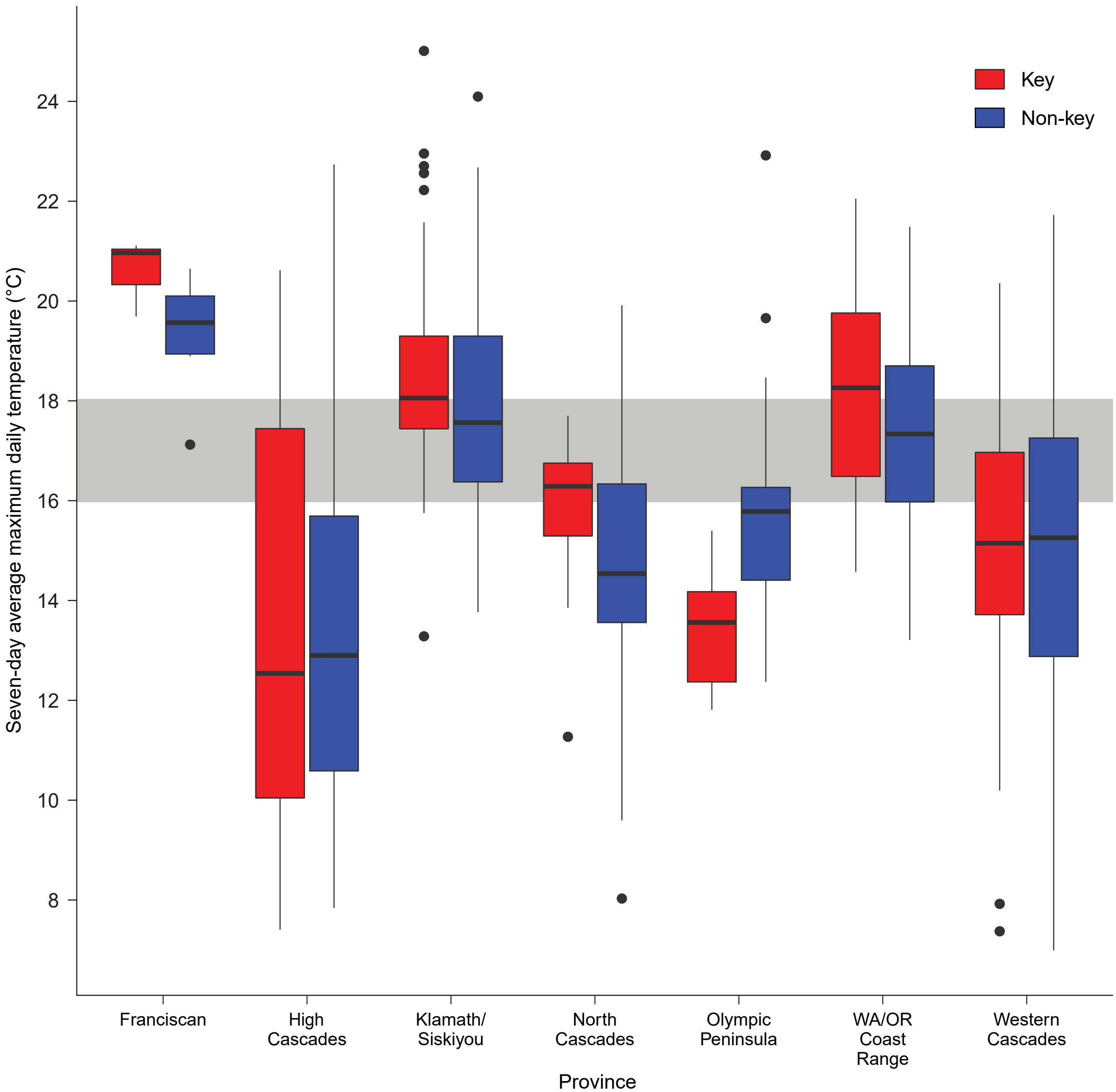

Figure 3.7—August 7-day average maximum daily water temperature (°C) by key and non-key watersheds within aquatic provinces, 2002–2018. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated.

forest growth in previously disturbed areas. The dynamic nature of cover is more apparent at the subwatershed level, where over one-third of the subwatersheds experienced a >5-percent change (positive or negative) in cover. We traced the most extreme localized changes in canopy cover within subwatersheds to stand-replacing wildfire and to a lesser extent forest harvest. Afforestation was most evident in previously disturbed areas or as emergence of forest cover in areas that were not forested at the start of the NWFP. At intermediate extents (provinces, key watersheds, and LUAs), changes in canopy cover were relatively minor (<5 percent). Such findings were not unanticipated, given the

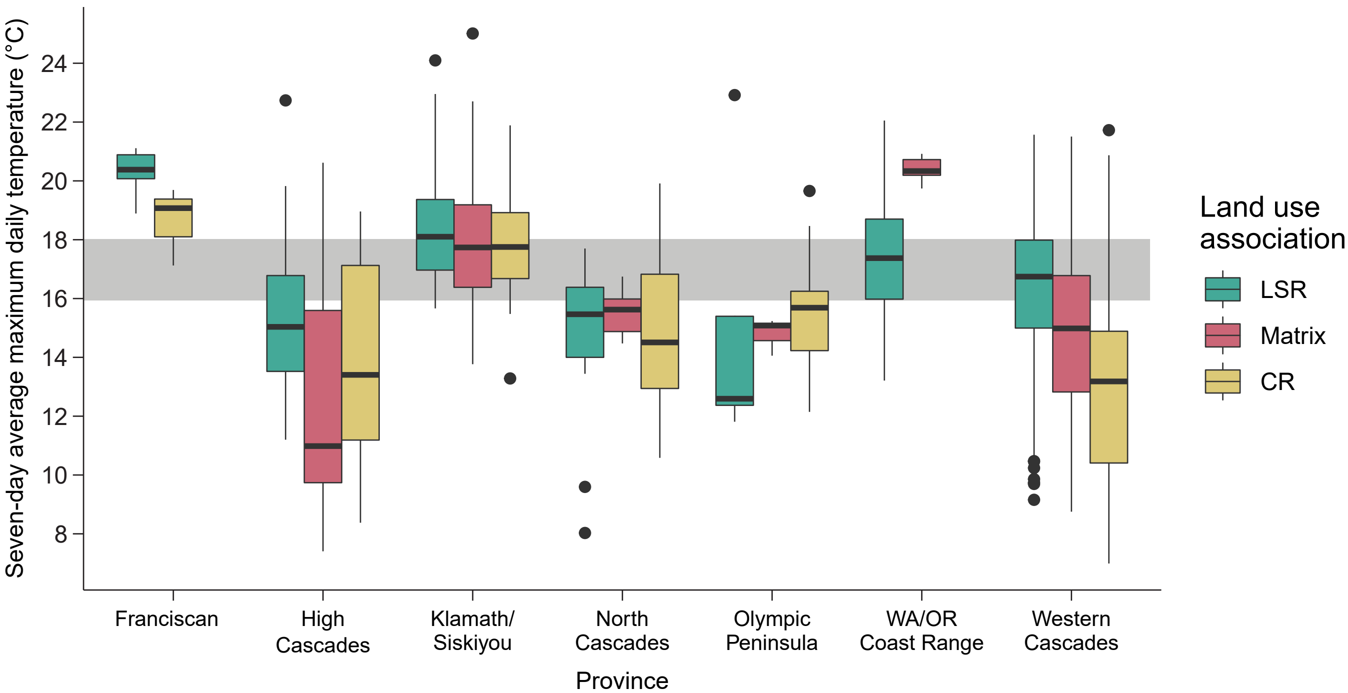

Figure 3.8—August 7-day average maximum daily water temperature (°C) by dominant federal land use allocation within aquatic provinces, 2002–2018. Medians (solid horizontal line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. CR = congressional reserve, LSR = late-successional reserve.

slow pace of forest recovery and influences of episodic processes that can accelerate loss or gain of forest cover in more localized cases (Davis et al. 2022).

Major temporal changes in temperatures do not seem to be evident, although longer time series (preferably 30 or more years) would be needed to evaluate annual trends rigorously (Arismendi et al. 2012, Isaak et al. 2016). Where temperatures are warmer within the NWFP, they can exceed levels suitable for coldwater salmon, trout, or other species. The most discernible pattern is that stream temperatures are warmer in warmer climates, such as northern California and western Oregon. The observed difference in temperature between LSR and congressional reserve land use allocations is potentially related to the differences in elevation of these areas (see fig. A1.7). Changes within RMAs were variable so we did not expect a consistent change or recovery in stream temperatures, especially considering limitations in the spatial and temporal distribution of observations. The major challenge ahead lies in attributing patterns of stream temperature to changing forest conditions, climate-related attributes, or other influences. It can be difficult to assign cause and effect in observational studies of stream temperature, even with detailed information on local processes that influence heat budgets (see "Temperature" in app. 5). Over time, local series of temperature data collected by the AREMP should prove invaluable for tracking management and climate-related changes, but additional work is needed to attribute local changes more rigorously in both canopy cover and stream temperature.

Chapter 4: Forest Conditions and Instream Large Wood

The condition of riparian and upslope forests may be described through assessment of a variety of characteristics, including forest stand structure, species diversity, age, and tree density (Davis et al. 2022). These elements of forest condition can strongly influence nearly all characteristics of streams, especially within riparian zones (Naiman et al. 2010, Vannote et al. 1980), but upslope areas as well (Naiman 1992, Wohl et al. 2019). The quantity, size, and location of large trees are of particular interest for watershed conditions. These trees in upslope and riparian forests potentially contribute large wood to streams, which in turn influence a full range of processes within streams, ranging from biogeochemical processes to creating habitats for fish, amphibians, and other species (Naiman et al. 2010, Swanson et al. 2020, Wohl et al. 2019). Here we consider old-growth structure, streamside large trees per hectare, and field assessment of instream wood. Field-measured densities of instream large wood are summarized across subwatersheds for instream responses. Forest condition indicators of old-growth structure index (OGSI) at 80 years and density of large trees per hectare are summarized within riparian management areas (RMAs) in subwatersheds across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area (see app. 1).

Old-Growth Structure Index

We chose the OGSI as defined by Davis et al. (2022) as an overall indicator of the potential availability of instream large wood. OGSI is based on the density of large live trees, diversity of live-tree size classes, density of large snags, and percentage of downed woody material cover, with specific thresholds for different forest types. Davis et al. (2022) summarized characteristics for old-growth forests at 80, 120, 160, and 200 years. We reasoned that the 80-year time horizon was most useful in tracking stand development due to the relatively short (25-year) time period of the NWFP to date and that trees can reach the size classes typically measured for instream wood by this age. Forests included in the OGSI 80 classification are a minimum of 80 years of age, but can also include those that are older in age or stand development. The OGSI 80 characteristic in RMAs was summarized in the year prior to NWFP implementation (1993) and for the last year of available vegetation data (2017) (see "Forest Characteristics" in app. 3 for additional methods). We used a minimum change threshold of 5 percent in OGSI 80 when reporting individual subwatershed scale changes.

Overall and by Province

Overall—

An overall increase in percentage of OGSI 80 in RMAs was apparent from 1993 (57 percent) to 2017 (61 percent) across the AREMP area (n = 1,972 subwatersheds), a net gain of 1,058 km². Eighty-two percent of the area remained stable between the two time periods (51 percent as OGSI 80 and 31 percent below the OGSI 80 threshold). Eighteen percent of the area changed condition: 11 percent developing into OGSI 80 and 7 percent losing OGSI 80 characteristics (fig. 4.1). Within individual subwatersheds, the percentage of OGSI 80 forest cover area within RMAs ranged widely (from 0 to 98 percent) across both time periods, 1993 and 2017 (fig. 4.2). Changes in OGSI 80 area by individual subwatershed between 1993 and 2017 ranged from a loss of 40 percent in the Rancherie Creek-Illinois River subwatershed in southwest Oregon, which was burned in the 2002 Biscuit Fire, to a gain of 56 percent in the Turner Creek subwatershed, a few alternating polygons of USDI Bureau of Land Management (BLM) and nonfederal lands in northwest Oregon. Out of the 1,972 subwatersheds assessed, 870 saw ≥5-percent increases in OGSI 80, while 204 saw ≥5-percent declines in OGSI 80.

◄ Aquatic and Riparian Effectiveness Monitoring Program crew member with an aggregation of instream wood at Hackleman Creek in Willamette National Forest, Oregon. Photo courtesy of Klynn Shelton.

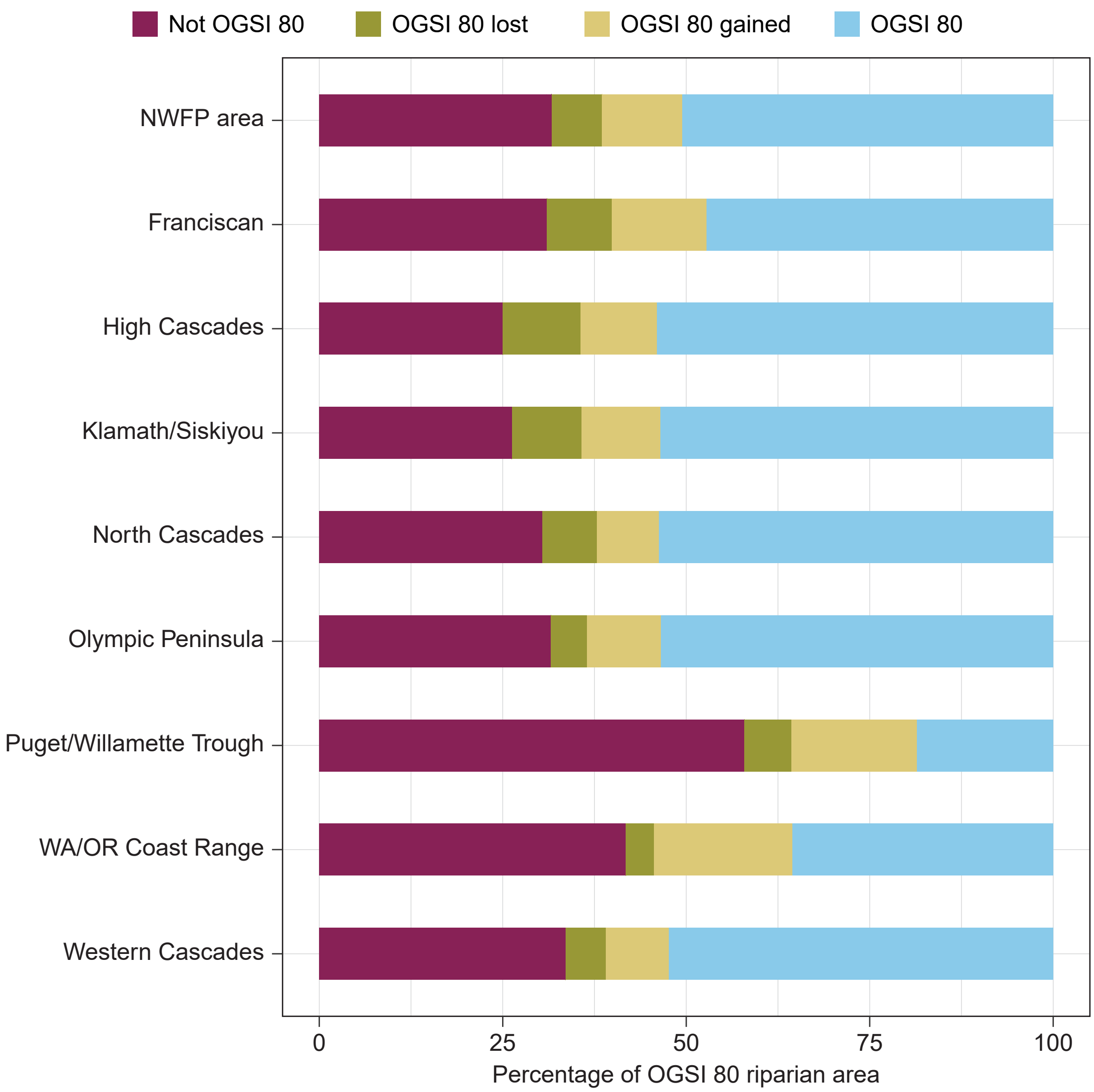

Figure 4.1—Change in percentage of federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) by aquatic province and the overall Northwest Forest Plan (NWFP) area. Change is assessed from 1993 through 2017.

Province—

The fraction of RMAs classified as OGSI 80 varied somewhat by province, but mean values were in the range of 50–60 percent among provinces between 1993 and 2017. Lower values occurred in the Puget-Willamette Trough (25–36 percent) and Washington-Oregon Coast Range (39 percent in 1993). The largest overall change in OGSI 80 was found in the Washington-Oregon Coast Range, with a 15-percent increase in the mean value (from 39 percent to 54 percent). Additionally, net increases were found in the Puget-Willamette Trough (11 percent) and Olympic Peninsula (6 percent) provinces. Changes in the other provinces were generally positive but smaller (0–4 percent) (fig. 4.3). Many of the overall changes were relatively small, but they reflect the differences between larger percentages of OGSI 80 gained and lost during the monitoring period. Following the changes cited above, the Washington-Oregon Coast Range lost the least (4 percent) and gained the most (19 percent); however, even provinces experiencing little net change saw larger losses and gains, e.g., the North Cascades 1-percent change was actually comprised of a 7-percent loss and 8-percent gain (fig. 4.3).

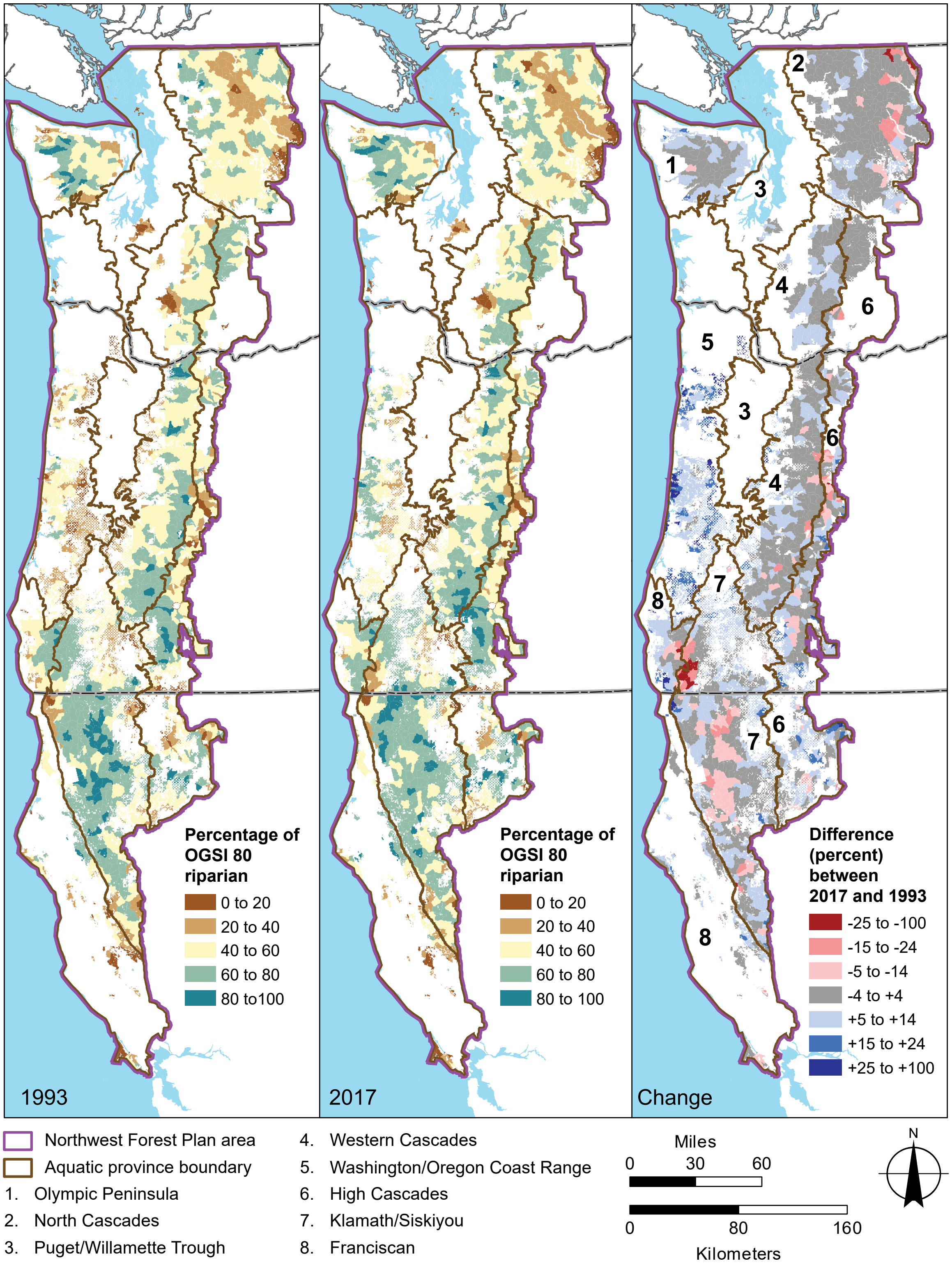

Figure 4.2—Percentage of federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) in the Aquatic and Riparian Effectiveness Monitoring Program area for 1993, 2017, and the difference between the two time periods.

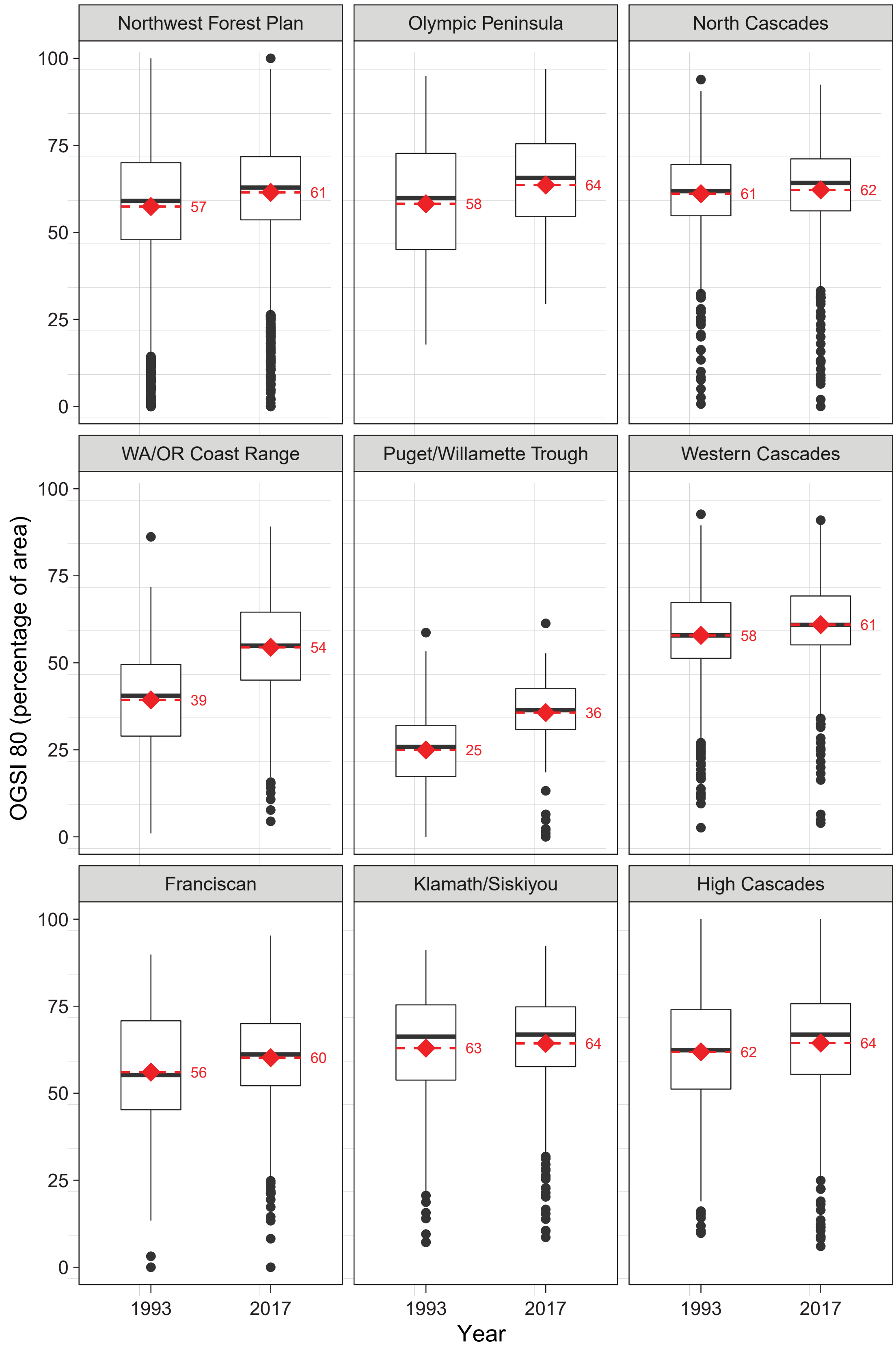

Figure 4.3—Percentage of the federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) by aquatic province and the overall Northwest Forest Plan for 1993 and 2017. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

Key Watersheds and Land Use Allocations

Key watersheds—

The overall mean percentage of the RMAs meeting the OGSI 80 criteria was higher in key watersheds (62 percent in 1993 and 64 percent in 2017) compared to non-key watersheds (55 percent in 1993 and 60 percent in 2017) (fig. 4.4). A larger increase was seen in non-key (5 percent) versus key watersheds (2 percent). These overall changes were the result of both gains and losses. Over the monitoring period, 10 percent of the key watershed area developed into OGSI 80, and 7 percent of the starting OGSI 80 was lost. In the non-key watershed area, the gain was 12 percent and the loss was also 7 percent (fig. 4.5).

Land use allocations—

The mean percentage of OGSI 80 area in RMAs was highest in the congressionally reserved LUA, which remained stable over the monitoring period (65 percent). The LSR LUA had the second highest values and experienced a 5-percent increase from 1993 (59 percent) to 2017 (64 percent). The percentage of OGSI 80 area was lowest in the matrix LUA, but it also showed a gain similar to LSR (from 49 percent to 55 percent) (fig. 4.6). OGSI 80 classifications associated with matrix and LSR LUAs showed losses (7 and 6 percent, respectively) and gains (13 and 11 percent, respectively) for a similar net increase (5–6 percent), whereas congressionally reserved lands remained stable (8-percent loss, 8-percent gain) (fig. 4.7).

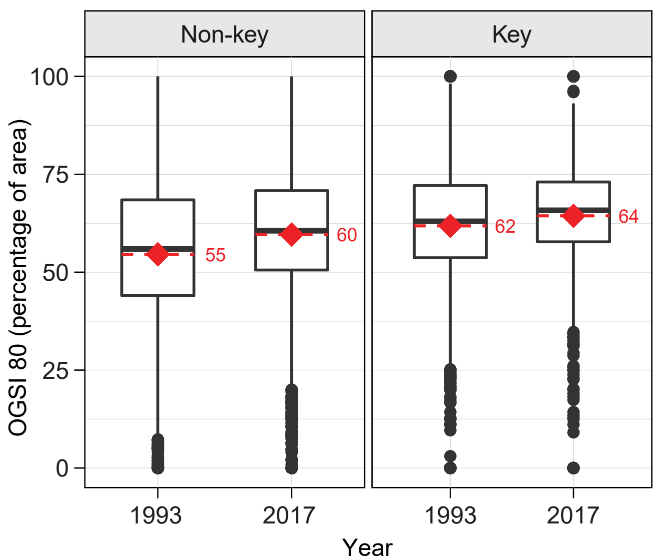

Figure 4.4—Percentage of the federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) for key and non-key watersheds for 1993 and 2017. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

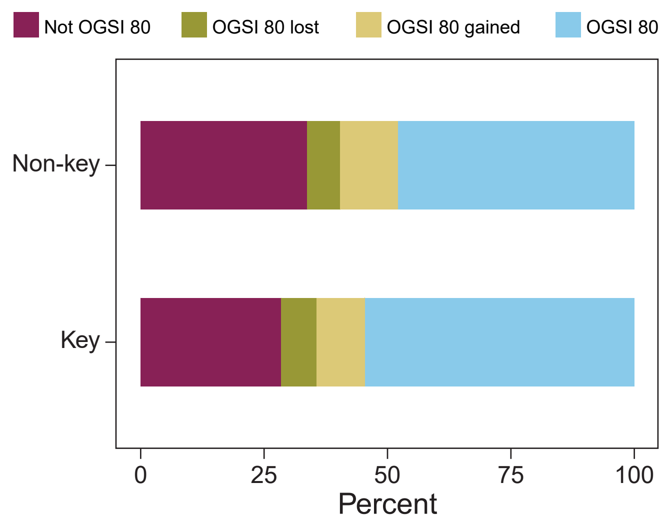

Figure 4.5—Change in percentage of federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) by key/non-key watersheds between 1993 and 2017.

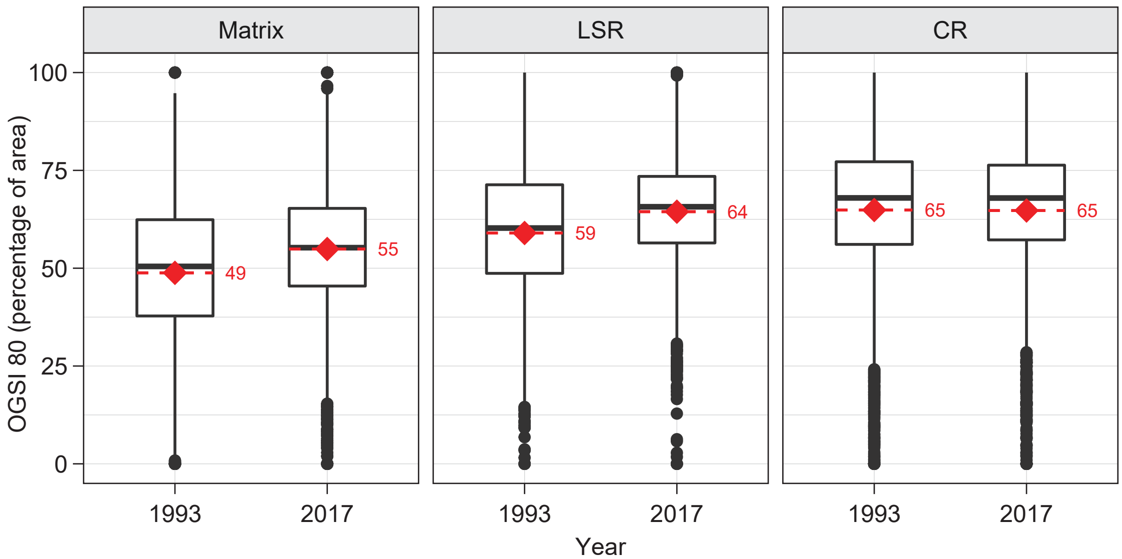

Figure 4.6—Percentage of the federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) by land use allocation in 1993 and 2017. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

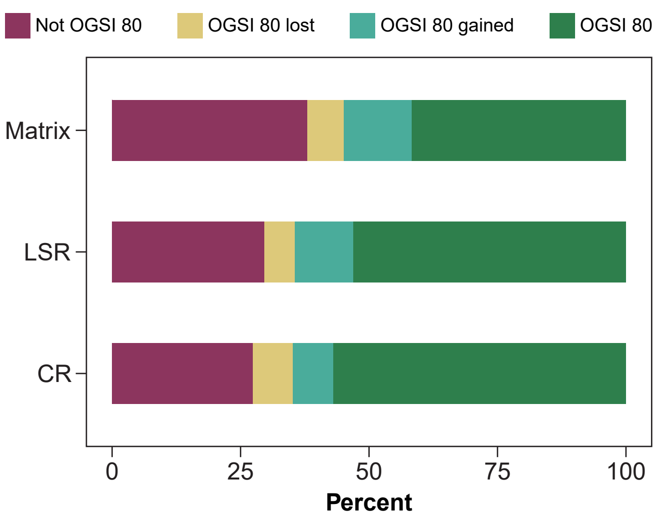

Figure 4.7—Change in percentage of federally managed riparian area meeting the old-growth structural index 80-year threshold (OGSI 80) by land use allocation between 1993 and 2017. CR = congressional reserve, LSR = late-successional reserve.

Density of Large Trees Near Streams

Quantifying the density of large trees near streams can provide an understanding of potential wood inputs from adjacent stream banks (see “Large Wood” in app. 5). The density (trees per hectare) of large trees (≥50 cm diameter at breast height [d.b.h.] of either conifer or hardwood) within 20 m (on each bank) of a fish-bearing stream was calculated and compared between 1993 and 2017 (for additional methods, refer to app. 3, “Forest Characteristics”). Changes in riparian density of large trees were summarized across the AREMP area, among provinces within the area, and across designated key watersheds and land use allocations. We used a minimum change threshold of five or more in the density of large trees per hectare when reporting individual subwatershed-scale changes.

Overall and by Province

Overall—

Across the AREMP area, the average density of large trees near fish-bearing streams increased from 1993 to 2017 (44.5 trees/ha to 46.3 trees/ha, respectively) (fig. 4.8). Out of the 1,972 subwatersheds assessed, 248 did not have fish-bearing streams and thus were not evaluated. For fish-bearing streams, 423 saw increases of ≥5 trees/ha, while 175 saw declines of ≥5 trees/ha (fig. 4.9). Within individual subwatersheds, mean values varied from 1 to 126 trees/ha. Changes in trees per hectare by individual subwatershed between 1993 and 2017 ranged from a loss of 45 trees/ha in Klondike Creek subwatershed in southwest Oregon, which was burned in the 2002 Biscuit Fire, to a gain of

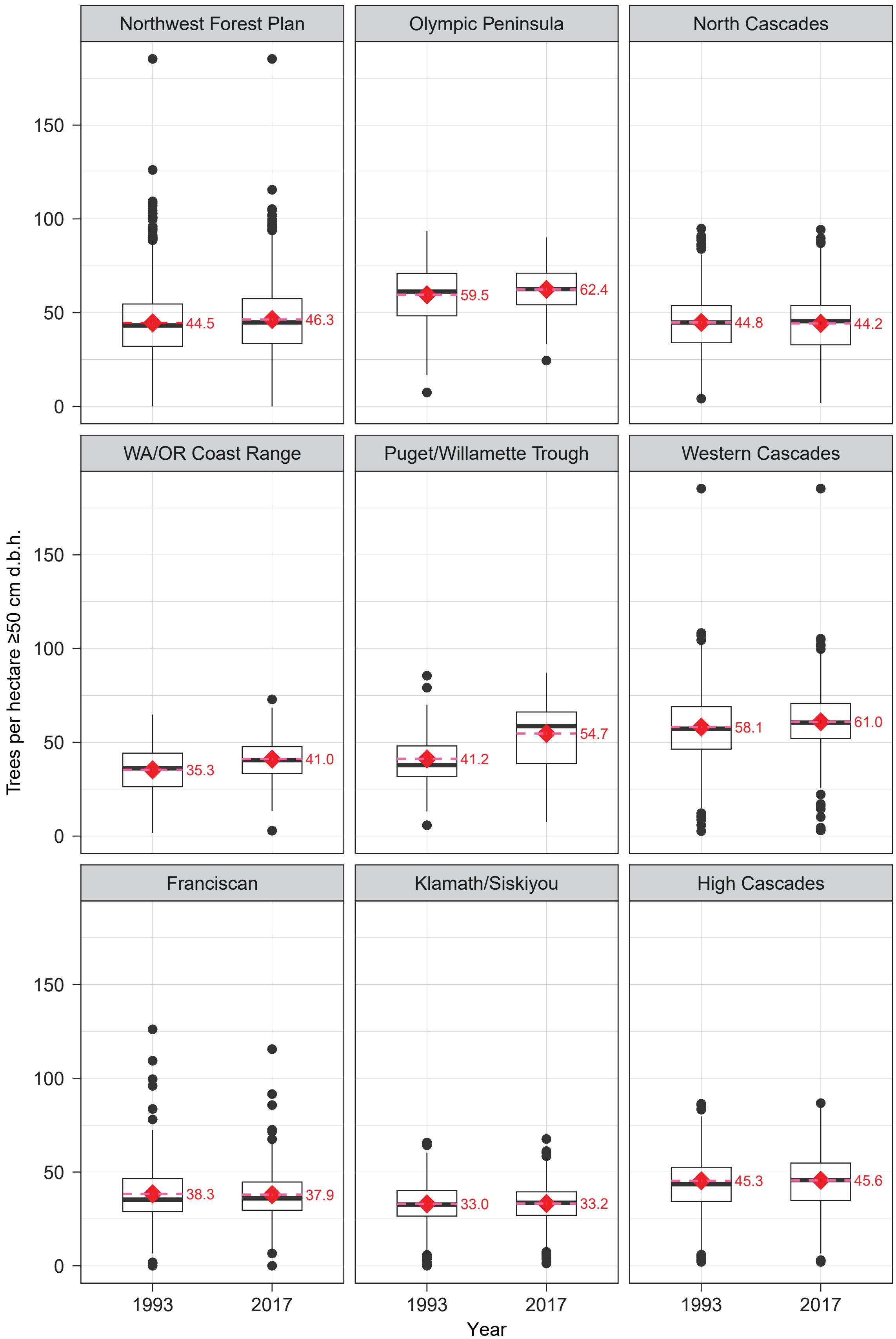

Figure 4.8—Average number of trees per hectare ≥50 cm diameter at breast height (d.b.h.) in a 20-m buffer on fish-bearing streams by aquatic province and overall Northwest Forest Plan for the 1993 and 2017 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

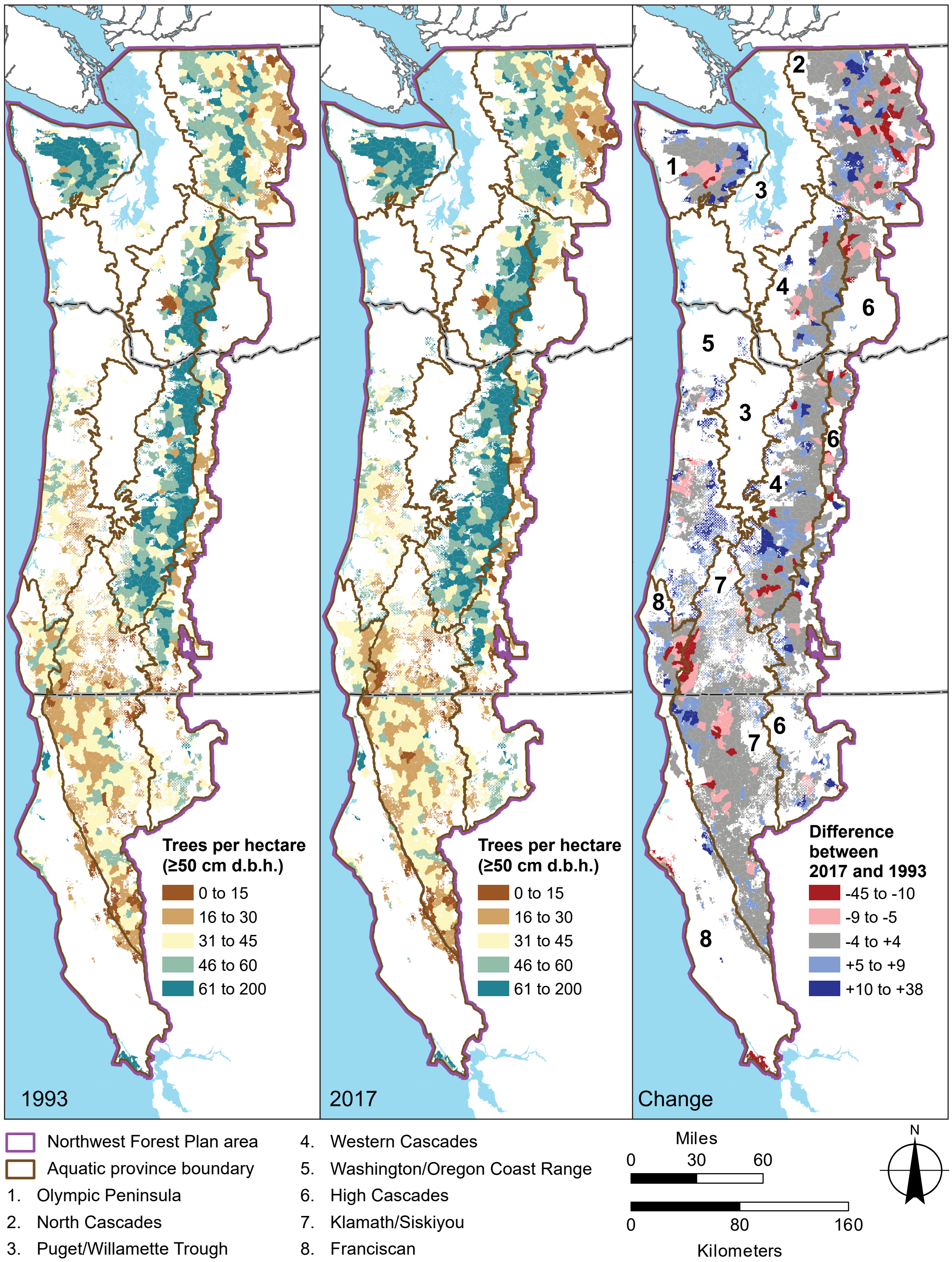

Figure 4.9—Average number of trees per hectare ≥50 cm diameter at breast height (d.b.h.) within 20-m of a fish-bearing stream by subwatershed for 1993, 2017, and the difference between the two time periods.

38 trees/ha in the Lower Willamina Creek subwatershed, a small amount of alternating polygons of BLM and nonfederal lands in the Oregon Coast Range province near the border of the Willamette Valley.

Province—

Although overall changes in the average density of large trees across the AREMP area were limited, the differences in responses (changes from 1993 to 2017) were more pronounced among provinces. The greatest increase in density of large riparian trees near fish-bearing streams was found in the Puget-Willamette Trough (+13.5 large trees/ha). However, <1 percent of the area analyzed fell in this province. The Washington-Oregon Coast Range had the second largest increase (+5.7 trees/ha), followed by the Olympic Peninsula and Western Cascades (+2.9 trees/ha each). These latter two provinces also had the highest absolute mean densities of large riparian trees (60–62 trees/ha). The remaining provinces experienced losses and gains of <1.0 tree/ha. Maps of individual subwatersheds indicate continued large declines in the density of large trees in the Klamath-Siskiyou and Franciscan provinces from the Biscuit Fire (2002). Similarly, occurrence of wildfire within the timeframe of the NWFP was associated with declines in the density of large trees in subwatersheds in eastside forests of the North and High Cascades provinces (fig. 4.9).

Key Watersheds and Land Use Allocations

Key watersheds—

In 1993, the average number of large trees near fish-bearing streams was higher in key watersheds (46.7 trees/ha) than in non-key watersheds (43.0 trees/ha). Large-tree density in key watersheds increased by 0.4 trees/ha, compared with a 2.8-trees/ha increase in non-key watersheds, reducing the difference in 2017 mean values to 47.1 trees/ha in key and 45.8 trees/ha in non-key watersheds (fig. 4.10).

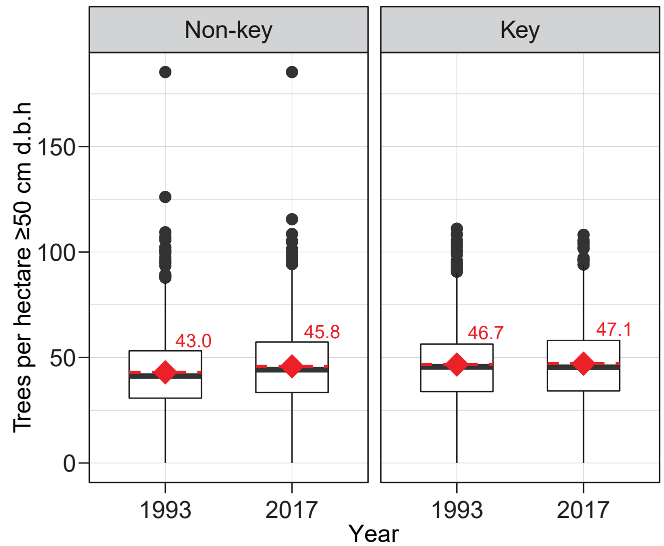

Figure 4.10—Average number of trees ≥50 cm diameter at breast height (d.b.h.) in a 20-m buffer on fish-bearing streams by key and non-key watersheds for the 1993 and 2017 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

Land use allocations—

At the beginning of the monitoring period, the highest mean density of large trees near fish-bearing streams was found in the congressionally reserved lands (49.0 trees/ha), followed by LSRs (45.2) and matrix (39.7) (fig. 4.11). In terms of trend over the monitoring period, this order was reversed with the largest gain occurring in matrix lands (+4.5 trees/ha), followed by LSRs (+1.8); there was a decrease (-1.3 trees/ha) in congressionally reserved lands. Mean values in 2017 followed the same initial pattern but were closer, with congressionally reserved lands (47.7 trees/ha) followed by LSRs (47.0) and matrix (44.2).

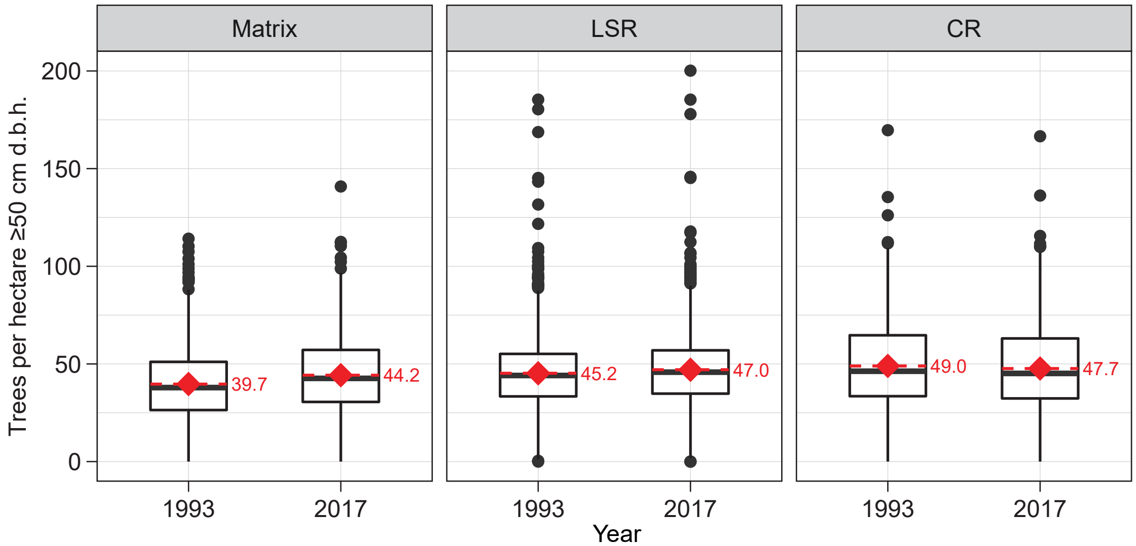

Figure 4.11—Average number of trees ≥50 cm diameter at breast height (d.b.h.) in a 20-m buffer on fish-bearing streams by land use allocation for the 1993 and 2017 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

Instream wood

Instream wood was surveyed at sample sites nested within randomly selected subwatersheds from 2002 until 2018 (see "Instream Wood" in app. 2 for field methods). Trends in the density of instream wood size classes relate to the complexity of many ecosystem processes and habitat for native aquatic species. Wood size classes described in this section (table 4.1) were summarized across the AREMP area, within ecological provinces, and for comparison between key and non-key watersheds, as well as LUAs (see app. 2 for analytical approaches). We considered trends in three wood size class densities (number of pieces per 100 m) from 2002 through 2018 (app. 2). Subwatershed-level trends by administrative unit for instream wood in size classes B, C, and D can be found in appendix 6. Size classes used here represent evenly spaced categories for ease of interpretation. Other classifications of instream wood (e.g., size of wood relative to stream size) may be considered in future efforts to consider the importance of supply and transport of wood throughout networks (app. 5). Complementary summaries of wood (e.g., total volume versus pieces, as summarized here) may also be useful in future efforts.

Overall and by Province

Overall—

Across the sampled sites in the AREMP area, overall survey results indicate that the density of wood pieces per length of stream consistently declined for the large size category, but they remained relatively constant for the intermediate and in some cases increased for the small size category (fig. 4.12). Across the AREMP area, the overall

Table 4.1—Definition of size classes for large wood using length and diameter of individual pieces counted

Size class	Minimum diameter	Maximum diameter	Minimum length
Large (D)	≥61.0 cm (24 inches)	NA	≥7.6 m (25 ft)
Intermediate (C)	≥45.7 cm (18 inches)	<61.0 cm (24 inches)	≥7.6 m (25 ft)
Small (B)	≥30.5 cm (12 inches)	<45.7 cm (18 inches)	≥7.6 m (25 ft)

NA = not applicable.

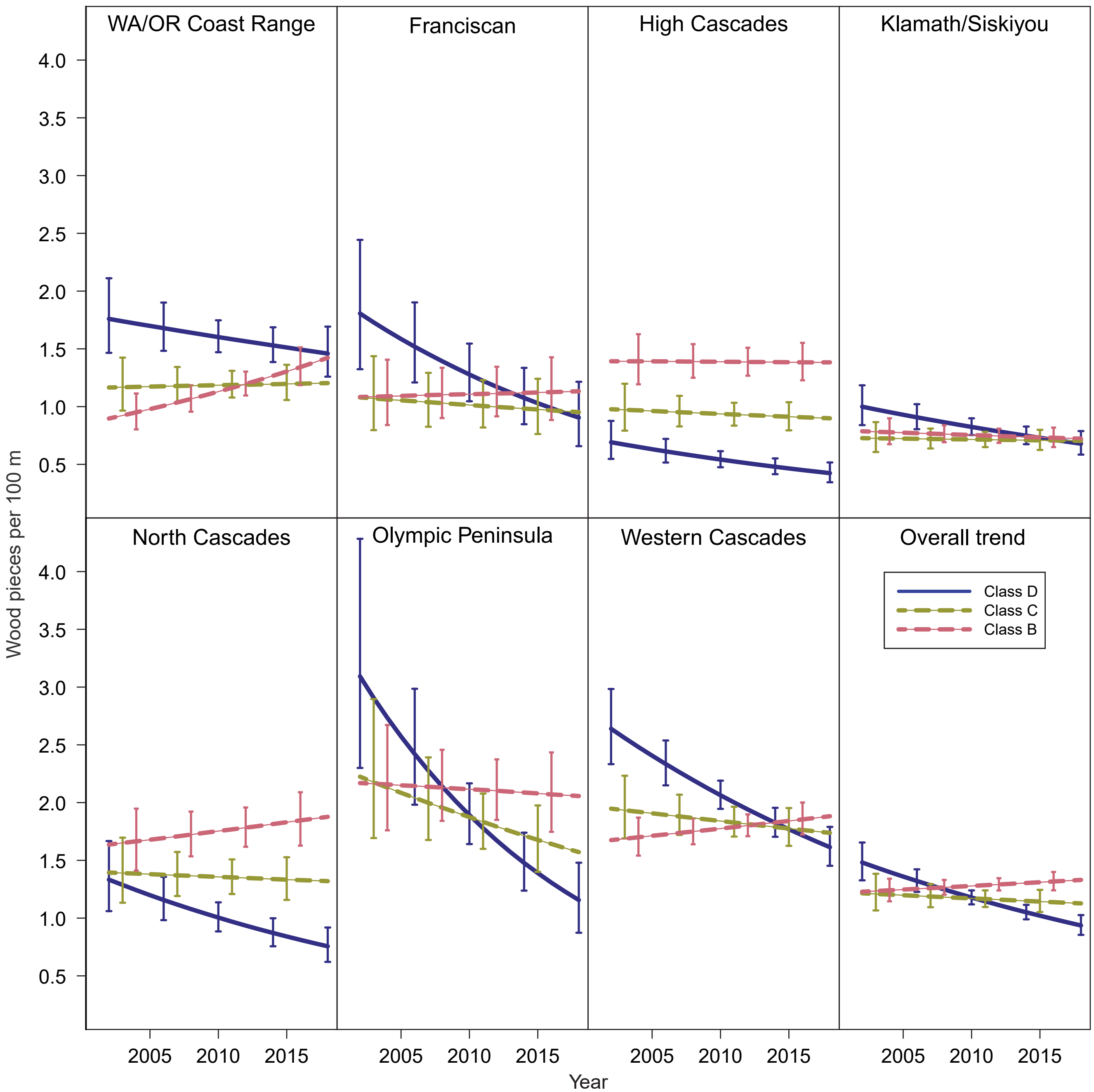

Figure 4.12—Trends in wood density (number of pieces per 100 m of stream) within size categories (see table 4.1) by aquatic province and monitored overall by the Aquatic and Riparian Effectiveness Monitoring Program for 2002–2018. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals.

trend in wood piece density was -24.9 percent per decade (95-percent credibility intervals from -32.6 to -16.4 percent), and 83 percent of subwatershed-level trends were negative (95-percent credibility intervals from 71 to 95 percent) for the largest size class D. These declines in wood density per length of stream contrasted with the small size class B.

There was weak probabilistic support for a positive trend in size class B (small size class) with an overall trend of +5.2 percent per decade (95-percent credibility interval from -4.0 to +15.3 percent), and 57 percent (95-percent credibility intervals from 42 to 73 percent) of subwatersheds showed positive trends in density in the smallest size class B.

Province—

All aquatic provinces experienced declines in density of the largest pieces of wood (class D) (fig. 4.12). For densities of the two smaller size classes of instream wood, provinces varied in terms of showing increases or decreases. For example, the Olympic Peninsula experienced declines in the intermediate size (class C) wood, while increases in the density of the smallest size category (class B) were evident in the Washington-Oregon Coast Range.

Key Watersheds and Land Use Allocations

Key watersheds—

Key watersheds exhibited greater density of instream wood than non-key watersheds for all size classes (fig. 4.13). Key and non-key watersheds followed the overall trends for size classes B, C, and D. Pronounced declines of 27 percent per decade in key watersheds (95-percent credibility intervals from -36 to -19 percent) and 23 percent per decade (95-percent credibility intervals from -32 to -14 percent) in non-key watersheds were observed in the largest size class D (fig. 4.13).

Land use allocation—

Differences were observed in density of instream wood and trends among LUAs. Within the larger wood size class (class D), LSRs harbored greater overall density of large wood in streams, whereas wood density in matrix and congressional reserves were similar and lower overall (fig. 4.13). Streams in congressional reserves had higher densities of instream wood in the smallest size class (class B). Within each LUA, the trend paralleled overall declining trends in density (number of pieces per 100 m) for the largest size (class D) of instream wood. Declines in density of larger sizes of wood (class D) were consistent across LUAs, with a 24-percent per decade decrease in LSRs, a 27-percent decrease in matrix, and a 29-percent decrease in congressional reserves (95-percent credibility intervals, respectively: -32 to -14 percent, -37 to -17 percent, and -39 to -17 percent). Changes over time were not evident for smaller size classes of large wood (classes B and C) in LUAs except for an increasing trend in the smallest size (class B) in LSRs.

Discussion

Trends in the density (pieces per 100 m) of instream wood indicate spatially variable patterns for smaller size classes of large wood (classes B and C) and consistent loss of larger pieces (class D) in streams across the AREMP area. This may be expected as historical (pre-NWFP) forest harvest near streams and landslide-prone areas plus active removals from stream channels across much of the AREMP area likely reduced large wood available for recruitment and retention (see "Large Wood" in app. 5). Alternatively, in congressional reserves where timber harvest is uncommon, disturbance events that deliver wood to streams such as wildfire have been suppressed (Moritz et al. 2014), potentially reducing natural recruitment from such disturbances for most of the past century.

Regeneration and recruitment of the largest size class of instream wood (i.e., ≥61.0 cm diameter) is expected to take centuries when left to passive forest recovery. Projects to add large wood to stream channels can provide a limited, immediate, local-scale solution (Benda et al. 2016, Jones et al. 2014, Roni et al. 2015). Recruitment of instream wood depends on local processes that are difficult to account for at larger scales. Recruitment may be attributed to episodic influences of wildfires or other disturbances that have variably occurred on the landscape during the 25 years since NWFP implementation. Overall, trends indicating losses of larger instream wood are consistent with the concept of slow attrition of pieces derived from older, larger trees recruited to streams prior to widespread forest harvest in the region and subsequent protections imposed in accordance with the NWFP.

Densities of larger (class D) instream wood were lower in congressional reserve streams relative to LSRs. This result may be expected if larger trees are less available to recruit to stream channels in congressional reserves. These areas are often higher elevation national parks and wilderness areas within the AREMP area (see fig. A1.7) where tree sizes may be smaller or trees are not broadly distributed (Davis et al. 2022, Waring and Franklin 1979). This explanation is consistent with the observation of higher densities of smaller (class B) instream wood in congressional reserve streams. More detailed analyses of past and present forest cover in the context of natural

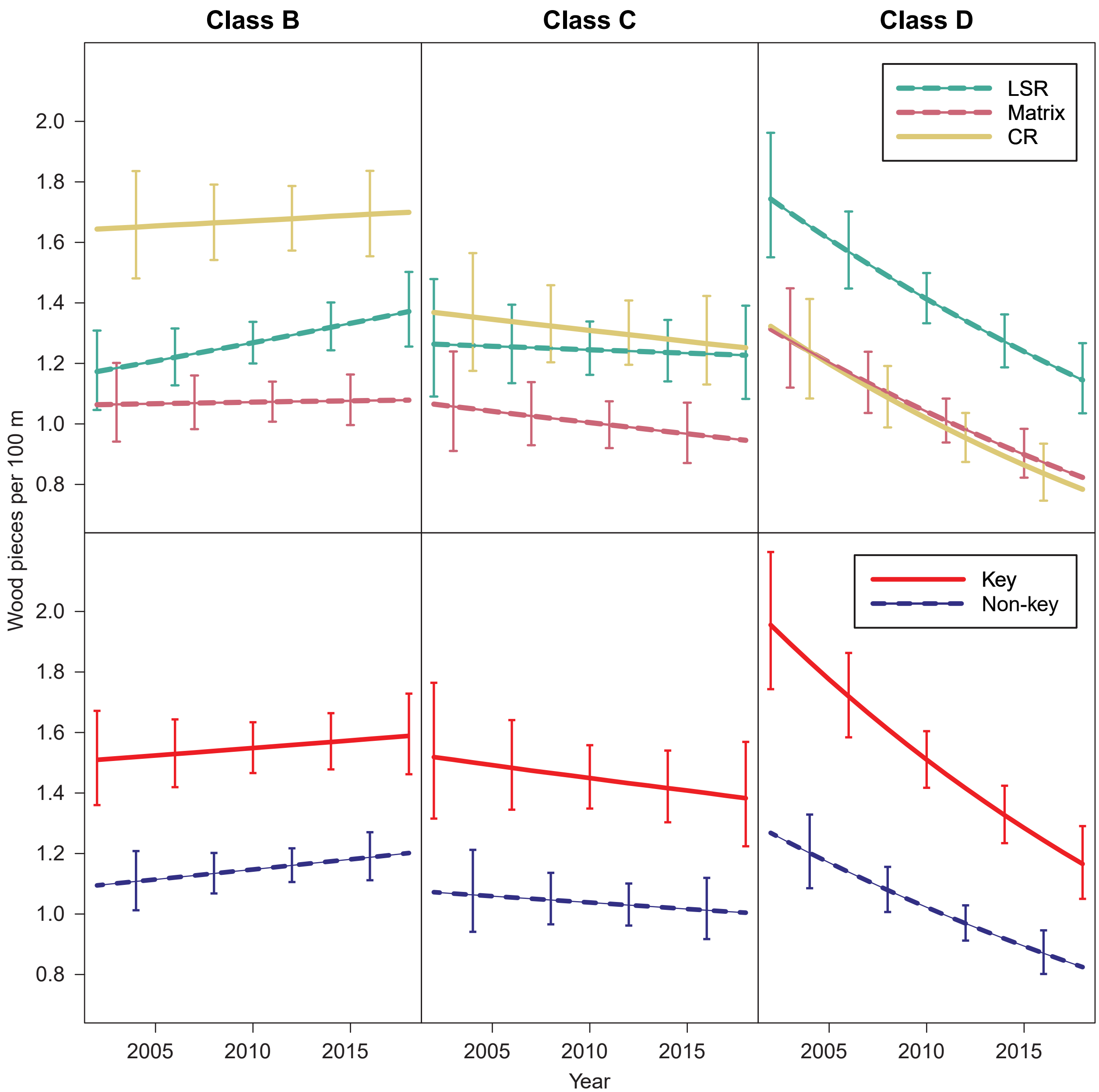

Figure 4.13—Trends in wood density (number of pieces per 100 m of stream) within size categories (see table 4.1) by key and non-key watershed and by land use allocation monitored by the Aquatic and Riparian Effectiveness Monitoring Program in 2002–2018. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals. LSR = late-successional reserve, CR = congressional reserve.

disturbance and legacies from historical removal of wood in streams (e.g., historical "stream cleaning" to remove large wood and log drives; Miller 2010) would be needed to more fully account for patterns observed here. A major challenge in addressing this question is lack of consistent documentation of historical activities that removed instream wood across the AREMP area.

Instream wood densities were significantly higher in key watersheds than in non-key watersheds. Key watersheds also had higher percentages of old-growth forest (OGSI 80) and more trees ≥50 cm d.b.h. next to fish-bearing streams. This finding is expected and consistent with the original criteria for identifying key watersheds, which were selected to serve as refuges for aquatic organisms, representing areas with the best existing or potential watershed condition (FEMAT 1993, Haynes et al. 2006).

Trends in availability of large wood within RMAs as indicated by development of OGSI 80 characteristics and presence of large trees near streams are also consistent with the slow dynamic of recovery expected for Pacific Northwest forests and consequences for instream wood. Continued losses of large wood may exceed gains from recruitment to streams into the future while local forests mature to larger sizes (Martens et al. 2020). Wildfire or other local disturbance events may be associated with changes in availability of riparian sources of wood (Davis et al. 2022). Modest increases in development of old-growth characteristics and presence of large trees appear to be higher in locations that likely experienced greater pressure from forest harvest before NWFP implementation. In other words, forests in areas with a history of more intensive forest harvest likely had more to gain (or conversely less to lose) in terms of development of stand characteristics considered herein (see Davis et al. 2022 for details on the development of large and old-growth characteristics in the NWFP area). Environmental factors, particularly precipitation and wildfire, also correlate respectively with the increases seen in riparian forests in the wetter, west-side provinces versus the more mixed losses and gains of the drier, east-side provinces.


SIMMS

Chapter 5: Roads, Landslide Risk, and Instream Fine Sediment

Roads are an essential component of the transportation infrastructure of federal lands, but they can also cause undesirable impacts to aquatic ecosystems (Black et al. 2012, Gucinski et al. 2001, Trombulak and Frissell 2000). Among the many potential direct and indirect effects of forest roads, one of the most prominent concerns is delivery of sediment to streams (see "Fine Sediment" in app. 5). The presence of some fine sediment along the channel bed is expected and benefits some species, such as native lamprey (Jones et al. 2020), but excess fine sediment deposition can be detrimental, such as when it reduces salmon egg-to-fry survival by clogging spawning gravels (Kemp et al. 2011). Accordingly, we quantified how the design and location of roads within watersheds could potentially influence sediment delivery. Road designs were evaluated using a GIS-based tool that estimates sediment delivered by roads to streams (see "Road Sediment Modeling" in app. 3). We also considered where roads were located on landscapes relative to their susceptibility to mass wasting via shallow landslides (see "Road and Slope Stability" in app. 3). Trends were evaluated using roads constructed or decommissioned since the start of the Northwest Forest Plan (NWFP) . Additionally, we present results from instream surveys of streambed sediment and particle size collected by the Aquatic and Riparian Effectiveness Monitoring Program (AREMP).

Although erosion from roads and infrequent events such as mass wasting can contribute fine sediment to streams, observed levels of fine sediment on streambeds may not simply correlate with these indicators (Al-Chokhachy et al. 2016), in part because most headwater streams have substantial capacity to transport fines. Ultimately, what is observed on the streambed is a product of numerous processes that can be difficult to quantify, and our intent here is not to produce a complete sediment budget (see "Fine Sediment" in app. 5), but rather to consider potential sources of sediment to streams, how they have changed over the course of the NWFP, and how observed fine sediment on streambeds has changed.

◀ An Aquatic and Riparian Effectiveness Monitoring Program crewmember checks a measurement of instream substrate in Quartzville Creek headwaters on Willamette National Forest. Photo courtesy of Morgan Holland.

Roads and Chronic Sediment

Sediment delivery to streams from road systems on federal land was estimated using the GRAIP Lite tool (Nelson et al. 2019) on the 1993 and 2019 road systems to evaluate relative changes since the NWFP was initiated. Estimated sediment delivery was summarized for each of the 12-digit hydrologic unit codes (HUC12) with at least 5 percent federal land within the AREMP area in 1993 and 2019, and the changes were calculated. Only roads on federal lands were assessed. Specific sediment delivery (the mass of sediment delivered to streams per unit area of subwatershed) was calculated by dividing the estimated sediment delivery values by the subwatershed area to account for subwatersheds of varying size (see "Road Sediment Modeling" in app. 3 for additional methods). Note that although absolute values of estimated sediment delivery are reported here, they should be interpreted as relative values because a constant base rate of erosion was applied across the AREMP area and rates were not adjusted for local geology, climate, or other influences (app. 3). Density of connected road length (i.e., estimated road length capable of delivering sediment to streams) was also calculated and reported here. We used a minimum change threshold of 0.5 Mg/yr/km^2 for sediment delivery when reporting individual subwatershed-scale changes. It is important to note that the processes tracked here are linked to roads and potential sediment delivery, and that there are a host of other processes that conspire to influence realized sediment budgets in streams (app. 5), including influences of roads on surface flow paths and runoff to streams (Kastridis 2020).

Overall and by Province

Overall—

Road decommissioning across the AREMP area has reduced the road length on federal lands by 8854 km (7.1 percent) since 1993. Connected road length declined by 1608 km (a 6.6-percent reduction) and the estimated sediment delivery (Mg/yr) was reduced by 4 percent based on these calculations. Subwatershed-scale estimates per unit-area for sediment delivery ranged from 0 to 12.7 Mg/yr/km² with a decrease in mean sediment delivery from 1.9 Mg/yr/km² for 1993 to 1.8 Mg/yr/km² for 2019 (fig. 5.1). Subwatershed values per unit-area for connected road length ranged from 0 to 1.6 km/km² (fig. 5.2). Connected road length mean was 0.24 km/km² for 1993 and decreased to 0.23 km/km² for 2019.

The majority (96 percent) of subwatersheds showed no change in estimated sediment delivery per km² (<0.5 Mg/yr/km²). However, 75 subwatersheds (4 percent) exhibited a net reduction in estimated sediment delivery, while one subwatershed had an increase (>0.5 Mg/yr/km²). The Susie Creek-Lyre River subwatershed on the Olympic Peninsula demonstrated the largest decrease in estimated sediment delivery (3.40 Mg/yr/km²) through decommissioning of the majority of the road network in the subwatershed (14.6 km).

Province—

Mean estimated sediment delivery and connected road length per area decreased from 1993 to 2019 across all the aquatic provinces. Relative decrease in mean sediment delivery and connected road length ranged from -0.4 to -5.8 percent and from -0.6 to -15.1 percent, respectively (figs. 5.3 and 5.4). By aquatic province, weighted mean sediment delivery was highest in the Klamath-Siskiyou (2.94 Mg/yr/km²) and lowest in the North Cascades (0.96 Mg/yr/km²). The Klamath-Siskiyou showed the largest absolute decrease in mean sediment delivery between 1993 and 2019 (0.15 Mg/yr/km²) and the Olympic Peninsula showed the largest relative decrease (-5.8 percent). The Olympic Peninsula had the lowest length of mean connected road (0.10 km/km²), and Puget-Willamette Trough had the highest connected road length (0.68 km/km²). The Washington-Oregon Coast Range province showed the largest decrease in mean connected road length per area (0.03 km/km²), and the Olympic Peninsula showed the largest relative decrease (-15.1 percent).

Key Watersheds and Land Use Allocations

Key watersheds—

Watersheds in the key class showed lower rates of estimated mean sediment delivery in the recent time period (1.28 Mg/yr/km²) when compared to watersheds in the non-key class (2.06 Mg/yr/km²) (fig. 5.5). Key watersheds showed a larger decrease (-7.5 percent) in mean sediment delivery rates between 1993 and 2019 when compared to the non-key watersheds (-2.7 percent). Key watersheds exhibited a 12-percent reduction in total road length compared to a 5-percent reduction in non-key watersheds. Connected road length per area exhibited a similar pattern to sediment delivery, with key watersheds having a lower mean value (0.19 km/km²) than non-key watersheds (0.25 km/km²) (fig. 5.6).

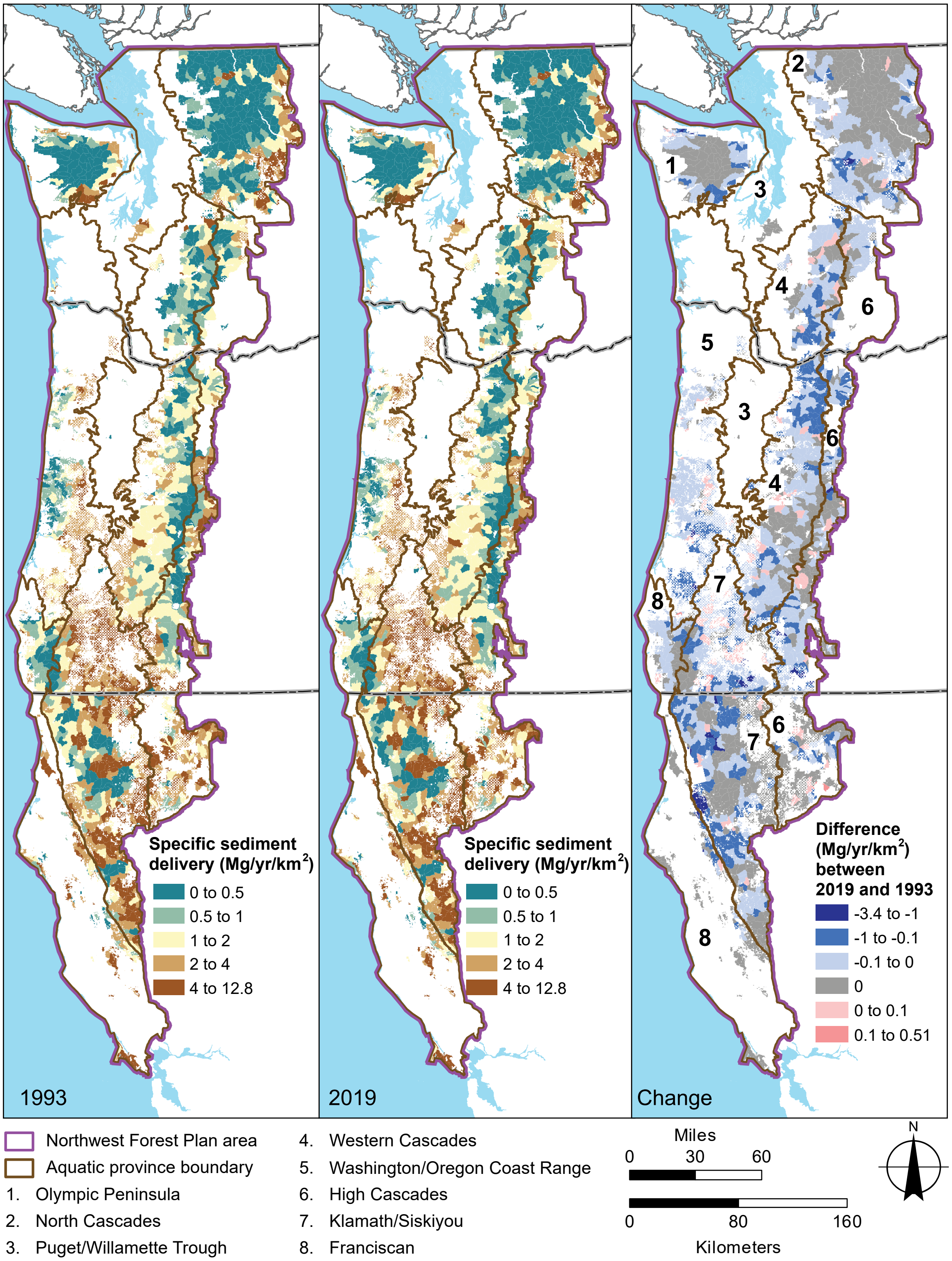

Figure 5.1—Modeled specific sediment delivery from GRAIP Lite summed per subwatershed for 1993, 2019, and the difference between the two time periods.

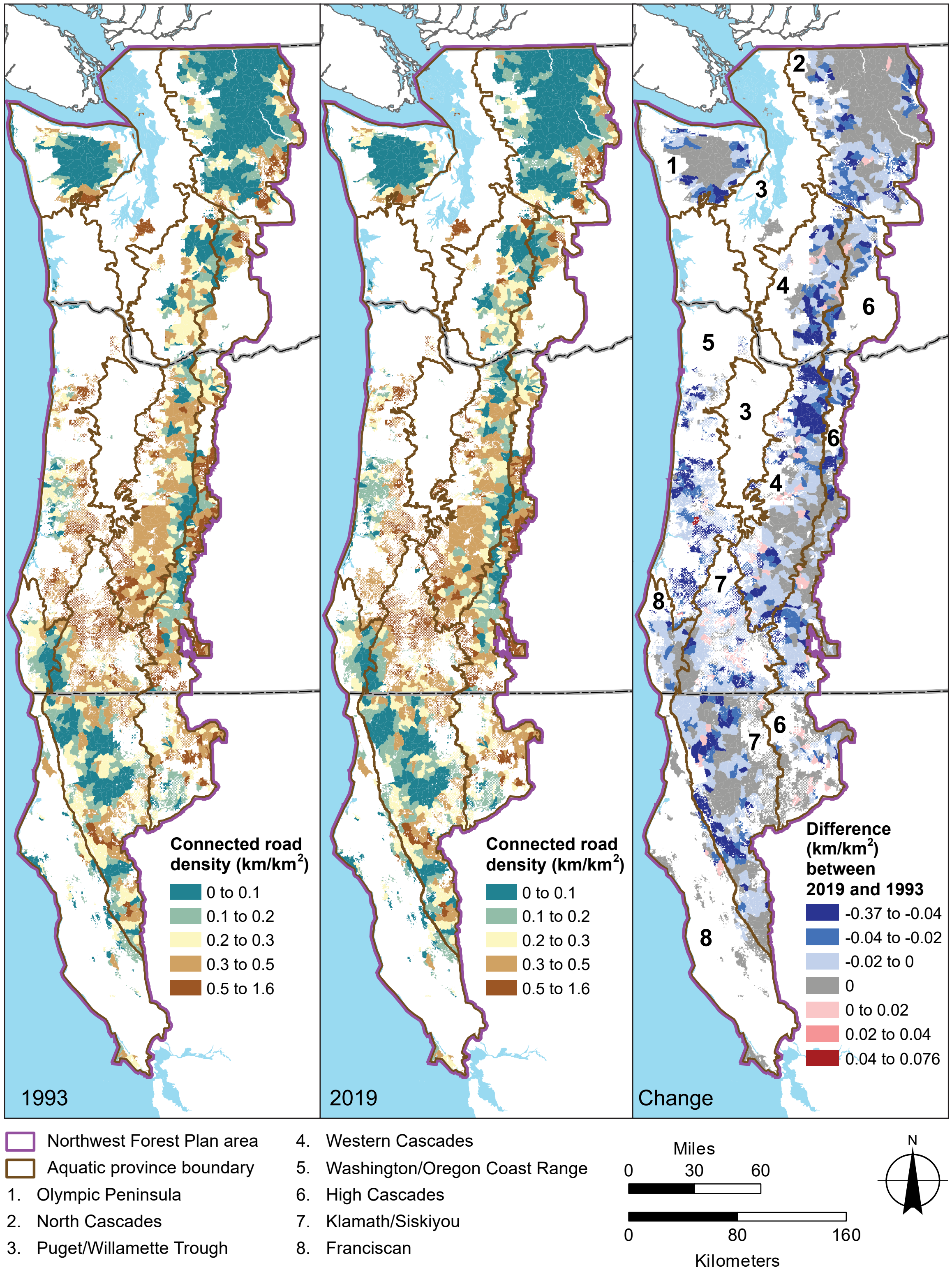

Figure 5.2—Modeled connected road length, weighted (km/km²), from GRAIP Lite, summed per subwatershed for 1993, 2019, and the difference between the two time periods.

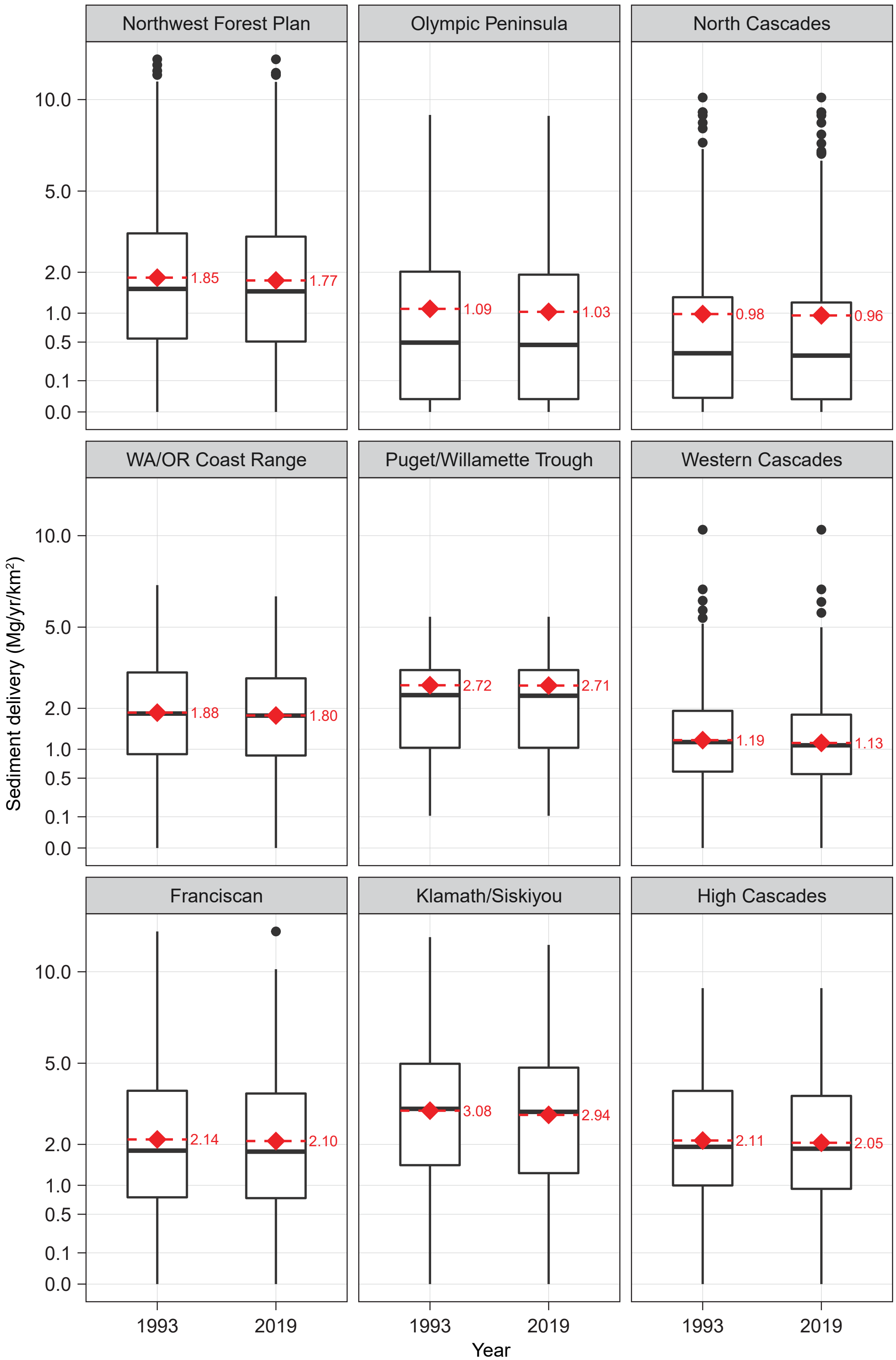

Figure 5.3—Modeled specific sediment delivery (Mg/yr/km²) from GRAIP Lite summed per subwatershed by aquatic province and overall (Northwest Forest Plan area) for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

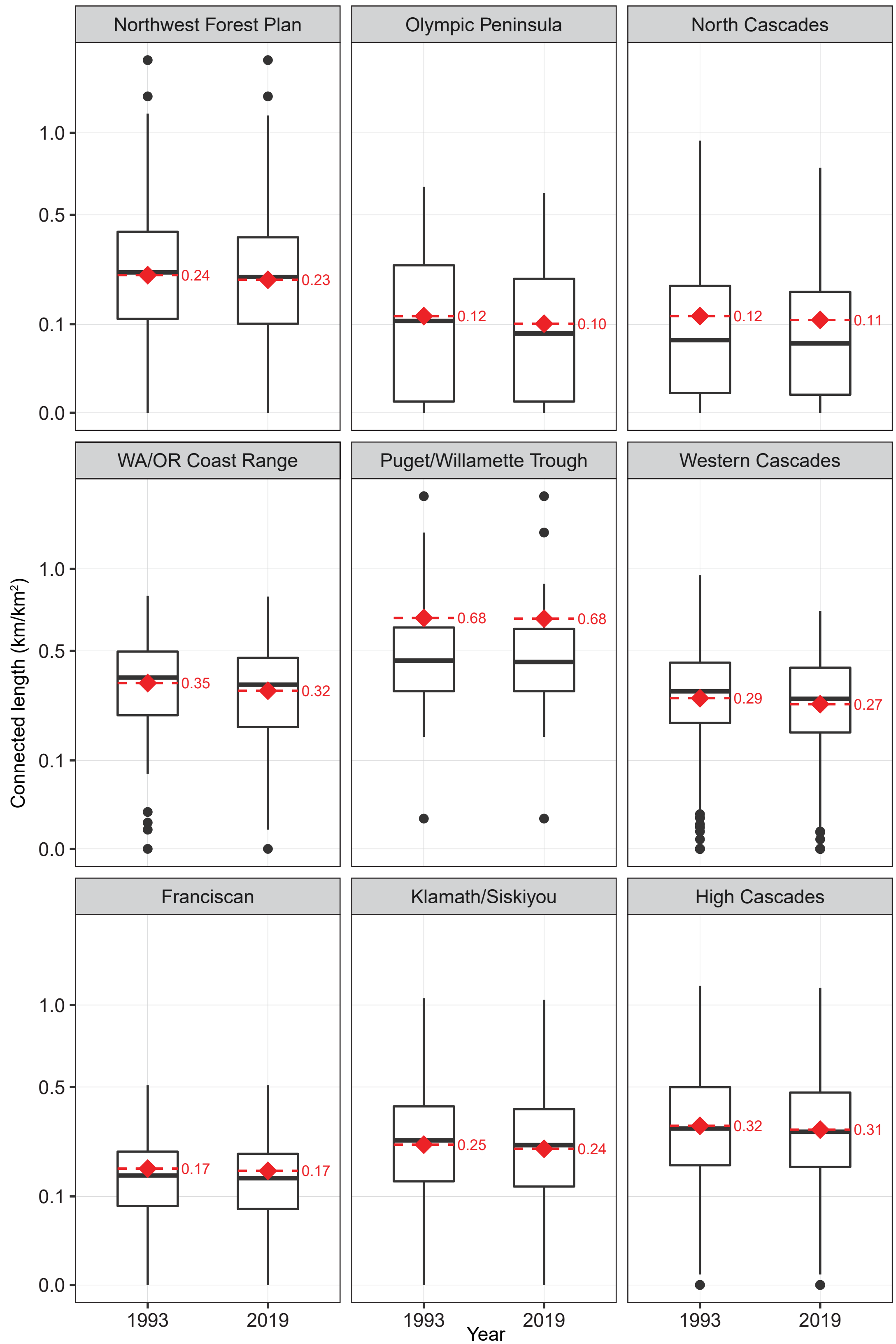

Figure 5.4—Modeled connected road length (km/km²) from GRAIP Lite summed per subwatershed with >5 percent federal land ownership by aquatic province and overall (Northwest Forest Plan area) for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

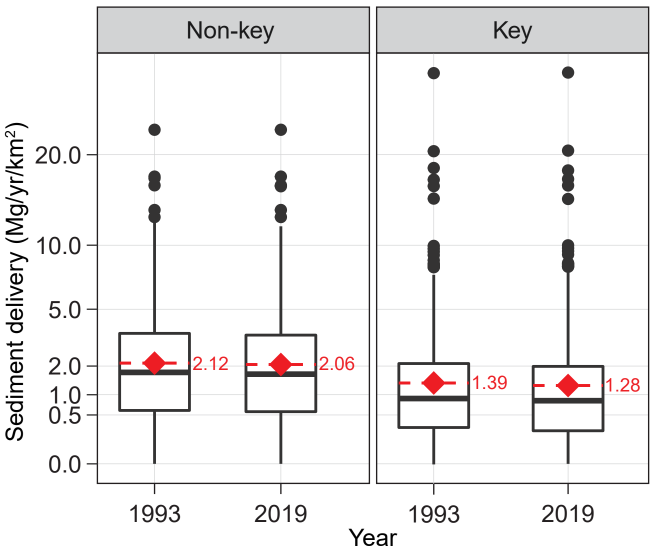

Figure 5.5—Modeled specific sediment delivery (Mg/yr/km²) from GRAIP Lite summed per subwatershed by key or non-key watershed for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

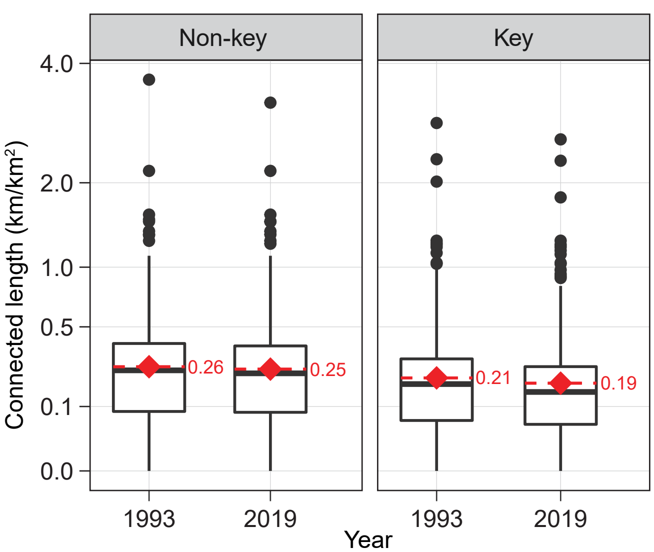

Figure 5.6—Modeled connected road length (km/km²) from GRAIP Lite summed per subwatershed by key or non-key watersheds for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

Land use allocation—

Between LUA groups, congressionally reserved areas had the lowest estimated mean sediment delivery in 2019 (0.50 Mg/yr/km²), and matrix lands the highest (2.95 Mg/yr/km²) in the most recent time period (fig. 5.7). Estimated sediment delivery values decreased in the LSR (-5.4 percent) and remained relatively unchanged (<5 percent) in matrix and congressionally reserved areas. Connected road length decreased in matrix (-5.6 percent) and LSR (-8.4 percent) classes and remained the same in congressionally reserved areas (fig. 5.8).

Roads and Shallow Landslide Risk

Shallow landslides can provide episodic inputs of sediments to streams, especially in landscapes that have high road densities. Roads on topography susceptible to shallow landslide events were assessed using a cohesion factor from the SINMAP (Stability Index MAPping) methodology (Pack et al. 1998, 2005). SINMAP calculates risk based on slope, topographic convergence, soil properties, and wetness. To simplify the analysis over this large area, we provided constants for most of the model parameters and focused on the minimum cohesion factor (C, a dimensionless value that ranges between 0 and 1) necessary to balance instability based on slope, topographic convergence, and contributing drainage area from a 30-m digital elevation model. In other words, subwatersheds with lower overall values for C (normalized by area here, C per km²) are those with road networks that are less susceptible to shallow landslide events. Further details on the analytical process can be found in appendix 3.

The analysis was run on the same roads system dataset used for the chronic sediment analysis above. All HUC12s with at least 5 percent federal land within the AREMP area were included, but overlay areas (HUC12-LUA and HUC12-key watershed) of <5 ha were dropped to reduce small area outliers. C values were summed for each analysis unit and then normalized by area (per km²) to account for watersheds of varying size and percentage of federal ownership. Values were calculated for 1993, 2019, and the difference between these two time periods (fig. 5.9). We used a minimum change threshold of 0.1 C for reporting individual subwatershed-scale changes.

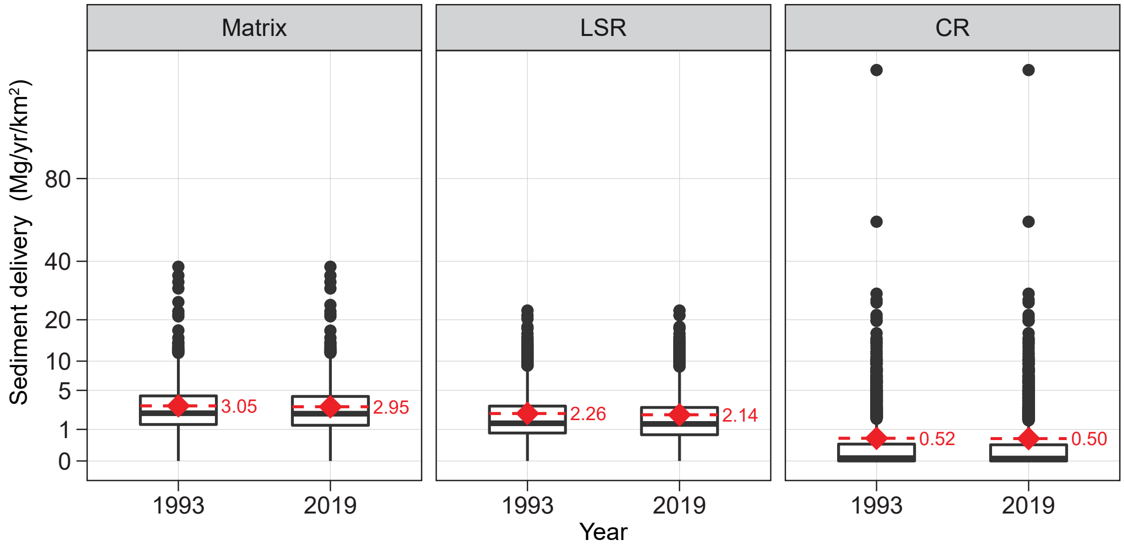

Figure 5.7—Modeled specific sediment delivery (Mg/yr/km²) from GRAIP Lite summed per subwatershed by land use allocation for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

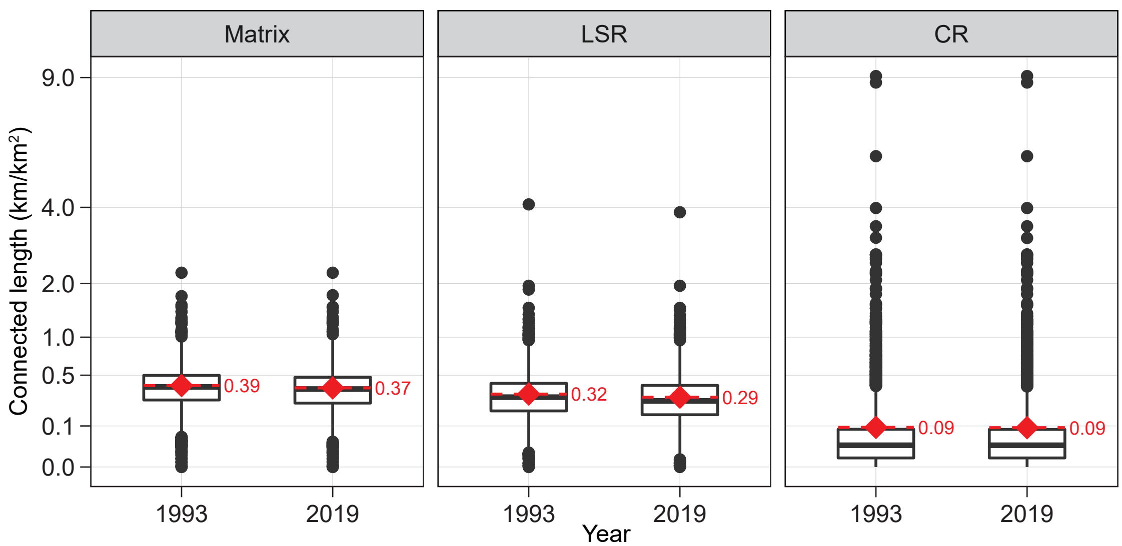

Figure 5.8—Modeled connected road length (km/km²) from GRAIP Lite summed per subwatershed by land use allocation for 1993 and 2019. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

Overall and by Province

Overall—

Subwatershed per km² C values (beginning in 1993) ranged from a low of 0 (133 watersheds) to a high of 4.7 per km² with a mean of 0.38. In the latest data (2019), an additional 7 subwatersheds showed no risk, the maximum risk had declined to 3.5, and average risk declined 11 percent to 0.34 (fig. 5.9). Out of the 1,972 subwatersheds assessed, 575 showed decreases in C (≤-0.1), indicating reduced landslide risk, whereas only 13 saw increases in C (≥0.1) (fig. 5.9). Similar to the chronic sediment results, the Susie Creek-Lyre River subwatershed in the Olympic Peninsula demonstrated the largest decrease in landslide risk (-1.7 C, 100 percent) through decommissioning of the majority of the road network in the subwatershed (14.6 km) (fig. 5.10).

Province—

By aquatic province, the highest mean values (i.e., highest risk) pre-NWFP were found in the Klamath-Siskiyou (0.53), the Western Cascades (0.49), and the Olympic Peninsula (0.49). The Olympic Peninsula achieved the greatest reduction over the monitoring period (-0.12, -24 percent), followed by the North Cascades (-0.06 C, -16 percent), and the Western Cascades (-0.04 C, -9 percent). The lowest values were found in the High Cascades (0.13, no change) and Puget-Willamette Trough (0.09 to 0.07, -13 percent) (fig. 5.11).

Key Watersheds and Land Use Allocations

Key watersheds—

The average shallow landslide risk was higher in key (0.40) than non-key (0.37) watersheds at the start of the assessment period (1993) (fig. 5.12). However, road decommissioning during the assessment period reduced risk in key watersheds by 17 percent (-0.07 C) compared to a 6.5-percent (-0.02 C) reduction in non-key watersheds.

Land use allocation—

In 1993, mean shallow landslide risk by subwatershed was lowest in congressional reserves (0.07), followed by matrix (0.54) and LSR (0.60) lands (fig. 5.13). Over the assessment period, LSRs showed the greatest reduction in risk (-0.09 C, -14 percent) compared to matrix lands (-0.04 C, -8 percent) and congressional reserves (-0.005 C, -8 percent). In 2019, shallow landslide risk in congressional reserves was still the lowest (0.07), but the difference between LSR (0.50) and matrix (0.51) had diminished.

Instream Sediment, Particle Size, and Pool-Tail Fines

Stream sediments reflect hydrogeomorphic characteristics and are directly tied to habitat needs for aquatic biota. Distributions of fine and coarse material reflect broad-scale patterns of underlying geology and disturbance processes, flow regimes, and local characteristics, including stream gradient and width. Characterization of sediment sizes and proportions to quantify trends relevant for management of forested systems requires both rigorous field methodology and carefully constructed statistical relationships. The AREMP collects instream sediment at regularly spaced transects within the bankfull channel and also at pool-tail crests (app. 2). These two measurements are intended to capture all sediments across habitats inclusively (transect) as well as the distribution of sizes specifically in potential salmonid-spawning habitat (pool-tail crest). Measurement in pool-tail crests were recorded at all sites regardless of their likelihood of being used by salmonids. Additionally, we considered trends in percentage of fines (≤2 mm b-axis) and the sizes (b-axis) of substrate particles in the 16th, 50th, and 84th percentiles (see app. 2).

Overall and by Province

Overall—

Overall, fine sediments declined in the transect-based sampling representative of all habitats, and in the pool-based sampling focusing on pool-tail crests. For transect fines, the odds that a sample was classified as fine sediment decreased by 48.6 percent per decade (with 95-percent credibility intervals from -59.2 to -30.5 percent). For pool-tail fines, the odds decreased by 20.4 percent per decade with moderate support (-40.8 to +6.3 percent). Most trends in transect fines within subwatersheds were negative (90 percent of watersheds, with 95-percent credibility intervals from 80 to 96 percent). However, pool-tail fines were more

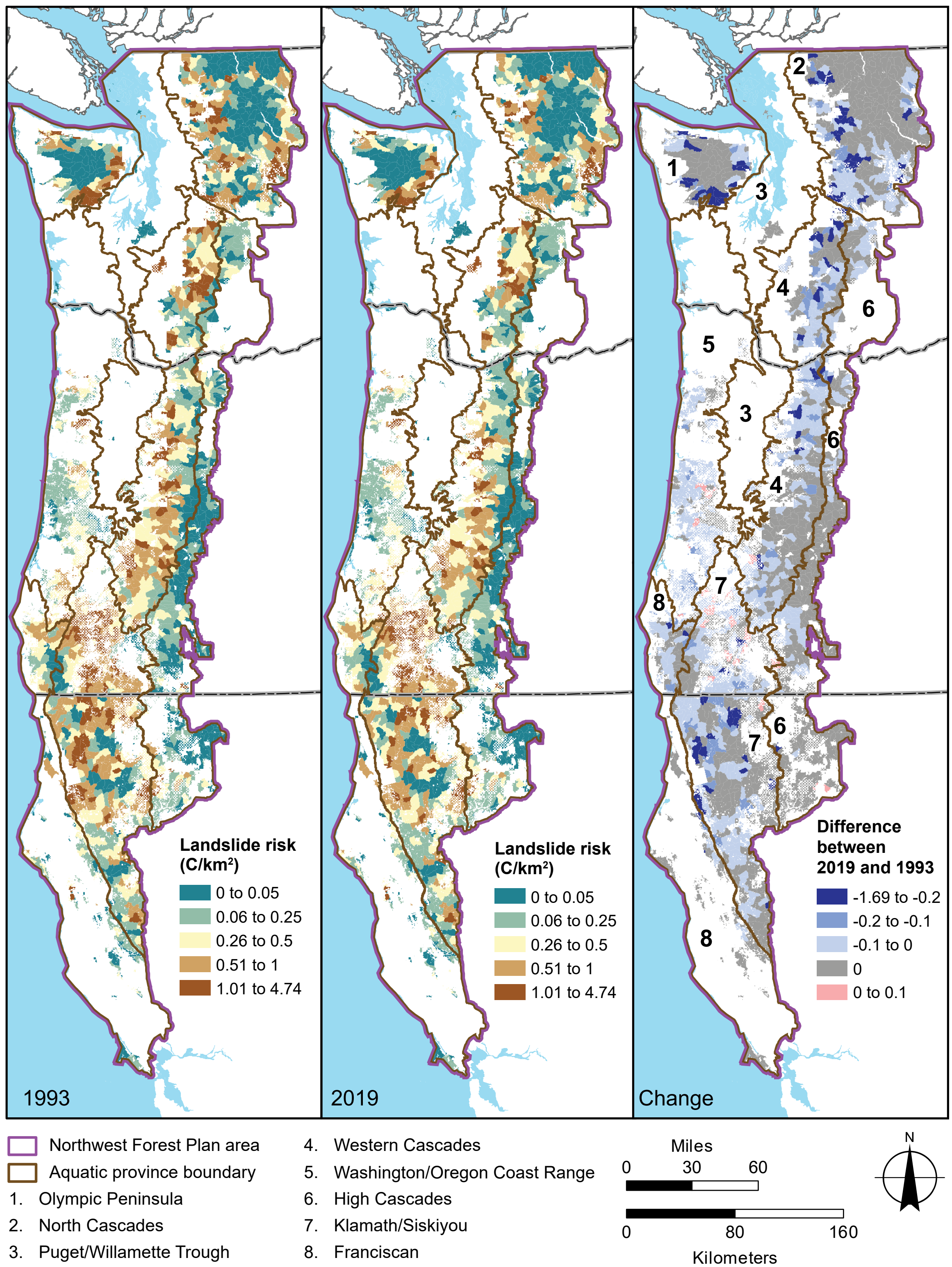

Figure 5.9—Shallow landslide risk from roads (sum of SINMAP cohesion factor [C] per km²) by subwatershed and aquatic province for 1993, 2019, and the difference between the two time periods.

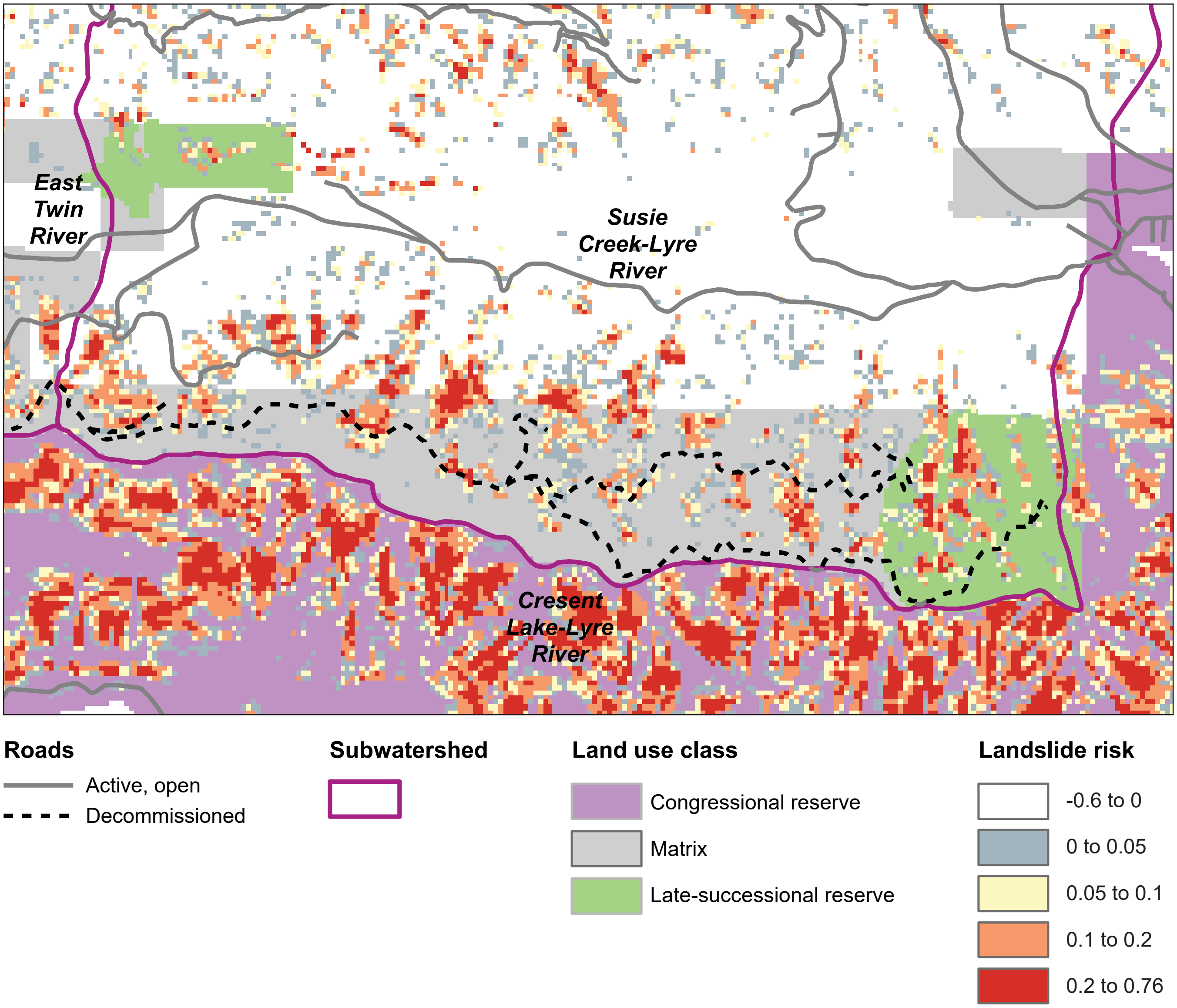

Figure 5.10—Susie Creek-Lyre River subwatershed in the Olympic Peninsula province, the subwatershed with the greatest decrease in landslide risk for the 1993–2019 time period. The decrease was due to the decommissioning of a very high-risk road, as shown in the figure by the large number of reddish pixels intersecting the road.

likely to include some increasing or flat trends across years at the individual subwatershed level (60 percent of subwatersheds were negative with 95-percent credibility intervals, from 42 to 75 percent) (fig. 5.14: Overall trend).

The patterns among substrate particle size distributions, the 16th, 50th, and 84th percentiles of the particle size distribution (D_{16}, D_{50}, and D_{84} responses, respectively), were supportive of a general convergence over the years toward intermediate particle sizes, in the 30–60 mm range (large gravel to small cobble, fig. 5.15). The changes were largely driven by declines in the size at the 84th percentile, which showed a declining trend (-32 percent per decade; 95-percent credibility intervals from -43 to -18 percent), and potentially by the 16th percentile, which demonstrated an increasing trend per decade of 43 percent with moderate support (95-percent credibility intervals from -16 to 144 percent).

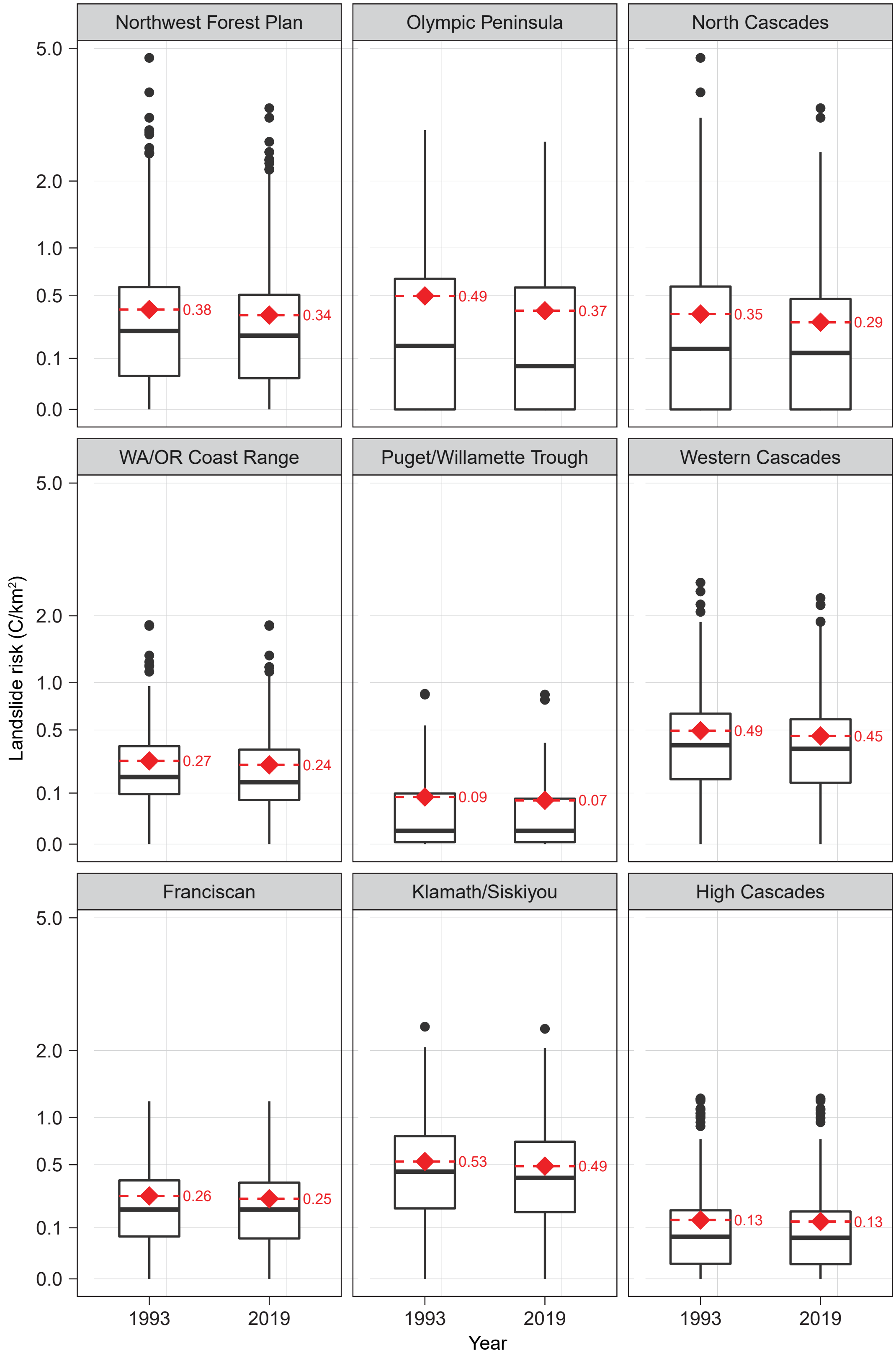

Figure 5.11—Shallow landslide risk from roads (sum of SINMAP cohesion factor [C] per km²) by subwatershed for each aquatic province and overall (Northwest Forest Plan area) for 1993 and 2019 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by subwatershed area.

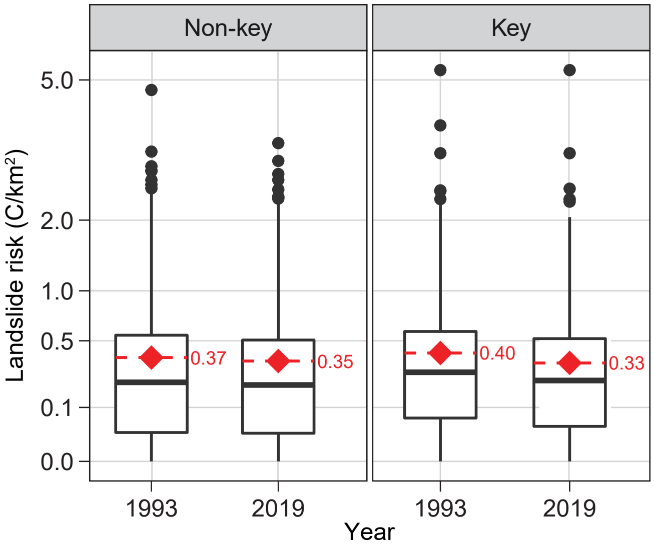

Figure 5.12—Shallow landslide risk from roads (sum of SINMAP cohesion factor [C] per km²) by subwatershed and key and non-key watersheds for 1993 and 2019 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class (key/non-key).

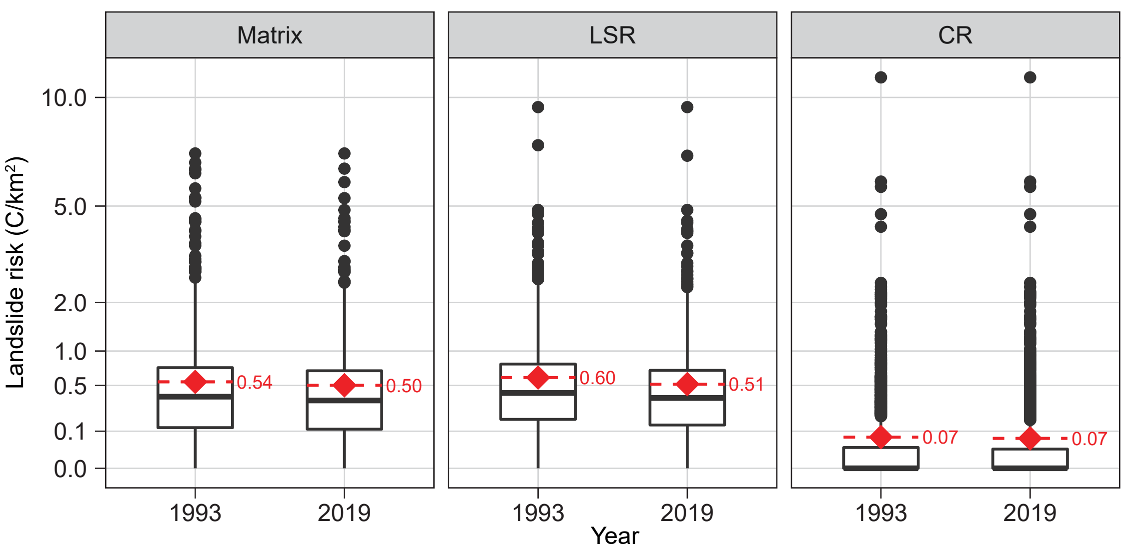

Figure 5.13—Shallow landslide risk from roads (sum of SINMAP cohesion factor [C] per km²) by subwatershed and land use allocation for 1993 and 2019 time periods. Medians (solid horizontal line), means (red diamond and red dotted line), interquartile ranges (box), interquartile range × 1.5 (whiskers), and outliers (points) are indicated. Values are weighted by the areas in each subwatershed in the designated class. LSR = late-successional reserve, CR = congressional reserve.

Province—

Declines in proportion of fine particle size material for the transect-based sampling were observed in all aquatic provinces, with the largest decline in the High Cascades. The highest proportion of fine material (≤2 mm) was observed in the Washington-Oregon Coast Range and the High Cascades, with the lowest in the Franciscan aquatic province (fig. 5.14). Pool-tail fines trend direction varied more than transect-based sampling between provinces.

The patterns among substrate particle size distributions amongst provinces followed the same trend as the overall

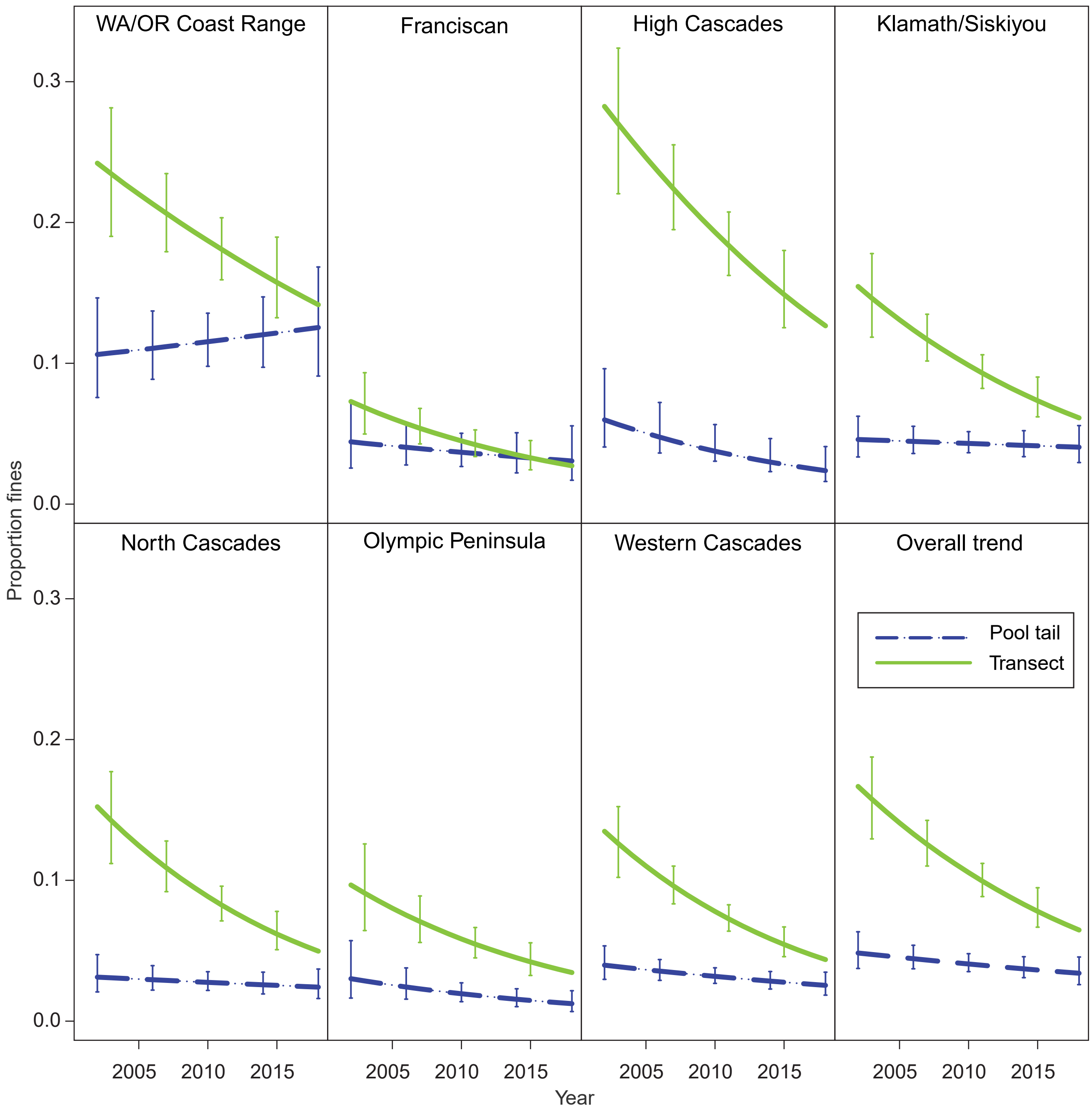

Figure 5.14—Trends in proportion of instream fine particle size material (b-axis diameters ≤2 mm) from transect-based and pool-tail crest sampling for the 2002–2018 time period by aquatic province and overall trend. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals.

trend and were supportive of a general convergence over the years toward intermediate particle sizes, in the 30–60 mm range (large gravel to small cobble) (fig. 5.15). The High Cascades showed the smallest overall distribution of particle sizes and the least amount of change through time.

Key Watersheds and Land Use Allocations

Key watersheds—

In key and non-key watersheds, there were declines in fine sediment (≤2 mm) across habitats (transect fines), and moderate support for declines in pool-tail fines over

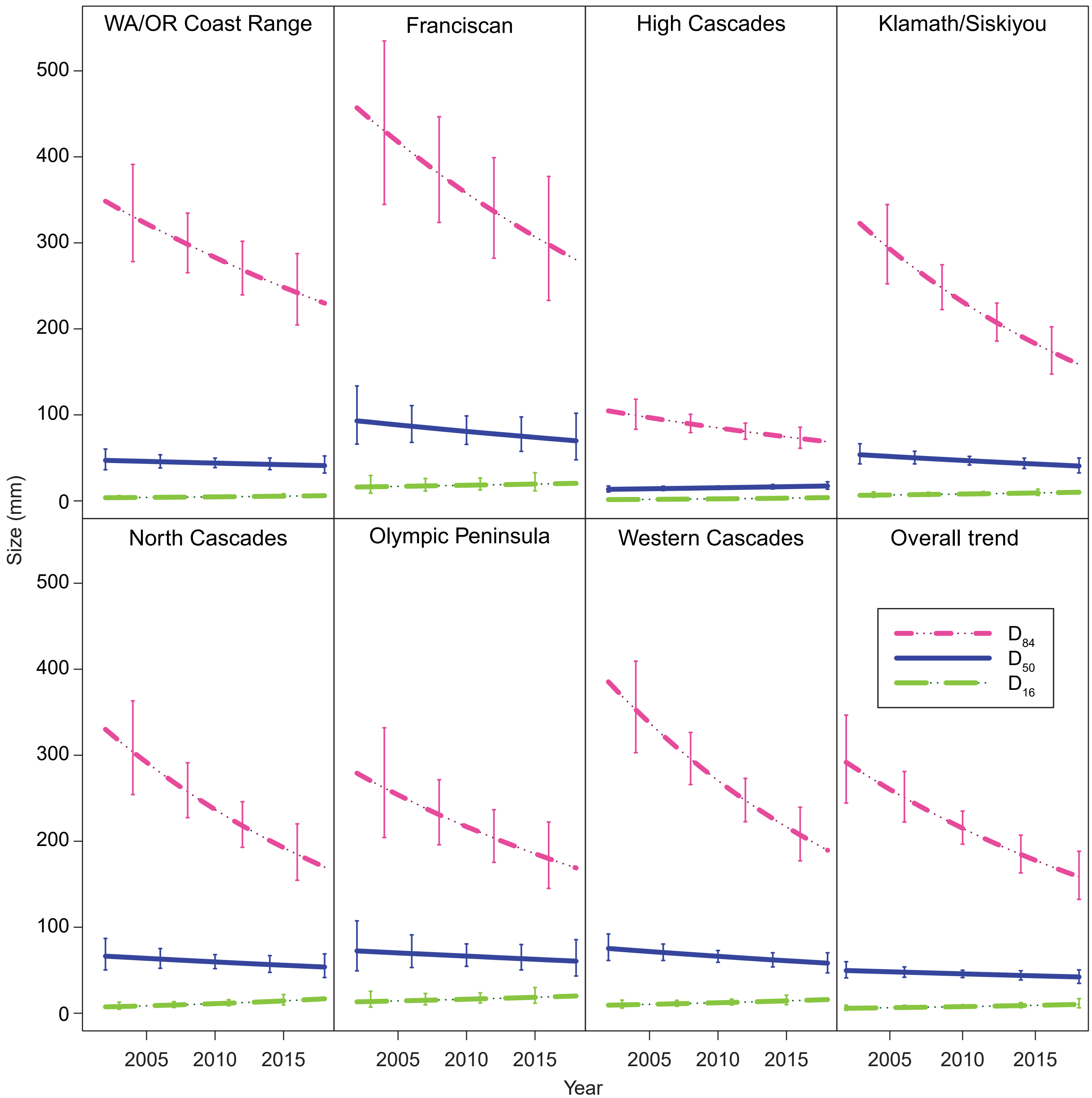

Figure 5.15—Substrate particle size (b-axis diameters) corresponding to the 16th (D_{16}), 50th (D_{50}), and 84th (D_{84}) percentiles of the particle size distribution for the 2002–2018 time period by aquatic province and overall trend. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals.

time (fig. 5.16). Key watersheds had lower proportions of transect fines than non-key watersheds, but there was not a difference in the proportion of pool-tail fines between key and non-key watersheds. Distributions of particle sizes indicated that key watersheds tended to have coarser material compared with non-key watersheds. As with patterns observed in the overall dataset, sediment particle sizes shifted in both key and non-key watersheds with fewer fine particles resulting in larger particle sizes documented at the 16th percentile of particle distribution (fig. 5.17). Smaller particle sizes at the 84th percentile of the distribution were also observed in both key and non-key watersheds.

Land use allocations—

Across LUAs, there were declines in fine sediment across habitats (transect fines). Congressional reserves and matrix LUAs demonstrated declines in pool-tail fines.

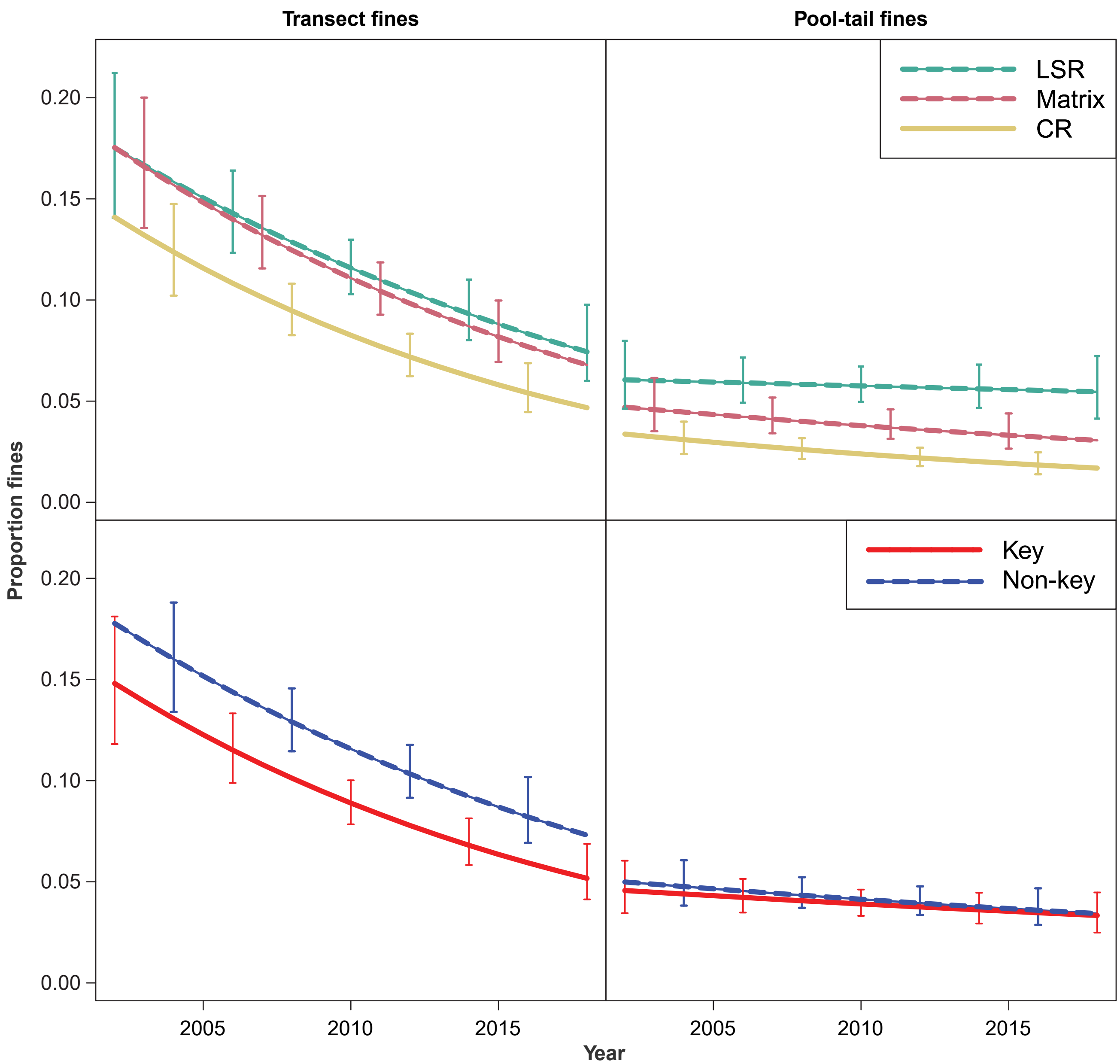

Figure 5.16—Trends in proportion of instream fine particle size material (≤2 mm) from transect-based or pool-tail crest sampling for the 2002–2018 time period by key and non-key watershed, and by land use allocation. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals. LSR = late-successional reserve, CR = congressional reserve.

Congressional reserves had lower percentages of transect and pool-tail fines compared with matrix and LSRs. LSRs contained the highest proportion of pool-tail fines across LUAs. Matrix lands were characterized by smaller particle sizes in all categories (D_{16}, D_{50}, D_{84}). The largest particle sizes at D84 were identified in LSRs.

Discussion

There have been substantial changes in active forest and road management since NWFP implementation (fig. 1.2). Focus on road management has resulted in a 7-percent net reduction of total road length. Changes in sediment risk related to roads tend to be patchy with activity focused in a smaller proportion of subwatersheds (52 percent). Where

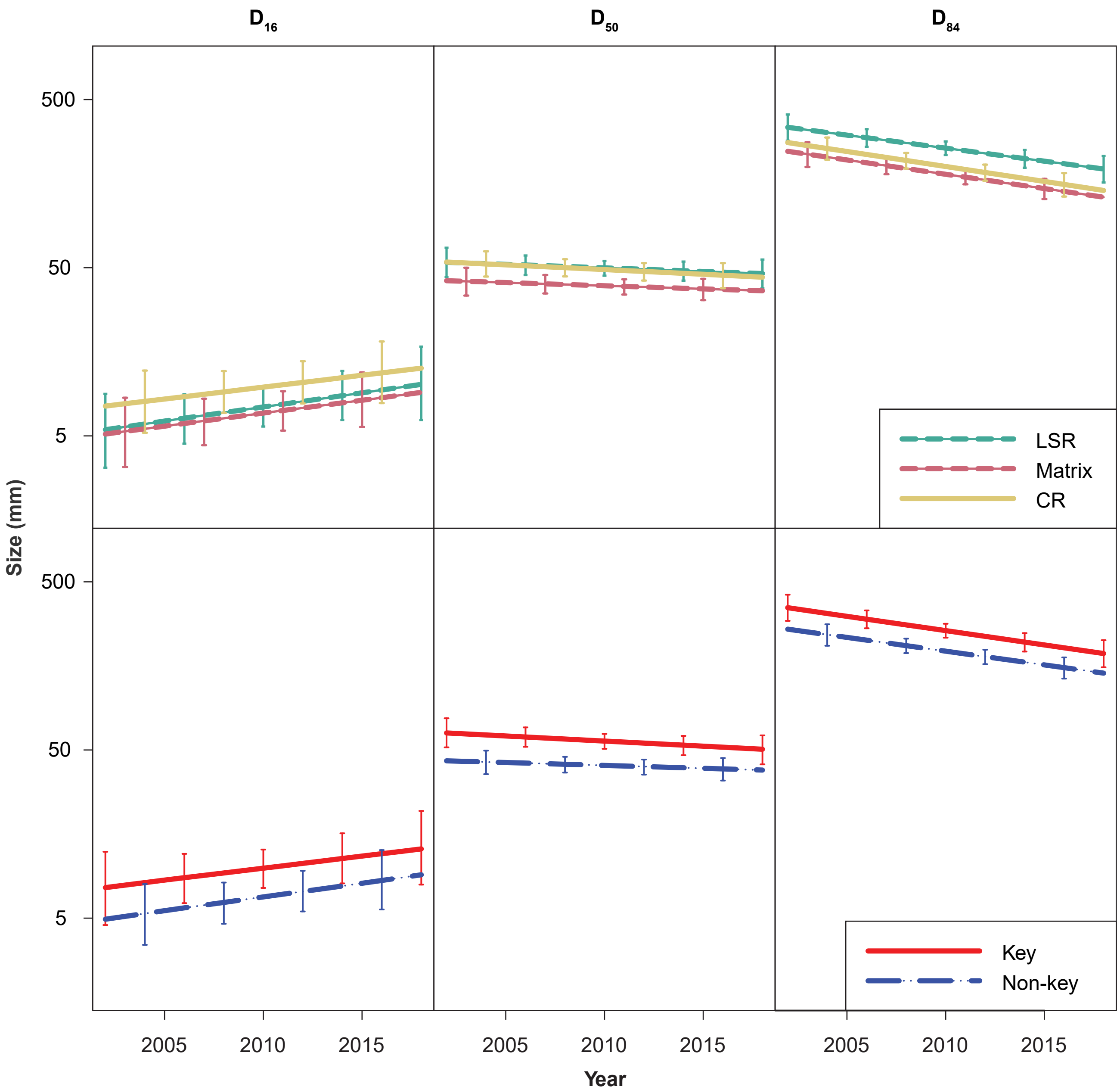

Figure 5.17—Substrate particle size (b-axis diameters) corresponding to the 16th (D_{16}), 50th (D_{50}), and 84th (D_{84}) percentiles of the particle size distribution for the 2002–2018 time period by key and non-key watershed, and by land use allocation. Lines represent average trends of sampled subwatersheds. Drop-lines indicate 95-percent credibility intervals. LSR = late-successional reserve, CR = congressional reserve.

road reductions occurred, road length was reduced 10 percent across treated subwatersheds. Decommissioning 7 percent of the road system reduced the percentage of the road system that could have been contributing to potential peakflow increases (i.e., connected road length) by 7 percent, reduced chronic delivery of fines by 4 percent, and reduced mass wasting risk by 11 percent. So collectively, the treatments were more effective at reducing landslide risk and peakflow risk than chronic fines. It should also be noted, importantly, that those models only capture the effects of road decommissioning. Finer level treatments (i.e., surfacing, improved drainage, and road storage) can also reduce risk and impact. Vegetation recovery in riparian management areas described in chapters 2 and 3 also supports the reduction in instream fine sediment observed (see “Fine Sediment” in app. 5).

Widespread declines were observed in instream fine sediment; modeled trends support declines in 90 percent of subwatersheds for transect fines and 60 percent for pool-tail fines. These results suggest both active (e.g., road modifications) and passive (e.g., forest management) measures have contributed individually and in combination to influence stream sediment to meet the objectives of the NWFP aquatic conservation strategy. Reductions in estimated episodic and chronic sediment delivery were more prevalent in key watersheds, reflecting the conservation priorities in the NWFP.

Stream sediment budgets include many factors that are difficult to assess (see “Fine Sediment” in app. 5). Accordingly, causal relationships between changes in riparian and upslope sediment delivery indicators and fine sediment dynamics in streams are challenging to establish (Al-Chokhachy et al. 2016). Given the vast number of unknown factors in play regarding processes influencing sediment, it is unlikely that instream fine sediment and changes in upslope road networks are directly tracking each other. Regardless of uncertainty concerning the specific processes in play, declines in fine sediment are consistent with expected outcomes associated with improvements in roads and vegetation management. Additional analysis of other components of sediment budgets outside of summer low-flows, such as suspended sediment, as well as key drivers (e.g., geology, stream discharge, gradient, sediment supply; Wise 2019) would be needed to evaluate fine sediment more comprehensively across watersheds. Furthermore, consideration of species adversely or beneficially (e.g., Kemp et al. 2011) influenced by fine sediment would provide additional relevant insights.

Aquatic and Riparian Effectiveness Monitoring Program crew members processing macroinvertebrate samples from Abbott Creek in the Rogue River-Siskiyou National Forest. Photo courtesy of Kiara McAdams.

Chapter 6: Biotic Responses

Three separate measures that characterize aquatic habitat connectivity and biological complexity were developed by the Aquatic and Riparian Effectiveness Monitoring Program (AREMP). These three measures are conceptually related; hence, they are presented together in this section. First, aquatic connectivity characterized by culvert inventories is presented. Culverts represent a common barrier to the movement of different life stages of fishes, amphibians, and other aquatic biota. The presented inventory targets culverts on federal lands. Second, macroinvertebrate survey results are presented. Macroinvertebrates can be a key indicator of habitat quality and have always been part of the AREMP inventory program. The third measure presented is new to the AREMP program as it represents emerging science and technology: environmental DNA, or eDNA. Since 2016, AREMP has been collaborating with scientists at the USDA Forest Service Pacific Northwest Research Station to research the viability of using eDNA to monitor for the presence of a diversity of aquatic biota including fishes, macroinvertebrates, insects, and pathogens (box 6.1). Collectively the three responses tracked by AREMP (aquatic habitat connectivity, macroinvertebrate surveys, with addition of more recent eDNA surveys) provide important information on biotic responses over the course of the Northwest Forest Plan (NWFP).

Aquatic Connectivity and Culverts

Stream culverts are critical to the maintenance of transportation systems and represent a growing management challenge, not only for road maintenance (Perrin and Dwivedi 2006), but also for maintenance of geomorphic and hydrologic processes, as well as passage of aquatic organisms (Clarkin et al. 2003, Hoffman and Dunham 2007). General guidelines for stream restoration (Roni et al. 2002) place a high priority on restoration of connectivity through culverts and other human-constructed barriers to movement of aquatic organisms. The U.S. General Accounting Office recognized this issue in the Pacific Northwest, reporting over 10,000 culverts on federal public lands, and that at least a quarter of those structures posed a barrier to upstream-migrating aquatic organisms (USGAO 2001). Federal land management agencies have invested substantially in restoring aquatic organism passage at barriers presented by road-stream crossings by replacing the structure or decommissioning the associated road. Federal agencies follow standard inventory, assessment, and design specifications (Clarkin et al. 2003, SSWG 2008), including updated designs intended for climate adaptation (Reagan 2015). This information has been translated into actions by federal land managers to upgrade, replace, or remove numerous stream culverts on federal lands since before the inception of the NWFP.

We created a summary of known culverts by assembling information contained within multiple existing databases of culvert inventory data and integrating it with our GIS roads layer. We then used this inventory to assess fish passage at culverts within streams considered potential fish habitat across the AREMP area. Limited historical data on culvert replacements was available, so we simply report on the status at this point in time. We also identified additional road-stream crossings where culverts are expected but not currently documented. Lengths of potential fish habitat upstream of culverts on federal lands were quantified and summarized across the AREMP area. It is important to clarify that this report considers federal lands only and not potential barriers on other lands that could influence connectivity. Crossings on streams with >100 km² in catchment area were assumed to be bridges and not included in this analysis.

For this analysis, we defined potential fish habitat as perennial stream reaches with <20-percent gradient. Work on culverts in the region indicates that the presence of fish in relation to stream gradient is subject to a number of uncertainties (Chelgren and Dunham 2015, Latterell et al. 2003), but detailed information needed to address them for this assessment was not available. For example, our definition of potential fish habitat did not incorporate locations of naturally occurring barriers to fish such as some waterfalls because spatial databases with those data are incomplete. Consequently, we adopted the simple and consistent criteria employed here, while acknowledging these uncertainties. For further details on our methods, see Culverts in appendix 3.

Summary information on stream culverts presented here includes: (1) the number of known stream culverts that are classified as barriers or partial barriers to fish movement, are passable, or have not been assessed for passage; (2) the stream length associated with habitats above known culverts; and (3) the numbers of additional unknown road-stream crossings based on spatial overlays. Further information about these unknown culverts is available in appendix 6.

Overall and by Province

Overall—

Across the AREMP area, culvert status is known on 3,193 culverts located in presumed fish habitat. Of these, 1,905 (60 percent) are considered complete barriers and 515 (16 percent) are considered partial barriers to fish passage, blocking 10 percent of the presumed fish habitat on federal lands (table 6.1; fig. 6.1). Overall, 773 (24 percent) of the assessed culverts are considered passable and provide fish passage to 4 percent of the total presumed fish habitat.

Our assessment of different sources of uncertainty revealed some significant gaps in knowledge regarding

Table 6.1—Stream culverts summarized by their level of passability and influence on the length (kilometers) of connected fish habitat for the aquatic provinces of the AREMP area

Aquatic province	Measure	Open[a]	Passable	Partial barrier	Complete barrier	Known total	Known unknowns[b]	Unknown unknowns[c]
Franciscan	count		29 (63%)	6 (13%)	11 (24%)	46	21	27
	km	2 988 (93%)	183 (6%)	38 (1%)	20 (1%)	3 229	55	131
High Cascades	count		78 (20%)	62 (16%)	253 (64%)	393	53	148
	km	4 931 (77%)	220 (3%)	287 (5%)	946 (15%)	6 384	217	1 706
Klamath/ Siskiyou	count		179 (23%)	169 (22%)	418 (55%)	766	142	399
	km	10 966 (87%)	484 (4%)	468 (4%)	592 (5%)	12 510	407	2 457
North Cascades	count		87 (23%)	41 (11%)	256 (66%)	384	13	76
	km	7 388 (88%)	213 (3%)	125 (2%)	619 (7%)	8 345	23	680
Olympic Peninsula	count		26 (26%)	11 (11%)	63 (63%)	100	26	65
	km	3 255 (92%)	63 (2%)	34 (1%)	182 (5%)	3 534	56	102
Puget-Willamette Trough	count		16 (47%)	6 (18%)	12 (35%)	34	9	65
	km	419 (93%)	19 (4%)	6 (1%)	8 (2%)	452	5	51
WA/OR Coast Range	count		191 (39%)	110 (22%)	195 (39%)	496	119	467
	km	5 998 (87%)	447 (7%)	186 (3%)	169 (3%)	6 800	227	1 539
Western Cascades	count		167 (17%)	110 (11%)	697 (72%)	974	156	596
	km	12 555 (84%)	633 (4%)	253 (2%)	1 568 (10%)	15 009	324	1 890
Total	count		773 (24%)	515 (16%)	1 905 (60%)	3,193	539	1,843
	km	48 500 (86%)	2262 (4%)	1 396 (3%)	4 104 (7%)	56 263	1 315	8 554

[a] Open = there is a bridge or no road crossing or bridge downstream, thus there is no count of culverts.

[b] Known unknowns = culverts present in the database with unknown passage status.

[c] Unknown unknowns = potential road-stream crossings in GIS that were not in any existing databases (Kellner and Hubbart 2017).

Percentages in parentheses are the given percentage of all known culverts.

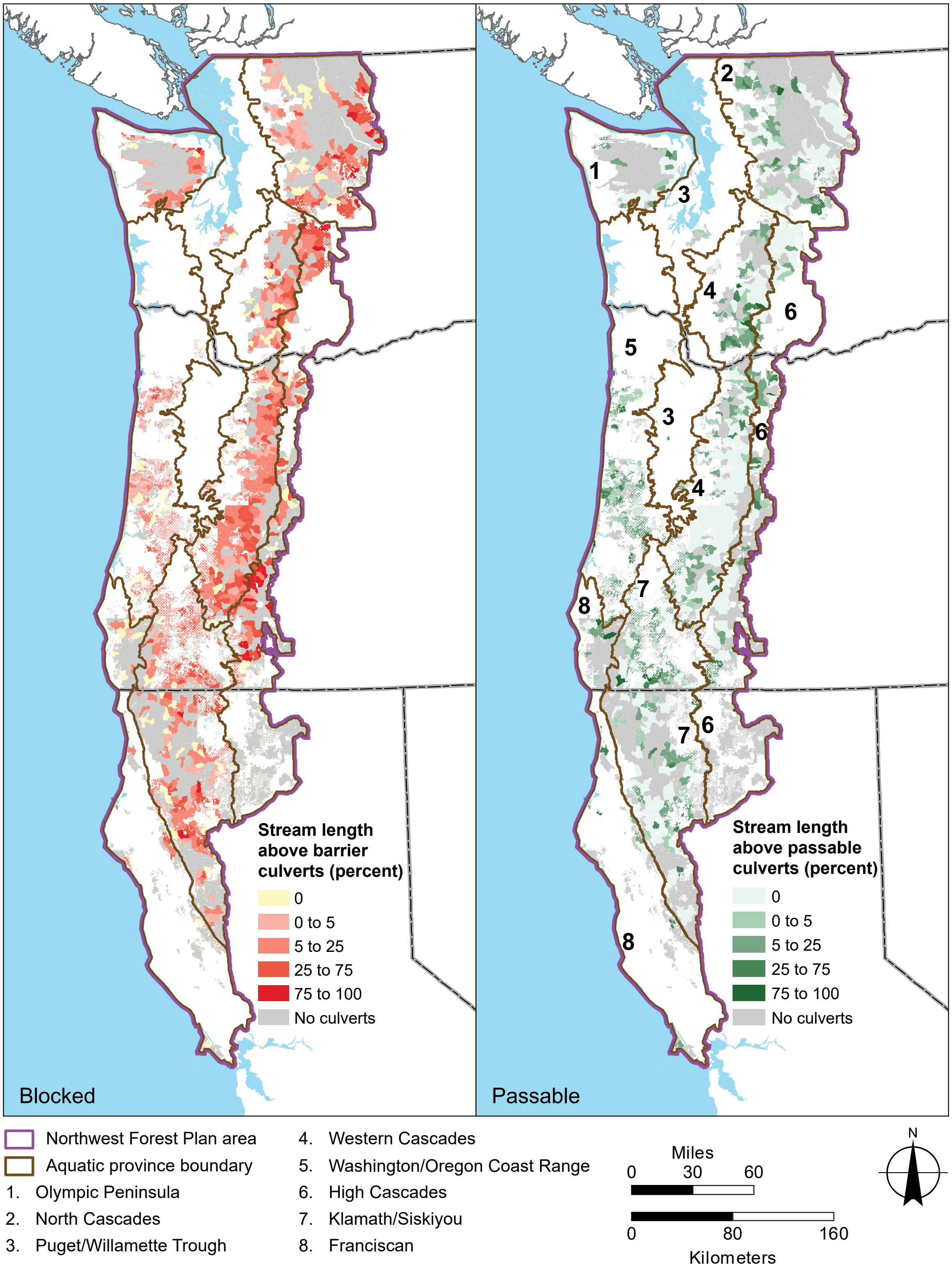

Figure 6.1— Percentage of stream length above known barrier and passable culverts by hydrologic unit code (HUC) subwatershed. HUCs containing no streams upstream of culverts are symbolized by a separate color.

passage status and unrepresented culverts. These total unknowns include known culverts without passage status and road-stream crossings not in databases. There are 539 culverts present in the database with unknown passage status, which may block an additional 2 percent of presumed fish habitat (see “Known unknowns” in table 6.1; fig. A6.12). Further, an additional 1,843 potential culverts may exist at road-stream crossings that are currently not represented in existing databases; these “unknown unknowns” could block up to 15 percent of total presumed fish habitat, 3 percent of which occurs above known barrier culverts (see “Unknown unknowns” in table 6.1; fig. A6.12). See appendix 3 for methods and caveats about potential culverts.

In summary, up to 24 percent of presumed fish habitat on federal lands in the AREMP area may be blocked by culverts. This includes stream length above culverts known to be barriers plus stream length above culverts of unknown status or a presumed road-stream crossing.

Province—

Among the aquatic provinces in the AREMP area, the Franciscan, Puget-Willamette Trough, and Washington-Oregon Coast Range have the highest percentage of known passable culverts (63, 47, and 39 percent, respectively), although they also have the highest percentage of culverts of unknown status and potential culverts compared to known culverts (known unknowns + unknown unknowns > known knowns) (table 6.1). The Washington-Oregon Coast Range and the Franciscan provinces have the highest percentage of potential fish-bearing stream length above passable culverts (7 and 6 percent, respectively). The Western Cascades and High Cascades have the highest percentage of culverts known to be partial or complete barriers (83 and 80 percent, respectively), which block 12 percent (Western Cascades) and 20 percent (High Cascades) of presumed fish habitat. The North Cascades province has the most complete information on culvert status. The High Cascades, Klamath-Siskiyou, and Washington-Oregon Coast Range provinces each have >20 percent of their potential fish habitat occurring above culverts of unknown status or potential culverts identified by road-stream crossings ([known unknown km + unknown unknown km] / known total > 20 percent).

Key Watersheds and Land Use Allocations

Key watersheds—

Key watersheds contain 39 percent of the presumed fish habitat of the AREMP area, and 40 percent of the presumed fish habitat above passable culverts occurs in key watersheds. Overall, there is no bias toward passable culverts in key watersheds, though there are differences among provinces in culvert passability and fish habitat above passable culverts. Fish habitat above passable culverts occurs more frequently, proportionately speaking, in key watersheds in the High Cascades, Franciscan, and Western Cascades provinces (table 6.2). In other words, in these provinces, there is a higher percentage of passable fish habitat in key watersheds than the percentage of total fish habitat in key watersheds. In the dataset of unknown but predicted road-stream crossings, key watersheds contain fewer potential predicted culverts (25 percent of unknown but predicted culverts) than would be expected based on their proportion of potential fish habitat. Overall, the percentage of presumed fish habitat upstream of culverts identified as movement barriers (i.e., partial plus complete barriers) was similar in key and non-key watersheds (9 and 10 percent, respectively). The percentage of habitat above passable culverts was 4 percent in both key and non-key watersheds.

Land use allocations—

Culvert passage status was not addressed by land use allocation (LUAs).

Discussion

We were able to develop the most comprehensive assessment to date across the AREMP area by assembling multiple and overlapping sources of information on stream culverts on federal land across states and agencies. Availability of data varied across agencies and areas. Of the 3,193 stream culverts that have been surveyed for their potential for fish passage across the AREMP area, 773 (24 percent) are considered passable. These passable stream culverts have existed since the NWFP inception and were initially designed for, or have since been replaced to meet design standards for, fish passage. There are many factors that influence the expected lifespan of a stream culvert (Reagan 2015), but we suspect that many culverts currently

Table 6.2—Stream culverts summarized by level of passability and influence on the length (km) of connected fish habitat for the aquatic provinces of the AREMP area, subdivided by key and non-key watersheds

Aquatic province	Subdivision	Open[a]	Passable	Partial barrier	Complete barrier	Total
Franciscan	Key	988 (91%)	84 (8%)	8 (<1%)	9 (<1%)	1090
	Non-key	2000 (94%)	99 (5%)	30 (1%)	11 (<1%)	2139
High Cascades	Key	1604 (72%)	147 (7%)	70 (3%)	399 (18%)	2219
	Non-key	3327 (80%)	72 (2%)	217 (5%)	548 (13%)	4164
Klamath/Siskiyou	Key	4611 (91%)	126 (3%)	171 (3%)	154 (3%)	5062
	Non-key	6355 (85%)	359 (5%)	297 (4%)	437 (6%)	7448
North Cascades	Key	4120 (90%)	68 (1%)	93 (2%)	314 (7%)	4596
	Non-key	3268 (87%)	145 (4%)	31 (1%)	305 (8%)	3749
Olympic Peninsula	Key	884 (91%)	19 (2%)	12 (1%)	58 (6%)	973
	Non-key	2372 (92%)	44 (2%)	22 (1%)	124 (5%)	2561
Puget-Willamette Trough	Key	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0
	Non-key	419 (93%)	19 (4%)	6 (1%)	8 (2%)	452
Washington-Oregon Coast Range	Key	1716 (88%)	117 (6%)	67 (3%)	57 (3%)	1957
	Non-key	4282 (88%)	330 (7%)	119 (3%)	113 (2%)	4843
Western Cascades	Key	5215 (85%)	334 (5%)	85 (1%)	554 (9%)	6187
	Non-key	7340 (83%)	300 (3%)	168 (2%)	1014 (12%)	8822
Total	Key	19 137 (87%)	895 (4%)	506 (2%)	1545 (7%)	22 083
	Non-key	29 363 (86%)	1367 (4%)	890 (3%)	2560 (7%)	34 179

[a] Open = a bridge or no road crossing downstream.

classified as passable by fish were replaced within the past 25 years and represent accomplishment under the aquatic conservation strategy. Additional investigation into specific records for individual stream culvert replacements or in-situ assessments to estimate the ages of existing structures without documentation would be needed to quantify patterns of stream culvert replacements more firmly across the AREMP area.

We provided an overall assessment of various sources of uncertainty in addition to analysis of the spatial distribution of barrier culverts. Specifically, we quantified the presence of stream culverts known to exist but are of unknown status (i.e., exist in a database, but not yet classified for fish passage), and road-stream crossings that may have culverts identified by GIS but have yet to be surveyed or included in an accessible database. The known barriers to fish passage block 10 percent of the total presumed fish habitat in the AREMP area. The addition of the unknown sites could block up to 14 percent more fish habitat. The High Cascades, Western Cascades, Klamath-Siskiyou, and Washington-Oregon Coast Range provinces have the largest need for fish passage data based on the amount of potential fish habitat above road-stream crossings of unknown status. AREMP engaged with field units to update culvert information and conduct additional culvert surveys to support this analysis to address both classes of uncertainty. Our assessment does not consider habitat blocked by culverts downstream of federal lands and therefore underrepresents the amount of blocked habitat. Not enough information was available to address the status of barriers downstream of federal lands. Further development of centralized and standardized databases and associated decision-support systems could improve identification of stream culverts in need of replacement, as well as their priority for replacement based on expected culvert life spans, value to the transportation network, and geomorphic, hydrologic, and ecological criteria (Moody et al. 2017, Perrin and Dwivedi 2006, Reagan 2015). Federal land managers within the region recognize these needs and, as a result of this assessment, a stronger collective effort to resolve uncertainties is emerging.

Macroinvertebrates

Streams that are monitored by the AREMP variably support fish (see app. 1), but all streams support populations of aquatic macroinvertebrates (aquatic insects, mollusks, and other taxa). AREMP has considered macroinvertebrates in the context of observed composition of species assemblages in a given sample relative to that expected from comparable least-disturbed or reference distributions (Miller et al. 2016, 2017) (see "Macroinvertebrates" in app. 2). The ratio of observed to expected (O/E) taxa should trend toward 1.0 as processes or states within a given location approach those in reference sites. Although the AREMP approach to applying reference conditions aligns with common bioassessments, there are well-known limitations with identification of reference conditions (e.g., the "E" in O/E), as well as how to consider uncertainties in describing patterns in macroinvertebrate assemblages (Damanik-Ambarita et al. 2018, Herlihy et al. 2020, Stoddard et al. 2006). Additional summary information that classifies O/E macroinvertebrates within administrative units is available in appendix 6.

Overall and by Province

Overall—

The overall trend in the O/E ratio increased at a rate of 2.7 percent per decade with weak probabilistic support (95-percent credibility intervals from -2.3 to +8.0 percent) (fig. 6.2). The majority of AREMP subwatershed trends were positive as the percentage of subwatershed trends that are increasing is 67 percent (120 subwatersheds, 95-percent credibility intervals from 33 to 95 percent). In some cases, O/E ratios exceeded a value of 1.0, which indicates sites with exceptionally high diversity of macroinvertebrates and, based on this approach, conditions that are considered to be equivalent to reference sites (Hawkins 2006).

Province—

Aquatic provinces followed patterns in the overall dataset, with weak support for increases in O/E in most subwatersheds in the Klamath-Siskiyou, Franciscan, and High Cascades provinces. We observed less support for increasing O/E trends in the other aquatic provinces (fig. 6.2).

Key Watersheds and Land Use Allocations

Key watersheds—

Key watersheds demonstrated higher O/E ratios than non-key watersheds (fig. 6.3). Both key and non-key watersheds showed weak support for consistent increases in O/E ratio over time, consistent with the overall trend.

Land use allocations—

Matrix and late-successional reserves (LSR) tended to have higher O/E values than congressional reserves. Increases in O/E ratios were most pronounced in matrix LUAs.

Discussion

Macroinvertebrates are often used as an indicator of watershed condition to integrate multiple components of stream habitat (e.g., temperature, sediment, dissolved oxygen) (Barbour et al. 2000). This metric appears to be improving by becoming more like least disturbed reference sites across the AREMP area. Improvements in macroinvertebrate O/E were greater in matrix lands, where watershed condition may have the greatest potential to improve. In contrast, congressional reserves showed little change over the monitoring period and now appear to have the lowest O/E score compared to other LUAs (although still quite high in absolute terms). This result may be due to more limited biological productivity in the generally higher elevation reserved areas (see fig. A1.7), the frequency of intermittent stream types, or the greater riparian vegetation losses they experienced. Challenges with application and interpretation of O/E scores (in absolute or relative terms) have been acknowledged in the region by other investigators (e.g., Ode et al. 2008). The Northwest Forest Plan science synthesis (Reeves et al. 2018) also acknowledged the limitations of employing reference conditions in bioassessments and called for new approaches to be considered, but to date more widely acceptable alternatives have not been developed.

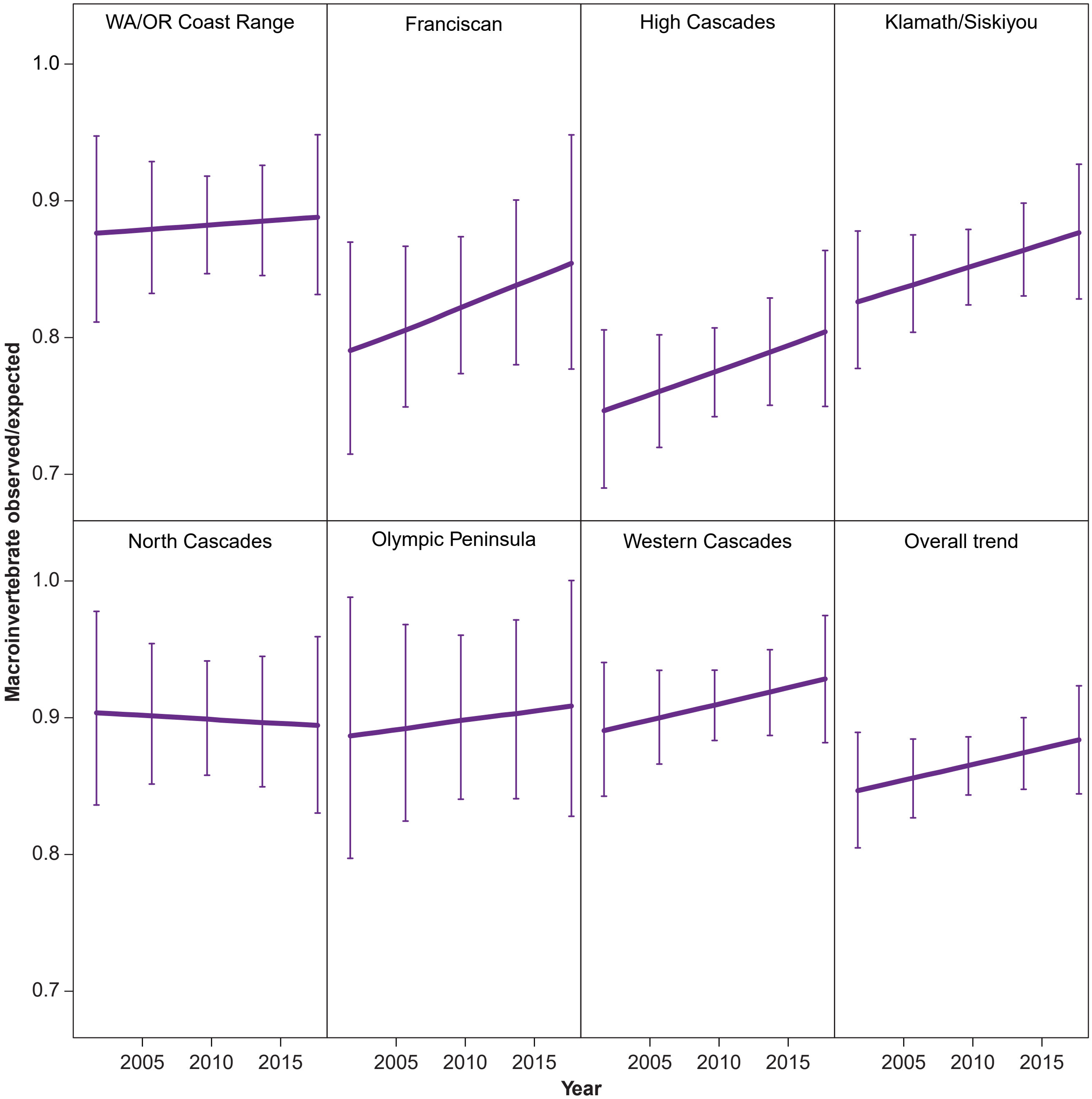

Figure 6.2—Estimated observed to expected ratios of macroinvertebrate assemblages sampled from sites monitored by the Aquatic and Riparian Effectiveness Monitoring Program for 2002–2018. Province-level average trends and the overall average trend are presented. Drop-lines are 95-percent credibility intervals.

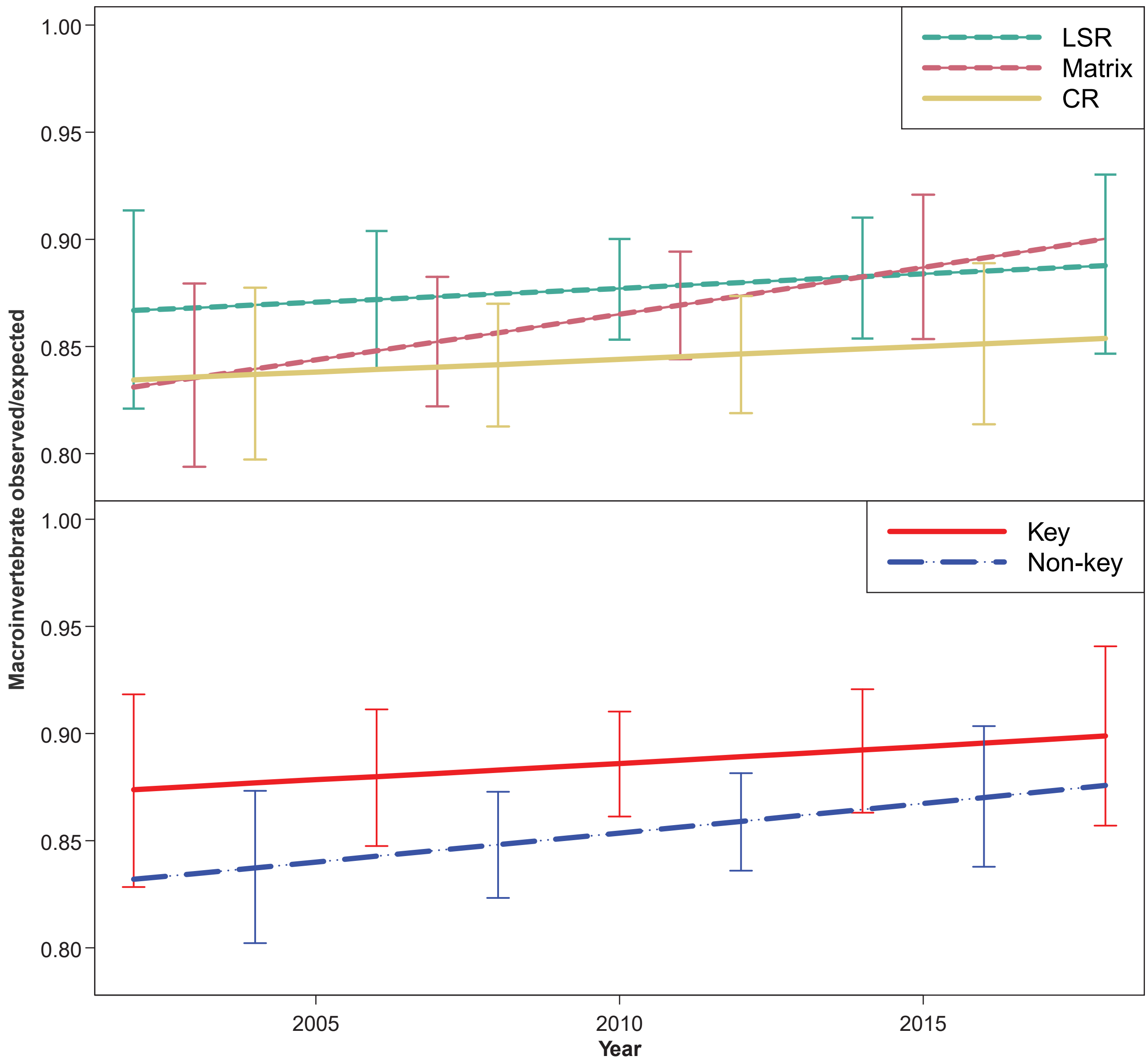

Figure 6.3—Estimated observed to expected ratios of macroinvertebrate assemblages sampled from sites monitored by the Aquatic and Riparian Effectiveness Monitoring Program for 2002–2018. Estimates in the top graph are referenced to three major land use allocations: congressional reserve (CR), late-successional reserve (LSR), and matrix lands. Estimates in the bottom graph are referenced to key and non-key watersheds. Bold lines are average trends. Drop-lines are 95-percent credibility intervals.

Box 6.1

Species Distributions

Understanding species distributions is a prerequisite for broad-scale conservation planning (Angermeier and Winston 1999), whether the question involves specific issues such as fish passage (Chelgren and Dunham 2015) or regional assessments of species status (Mims et al. 2018). Species distributions are difficult to track as they can be dynamic and difficult to quantify with conventional methods of sampling (Chelgren and Dunham 2015). Recent development of environmental DNA (eDNA) as a means of quantifying species distributions represents a new opportunity to address this ongoing question (Penaluna et al. 2021). Like other methods for determining species distributions, eDNA does not offer perfect detectability. However, considered in a statistical context in conjunction with conventional survey data or even information from expert opinion, data from eDNA offer a powerful means of estimating the probability of species presence and the associated distribution (Peterson and Dunham 2003).

eDNA refers to any DNA collected directly from the environment rather than directly from an organism. This DNA is usually contained in cells released from the organism from any of multiple sources, such as sloughed skin, mucous, waste, gametes, or decaying matter. eDNA surveys allow for the non-invasive detection of organisms in the environment, including those that may be present at low abundance, such as rare, threatened, or endangered species; newly arrived invasive species; or species that may be difficult to detect with traditional surveys. eDNA collection mitigates many of the logistical hurdles of traditional surveys, often requiring reduced field crews, reduced safety concerns, and reduced permitting because animals are not being directly handled. eDNA sampling at AREMP sites began in 2016 and has been ongoing since. It occurs through a partnership between the AREMP and the USDA Forest Service Pacific Northwest Research Station (lead scientists: Brooke Penaluna and Rich Cronn).

Three eDNA samples were collected from each site within a watershed. The presence of multiple animals, including fish, amphibians, and mammals, was determined using a metabarcoding approach that allows for the simultaneous detection of multiple species (Hauck et al. 2019, Weitemier et al. 2021). Analysis of the complete dataset is ongoing. An example of processed data from the Elk Creek watershed in the USDI Bureau of Land Management Coos Bay District from 2016 is presented here (fig. 6.4).

Among six sites in the Elk Creek watershed, eDNA detected evidence of seven fish species from three genera, including four salmonids, one sculpin, and two dace. eDNA also detected evidence of beaver and multiple amphibians, including the Oregon slender salamander and the coastal giant salamander. This sampling also provided evidence of site partitioning within the watershed, with some sites containing fishes only, amphibians only, or a mix of both (fig. 6.5). Over time, additional data collection across the AREMP area will improve understanding of species distributions, as illustrated by this example.

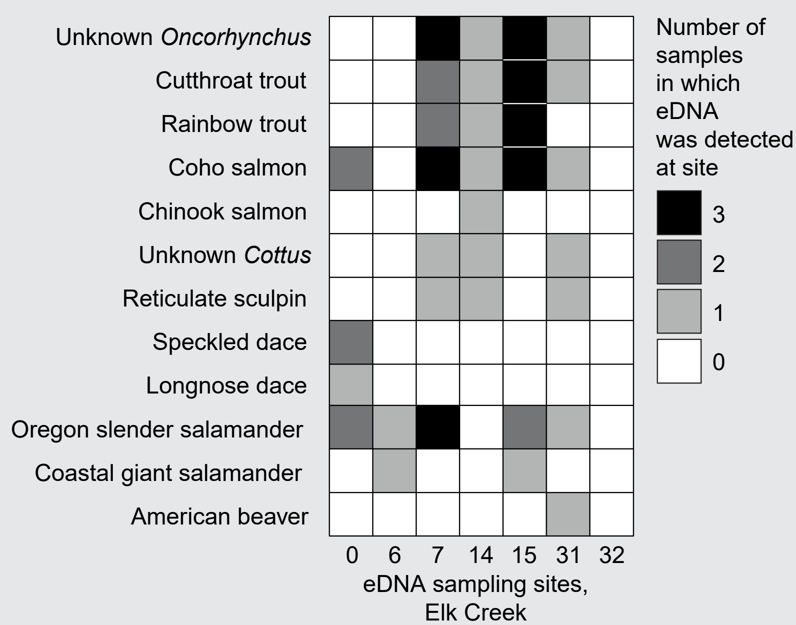

Figure 6.4—eDNA detection of organisms in Elk Creek, Oregon, Bureau of Land Management Coos Bay District. Rows represent species detected at one or more of six sites, represented by columns. Shading indicates whether a species was detected in zero (white), one (light gray), two (dark gray), or three (black) samples at that site.

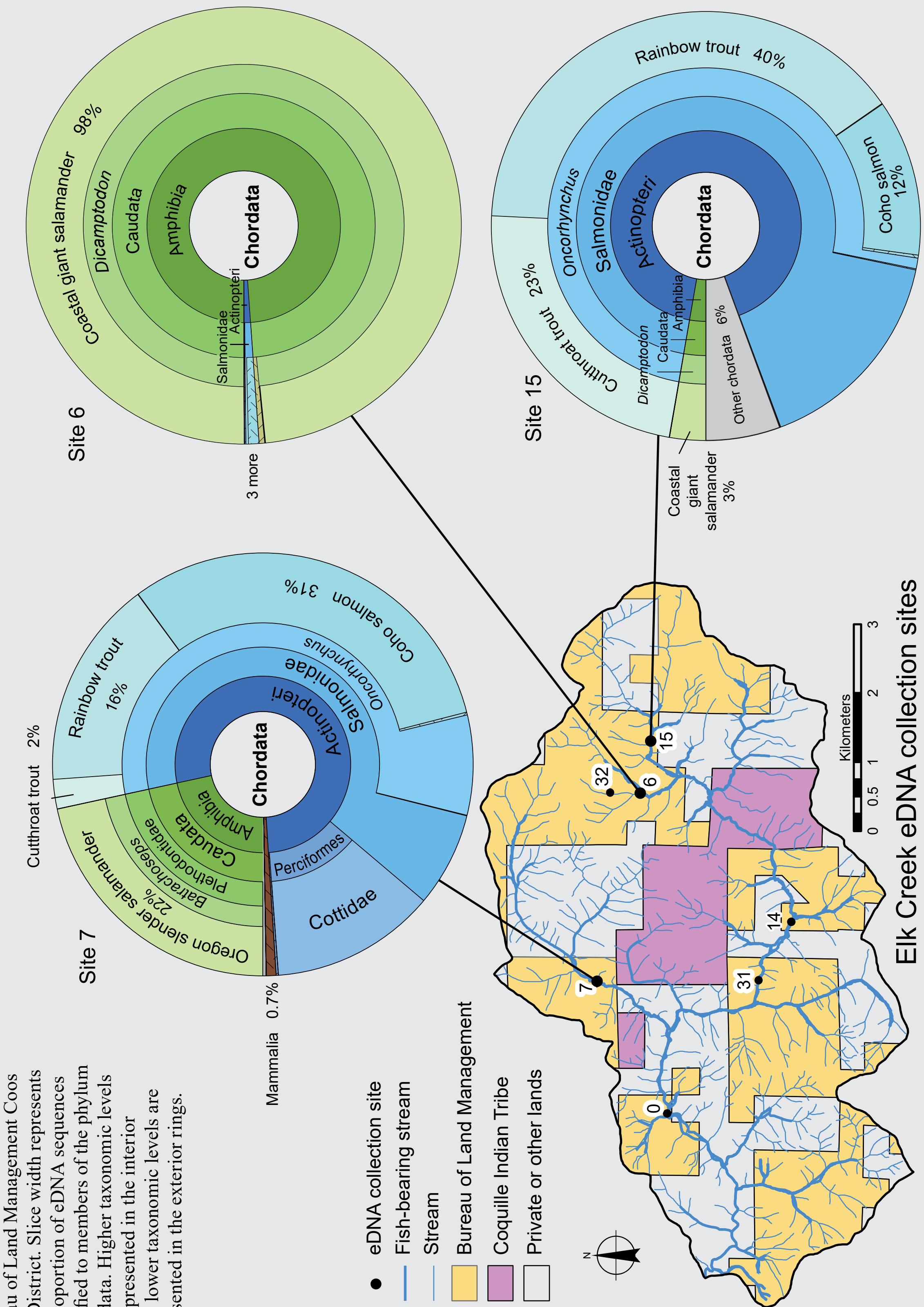

Figure 6.5—eDNA signal strength by species at Elk Creek, Oregon, Bureau of Land Management Coos Bay District. Slice width represents the proportion of eDNA sequences classified to members of the phylum Chordata. Higher taxonomic levels are represented in the interior rings; lower taxonomic levels are represented in the exterior rings.

Foster Bar-Rogue River watershed. Photo courtesy of Morgan Garay.

Chapter 7: Integrated Watershed Condition

Conceptually, the broad term of "watershed condition" is often used with the intention of capturing characteristics that describe biotic and abiotic functions. Therefore, no one indicator or response is likely to completely characterize watershed conditions as stream networks operate as integrated systems. Capturing the integrated overall condition of a watershed is challenging. Various approaches have been proposed, each intended to capture the integration inherent in an assessment of watershed conditions. For example, Carlisle et al. (2008) suggested that if one of the parameters of interest in a sampling unit (reach or watershed) is outside the threshold value for suitable conditions, the unit as a whole is outside the suitable range. Miller et al. (2017) acknowledged that reliance on a single biological metric can lead to erroneous interpretation of the biological condition of a watershed (e.g., Barbour et al. 1999); they suggested that the findings of several responses can be used as multiple lines of evidence to look at watershed condition trends. Reeves et al. (2004) used an expert opinion approach with logic models (Reynolds et al. 2014) to develop a process that allowed for integration of the parameters of interest to monitor watershed conditions on federal lands in the Pacific Northwest of the United States. The integration process was transparent, but there was no clear way to assess the statistical significance of trends in the overall conditions. Here, we employed a multivariate statistical approach to address multiple indicators collectively as descriptors of watershed conditions. A multivariate approach allows for an assessment of how patterns in multiple conditions change over time in contrast to the univariate response results presented elsewhere in this report. We selected a mix of instream and upslope responses to analyze, including macroinvertebrate O/E, percentage of fines measured across habitats, and large wood densities as indicators of instream condition, and road density, percentage of canopy cover, and density of large trees (≥50 cm in diameter at breast height [d.b.h.]) from the upslope/riparian characteristics (see table A4.1).

We used principal components analysis (PCA), a multivariate ordination of samples by Euclidean distances describing linear relationships among variables (McCune and Grace 2002). We used a separate multi-response permutation procedure (MRPP) to determine the significance (p-value) and effect size (A: $-1 < A < 1$) among categorical groups. The larger the effect size above zero ($A > 0$), the greater the difference between groups. However, when the effect size is small ($A < 0.05$) the ecological significance of the result should be interpreted with caution because small p-values can result with large sample sizes (McCune and Grace 2002). As such, we focus our results on group differences that are both statistically significant ($p < 0.01$) and with effect sizes (A) greater than 0.05. We conducted multiple different PCAs and MRPPs to examine watershed conditions from the entire dataset of 406 samples drawn from the 219 subwatersheds collected in time periods 1 and 2 together (2002–2009 and 2010–2017, respectively). We also analyzed subsets of these data grouped by first and second time periods, aquatic provinces (excluding the Franciscan Province due to low sample size), key and non-key watersheds, and federal LUAs (see table A4.2 for sample sizes). We used an MRPP to test for significant differences and effect sizes between groups and PCAs for understanding what variables could be driving them.

Overall and by Province

Overall—

Strong differences in watershed conditions were found among provinces ($A = 0.23$) (table 7.1). Weak differences ($A < 0.05$) were observed between key and non-key watersheds and among LUAs in the combined sample set. MRPP test results indicated no differences among watershed conditions from the first to the second survey period when all provinces were combined.

In the multivariate PCA of all 406 samples, the first and second axes (i.e., principal components) were significant and explained 60.3 percent of the variability in the relationships among the instream and upslope watershed conditions (table 7.2, fig. 7.1). The first PCA axis was defined by riparian canopy cover and riparian

Table 7.1—MRPP results from all samples together and by subsets of data grouped by aquatic province, key and non-key watersheds, and federal land use allocations

Sample group	Sample size	Time 1/Time 2	Key/non-key	Matrix/LSR/CR	Aquatic provinces
All samples[a]	406	<0.01 (0.052)	0.02 (<0.001)	0.04 (<0.001)	**0.23 (<0.001)**
High Cascades	54	-0.01 (1)	-0.01 (0.574)	0.02 (0.157)	NA
Klamath/Siskiyou	110	<0.01 (0.316)	-0.01 (0.968)	0.03 (0.004)	NA
North Cascades	42	-0.01 (0.655)	**0.19 (<0.001)**	0.03 (0.156)	NA
Olympic Peninsula	17	-0.03 (0.648)	0.02 (0.279)	0.08 (0.133)	NA
Washington-Oregon Coast Range	44	**0.12 (<0.001)**	**0.08 (0.001)**	-0.01 (0.430)	NA
Western Cascades	129	0.02 (0.010)	**0.05 (<0.001)**	**0.13 (<0.001)**	NA
Key	151	-0.01 (0.734)	NA	**0.08 (<0.001)**	NA
Non-key	255	0.01 (0.045)	NA	0.01 (0.058)	NA
LSR	164	0.01 (0.025)	0.02 (0.004)	NA	NA
Matrix	143	0.01 (0.037)	0.01 (0.051)	NA	NA
CR	99	<0.01 (0.235)	0.03 (0.006)	NA	NA

The larger the group difference effect size (A), the greater the difference between groups. Bolded results with both A > 0.05 and p < 0.01 (to account for multiple tests, in parentheses) are considered significant. Note that values of A < 0.05 indicate small differences between groups, even though they may be statistically significant owing to large sample sizes.

CR = congressional reserve, LSR = late-successional reserve, NA = not applicable.

[a] 10 sites in the Franciscan Province were included in the "All samples" analysis, but were not analyzed independently owing to small sample size.

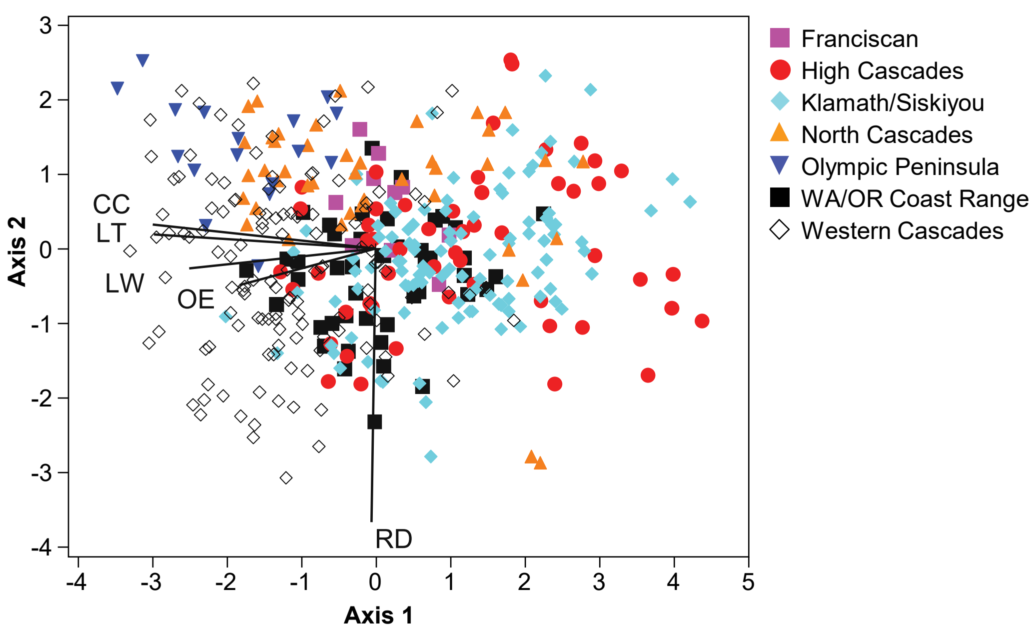

Figure 7.1—Principal components analysis ordination of all watershed samples (n = 406) grouped by aquatic provinces. Vectors indicate explanatory (highly correlated) habitat variables. Non-explanatory variables (percentage of fines) are not present in the figure. Samples from both time periods 1 and 2 are included. RD = road density, OE = observed to expected macroinvertebrate values, LT = riparian large trees, CC = canopy cover, LW = large wood.

Table 7.2—PCA results from all samples together and by subsets of data grouped by aquatic province, key and non-key watersheds, and federal land use allocations

Sample groups	PCA axis	Eigenvalue	p-value	Variance	Road density	Fines (log)	Canopy cover	Riparian large trees	Large wood (log)	Macroinvertebrate O/E
				Percent		*- - - Percent - - -*				
All samples	1	2.45	**0.001**	40.75			-0.81	-0.81	-0.75	-0.64
(n = 406)	2	1.18	**0.005**	19.58	-0.90					
High Cascades	1	2.42	**0.001**	40.37		0.64	-0.72	-0.68	-0.61	-0.80
(n = 54)	2	1.30	0.172	21.72	0.85	0.52				
Klamath/Siskiyou	1	2.29	**0.001**	38.24	-0.59		-0.79	-0.66	-0.65	-0.65
(n = 110)	2	1.11	0.864	18.48		-0.80				
North Cascades	1	2.40	**0.001**	39.93			0.92	0.90	0.79	
(n = 42)	2	1.66	**0.001**	27.64	0.78	0.81				0.60
Olympic Peninsula	1	2.18	0.124	36.30	0.86		-0.64	-0.89		
(n = 17)	2	1.50	0.290	24.96		-0.86	0.62			
WA/OR Coast Range	1	1.96	**0.006**	32.64		0.76	-0.60	-0.55	-0.57	-0.57
(n = 44)	2	1.36	0.129	22.68	-0.83			0.52		-0.59
Western Cascades	1	1.92	**0.001**	31.91	0.59		-0.89	-0.84		
(n = 129)	2	1.63	**0.001**	27.20	0.54	-0.53			0.70	0.73
Key	1	2.36	**0.001**	39.26			-0.86	-0.91	-0.54	-0.53
(n = 151)	2	1.30	**0.002**	21.60	-0.79				-0.54	
Non-key	1	2.53	**0.001**	42.21			0.80	0.76	0.79	0.65
(n = 255)	2	1.06	0.949	17.59		0.82				
LSR	1	2.37	**0.001**	39.44			-0.81	-0.81	-0.75	-0.51
(n = 164)	2	1.11	0.715	18.47	-0.85					
Matrix	1	2.92	**0.001**	48.72	-0.53	0.50	-0.78	-0.79	-0.74	-0.78
(n = 143)	2	0.09	1.000	15.62	-0.71	-0.63				
CR	1	2.41	**0.001**	40.08			-0.86	-0.88	-0.75	-0.56
(n = 99)	2	1.21	0.259	20.15		-0.87				0.52

PCA results for the first two axes include the eigenvalue, p-value (bolded if $p < 0.01$), percentage of variance explained, and the instream and upslope variables correlated with the ordination axes ($r > |0.5|$).

O/E = observed to expected; CR = congressional reserve; LSR = late-successional reserve.

tree densities, along with instream wood abundance and macroinvertebrate O/E compositions. The second PCA axis was defined by road density, which represents watershed-scale human disturbance. Instream sediment (percentage of fines) was not a driver of either axis.

The aquatic provinces were visibly different from each other in multivariate space (fig. 7.1) and as indicated in the MRPP analysis (table 7.1), thus motivating a look at variation among and within provinces. Differences are most obvious with the Olympic and Western Cascade provinces having higher canopy cover and riparian tree densities, especially compared to the Klamath-Siskiyou and High Cascades. The Olympic and North Cascade provinces also have relatively low road density.

Province—

Each aquatic province was analyzed separately. No field samples were taken in the Puget-Willamette Trough, so this AREMP aquatic province is not represented. The 10 sites in the Franciscan Province were included in the "All data" analysis, but not as an independent province due to small sample size. Of the remaining provinces analyzed, only the Washington-Oregon Coast Range showed a significant difference between the first and second time periods (A = 0.12) (table 7.1). The ordination plots per province (fig. 7.2) show that most of the provinces have diverse watershed conditions, regardless of the survey time period.

The first PCA axes were usually significant (Olympic Peninsula province was not significant), but the second axes were only significant for the North Cascades and Western Cascades provinces (table 7.2). Variability among instream and upslope subwatershed attributes explained by the first two axes ranged from 55 percent in the Washington-Oregon Coast Range to 68 percent in the North Cascades. Among the provinces, the first PCA axis was most often defined by riparian tree densities and canopy cover, closely followed by instream large wood and macroinvertebrate O/E values (table 7.2), similar to the overall PCA. The second PCA axis was most often defined by instream sediment (percentage of fines) or road density, although these habitat variables rarely correlated with each other in the same province.

Key Watersheds and Land Use Allocations

Key watersheds—

PCA and MRPP results comparing key versus non-key watersheds show little difference between these groups in the combined dataset, including all provinces (table 7.1, fig. 7.3). However, differences between key and non-key watersheds are found in the North Cascades, Western Cascades, and Washington-Oregon Coast Range provinces (table 7.1). These differences are primarily associated with greater percentages of canopy cover and large tree densities in key compared with non-key watersheds.

The combined group of key watersheds from all provinces did not change noticeably between time periods, while non-key watersheds appear to have small, not statistically significant differences (A = 0.01, p=0.045) between time periods.

Land use allocations—

The distribution of land use allocation (LUA) designations in sampled subwatersheds is not equal across aquatic provinces in the AREMP area. Watersheds where matrix lands dominate (greatest percentage of area per subwatershed) are the Western Cascades, High Cascades, and Klamath-Siskiyou provinces (see table A4.2). Watersheds where congressional reserves dominate are the North Cascades and Olympic provinces. In the analysis with all provinces combined, MRPP identified small differences (A = 0.04) among the three LUAs, with the greatest difference found between watersheds dominated by matrix and congressional reserve lands driven by higher road densities in matrix watersheds and lower road densities in congressional reserve watersheds (fig. 7.4). Strong differences (A = 0.13) between matrix and congressional reserve watersheds were found in the Western Cascades province where congressional reserve watersheds were also associated with greater canopy cover and large tree densities (fig. 7.4).

In the individual LUA analyses with all provinces combined, late-successional reserve (LSR) and matrix watersheds show small, not statistically significant, differences (A = 0.01, p > 0.01) between time periods 1 and 2. Both LSR and matrix had lower road densities in time period 2. Differences between LUAs in key watersheds were again more prominent (A = 0.08) where matrix watersheds had higher road densities and congressional

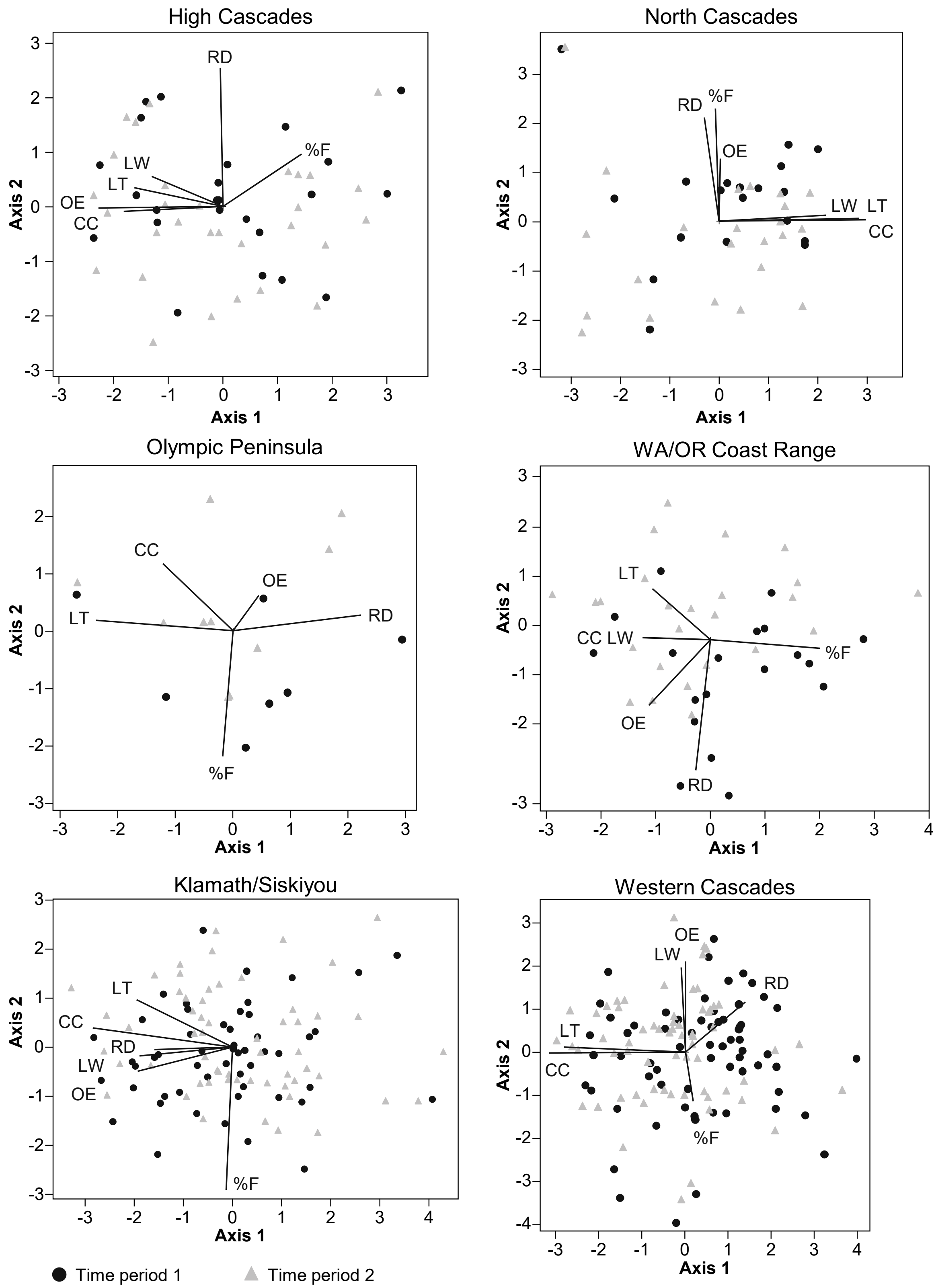

Figure 7.2—Principal components analysis ordinations by aquatic province (see table A4.2 for sample sizes) grouped by time periods 1 (black circle) and 2 (grey triangle) with vectors indicating explanatory habitat variables. Franciscan province is not included because of low sample size (n = 10). RD = road density, OE = observed to expected macroinvertebrate values, LT = riparian large trees, CC = canopy cover, LW = large wood, %F = percentage of fines.

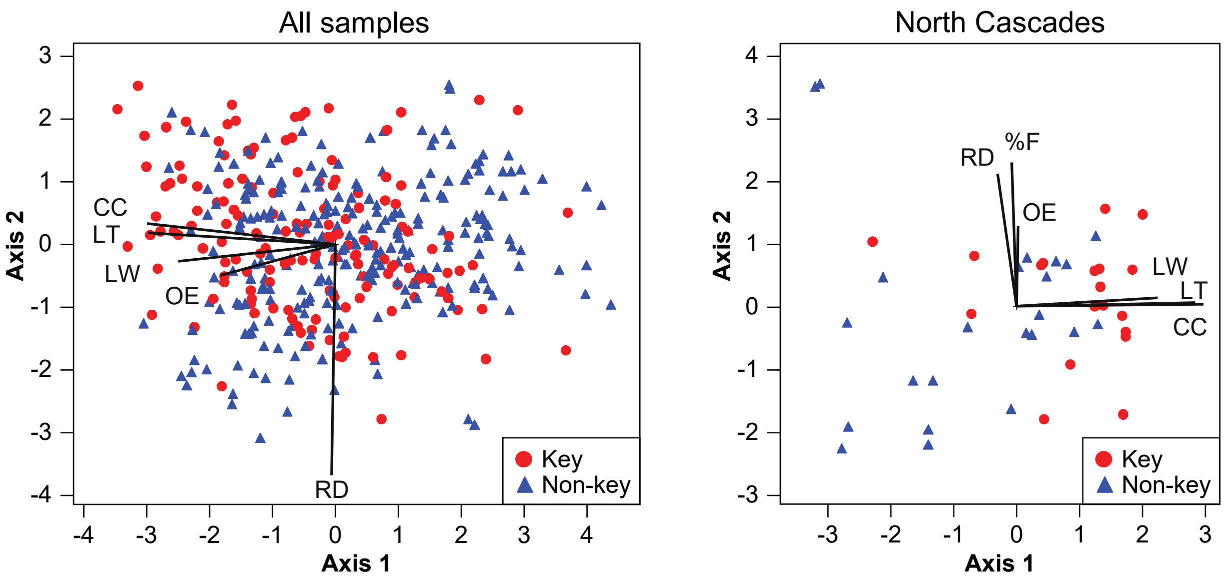

Figure 7.3—Principal components analysis ordinations for all watershed samples (n = 406) and North Cascades province (n = 42) grouped by key and non-key watersheds. The North Cascades province showed the strongest group dissimilarity (multi-response permutation procedure, A = 0.19). RD = road density, OE = observed to expected macroinvertebrate values, LT = riparian large trees, CC = canopy cover, LW = large wood, %F = percentage of fines.

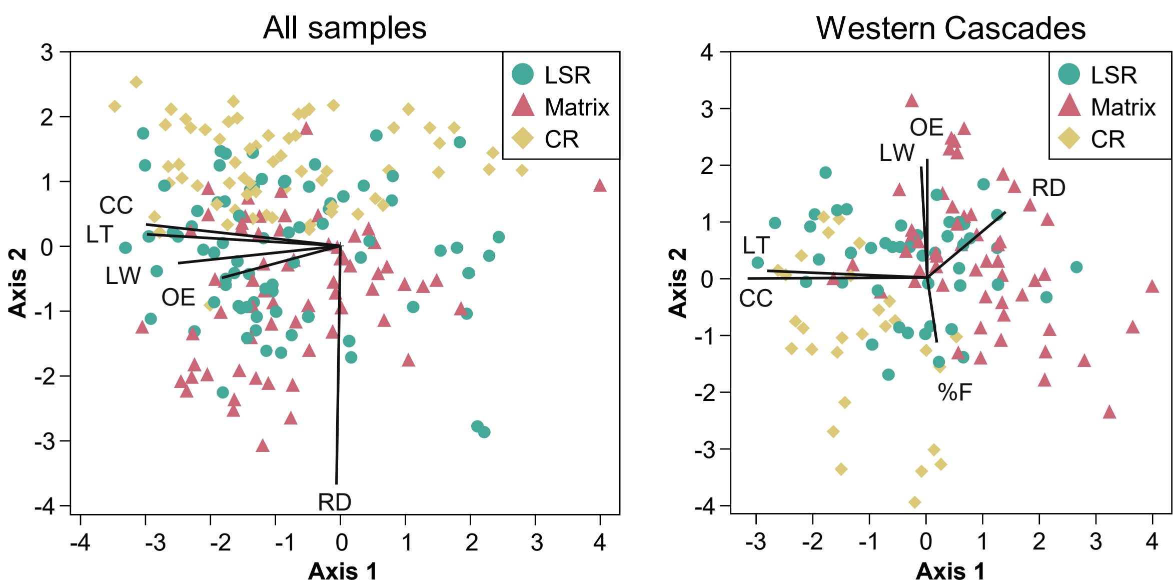

Figure 7.4—Principal components analysis ordinations for all watershed samples (n = 406) and Western Cascades province (n = 129) grouped by federal land use allocation. The Western Cascades showed the strongest group dissimilarity (multi-response permutation procedure, A = 0.13). RD = road density, OE = observed to expected macroinvertebrate values, LT = riparian large trees, CC = canopy cover, LW = large wood, %F = percentage of fines.

reserve watersheds had greater canopy cover and large tree densities, regardless of time period.

Discussion

Watershed condition is not explicitly defined, but it is generally accepted to comprise multiple physical and biological attributes supporting the function of a watershed. We combined a subset of instream and upslope metrics to represent related processes in sampled watersheds, such as selecting road density to represent the road-related sediment risk factors discussed in previous chapters. Our integrated assessment of watershed condition was a multivariate ordination approach allowing for comparisons of different groupings of watersheds using a suite of variables. This approach is intended to allow us to better interpret the correlation among variables simultaneously providing insights regarding the complex interaction between land management and aquatic conditions. Our analysis was tailored to test targeted management questions such as differences between LUAs and effectiveness of key watershed designations. This multivariate approach provided integration across datasets and spatial extents to better interpret broad-scale patterns across the AREMP area.

The overall suite of watershed conditions differed between aquatic provinces and LUAs. There were no overall statistical differences between the first and second time periods across the AREMP area when provinces were combined. However, we did find a difference between time periods in the Washington-Oregon Coast Range province, which showed large gains in vegetation-related attributes and a reduction in roads between the two time periods consistent with results from the individual metric analyses. This aquatic province has historically experienced intensive timber harvest and has a wetter climate conducive to rapid tree growth, possibly contributing to observed differences over time. Thus, our ability to detect changes over time in an integrated assessment of watershed conditions will vary by aquatic province because of legacy conditions, ongoing disturbance regimes, current vegetation conditions, recovery rates, and amounts of active management.

There was considerable overlap in conditions between key and non-key watersheds across the combined provinces within the AREMP area. Where differences between key and non-key watersheds were observed, those differences were associated with greater percentage of canopy cover and riparian tree densities in key versus non-key watersheds. Multivariate analysis also found a slight difference between time periods 1 and 2 in non-key watersheds, but not in key watersheds. Similar to the individual response analyses, non-key watershed vegetation attributes increased more than key watersheds over time, which is consistent with lower initial conditions in non-key watersheds at the onset of the Northwest Forest Plan (NWFP), indicating recovery of watershed conditions. Analysis in multivariate space shows the alignment of several indicators of upslope condition and instream condition over time, and the differentiation between key and non-key watersheds.

Differentiation among LUAs was identified in the analysis, with limited overlap among matrix or congressional reserve watersheds, driven primarily by greater road densities in matrix lands. Some of the patterns in individual metrics observed earlier in this report did not emerge to differentiate among LUAs in the multivariate analysis when all provinces were combined. This could indicate that the differing physical characteristics within aquatic provinces masked differences among LUAs.

A strong result of the individual metric analyses, mirrored by the multivariate analysis, was the role of road densities. Matrix lands at the time of NWFP implementation had the highest impact levels in road-related metrics of LUAs. Trends in these road-related impacts decreased over time for matrix and late-successional reserve LUAs. This likely reflects the reasons for the original designations of watersheds into these two classifications and ongoing management action over the duration of the NWFP. This is consistent with observations by Steel et al. (2017) who found relationships between land ownership and salmonid habitat characteristics in western Oregon that linked both to the spatial location of those ownerships and inherent characteristics of the environment. While sites within the AREMP area are dominated by federal ownership, matrix lands tend to be those areas that have experienced more timber harvest in the past and are more likely to experience harvest since NWFP implementation. Congressional reserves are often in more remote, higher elevation locations and have experienced less harvest and other management actions. These differences influenced both their original designation and ongoing management, resulting in the overall differences in observed watershed conditions. However, the analysis indicated federal land allocations are, in some ways, becoming more similar over time as differentiation among designations becomes less clear.

The multivariate analysis provided a new way to think about watershed condition. The differences between provinces observed using multivariate techniques likely counterbalanced one another at the scale of the AREMP area, masking differences when analysis was completed without province-scale stratification. Comparisons within a province were effective at demonstrating not only which characteristics defined the province, but also what important changes were occurring that drive watershed condition. Shifts in condition, particularly associated with LUAs, were evident in this analysis and could be jointly tied to both instream and upslope processes, enhancing our understanding of linkages throughout watersheds. This analysis also supports an ongoing overall trend of improved conditions on federal lands as differentiation among watersheds from different LUAs declines.

Headwaters of the Upper Fish Creek watershed in the Umpqua National Forest. Photo by an Aquatic and Riparian Effectiveness Monitoring Program crew member.

Chapter 8: Conclusions

This report has marshaled insights from field data collection, spatial datasets, and a host of landscape models to evaluate the status and trends of aquatic resources in streams across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area over the past 25 years, since inception of the Northwest Forest Plan (NWFP). Despite their limitations, these approaches collectively provide important insights into observable patterns, as well as information for assessing driving processes behind them. We summarize major findings here and conclude with a vision for steps to further process and present information from this effort and thoughts on future directions for monitoring AREMP area aquatic ecosystems. AREMP data provides a foundation from which to assess future conditions as changes occur under the new USDI Bureau of Land Management and USDA Forest Service land management plans, as well as planning processes implemented by other federal agencies in the NWFP area.

We considered three major human-related drivers of status and trends in aquatic resources that reflect prescriptions from the NWFP aquatic conservation strategy: forest conditions, roads, and road-stream crossings. Forest conditions were evaluated within a refined definition of riparian management areas. As may be expected, changes in forest condition (i.e., forest structure, densities of large trees, and canopy cover in riparian management areas) in the time following NWFP implementation (1994–2017) have been incremental when viewed across the AREMP area, and they are often highly variable when considered among subwatersheds, likely owing to the influences of numerous local factors (e.g., intensive restoration efforts, severe wildfire, differences in initial condition, or conditions that influence afforestation). Collectively, these changes indicate trends toward what would be considered desirable conditions within the NWFP aquatic conservation strategy. Canopy cover and large-tree densities near streams incrementally increased, and forest structure more closely resembles characteristics of old-growth forests at 80 years. Similarly, assessments of connected road length, estimated chronic sediment delivery, and shallow landslide risk from forest roads indicated incremental reductions. Finally, although we were unable to determine the status of road-stream crossings prior to the NWFP, we were able to identify hundreds of crossings that were likely replaced to allow for improved passage of fish, security of road transportation networks, and improved geomorphic functioning. Through these assessments, AREMP was able to document management actions completed at a scale resulting in discernible changes in aquatic conditions. We identified a host of road-stream crossings with no passage status information; these crossings can be targeted for future assessments to focus restoration efforts. Overall, these findings indicate that a quarter century of broad-scale forest recovery combined with targeted forest, road, and stream management under the NWFP have led to actions and observable outcomes that improve watershed condition.

Changes in human-related drivers were accompanied by corresponding changes in instream conditions that signal improvements. These included widespread declines in fine sediment observed in stream channels, improved biotic condition as indicated by macroinvertebrates, and some increases in quantity of the smaller size class of instream wood (number of pieces per channel area). For instream wood, we also observed declines in the quantity of the largest size class, presumably as recruitment of these largest pieces mainly occurred prior to the extensive removal of old-growth forests preceding the NWFP. Recovery of larger wood recruitment from older trees occurs on time scales that far exceed the 25-year period of this report. If trends in forest and watershed conditions continue along current trajectories, we would expect eventual recovery of natural large wood recruitment processes, likely over longer temporal extents (Martens et al. 2020). We were unable to evaluate changes in temperature due to the nature of sampling and short durations of available time series. However, we were able to quantify where temperatures are suitable for supporting coldwater species, with important patterns of local and latitudinal variability noted (e.g., warmer temperatures in the southern margins of the AREMP area). Temperature data collected by the AREMP comprises a critical foundation for future evaluation of how it responds to both forest management and changing climatic conditions.

We used a multivariate statistical approach to collectively evaluate a subsample of indicators monitored by the AREMP. Insightful differences emerged among aquatic provinces and management allocations. Some statistically significant trends related to provinces and land management were observed over time, but area-wide trends were not detected. For example, the Washington-Oregon Coast Range province that experienced high levels of historical forest harvest and has growing conditions that facilitate more rapid forest recovery exhibited positive trends. Conditions across land use allocations are generally becoming more similar to the least modified land use allocation (Congressional reserves) over time, indicating potential recovery of lands historically subjected to more intensive forestry practices.

Detailed analyses of all responses in this report were conducted within land use allocations (LUAs) and across designated key and non-key watersheds, as delineated by the NWFP aquatic conservation strategy. By partitioning analyses into these divisions, we were able to see how conditions changed across lands with contrasting management prescriptions (e.g., matrix lands versus congressional reserves) (see table A1.1) and specific watersheds (key watersheds) delineated for their high-quality water and potential benefits to fish. Initial conditions and rates of change differed between LUAs and key watershed designations. Key watersheds were often in better condition at the initiation of the NWFP than non-key watersheds, reflecting their identification as important fish habitat and water sources. Often these more detailed assessments provided insights that were not evident through a wholesale assessment of the AREMP area as a single unit. Linking these management allocations to ecological status and trends is challenging due to the variety of possible drivers, such as differing prior conditions, influences of nonfederal lands, changes owing to forest fires, or other landscape changes, as well as changes attributable to localized active restoration on the landscape. More comprehensive evaluations of the drivers behind these patterns require additional study.

This is the first AREMP monitoring report to explicitly address climate change, which influences the entire area and interacts with regional management prescriptions as well as local restoration activities to influence outcomes (Spies et al. 2019). We evaluated climate-related changes through changes in a hydrologically relevant drought index and modeled stream discharge. Both indicated overall trends that would be expected from warming climatic conditions across the landscape with variability among provinces and over time. The trends in these responses paralleled observed declines in surveyed wetted widths. Although the three lines of evidence considered here point to the potential pervasive influences of a changing climate on aquatic ecosystems across the AREMP area, it is important to recognize that identifying causal relations would require more intensive study. This caveat aside, evidence in this report supports the notion that both changes in forest management and climate change are likely working to influence observed outcomes for watershed condition. Accordingly, information from this effort should provide invaluable information for follow-up regional climate vulnerability assessments, climate adaptation planning, and development of more localized plans to address land uses and climate change (e.g., Halofsky et al. 2018).

Assembling the information presented herein required considerable effort, but now the AREMP is effectively poised to address the growing urgencies facing water resources, aquatic ecosystems, and native fishes on federal lands in the future. Additional data releases and analyses of results provided herein are planned for the immediate future to provide more detailed and quantitative interpretations of watershed condition and the influences of landscape and climate change. Some modification of monitoring protocols may be warranted in response to emerging management questions, such as streamflow permanence, biological invasions, and carbon sequestration and storage. Incorporation of additional monitoring questions is consistent with a long-term monitoring program designed to respond to emerging topics and is in keeping with past practices of the AREMP, such as the addition of thermal and biodiversity monitoring (e.g., eDNA). As the Pacific Northwest's ecosystems and society witness more changes in the 21st century, broad-scale monitoring programs such as the AREMP will continue to play a vital role in understanding the effects of federal land management on water resources that are critical to the region for drinking water, irrigation, fisheries, recreation, and biodiversity.

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U.S. Equivalents

When you know:	**Multiply by:**	**To find:**
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Hectares (ha)	2.47	Acres
Square kilometers (km^2)	0.386	Square miles
Kilometers (km)	0.621	Miles
Trees per hectare	0.405	Trees per acre
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit

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Appendix 1: Study Area

Overview

The Northwest Forest Plan (NWFP) covers about 99,000 km² (24.5 million ac) of federal lands in western Washington, Oregon, and northwestern California. About 79 percent of the NWFP area is managed by the USDA Forest Service, 11 percent is managed by the USDI Bureau of Land Management (BLM), 9 percent is managed by the National Park Service, and <1 percent is managed by the USDI Fish and Wildlife Service (as national wildlife refuges) and the Department of Defense (fig. A1.1). Within these jurisdictions, the NWFP and its aquatic conservation strategy (USDA FS and USDI BLM 1994a) recognize several land management classifications, including riparian management areas, key watersheds, congressional reserves, late-successional reserves, and matrix lands (table A1.1). In addition, physiographic provinces were also used in the ecological design of the NWFP and continue to be used for reporting purposes. These classifications are described in subsequent sections.

Aquatic Provinces

The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) monitors the status and trends in watershed conditions across the NWFP region. Ecologically, the resulting AREMP area can be considered in terms of physiographic provinces. Physiographic provinces were initially used by the Forest Ecosystem Management Assessment Team (FEMAT 1993) to produce the NWFP aquatic conservation strategy and other assessments (Thomas et al. 2006); they were intended to describe spatial variability in processes that drive the functioning of terrestrial and aquatic ecosystems. Physiographic provinces used here were derived from aquatic province boundaries based on Bryce et al. (1999) for Oregon and Washington, Bailey et al. (1994) for California, and the USDA Forest Service Pacific Northwest Region sixth-field hydrologic unit code (HUC) watershed layer for the Cascade Range crest. Overall, the NWFP is represented by eight aquatic physiographic provinces: Olympic Peninsula, North Cascades, Puget-Willamette Trough, Western Cascades, Washington-Oregon Coast Range, High Cascades, Klamath-Siskiyou, and Franciscan (fig. A1.1). Most of these provinces include substantial federal ownership, with the exception of the Puget-Willamette Trough, which is predominantly private (see fig. A1.1, A1.2).

Land Use Allocations

Standards and guidelines for the management of federal lands in the NWFP area are linked to land use allocations (LUAs). There are eight LUA categories that were recognized in the NWFP. For the purpose of assessing effectiveness of the NWFP for aquatic ecosystems, the AREMP categorizes these classes into four groups that reflect major differences in how lands are managed (table A1.2): congressional reserves, late-successional reserves (LSRs), riparian management areas (RMAs), and matrix.

Key and Non-Key Watersheds

Key watersheds were defined under the NWFP aquatic conservation strategy as areas intended to “serve as refuge for aquatic organisms, particularly in the short term for at-risk fish populations, to have the greatest potential for restoration or to provide sources of high-quality water” (Haynes et al. 2006). Habitat-focused key watersheds were designated as Tier 1, and water quality-focused key watersheds as Tier 2; however, for this report, no distinction was made between the tiers.

Key watersheds are independent of the LUAs in the NWFP, thus their designations overlay the other LUAs. Key watershed delineation began prior to the development of the interagency standard fifth- and sixth-field watershed boundaries, so their boundaries are not always coincident. For the analyses presented in this report, actual key watershed boundaries were used (as opposed to assigning key watersheds to subwatershed boundaries as done in past reports). Key watersheds cover ~37 000 km² of the ~99 000 km² area of the NWFP.

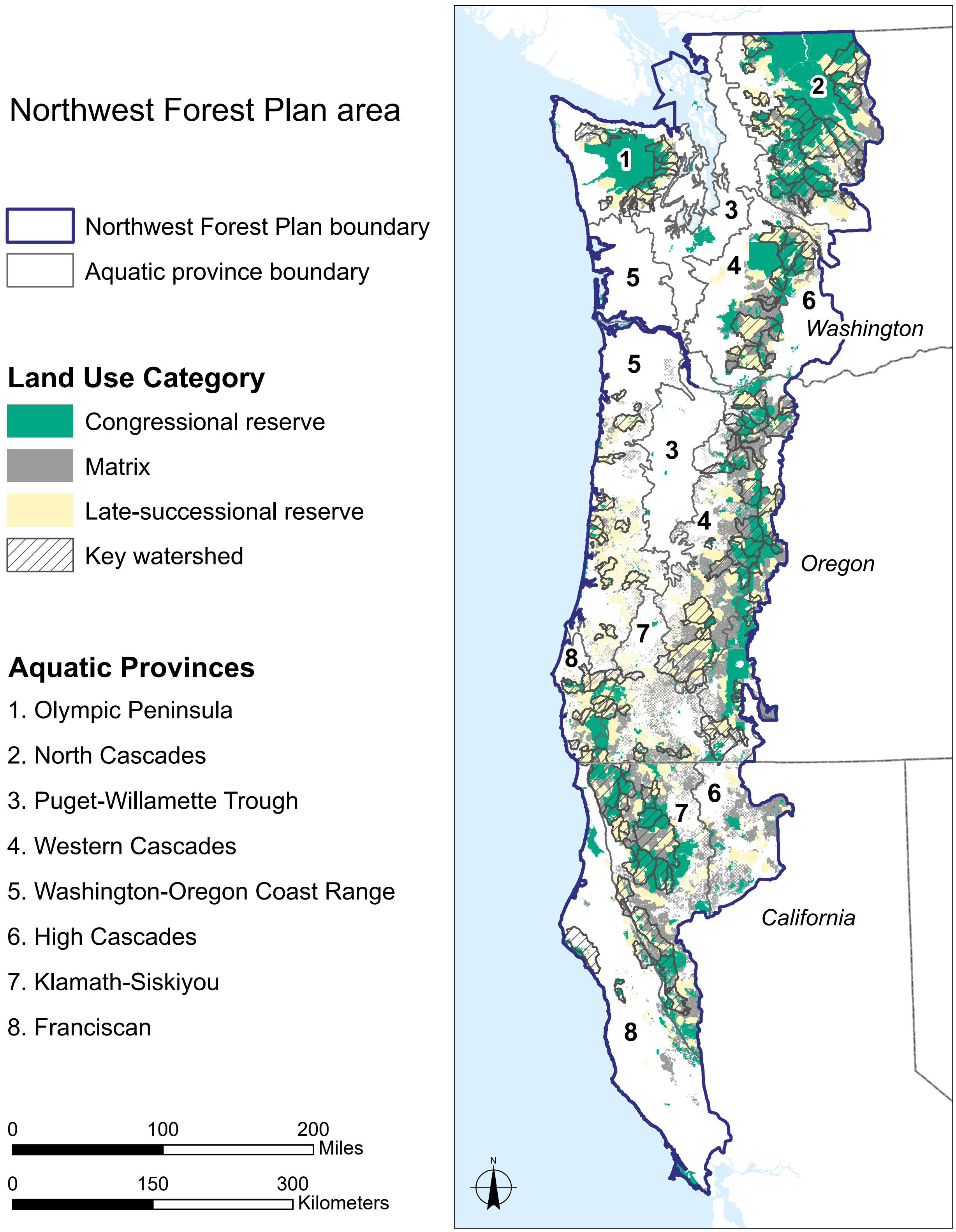


Figure A1.1—Northwest Forest Plan extent (Aquatic and Riparian Effectiveness Monitoring Program area), aquatic provinces, land use allocations, and key watersheds.

Table A1.1—Land areas, percentages, and numbers of Aquatic and Riparian Effectiveness Monitoring Program-sampled subwatersheds within classes, provinces, riparian management areas, land use allocations, and key watershed designations.

		Subdivision		Riparian		Subwatershed	
Division	**Subdivision**	**Area**	**Percentage**	**Area**	**Percentage**	**Sampled**	**Percentage**
		km²	*Percent*	*km²*	*Percent*	*Number*	*Percent*
NWFP							
	NWFP	100 128	100	28 049	28	219	100
Province							
	Klamath/Siskiyou	23 459	23	6620	28	60	27
	Western Cascades	21 398	21	7433	35	66	30
	North Cascades	19 831	20	4435	22	23	11
	High Cascades	16 706	17	2805	17	30	14
	Olympic Peninsula	6166	6	1625	26	10	5
	WA/OR Coast Range	6075	6	3332	55	25	11
	Franciscan	5769	6	1609	28	5	2
	Puget-Willamette Trough	724	0.7	189	26	0	0
Land use allocation							
	Congressional reserves	37 718	38	8821	23	54	25
	Late-successional reserves	31 350	31	10 499	33	89	41
	Matrix	31 060	31	8730	28	76	35
Watershed class							
	Non-key	62 740	63	17 133	27	138	63
	Key	37 387	37	10 915	29	81	37

Subwatersheds were not sampled in the Puget-Willamette Trough province as it represents <1 percent of the NWFP area.
NWFP = Northwest Forest Plan.

Hydrologic Units

Assessing the effects of land cover on streams generally relies on the delineation of hydrologic units, which are subdivisions of the landscape that drain to a particular stream network. This analysis used the National Watershed Boundary Dataset (Seaber et al. 1987; USGS and USDA 2013). The AREMP monitoring plan chose hydrologic unit code HUC12 (subwatersheds) as its basic analysis unit; it was the finest level of resolution of boundaries at the time (Reeves et al. 2004). It should be noted that these are not true complete watersheds because often the outflow of one HUC12 will be the inflow to another HUC12, so they are not independent (Omernik et al. 2017). However, true watersheds can vary extensively in area, and the National Watershed Boundary Dataset was chosen to provide a more homogeneous division of the landscape in terms of unit areas.

Selection of Subwatersheds for Upslope and Riparian Analysis

As with the 20-year report, all subwatersheds with ≥5 percent federal ownership were selected for upslope and riparian analysis. This threshold was chosen to be consistent with the USDA Forest Service national watershed assessment guidelines (USDA FS 2011a, 2011b) and to include more BLM lands, many of which occur in the highly fragmented “checkerboard” of public/

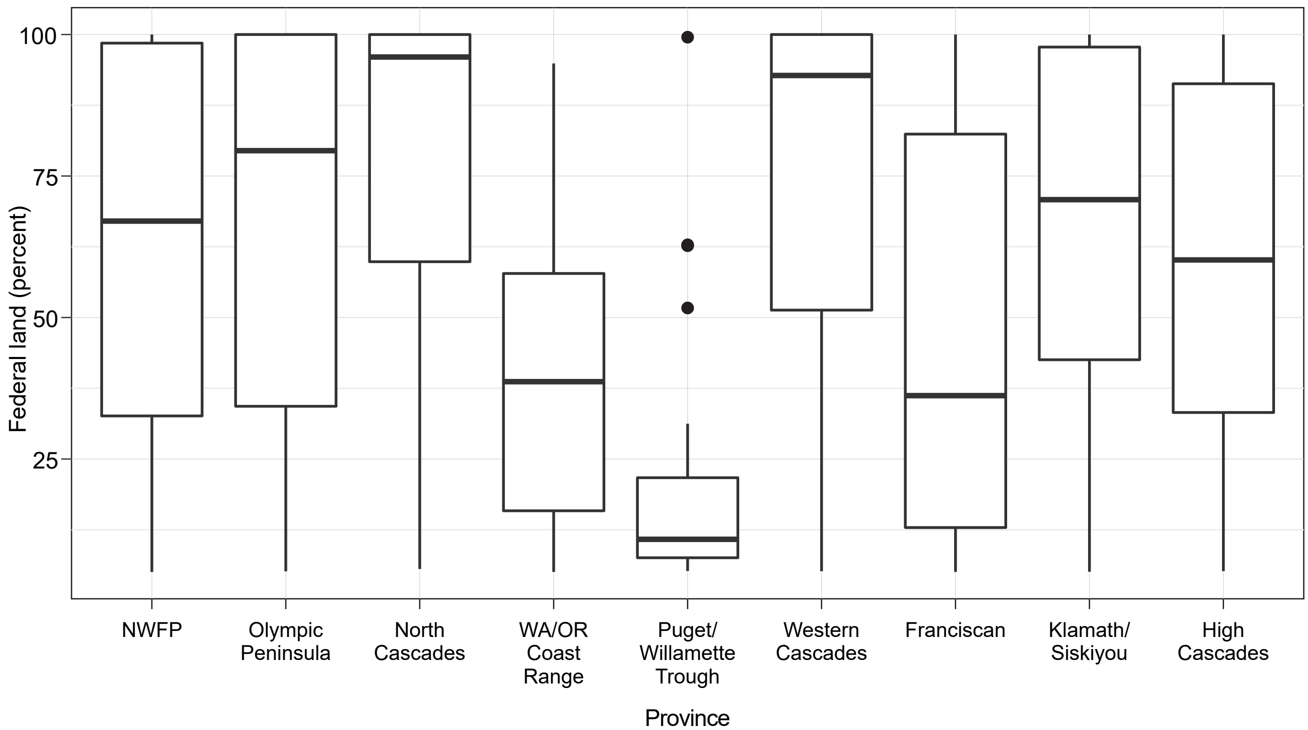


Figure A1.2—Percentage of federal land per subwatershed by aquatic province, for subwatersheds (n = 1,972) within the footprint of the Northwest Forest Plan (NWFP) with ≥5 percent federal ownership.

Table A1.2—Land use allocation (LUA) groups within the Northwest Forest Plan

Land use allocation	LUA group	Description
Congressionally reserved areas	Congressionally reserved	Lands reserved by the U.S. Congress, such as wilderness areas, wild and scenic rivers, and national parks and monuments.
Administrative withdrawn areas	Congressionally reserved	Areas identified in local forest and district plans; they include recreation and visual areas, back country, and other areas where management emphasis does not include scheduled timber harvest.
Late-successional reserves	Late-successional reserves	Lands reserved for the protection and restoration of late-successional and old-growth forest ecosystems and habitat for associated species, including marbled murrelet reserves (LSR3) and northern spotted owl activity core reserves (LSR4).
Managed late-successional areas	Late-successional reserves	Areas for the restoration and maintenance of optimum levels of late-successional and old-growth stands on a landscape scale where regular and frequent wildfires occur. Silvicultural and fire hazard reduction treatments are allowed to help prevent older forest losses from large wildfires or disease and insect epidemics.
Adaptive management areas—reserved	Late-successional reserves	Identified to develop and test innovative management to integrate and achieve ecological, economic, and other social and community objectives. Emphasis on restoration of, and management as, late-successional forests.
Adaptive management areas—non-reserved	Matrix	Identified to develop and test innovative management to integrate and achieve ecological, economic, and other social and community objectives. Some commercial timber harvest was expected to occur in these areas, but with ecological objectives.
Matrix	Matrix	Federal lands outside of reserved allocations where most timber harvest and silvicultural activities were expected to occur.
Riparian management areas	Riparian management areas	Protective buffers along streams, lakes, and wetlands designed to enhance habitat for riparian-dependent organisms, provide good water-quality and dispersal corridors for terrestrial species, and provide connectivity within watersheds.

Source: USDA FS, n.d.

private lands. Only the federal portion of subwatersheds was included when determining subwatershed condition because federal agency land managers have no jurisdiction over management of nonfederal lands. Overall, the NWFP area contains 2,810 subwatersheds, of which 2,039 contain some land that is federally owned, and of that number 1,972 have at least 5 percent federal ownership by area. The 5-percent ownership criterion excludes about 1 percent of the federal lands within the NWFP area from this analysis. For reporting purposes, we further subdivided these subwatersheds by the NWFP LUAs and key/non-key watersheds and focused on riparian management areas (as described in the next section).

Riparian management areas

RMAs were delineated on federal lands using USDI Geological Survey National Hydrography Dataset (NHD) Flowline (1:24,000 high resolution), NHD Area, NHD Waterbody (USGS 2019b), National Wetlands Inventory (USFWS 2018), and fish distribution layers from different agencies (AREMP 2019, USDA FS 2017, USDI BLM 2019c). RMA widths were mapped according to the following standards from the NWFP aquatic conservation strategy:

- Fish-bearing streams—a distance equal to the height of two site-potential trees, or 300-ft distance, whichever is greatest. Applies to all fish-bearing streams.
- Non-fish-bearing perennial streams—a distance equal to the height of one site-potential tree, or 150-ft distance, whichever is greatest.
- Nonfish-bearing intermittent streams and all others, including ephemeral and not coded (e.g., FCODE = 46000)—extension from the edges of the stream channel to a distance equal to the height of one site-potential tree, or 100-ft distance, whichever is greatest.
- Constructed ponds and reservoirs, and wetlands >1 ac—a distance equal to the height of one site-potential tree, or 150-ft distance from the edge of the wetland >1 ac, whichever is greatest.
- Lakes and natural ponds—a distance equal to the height of two site-potential trees, or 300-ft distance, whichever is greatest.
- Wetlands <1 ac—extension from the edges of the stream channel to a distance equal to the height of one site-potential tree, or 100-ft distance, whichever is greatest.

Site potential tree height was available for Forest Service (USDA FS 2019b) and BLM lands (USDI BLM 2017), so it was incorporated into the buffer widths. These data were not available for other federal lands, so the default minimum values were used. It was not feasible to model some aquatic conservation strategy criteria at this scale, including 100-year floodplains, unstable areas, and inner gorges. Additionally, all buffer width distances are horizontal (2-D) distance and not slope distance.

Selection of Subwatersheds and Sites for Field Sampling

Selection of subwatersheds and sites for AREMP field sampling began with a hierarchical selection of subwatersheds and sites within them, following sample survey designs developed for the Environmental Protection Agency National Rivers and Streams Assessment (Gallo et al. 2005, Paulsen et al. 2008, Stevens and Olsen 2004, Stevens et al. 2007). To be included in the sample draw, a subwatershed had to include at least 25 percent federal land ownership along the length of its major streams (as delineated by the 1:100,000 NHD, USGS 2019a) (Gallo et al. 2005), a percentage considered to represent a significant contribution of federal lands to watershed condition (Reeves et al. 2004). Within subwatersheds, the sample selection procedure resulted in an ordered list of sites (points). The selection included an "oversample" to account for the possibility of excluding initially selected sites (e.g., lack of access, safety, logistical considerations) (fig. A1.3).

Over the 17 years of field sampling by the AREMP (2002–2018) covered in this report, several changes to the original sample design were needed. The initial sample included 250 HUCs, with a plan of sampling 50 each year and repeating visits every 5 years (Reeves et al. 2004). Within each HUC, 80 sites were selected in first- through third-order streams (Strahler 1957). Full implementation of the original sampling design for the AREMP was not possible due to limited resources and logistical constraints. Currently, the AREMP samples 219 of the 250 originally selected subwatersheds. Removal of subwatersheds from

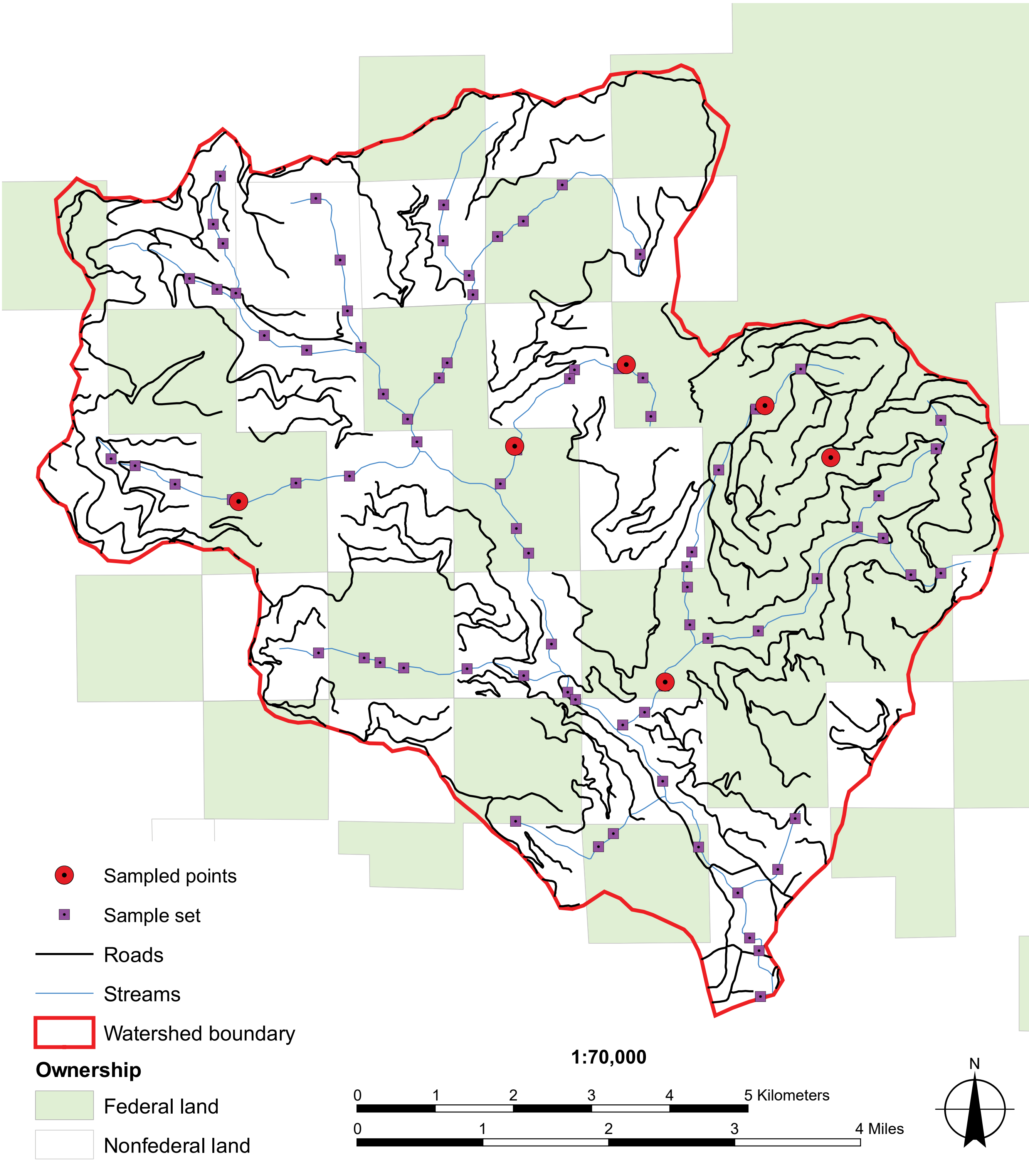


Figure A1.3—Site selection within a subwatershed sampled by the Aquatic and Riparian Effectiveness Monitoring Program (from Gallo et al. 2005) with patterns of ownership and sample data (Stevens and Olsen 2004).

field sampling by the AREMP occurred for a host of reasons, including safety considerations (e.g., presence of illegal activities, wildfire hazards), inaccessibility (e.g., lack of road access or reserved for cultural reasons), environmental conditions (e.g., dry stream channels), and revision of subwatershed boundaries. In 2011, HUC boundaries were updated, resulting in some HUCs being split. In cases of split HUC boundaries, the newly delineated HUC with the greatest number of sampled points was retained. Due to funding limitations, repeat sampling was extended from 5- to 8-year intervals. In practice, sampling intervals varied around a median of 8 years (fig. A1.4), with some sampling repeated in consecutive years to informally assess sites for annual variability in measured responses. In addition to these intervals, in some cases, different crews visited the same sites within the same year for the purpose of evaluating repeatability of field measurements. Repeat visits within the same year occurred on average in 11 percent of subwatersheds per year. All of this information was used in statistical models for evaluating trends in measured responses

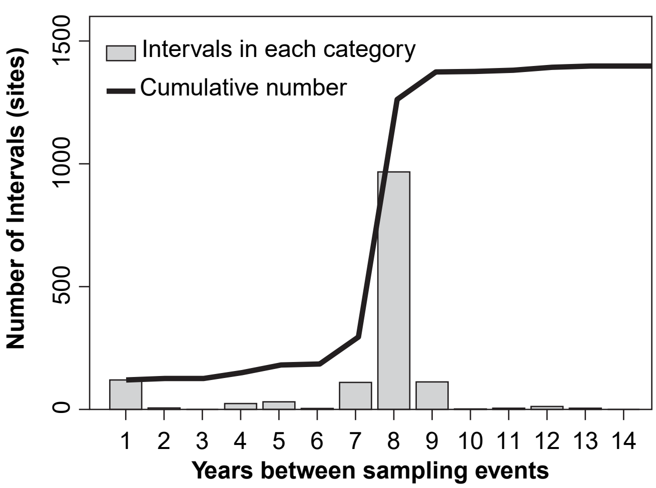


Figure A1.4—Distribution of sampling intervals (number of years between revisits) for sites sampled within subwatersheds monitored by the Aquatic and Riparian Effectiveness Monitoring Program, 2002–2018. The absolute number of revisits within each year category are shown, as well as the cumulative number of sites with respect to sampling intervals.

(app. 2). Repeat visits to sites within subwatersheds in the same year are referred to in subsequent sections as quality assurance/quality control (QA/QC) surveys. As with sampling intervals, available resources and other logistical constraints limited the number of sample sites per subwatershed to a range of 4 to 11 sites.

Types of Streams Sampled by the AREMP

Subwatershed size

For the 219 HUC12 subwatersheds sampled by AREMP, subwatershed size ranged from 33.6 to 193.2 km². The mean subwatershed size was 82.7 km². Across the AREMP sampled subwatersheds, size varied somewhat by aquatic province (fig. A1.5). The North Cascades aquatic province had the largest median subwatershed size (89.5 km²) followed by the Franciscan (86.8 km²), and High Cascades (83.4 km²). The Washington-Oregon Coast Range had the smallest median subwatershed size (60.4 km²), and the Western Cascades had the largest range in size (33.6 to 193.2 km²).

Subwatershed elevation

AREMP-sampled subwatersheds ranged in mean elevation from 108.8 to 2068.7 m with a mean elevation of 1002.1 m. High Cascades province had the highest mean elevation (1383.5 m) followed by the North Cascades province (1273.7 m) (fig. A1.6). The Washington-Oregon Coast Range had the lowest mean elevation (335.5 m). Across LUA classes, subwatersheds classified as congressionally reserved (dominant federal LUA class) had higher mean elevations (1250.8 m) than those classified as matrix (1032.0 m) or LSR (825.6 m) (fig. A1.7). Mean values between key and non-key watersheds were similar: 987.3 m and 1000.8 m, respectively (fig. A1.8). However, non-key watersheds had a larger range in mean elevation (108.8–2068.7 m) than key watersheds (164.9–1812.6 m).

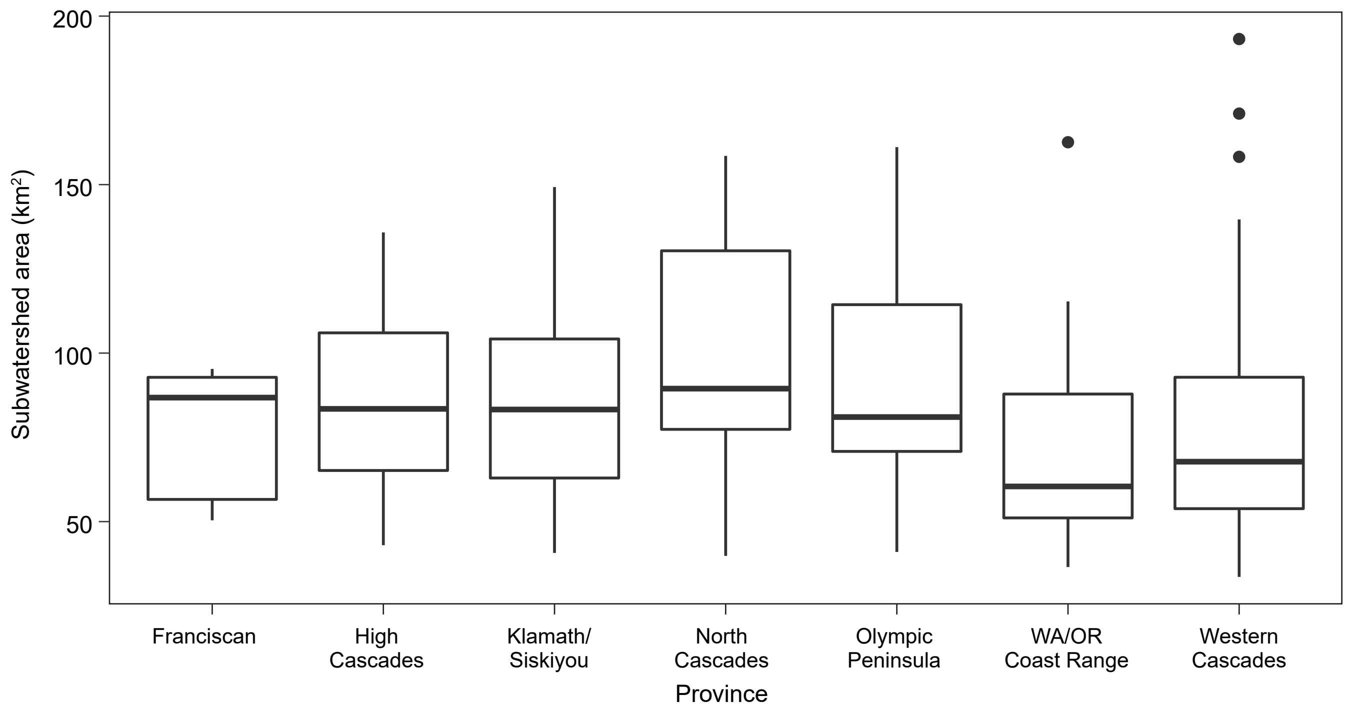


Figure A1.5—Distribution of subwatershed area by aquatic province. Horizontal lines inside of boxes represent median values, outer edges of the boxes represent the 75th and 25th quantiles, whiskers indicate the interquartile range × 1.5, and the points represent outliers. Note that the Puget-Willamette Trough subdivision is not included as it represents only <1 percent of the NWFP area (table A1.1).

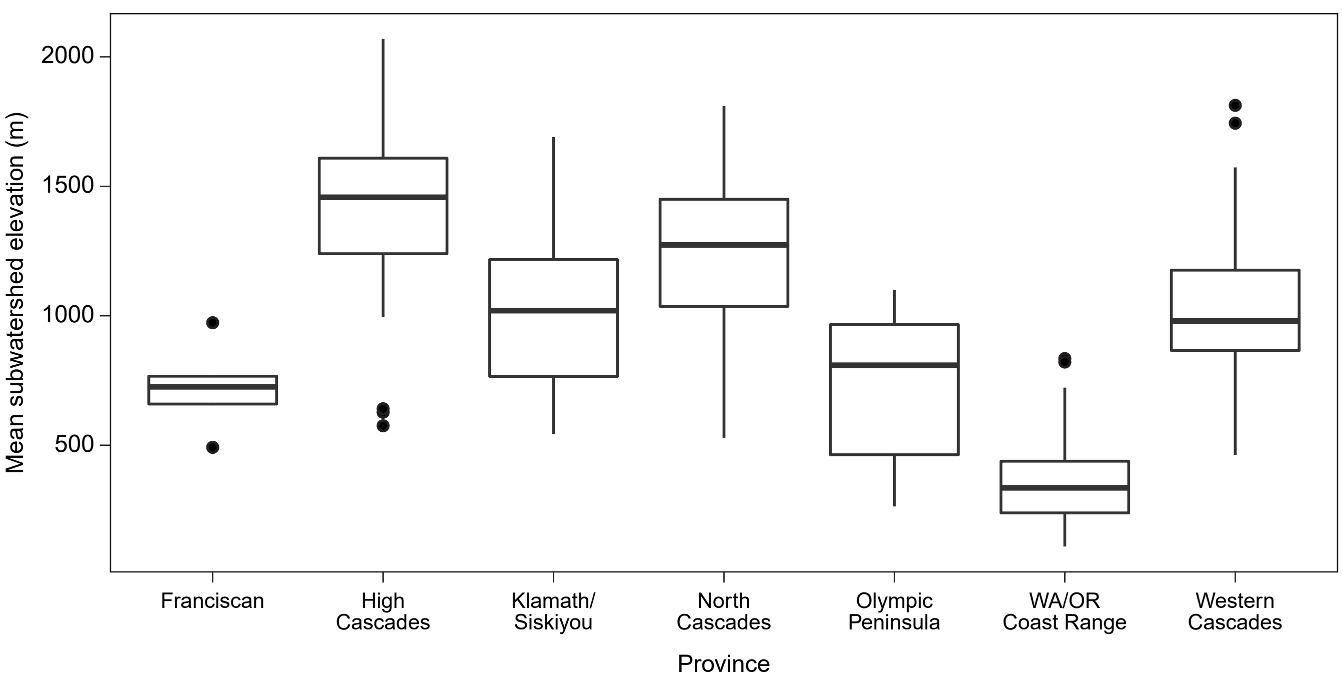


Figure A1.6—Distribution of mean subwatershed elevation by aquatic province (values for the mean of all elevations within a watershed within each aquatic province). Horizontal lines inside of boxes represent median values, outer edges of the boxes represent the 75th and 25th quantiles, whiskers indicate the interquartile range × 1.5, and the points represent outliers. Note that the Puget-Willamette Trough subdivision is not included as it represents only <1 percent of the NWFP area (table A1.1).

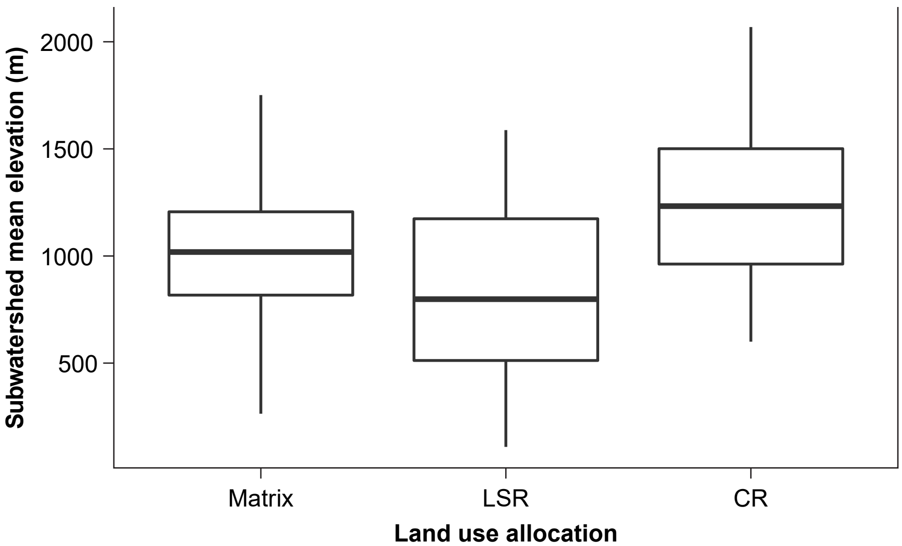


Figure A1.7—Distribution of mean subwatershed elevation by dominant federal land use allocation. LSR = late-successional reserve, CR = congressional reserve. Horizontal lines inside of boxes represent median values, outer edges of the boxes represent the 75th and 25th quantiles, and the whiskers indicate the interquartile range × 1.5.

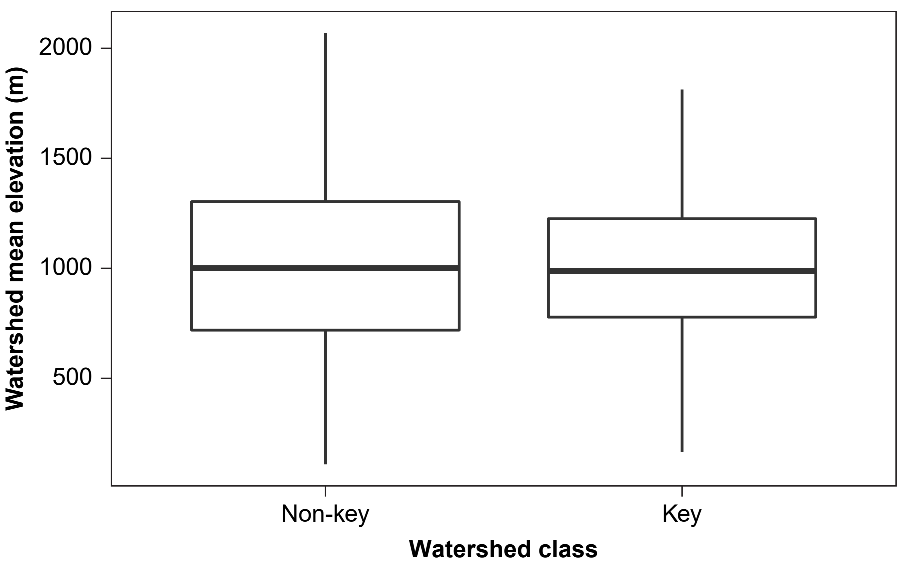


Figure A1.8—Distribution of mean watershed elevation by watershed class. Horizontal lines inside of boxes represent median values, outer edges of the boxes represent the 75th and 25th quantiles, and the whiskers indicate the interquartile range × 1.5.

Site Level: Order, Gradient, Intermittent/ Perennial, Fish-Bearing

AREMP instream sampling sites encompass the wadeable hierarchy of stream types from first-order, high-gradient streams to higher order, low- and mid-gradient streams. The sampling sites encompass fish-bearing and non-fish-bearing streams across varying levels of stream permanence.

Stream order is a way of classifying streams based on their position in a river network with increasing stream order correlated with larger watershed sizes. According to Strahler (1957) stream order classes from the NHDPlus 1:100000 resolution stream network, AREMP-sampled stream sites ranged in order from 1 to 7 (fig. A1.9). Sites classified as stream order 3 and 4 were the most frequently sampled, comprising 59 percent of sites sampled. Across aquatic provinces, LUAs, and key/non-key watershed classes, the distribution of stream orders sampled were similar. The distribution of stream reach gradient across sampled sites ranged from 0 to 46.1 percent, with a mean of 7.1 percent. Seventy-five percent of sites had a gradient <10 percent. Mean stream reach gradient values were similar across LUAs and between key and non-key watersheds. Mean and maximum gradient values varied by aquatic

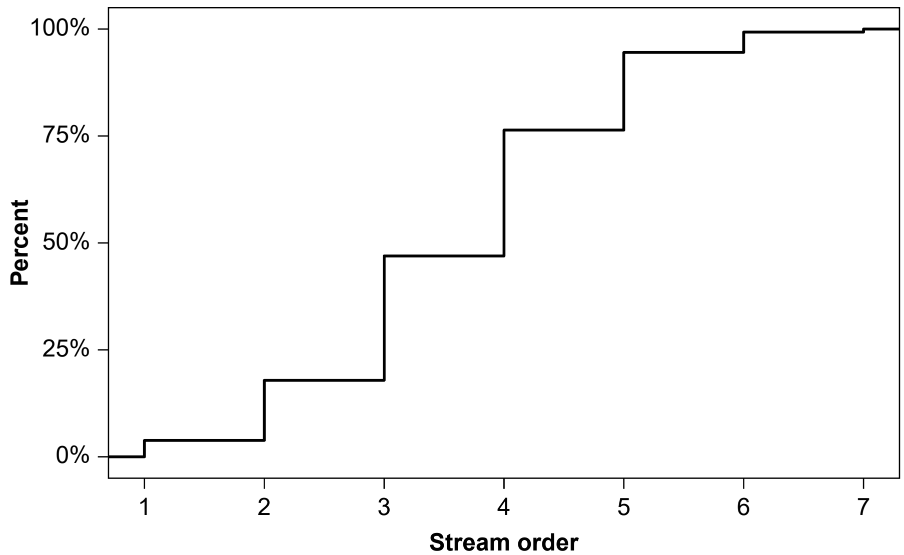


Figure A1.9—Cumulative distribution of Strahler (1957) stream order values for the Aquatic and Riparian Effectiveness Monitoring Program sites derived from stream reaches on the NHDPlus 1:100000 stream network.

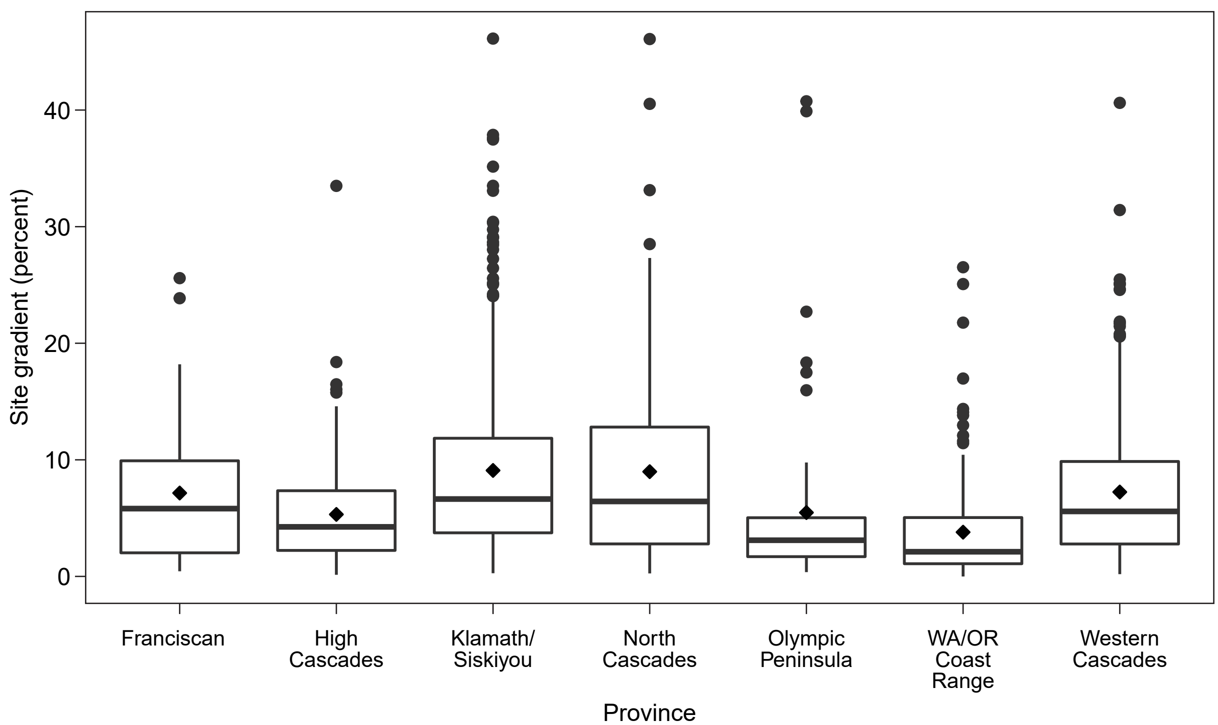


Figure A1.10—Distribution of site gradient measured in the field by aquatic province. Diamonds represent mean values. Horizontal lines inside of boxes represent median values, outer edges of the boxes represent the 75th and 25th quantiles, whiskers indicate the interquartile range × 1.5, and points beyond these are outlying observations. Note that the Puget-Willamette Trough subdivision is not included as it represents only <1 percent of the NWFP area (table A1.1).

province (fig. A1.10). The Klamath-Siskiyou (9.1 percent) and North Cascades (9.0 percent) had the highest mean and maximum values. The Washington-Oregon Coast Range (3.8 percent) showed the lowest mean values.

The majority of sites sampled were classified as perennial (85 percent) based on field measures of wetted width. Fish-bearing or non-fish-bearing reaches were categorized from the fish distribution layer assembled by AREMP (see “Riparian management” areas above). Non-fish-bearing streams comprised 30 percent of sites. We combined the measure of stream permanence with the fish-bearing category to derive four categories: intermittent without fish, perennial without fish, intermittent with fish, and perennial with fish. Perennial with fish comprised the majority of stream reaches with 63 percent of the sites. The next most common stream class sampled was perennial without fish (22 percent). Intermittent streams with fish (7 percent) and without fish (8 percent) were the least common stream reach types. Commonly used stream classification systems (e.g., Montgomery and Buffington 1988, Rosgen 1994) use stream gradient along with other morphological features (e.g., sinuosity, width-to-depth ratio, entrenchment) to categorize streams into homogenous stream types. Based on gradient categories from Rosgen (1994), the majority of stream sites sampled would be classified as a stream type that has a gradient from 4 to 10 percent (fig. A1.11).

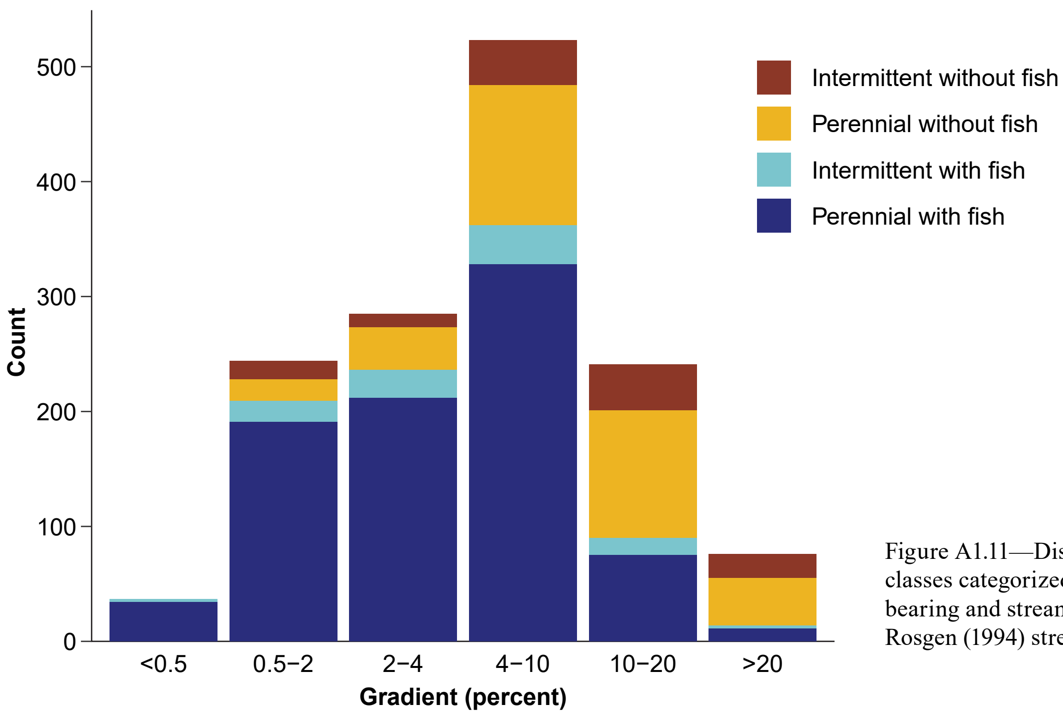


Figure A1.11—Distribution of common stream classes categorized by fish-bearing or non-fish-bearing and streamflow permanence across Rosgen (1994) stream gradient categories.

Appendix 2: Field-Based Instream Measurements—Methods and Analysis

Field Protocols

Temperature

Stream temperatures have been monitored across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area since 2002. Temperature loggers were deployed in the spring and downloaded in the fall for the first 9 years of monitoring (2002–2010). Beginning in 2011, temperature loggers were deployed on a year-round basis at most sites in Oregon and Washington. Exceptions include locations where temperature loggers cannot be easily accessed or are likely to fail in winter (e.g., due to washout of loggers). In California, temperature data loggers have been deployed in the spring and downloaded in the fall since monitoring began. Temperatures were recorded hourly and summarized as the 7-day moving average of the daily maximum temperatures. The average 7-day maximum temperature was calculated for the month of August (the most commonly available time period across sites).

Temperature loggers were deployed at the lowest point on federal lands within each sampled subwatershed whenever possible. If the federal land holdings were discontinuous, the water temperature logger was placed in the downstream-most continuous portion of federal land surveyed and downstream of all instream survey sites and tributaries. If the subwatershed was a composite watershed (a drainage basin that has upstream flow delivery from outside the basin) and the mainstem was not sampled, the thermograph was placed in the tributary with the most survey sites at the downstream-most point on federal land (fig. A2.1). Temperature loggers were housed inside a heavy-duty PVC pipe section and cabled or affixed with epoxy to a solid feature such as a boulder, tree, or bridge abutment. (fig. A2.2).

Wetted and Bankfull Stream Width

Wetted widths of streams were measured at transects, following methods described by Platts et al. (1983). Coinciding field measurements of bankfull stream widths (Harrelson et al. 1994) were collected at all sites at systemically spaced cross-section transects from 2002 to present day. At least five measures of bankfull width have been collected during every survey from 2002 to present.

From 2002 to 2003, the total number of bankfull widths collected at each site differed depending on whether a site was classified as constrained or unconstrained. Constrained reaches were identified as having a gradient >3 percent, while unconstrained reaches were identified as having a gradient <3 percent. In constrained reaches, only 5 measures of bankfull width were collected at transects, while in unconstrained reaches, 11 measures of bankfull width were collected at transects. Since 2004, 11 bankfull width measurements were collected at all transects regardless of whether the reach was classified as constrained or unconstrained.

Instream Wood

Instream large wood was defined by measures of diameter and length of pieces within the bankfull channel of sampled sites. Large wood diameter was measured (or estimated) at one-third of the total length of the piece from the base or largest end for pieces with diameter starting at 30 cm (since 2012, wood with diameters starting at 15 cm were also included). Survey crews measured the first ten pieces in a site, then every fifth piece up to 35 pieces, then every tenth piece is measured. The unmeasured pieces are estimated. Large wood definitions were designed to be consistent with criteria used by the Oregon Department of Fish and Wildlife's stream habitat surveys (Moore et al. 2014). Measurements of large wood were collected in all sites sampled across the 2002–2018 time period. As some measurements of wood evolved over time, we relied on the most consistent and longest tenured measures of wood from field surveys conducted by the AREMP. Data were summarized at the subwatershed scale as three wood size class densities (number of pieces/100 m) from 2002 through 2018 defined by length and diameter (table 4.1).

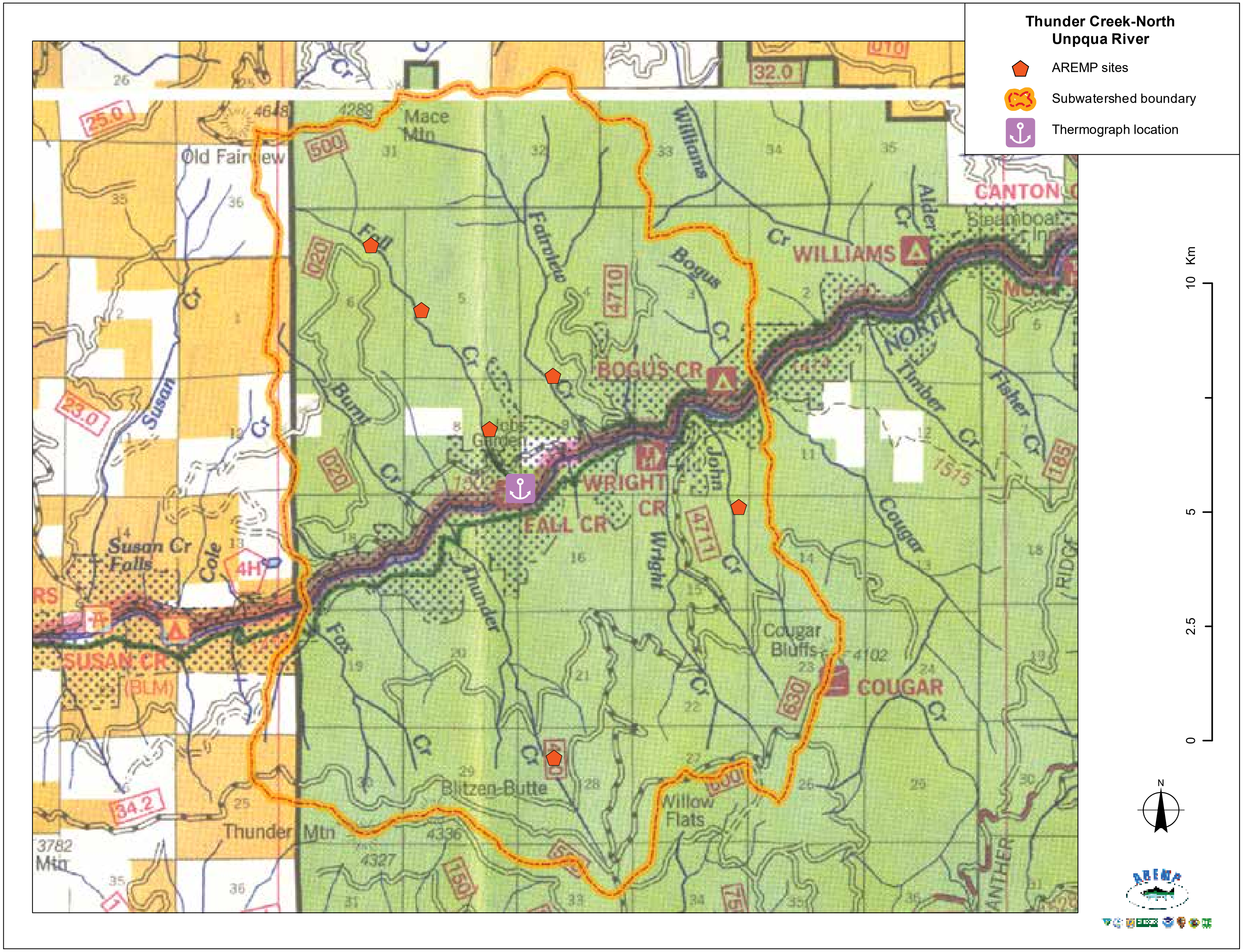


Figure A2.1—Example of temperature logger placement in a subwatershed with upstream flow delivery from outside of its boundary (a composite watershed). In these cases, temperature loggers were located in the tributary sampled with the most surveyed sites at the downstream-most point of federal land. In this example, the tributary is Fall Creek in the middle section of the North Umpqua River, Umpqua National Forest, Glide, Oregon.

Figure A2.2—Typical thermograph placement with the housing attached via epoxy to a rock; the thermograph goes inside. Photo courtesy of the Aquatic and Riparian Effectiveness Monitoring Program.

Fine Sediment

Sediment on the streambed of each site was measured based on a protocol modified from the Environmental Protection Agency's Environmental Monitoring and Assessment Program (Peck et al. 2001). Measurements of substrate particles were collected at equally spaced transects in all sites, beginning 2002. Number of transects, points per transect, number of particles per point, and total number of points varied across years (table A2.1). Substrate particle sizes were measured along the second-longest, or b, axis (USDA FS 2012). Particle sizes within each site were summarized as the proportion of the particles identified as fines (particles ≤2 mm), median particle size (D_{50}, mm), and the particle size at which 16 percent (D_{16}) and 84 percent (D_{84}) of the material was smaller. Pool-tail fines (≤2 mm) were assessed with a 14- by 14-inch grid with 49 evenly distributed intersections along the wetted channel following the shape of the pool tail crest. The number of gridded samples within a pool tail crest was three from 2002 through 2012, three to seven in 2013, and three from 2014 through 2017. The number of pools at a site had a median of 5 and ranged from 0 to 36 pools.

Macroinvertebrates

Benthic macroinvertebrates were sampled using protocols described by Hawkins et al. (2003). Benthic macroinvertebrate samples were collected at all sites with flowing water. Two sub-samples were collected at the first four fast-water riffle habitats encountered at the site for a composite of eight sub-samples. Sub-sampling consisted

Table A2.1—Sampling approach for quantifying substrate size at sites sampled by the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) for three time periods from 2002 through 2018

Years	Transects	Points per transect	Particles per point	Total particles
2002–2003	11	11	1	121
2004–2012	21	5	1	105
2013–2018	20	5	3	300

of macroinvertebrate collection using a 1 m² kick-net. Macroinvertebrates that were collected in the kick-net sampling were preserved in 95 percent alcohol before being sent to a lab for processing.

Macroinvertebrate data were summarized at sites using an observed-to-expected (O/E) index developed by Miller et al. (2017) and based on reference ("least human disturbed" [see Miller et al. 2016]) conditions. Reference site data from the AREMP, state agencies, Utah State University, and the U.S. Environmental Protection Agency were used to develop a River Invertebrate Prediction and Classification System (RIVPACS) type O/E ratio (Clarke et al. 2003) for the AREMP area. The O/E modeled ratio compares the taxa richness at an observed site to a similar reference site. The O/E modeled ratio does not explicitly account for increases in tolerant taxa found at observed sites and therefore could be over-estimating stream condition. Two labs were used for processing samples between 2002 and 2018. One lab, the National Aquatic Monitoring Center,[1] conducted the majority of the processing with comparable species identification over time. A second lab, ASci Corporation,[2] processed samples in 2005 and 2006. Preliminary assessment of macroinvertebrate species abundance indicated a difference between the two labs contracted to process samples. This difference could be controlled in the modelled analysis of ratio of observed to expected numbers of macroinvertebrates (O/E) by including a day of year covariate and an effect of the alternate laboratory that was used in 2005 and 2006.

[1] Utah State University, Department of Watershed Sciences, 5210 Old Main Hill, Logan, UT 84322.

[2] Environmental Testing Laboratory, 4444 Airpark Boulevard, Duluth, MN 55811.

Analysis of Status and Trends

Temperature

Box plots were generated with different combinations of sites over time using all available temperature data for the month of August (summarized as average maximum 7-day water temperature). No statistical tests or comparative analyses were completed with water temperature data due to variable and low sample sizes (particularly before 2011) and limited data in California. Average maximum 7-day air temperature was displayed with average maximum 7-day water temperature so that differences over time and between LUAs, aquatic provinces, and key and non-key watersheds could be explored.

Wetted Width/Instream Wood/Fine Sediment/Macroinvertebrates

Model description—
Our objective for developing a statistical model of instream responses (relative wetted widths, instream wood, sediment, and macroinvertebrates) was to measure trends over time that account for change in spatial variation of responses. Although AREMP uses well-vetted, standardized field protocols (Roper et al. 2010), surveys of instream conditions cannot be conducted without variation among observers (Bunte et al. 2012, Harman et al. 2008). Furthermore, it should be expected that instream conditions exhibit considerable spatial variability in terms of change over time because of numerous influences linked to climate, landforms, vegetation, natural disturbance, and land management prescriptions. Accordingly, we developed a model framework that can account for these influences.

Overall, the hierarchical modeling approach we used can be represented as a single model having two components: a component that encapsulates true variation in the response variables in time and space, and a component that represents numerous sources of variation associated with the imperfect observation process. The statistical estimation method we used was a Markov Chain Monte Carlo with Bayesian estimation. Program R and OpenBUGS were used in the analysis (R Core Team 2020).

Across all instream responses, the model component representing true variation is identical. We included model terms to account for time (year) and for variability among provinces, subwatersheds within provinces, and sites within subwatersheds. The observation model component, however, is slightly different for each response to account for slight differences in sampling designs, and some covariates are included to account for important or potential influences that are not of primary interest (e.g., sources of variation attributed to observation). Also, link functions differ for various response types, such as binomial, count, and continuous normal and log-normal responses.

True variation component—

The model component representing true variation for response *y* at site *s* and year *t*, free from observation error, can be represented as a regression equation:

$$y_{s,t} = \alpha_{s,1} + \alpha_{s,2} \cdot Year_t + \epsilon_t + \epsilon_{s,t} \,.$$

Here, $\alpha_{s,1}$ and $\alpha_{s,2}$ are intercepts and regression slopes across year of observation ($Year_t$), respectively, and ϵ_t, and $\epsilon_{s,t}$ are zero-centered normal random effects of year, and site by year, respectively, each with their own estimated variance. In ϵ_t and $\epsilon_{s,t}$, years are treated as discrete. The intercepts and regression slopes across year of observation are themselves functions of land-use type variables and random subwatershed (*w*) effects $\delta_{w,1}$ and $\delta_{w,2}$:

$$\alpha_{s,1} = \beta_{p,1} + \beta_{p,2} \cdot Matrix_s + \beta_{p,3} \cdot Nonfed_s + \beta_{p,4} \cdot LSR_s + \delta_{w,1}$$

$$\alpha_{s,2} = \beta_{p,5} + \beta_{p,6} \cdot Matrix_s + \beta_{p,7} \cdot Nonfed_s + \beta_{p,8} \cdot LSR_s + \delta_{w,2} \,.$$

The random intercepts and coefficients in $\beta_{p,1:8}$ are assumed to be multivariate-normally distributed and are representative of AREMP province *p*:

$$\beta_{p,1:8} \sim MVNormal\left(\mu_{1:8}, \Sigma^{(\beta)}_{1:8,1:8}\right) \,.$$

The estimated parameters $\mu_{1:8}$ and $\Sigma^{(\beta)}_{1:8,1:8}$ represent the average effects and variance-covariance matrix for the seven AREMP provinces analyzed. The random subwatershed effects are bivariate-normal:

$$\delta_{w,1:2} \sim MVNormal\left(0_{1:2}, \Sigma^{(\delta)}_{1:2,1:2}\right) \,.$$

Finally, the land-use type variables *Matrix*, *Nonfed*, and *LSR* are remotely derived continuous measures of the proportional makeup of matrix, nonfederal and late-successional reserve lands, respectively, in the contributing drainage area for sites. In the absence of matrix, nonfederal, and late-successional reserve lands, the intercept would represent congressional reserve lands.

The interpretation of $y_{s,t}$ is the true site-level average for a given response in year *t*, free from observation error. Note that the $y_{s,t}$ are estimates, not observed data or summary values treated as data. The actual observed data were modeled as a function of the $y_{s,t}$ plus further variables that affect the observation process, described below.

Observation component—

Picking up from $y_{s,t}$, the model for instream responses is completed by adding several important covariates affecting observation: sampling variation attributed to site-surveys (e.g., which would include differences between observers and differences in the exact placement of transects) and a fixed effect for independent, follow-up site surveys that were sometimes conducted within the same year for quality assurance, referred to here as *QAQC* surveys.

Every model for instream responses has a site-survey (*u*) random effect ε_u and a fixed *QAQC* effect measuring systematic differences associated with the follow-up surveys. In the equation below, *QAQC* is a binary indicator that the given site survey was the second, independent full survey of the site conducted within the year, and γ_1 is its coefficient:

$$\mu_u = y_{s,t} + \gamma_1 \cdot QAQC + \varepsilon_u \,.$$

One or two other features were included in the observation component for some responses where needed.

For the transect fines response derived from the pebble count sampling, we controlled for measured bankfull width at the transect level during each site survey ($stBankfull_{tran}$) and included transect-level random effects τ_{tran} within site surveys:

$$logit\,(P_{tran}) = y_{s,t} + \gamma_1 \cdot QAQC + \varepsilon_u + \gamma_{2,s} \cdot stBankfull_{tran} + \tau_{tran} \,.$$

Here, the coefficients $\gamma_{2,s}$ are random effects of site that have an estimated mean and variance. The ε_u, the τ_{tran}, and the standardized bankfull covariate $stBankfull_{tran}$ are all centered at zero for any site and year, and because $QAQC$ = 1 on the second survey of the year and 0 for the first survey, then $y_{s,t}$ represents the average condition at the site at the time of the first survey of the year. The actual transect fines response was then the number of particles that qualified as fines (particles ≤2 mm) of those measured at the transect on the given site survey. This number of particles was modeled as a binomial outcome having probability P_{tran} and the total number of samples at the transect as the binomial index.

For D_{16}, D_{50}, and D_{84}, which were also from the transect-based pebble count sampling, the model was similar except that the log-normal distribution (with log link function) was used in place of the binomial distribution, and the particle size that was the 16th, 50th or 84th percentile was the response, respectively. Very small particles classified as silt (<0.03 mm) or sand (>0.03 and <1 mm), and the largest particles classified as bedrock (>4096 mm) were treated as latent continuous values falling within size category bins. Because sand, silt, and bedrock were not precisely measured but were known to fall within width bins, their precise values were imputed to fall within the appropriate size range as part of the Markov Chain Monte Carlo procedure.

For the wood count responses, we tabulated data as total counts of qualifying pieces within size classes for each site survey. The data for each of the three wood size classes were analyzed separately. Subscripts for size class are suppressed here for simplicity. For the wood count responses, we included a standardized bankfull width covariate $stBankfull_u$, and the wood piece counts N_u for site survey u were modeled with the Poisson distribution and log link function. The standardized bankfull width covariate used here, $stBankfull_u$, was a site survey average bankfull width centered on the subwatershed average. The coefficients $\gamma_{2,w}$ for $stBankfull_u$ were random effects at the subwatershed level, having an estimated mean and variance. We accounted for the length of the study reach ($ReachLength_s$) as a factor in the Poisson expectation such that the linear combination modeled wood density per length of stream:

$$N_u \sim Poisson(ReachLength_s * \lambda_u),$$
$$\log(\lambda_u) = y_{s,t} + \gamma_1 \cdot QAQC + \varepsilon_u + \gamma_{2,w} \cdot stBankfull_u.$$

For the macroinvertebrate O/E ratio response, the data were tabulated as a single ratio for each site survey. We included a day of year covariate ($stDay$) and an effect of the alternate laboratory that was used in 2005 and 2006 (Lab). The $stDay$ covariate was centered on July 25th, the average survey day of year in the sample, and $\gamma_{2,w}$ is its coefficient. The $\gamma_{2,w}$ are subwatershed-level random effects having an estimated mean and variance. The variable Lab was a binary indicator having value one in years 2005 and 2006, and zero otherwise, and γ_3 is its coefficient. The O/E ratio required further attention to conform to a standard probability distribution. We treated the number of taxa observed ($N_{u,\ Observed}$) as a count response (Poisson). The number expected, ($N_{u,\ Expected}$), which is non-random, became a factor in the Poisson expectation so that λ represented the expectation for the O/E ratio, yet $N_{u,\ Observed}$ was the actual response variable:

$$N_{u,Observed} \sim Poisson(N_{u,Expected} * \lambda_u),$$
$$\log(\lambda_u) = y_{s,t} + \gamma_1 \cdot QAQC + \varepsilon_u + \gamma_{2,w} \cdot stDay + \gamma_3 \cdot Lab.$$

For the wetted width response W_{tran}, we used a normal model for transect-level wetted width measurements that were divided by a site-level average bankfull width benchmark B_S. Data were tabulated at the level of the transect and were modeled as normally distributed. Transect-level wetted widths were divided by the all-time average measured bankfull width at the site B_S to achieve normality. Response data at dry transects were treated as latent and their values imputed in the Markov Chain Monte Carlo procedure to be negative. We included in the model a day of year effect, which was a random effect at the province level having an estimated mean and variance and a transect ID random effect τ_{tran}, common across years, that identified the location of the transect measurement along the surveyed length of stream.

$$\frac{W_{tran}}{B_s} \sim Normal(\mu_{tran}, \sigma^2)$$

$$\mu_{tran} = y_{s,t} + \gamma_1 \cdot QAQC + \varepsilon_u + \gamma_{2,p} \cdot stDay + \tau_{tran}$$

For the pool-tail fines response, the number of particles ≤2 mm, there were no further additions to μ_u , beyond the random survey effect ε_u and the $QAQC$ effect. The response for pool-tail fines was binomial at the site-survey level and the random survey effects ε_u were beta-distributed. The precision T associated with the random effects was an estimated parameter. The beta random effects approach was necessary, over the alternative normal random effects approach, due to some site surveys having zero observed fines. More general models, including pool-level random effects, which are justified because of the profound level of spatial clustering, were not feasible. Attempts to include pool-level random effects failed to converge because there was a large amount of variation in fines between pools within a site survey and many cases had zero fines at the pool level. Further, convergence, even using the beta random effects approach, required restrictive prior distributions on the values of the random effects to avoid numeric overflow because some site surveys resulted in no samples being classified as fines. The number of samples classified as fines X_u for site survey u was modeled as a binomial outcome having probability P_u and the total number of samples M_u in the site survey as the binomial index:

$$X_u \sim Binomial(P_u, M_u)$$

$$P_u \sim Beta\left(T \cdot P_{s,t}, T \cdot \left(1 - P_{s,t}\right)\right)$$

$$logit\left(P_{s,t}\right) = y_{s,t} + \gamma_1 \cdot QAQC\ .$$

Quantitative results statements (e.g., the subwatershed, province, key watershed, LUA-level and overall summary trends, and intercepts that we report and portray in figures) are derived from the joint posterior distribution of the model parameters (intercepts, coefficients, and random effects, etc.) described above for each response. The joint posterior distributions of the model parameters were themselves 10,000 samples taken every 500th iteration from the Markov Chains after chains had converged to stability. We used these samples with the standard method for determining the posterior probability of an arbitrary function of model parameters in Bayesian inference (Gelman et al. 2014), which is straightforward with accumulated samples from the Markov Chains. To generate the posterior distribution of any function of model parameters, such as subwatershed-level decadal trend, the desired function is computed independently using the collection of parameter values from a single iteration of the Markov Chain. The accumulated sample of 10,000 functional results, one result per iteration, are then simply accumulated and summarized. The accumulation of results embodies the variances and covariances among the original parameters and can be summarized by its percentiles or other properties as samples from a probability distribution (e.g., by its mean, median, and 2.5th and 97.5th percentiles). The 2.5th and 97.5th percentiles are the credibility intervals reported in the results, and point estimates are posterior medians. Probability statements such as the probability that a trend is negative or positive are based on the percentage of the posterior distribution of the average slope (e.g., subwatershed average or overall average) that is less than or greater than zero, respectively (fig. A2.3).

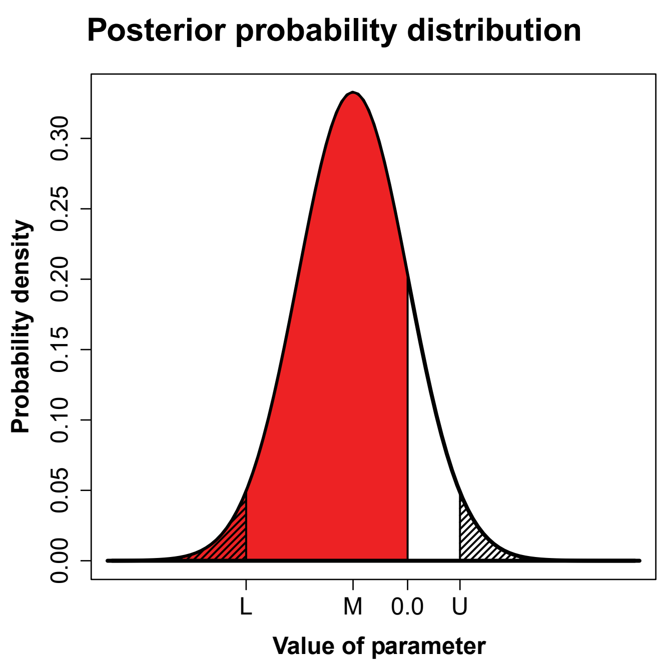


Figure A2.3—Plotted is an example posterior probability density. We report the posterior median (the value along the x-axis at M) as our point estimate with 95-percent credibility intervals from lower (L) to upper (U). The cross-hatched regions under the curve are 2.5 percent in area, both summing to 5 percent of the total area under the curve. In this example, the probability the parameter is negative is 84 percent; the area under the curve to the left of zero (red).

Following this method, for example, we derived overall proportional decadal trends for all but the wetted width response as follows. For Markov Chain Monte Carlo iteration z, we computed:

$$\ell_z = 100 * \left(exp\left(\frac{1}{N}\sum_s^N\left(\alpha_{s,2,z}\right)\right) - 1\right),$$

where $\alpha_{s,2,z}$ is the z^{th} Markov Chain sample of the site-specific coefficient of year, and N is the total number of sites. Here we add a subscript z to the linear combination $\alpha_{s,2}$, described in the modeling section above, in order to index the Markov Chain sample z. The value $\alpha_{s,2,z}$ is itself computed by:

$$\alpha_{s,2,z} = \beta_{p,5,z} + \beta_{p,6,z} \cdot Matrix_s + \beta_{p,7,z} \cdot Nonfed_s + \beta_{p,8,z} \cdot LSR_s + \delta_{w,2,z}.$$

Within the equation for ℓ_z, the average slope $\frac{1}{N}\sum_s^N(\alpha_{s,2,z})$ is multiplied by 10 (years) to reach a decadal scale, and it is exponentiated because of the log or logit link functions used for most responses. The result is multiplied by 100 to achieve a proportion. The estimates of proportional decadal trends with 95-percent credibility intervals are the median and 2.5th and 97.5th percentiles of ℓ overall iterations.

We maintained consistent prior distributions across the various responses, with the exception being the logistic models for transect and pool-tail fines. In the description above, we use the term "fixed effect" in cases where simple non-hierarchical priors were placed on coefficients, and the term "random effect" in cases having a hierarchical structure, i.e., the respective parameter was drawn from a distribution, where the distribution then had estimable parameters. In the case of the logistic models, we used the default prior distribution for logistic regression coefficients arrangement described by Gelman et al. (2014), specifically a t-distribution with a mean 0, scale 2.5, and 7 degrees of freedom. This was replaced with a diffuse Normal prior having mean 0 and precision 1.0e+4 for fixed effects in the non-logistic situations. For all univariate-Normal random effects, we used the hierarchical half-Cauchy arrangement (Gelman et al. 2014) for collections of random effect standard deviations. In this arrangement, the standard deviations were modeled as arising from a half-Cauchy distribution with mean 0 and estimated standard deviation, itself given a Uniform(0,3) prior distribution. The variance matrices for multivariate-Normal random effects were always given priors of the Scaled-Inverse-Wishart($\mathbf{I}_d$,d+1) form that employed parameter expansion (Gelman et al. 2014). Here, $\mathbf{I}_d$ is the identity matrix and d is the dimension of the multivariate-Normal distribution.

In determining adequacy of model fit and model suitability, there are multiple considerations. First, it is important to match the statistical model with the sampling design that has been implemented in terms of hierarchical levels employed in field sampling. Without this matching of hierarchical structures in the sampling design and the statistical model, the replication that is present at various hierarchical levels would not be appropriately represented. Second, although the model we have chosen may not be the simplest that one could have envisioned, because there was a large number of hierarchical levels in the AREMP sampling design, we did still strive for simplicity overall. Our objective was one overall hierarchical model design that represented the various hierarchies present in the sampling design and could be applied to all the different instream responses. Model assumptions, such as the normality of random effects, were judged to be adequately met by graphical examination of estimated random effects using posterior samples. Many other model formulations are feasible and may produce differing insights. We consider inference condition on the models employed. Future efforts incorporating additional years of field data will undoubtedly provide opportunity for exploring alternative models of changing conditions that are not at this time practical owing to the small number of subwatersheds having more than a single rotation (8-year span).

Appendix 3: Upslope and Riparian Indicators—Methods and Analysis

Drought

The American Meteorological Society (2019) Glossary of Meteorology defines drought as "a period of abnormally dry weather sufficiently prolonged for the lack of water to cause serious hydrologic imbalance in the affected area." Although this definition appears to be simple, there has been a proliferation of drought indicators and indices that are now in use across the world (Svoboda and Fuchs 2017). Part of the motivation for developing these diverse measures is that drought impacts can be manifested in many ways (van Loon 2015), and thus there are many drought indicators and indices. For example, droughts can be thought of in terms of meteorological, hydrological, agricultural, and socioeconomic impacts (Wilhite and Glantz 2019), as well as social-ecological impacts (Crausbay et al. 2017). We were interested in applying an index that was based on changes to several indicators (Svoboda and Fuchs 2017), including precipitation, temperature, and water balance: the Standardized Precipitation Evapotranspiration Index (SPEI) (Vicente-Serrano et al. 2010). Vose et al. (2016) reviewed drought in forests and rangelands and noted that this index may be superior in capturing drought impacts in summer—when Aquatic and Riparian Effectiveness Monitoring Program (AREMP) surveys are conducted and direct indicators of drought such as wetted stream widths and stream temperatures are tracked. Furthermore, analysis of changing drought conditions, such as those associated with droughts in California, have identified temperature as playing a key role in exacerbating impacts when precipitation deficits occur (Diffenbaugh et al. 2015).

The SPEI is fundamentally based on estimating the site-specific difference between precipitation and potential evapotranspiration and is thus an index of the relative surplus or deficit of water in a given location and time (Vicente-Serrano et al. 2010). Values of the SPEI are standardized, such that the average value is 0 with a standard deviation of 1 (Vicente-Serrano et al. 2010). This provides a common scale and simple basis for comparisons. Negative values of the SPEI indicate water deficits. The SPEI units can be roughly interpreted as follows: (1) non-drought conditions correspond to values >-0.5; (2) mild drought conditions correspond to values between -1 and -0.5; (3) moderate drought corresponds to values between -1.5 and -1; (4) severe drought corresponds to values between -2 and -1.5; and (5) extreme drought is any value <-2 (McKee et al. 1993, Paulo et al. 2012). We do not report these categories but note them here so readers understand what a potential unit change in the SPEI means over the course of a decade as indicated by trends reported herein.

We calculated the SPEI following methods outlined by Vicente-Serrano et al. (2010), as implemented in R. SPEI is a scalable drought metric, and we summarized responses over the months of July, August, and September to correspond with AREMP field measurements. During these months, water deficits may be expected as seasonal precipitation is usually reduced and evaporative demand is increased (Coble et al. 2020). The essential inputs for calculating SPEI were extracted from Daymet (Thornton et al. 2016). Daymet is a gridded (1- by 1-km) daily dataset available from 1980 to present for North America. The Daymet variables used in this analysis include precipitation, maximum air temperature, minimum air temperature, and day length. Daily values were estimated for all included sites from 1980 through 2018 through the use of the R package Interface to the Daymet Web Services (Hufkens et al. 2018). This common application of Daymet data access allows for extraction from any given pixel within the gridded Daymet dataset corresponding to a provided location in latitude/longitude format. Data from each AREMP temperature site were pulled to correspond directly to the 1- by 1-km square grid that the location resides in and directly exported data from the Daymet dataset. The significance of trends in the SPEI within each location was calculated with non-parametric Mann-Kendall trends, and the magnitude of trends was estimated with a non-parametric Sen slope (Arismendi et al. 2012, Helsel and Hirsch 2002).

We selected 374 sites for calculation of the SPEI to include all subwatersheds where AREMP has deployed stream temperature loggers. Overall, our objective in this assessment was to provide a context for determining if the severity of drought was increasing or decreasing at these locations over different time periods corresponding to the full available series (1980–2018), pre-Northwest Forest Plan (NWFP) (1980–1993), and post-NWFP implementation (1994–2018). Accordingly, interpretations of our results should consider the fact that assessments of the SPEI should be expected to vary somewhat depending on the time series selected, locations selected, and time scales considered.

Discharge

Stream discharge is a crucial measure for assessing the effects of climate on watershed response as it integrates the upstream effects of multiple processes (app. 5). Measuring stream discharge in many different streams across a large spatial area, with a complete temporal record, presents numerous difficulties. Alternatively, a distributed parameter hydrologic model can simulate stream discharge in ungauged basins using available stream discharge observations for calibration.

Stream discharge was simulated using the distributed parameter, physical process-based, deterministic Precipitation-Runoff Modeling System (PRMS) developed and maintained by the USDI Geological Survey (USGS) (Markstrom et al. 2015). In PRMS, meteorological forcing data (e.g., air temperature and precipitation) are applied to a model domain and routed through interacting modules (i.e., hydrologic processes) to simulate hydrologic conditions. Each PRMS model domain is discretized into hydrologic response units (HRUs) that route incoming precipitation through surface, subsurface, and groundwater zones that are then routed to stream segments. Stream segments route streamflow from upstream HRUs and segments to downstream segments and eventually the model domain outlets. Model parameters are distributed across a model domain by HRU or stream segment, allowing each unit to be parameterized individually to represent unique conditions, such as aspect, vegetation, soil properties, and elevation. Simulated hydrologic conditions are output at user-defined time intervals, often daily, for user-defined variables of interest (e.g., streamflow, evapotranspiration, and snowpack).

The National Hydrologic Model for use with PRMS (NHM-PRMS) (Regan et al. 2018) is an application of PRMS across the conterminous United States (CONUS) with a consistent geospatial discretization and consistent methods for calculating and calibrating model parameters at CONUS-scale. The NHM is discretized into HRUs, stream segments, and points of interest (POIs) based on the Geospatial Fabric (GF) (Viger and Bock 2014). Discretization of HRUs and stream segments is determined by the POIs, which includes stream gage locations, major confluences, and elevational break points. Default PRMS parameters were developed for every component of the GF (Viger 2014), and an uncalibrated CONUS-scale application of NHM-PRMS was simulated for water years 1980–2016 (a water year defined as October 1 of previous year through September 30 of calendar year; Driscoll et al. 2018). Parameters were further calibrated using remote-sensing baseline datasets and headwater streamflow datasets, resulting in additional CONUS-scale applications of NHM-PRMS with select output variables (Hay 2019, Hay and LaFontaine 2020). Water years 1980 and 1981 are considered model initiation and spin-up years and were excluded from analyses.

Stream discharge was summarized for points adjacent to sampled AREMP subwatersheds by locating the nearest downstream predictions at POIs from the most recent national NHM-PRMS simulation (Driscoll et al. 2018, Hay 2019, Hay and LaFontaine 2020). Stream discharge from the PRMS variable seg_outflow, in units of cubic feet per second, was converted to millimeters per day and summarized across the NWFP area, as well as among provinces within the NWFP area. We relied on summaries of annual discharge for the purposes of this report. Trends for the median annual estimated discharge were analyzed using non-parametric Mann-Kendall and Sen’s estimator (Arismendi et al. 2012, Helsel and Hirsch 2002). This analysis was used to determine whether there was a positive or negative trend in stream discharge for 1982–2016, 1982–1993, and 1994–2016.

Forest Characteristics

The three forest characteristics indicators used in this report were summarized from data created for the NWFP late-successional and old-growth (LSOG) monitoring program. These data were derived using a gradient nearest neighbor (GNN) imputation approach that matches forest inventory plots to multispectral forest surface reflectance captured by Landsat satellites (Ohmann et al. 2011). This process provides a complete spatial coverage for estimated characteristics with a 30 m grid resolution. As with the LSOG program, a nonforest mask was applied to exclude urbanized areas, major roads, agricultural areas, water, lands above tree line, snow, rock, and other nonforested features. Further details are described in the most recent LSOG report (Davis et al. 2022).

For summarizing each characteristic, we extracted only those pixels falling within the riparian management areas (see app. 1) and calculated mean values by subwatershed (12 digit hydrologic unit code, or HUC12). These subwatershed means were used for generating the maps and other figures. In summarizing HUC-level results to broader areas (e.g., provinces), values were weighted by the riparian area in each HUC.

Canopy Cover

Canopy cover is the percentage of ground area that is directly covered with tree crowns. In the GNN dataset, this remotely sensed estimate includes all live trees and is a percentage value calculated using methods in the Forest Vegetation Simulator for Forest Inventory and Analysis plots, or the sum of ocular estimates for USDA Forest Service ecology plots.

Old-Growth Structure Index

The LSOG program's old-growth structure index (OGSI) is computed by comparing four stand attributes (abundance of large live trees, density of large snags, percentage cover of downed woody material, and diversity of live tree sizes) to regression equations on age from regional plot data. OGSI can either be reported as a continuous value between 0 and 100 or as a binary indicator for specific age thresholds. The LSOG program calculated these binary thresholds for forests at 80, 120, 160, and 200 years. For this report, the 80-year threshold was used (1) because at this age, trees in the dominant NWFP forest types are reaching the diameter thresholds used for instream large wood monitoring (≥12 in), and (2) because of the relatively short time period (25 years) of the NWFP to date. The final metric used was the percentage of riparian forest-capable area meeting the OGSI-80 threshold.

Density of Large Trees Near Streams

Research has established strong links between the abundance of large woody debris in streams and the quality of fish habitats (Bisson et al. 1987). Data from the Cascade and Coast Ranges of Oregon and Washington found that >70 percent of fallen trees in streams originated within 20 m of the stream (McDade et al. 1990). Therefore, for this report, we calculated an additional large woody debris supply indicator based on the density of large trees within 20 m of a fish-bearing stream. LSOG data contain large tree counts per hectare at 25 cm intervals; we chose the 50 cm (20 in) threshold because it was closest to our medium (18 in) and large (24 in) thresholds for instream wood counts.

Road Layer Development

As with past reports, we only report on "system" roads (i.e., those designated to be maintained over the longer term) and changes due to decommissioning and new construction, because temporary roads and road improvements like "storm proofing" are not tracked in the regional corporate data.

A layer of Forest Service roads was created by combining the main national Forest Service road networks (table A3.1) and clipping them to the boundaries of the Forest Service Pacific Southwest and Pacific Northwest Regions. A layer of non-Forest Service roads was created by clipping the existing non-Forest Service road networks to the non-Forest Service federal land footprint in the NWFP area, and then combining them into one layer. While we used road layers produced by the USDI Bureau of Land Management (BLM), they covered all of Oregon and Washington, making them suitable for all non-Forest Service federal lands in those states (table A3.1). The Caltrans road layers were used to cover non-Forest Service lands in California.

The Forest Service and non-Forest Service road layers were combined to create an all-NWFP federal land road

Table A3.1—Road network data sources combined to create a road network for federal lands in the Northwest Forest Plan area

Forest Service[a]	Bureau of Land Management[b]	Caltrans[c]
RoadCore_existing	GTRN_PUB_ ROADS_ARC	D01
RoadCore_ decomissioned	Highways_arc	D02
RoadCore_converted		D03
Road		D04

[a] USDA FS 2019a.

[b] USDI BLM 2019b.

[c] Caltrans 2019.

layer (table A3.1). The combined road layer was projected from the Geographic NAD83 coordinate system to UTM NAD83 for the most accurate representation of length.

Prior to the two merging efforts, roads that overlapped one another (shared line segments) across layers were reviewed and unselected so as not to create duplication of roads. However, duplication of road features within layers occurred, but was not discovered until after the Geomorphic Roads Analysis and Inventory Package (GRAIP) Lite processing was completed. Duplication also occurred as an outcome of geoprocessing. Line features lying on a border between land use polygons were duplicated and assigned to each neighboring polygon during the intersect process. About 0.86 percent of the network (1076 km) were duplicate roads in 1993 and 0.9 percent (1046 km) in 2019.

A large proportion (~45 percent) of the total road duplication was concentrated in nine HUC12 subwatersheds in the Klamath National Forest. Between 25 and 50 percent of the total road length in these subwatersheds consisted of duplicated roads. Given the impacts of these nonexistent, excess roads on GRAIP Lite metrics, we removed the duplicate roads in these subwatersheds.

The road network attribute data were adapted to specifications needed by the GRAIP Lite sediment model. Road network attributes for operation maintenance level, route status, and road surface were standardized by translating to Forest Service attribute categories (table A3.2).

For all road-related analyses, the road construction attributes were summarized into two categories: built before 1994 (Built_94 = 0) or built during or after 1994 (Built_94 = 1). The road decommissioning attributes were also summarized into three categories: not decommissioned, decommissioned after 1993, or decommissioned prior to 1994. The road construction

Table A3.2—Non-Forest Service road network data source attributes used to create surface type, route status, and maintenance level inputs for GRAIP Lite road sediment modeling

Input type	Bureau of Land Management layer attribute	Forest Service layer attribute
Surface	SURFACE = “Aggregate”	SURFACE_TYPE = “AGG – CRUSHED AGGREGATE OR GRAVEL”
	SURFACE = “Bituminous”	SURFACE_TYPE = “BST – BITUMINOUS SURFACE TREATMENT”
	If SURFACE <> “Aggregate” or “Bituminous” and SURFACETYPE = “Natural Improved”	SURFACE_TYPE = “IMP – IMPROVED NATIVE MATERIAL”
	If SURFACE <> “Aggregate” or “Bituminous” and SURFACETYPE = “Natural Unimproved”	SURFACE_TYPE = “NAT– NATIVE MATERIAL”
Route	CLOSURESTAT = “Decommission”, “Full Decommission”, or “Obliteration”	ROUTE_STATUS = “DE - DECOMMISSIONED”
Maintenance	MaintLvl = {“Maintenance Level 1”, “2”, “3”, “4”, “5”}	OPER_MAINT_LEVEL = {“1”, “2”, “3”, “4”, “5”}
	highways_arc (all features)	OPER_MAINT_LEVEL = “5”

Note: non-Forest Service road layer attributes were modified to match Forest Service naming conventions for GRAIP Lite processing.

and decommissioning attributes were updated based on the Forest Service road infrastructure database (INFRA). The road system in the first time period (1993) shows the roads as they existed prior to the NWFP, with no roads built later than 1993 or decommissioned before 1994. The road system in the latter time period (2019) shows the roads as they existed when the geographic information system (GIS) layer was made, which included all roads not decommissioned at any time.

Road Sediment Modeling

To address roads as sources of fine sediment to streams across the NWFP area, we employed a GRAIP desktop extension named GRAIP Lite (Nelson et al. 2019). GRAIP Lite is a GIS-based procedure to estimate the potential of forest roads to produce and deliver sediment and is based on readily obtained spatial data on roads, topography, and assumptions regarding base road surface erosion rates (Nelson et al. 2019). The model was run at the scale of individual road segments and results for each segment were aggregated up to the subwatershed scale. All HUC12s with ≥5 percent federal land within the AREMP area were included, but overlay areas (HUC12-LUA and HUC12-key watershed) <5 ha were dropped to reduce small area outliers. As employed here, GRAIP Lite estimates of fine sediment produced by forest roads are considered to be relative within each subwatershed. Realized values of fine sediment production in the field may differ due to unknown variability in base road surface erosion rates and characteristics of road networks on the ground versus what can be determined with existing spatial data. These uncertainties notwithstanding, GRAIP Lite is a feasible and powerful tool for evaluating potential changes in sediment production from forest roads (Nelson et al. 2019).

Sediment production from forest roads (kg/yr) was estimated using GRAIP Lite in 1993 and 2019 to evaluate relative changes since the NWFP was initiated. Overall, GRAIP Lite estimates sediment production on a road segment scale as follows:

$$E = BRSV,$$

where E is the total sediment production (kg/yr), B is the base rate (kg/yr/m), R is the elevation difference between road segment ends (m), S is the surface type factor (dimensionless), and V is a vegetation factor (dimensionless).

Road GIS features were categorized into five Forest Service operational maintenance levels (ML) using data on forest infrastructure (USDA FS 2019) and then split into shorter segments for processing in GRAIP Lite. The ML values of 77,000 road segment field observations were used to create the length classes (Nelson et al. 2019) (table A3.3). Note that some segments were necessarily shortened based on where the ridgelines and streams were located. Measurements of road segments proceeded uphill from the low point at the stream or junctions.

For each road segment, delivery of sediment to a nearby stream (D) was calculated as sediment production by the road segment multiplied by the fractional sediment delivery (C, defined below):

$$D = EC.$$

This approach strongly weights sediment production from stream-proximate road segments. Subwatershed sediment delivery is the sum of the road segment sediment deliveries.

Table A3.3—Description of operational maintenance level and corresponding road segment lengths used to split road features in Aquatic and Riparian Effectiveness Monitoring Program (AREMP) subwatersheds

Maintenance level	Description	Length
		meters
1	Roads that have been decommissioned or stored. These are generally out-sloped, recontoured, or have frequent waterbars.	50
2	High-clearance roads with minimal engineering.	50
3–4	Passenger car roads with culverts and engineered following agency specifications (e.g., USDA FS 2006).	100
5	High-traffic mainlines that are typically paved or aggregate surfaced and dust abated.	200

Source: USDA FS 2021.

Derivation and Processing of GRAIP Lite Parameters

Details of each input to GRAIP Lite defined above are briefly described below.

Base rate (*B*, kg/yr/m) was based on a uniform value of 79 kg/yr/m of vertical drop based on a field study in western Oregon (Luce and Black 1999). Because the base rate is assumed to be uniform in this application, estimates of sediment production from GRAIP Lite vary in relation to other model parameters (*R, S, V, L*) that varied for each road segment.

The **elevation difference** between road segment ends (*R*, m) was based on a 30 m resolution digital elevation model (DEM) mosaiced from the USGS National Elevation Dataset (USGS 2017). Each end-point elevation was bilinearly interpolated between the four proximal DEM center points.

The **surface type factor** (S) was also derived from data on forest infrastructure and represented by combinations of traffic level (none, low, medium, high) and surface type (crushed rock, native, paved, not a road) (Nelson et al. 2019).

The **vegetation factor** (*V*) was calculated as 1 - 0.86x where x is the fraction of the road where flow path vegetation is >25 percent, and x is a function of ML and surface type.

Fractional sediment delivery of road segments (*C*) was based on distance to stream conditioned on road segment length. Road segments resulting from the maintenance level splitting process were classified as short (≤30 m), medium (>30 m and ≤75 m), or long (>75 m). A logistic regression on over 77,000 field observations of road-stream connection from the region (Nelson et al. 2019) was used to create separate probability curves for each road segment length category, resulting in a stream connection probability value for a given stream distance. These probability functions were used to estimate the fractional sediment delivery at each road discharge point.

Data processing used GRAIP Lite for ArcGIS™ (downloaded from Esri ArcHydro [http://downloads.esri.com/archydro/archydro/setup/]) (Maidment 2002). The input data for the roads were used as received and run through the GRAIP Lite model using the default settings. Automated road quality assurance/quality control (QA/QC) tools were used to remove some duplicate road segments, remove overlaps, and break loops (Esri 2019). Sediment delivery was predicted for sediment leaving the lower end of each road segment at a drain point and aggregated across each subwatershed within the AREMP area. The principal model outputs are connected road length (km/km^2), road fine sediment delivery (Mg/yr), and specific fine sediment delivery (Mg/yr/km^2) to the fluvial system. Areas used in calculations include the federally managed portions of the subwatersheds only.

For summarizing connected road length and specific sediment delivery outputs, we calculated mean values by subwatershed (HUC12) containing ≥5 percent federal land. These subwatershed means were used for generating the maps and other figures. In summarizing HUC-level results to broader areas (e.g., provinces), values were weighted by the analysis unit area (land use allocation [LUA], key watershed) in each HUC.

Roads and Slope Stability

We estimated the risk from roads of increasing shallow landslides based on topography, such as steep colluvial hollows prone to such failures. Our slope stability metrics were developed by modifying the method used in SINMAP (Stability Index MAPping) (Pack et.al. 2005), which calculates a factor of safety for slope stability. Based on the SINMAP model, we calculated the minimum cohesion value (C) that would result in a stable slope given other fixed parameters. Higher values of a dimensionless C, which can range between 0 and 1, imply less stability and lower values imply greater stability. The road layer is then intersected with the resultant grid of C values to identify where roads cross ground of varying stability. This approach is not intended to assess road-related mass wasting risks associated with other processes (e.g., earthflows and deep-seated rotational failures), which would need to be assessed by other methods.

Given the assumptions for soil density, internal friction angle, and the transmissivity-recharge ratio, locations mapped with higher C values are predicted to be more unstable in the absence of soil or root cohesion. The soil density and internal friction angle were given the same values as have commonly been used in shallow slope stability model (SHALSTAB) assessments in the AREMP area in the past (e.g., Dietrich et al. 1998). We set a transmissivity-recharge ratio of 500 m (or log(q/T) = -2.7 as used in SHALSTAB analyses), which represents a conservative criterion for instability in cohesionless soil relative to that suggested by Dietrich et al. (1998), which is a log(q/T) threshold of -2.8. In effect, the criterion we used results in slightly more landscape classified into a "high hazard" category relative to use of values suggested by Dietrich et al. (1998). From there, the calculated dimensionless C value is a metric of how far beyond the threshold a particular site is. Each grid cell with a value of C (meaning that some cohesion is conceptually necessary for it to be stable) that is crossed by a road is summed across a watershed. A larger sum of C values would represent a larger potential risk of mass wasting events associated with roads within the watershed. Since the amount of federal ownership in each watershed can vary greatly, we divided the sum by the federal watershed area to provide a per acre risk density measure. This index provides a value that could be used to compare one watershed with others, but it is important to recognize that it is only an index value. Depending on local variations in climate (e.g., precipitation intensity), drainage density, and soil density and internal friction angle, a local stability threshold may be greater than or less than the base threshold value applied here. These index values only provide a coarse estimate of stability risks, and local knowledge would be needed to assess actual risk issues associated with the road network.

The following equations describe how C values were calculated. We start with the infinite slope stability equation (Sidle et al. 1985):

$$SS = \frac{C + \cos\theta \tan\Phi\,(1 - wr)}{\sin\theta},$$

where *SS* is infinite slope stability model factor of safety, Θ is slope angle, Φ is the internal friction angle of the soil, w is the relative wetness, and r is the water to soil density ratio. A location is considered stable if $SS > 1$. Assuming a water-to-soil density ratio of 0.625 (a soil density of 1600 kg/m^3) and an internal friction angle of 45 degrees, we solve for the minimum dimensionless cohesion value (C) that would result in a stable slope:

$$C > \sin\theta - \cos\theta \tan\Phi\,(1 - wr).$$

Slope and relative wetness become the key variables, where relative wetness is determined by the area estimated to send water to a particular pixel ("contributing area") and the transmissivity-recharge ratio, which indicates how much contributing area is necessary to bring a soil to saturation during a characteristic event.

For summarizing minimum cohesion values (C) in box plots, we calculated mean values by subwatershed (HUC12) containing ≥5 percent federal land. These subwatershed means were used for generating the maps and other figures. In summarizing HUC-level results to broader areas (e.g., provinces), values were weighted by the analysis unit area (LUA, key watershed) in each HUC.

Culverts

To develop a GIS dataset of culverts on federal land in the NWFP area, we combined the California Fish Passage Assessment Database (CCC 2019), Oregon Fish Passage Barriers dataset (ODFW 2016), Washington Department of Fish and Wildlife (WDFW) Fish Passage dataset (WDFW 2019), BLM Oregon Fish Passage Barrier dataset (USDI BLM 2019a), Forest Service Pacific Northwest Region Culvert Locations dataset, and an assortment of park-specific National Park Service culvert datasets obtained in October 2019 (Crater Lake, Mount Rainier, North Cascades, Olympic National Parks, and Whiskeytown National Recreation Area). Since 2018, we were able to fill in additional data by conducting culvert surveys (almost 300 sites) and gathering information from local units (67 additions on Forest Service land, 62 additions in the BLM Eugene District, 144 additional sites from WDFW surveys, and 63 California Fish Passage Assessment Database updates) to fill in missing data at 336 additional crossings on Coos Bay and northwest Oregon BLM districts, and Okanogan-Wenatchee, Gifford Pinchot, Mount Hood, Siuslaw, Umpqua, Klamath, Six Rivers, and Shasta-Trinity National Forests to prepare for this analysis. Only roads and culverts on federal lands were considered for this analysis.

Some points in culvert databases appear to be bridges rather than culverts. To identify these points, we first filtered for attributes that contained "bridge" in the name or comment and classified these as confirmed bridges. Bridge datasets and the watershed area above stream culvert locations were then used to identify possible bridges within culvert datasets. Culvert locations that were within 100 m of a known bridge were classified as a potential bridge. Likewise, culvert locations with an upstream watershed area of ≥100 km² were classified as a potential bridge. This area threshold was determined by calculating the median watershed area above known bridges. Road crossings over large streams are more likely to be bridges than culverts.

The total length of blocked streams above culverts was determined by using the Barrier Analysis Tool (BAT) (Hornby 2010). Culvert data were filtered by selecting only those culverts within 100 m of potential fish habitat streams. The potential fish habitat stream layer was created by starting with high-resolution (1:24k) National Hydrography Dataset streams and filtering to stream reaches with perennial or artificial path FCode status and <20 percent gradient. Although only culverts on federal land were analyzed, the stream layer used in the BAT was kept intact across all ownerships so connectivity above and below barriers could be determined. Only streams on federal land were included when determining the length of stream above culverts.

We also conducted a GIS analysis of road crossings of fish-bearing streams. To determine where there may be crossings that are not currently included in existing barrier databases, we used GIS road and fish-bearing stream layers to find intersections that are not within 100 m of a known stream culvert or bridge. Some of these intersections were expected to be a result of imprecise georeferencing in spatial data layers that indicate an intersection where none exist. Others are real road-stream crossings, and some of these contain culverts. As part of this analysis, road-stream intersections were reviewed in ArcGIS to remove as many false positives from imprecise georeferencing as possible. In a field verification effort on the Gifford Pinchot National Forest, 137 potential road-stream crossings were visited. Of these, 8 were not real crossings (false positives), 60 were determined to be passable culvert and bridge crossings, and 69 were culverts deemed likely to be partial or total barriers to fish passage. This analysis only considered fish-bearing streams and road-stream crossings on federal lands. Of the identified road-stream crossings located throughout Klamath, Six Rivers, Shasta-Trinity, Mendocino, and Modoc National Forests, 383 were added to the California Fish Passage Assessment Database as new records with unknown passage status.

Appendix 4: Methods for Assessing Overall Watershed Condition

Overall watershed condition was evaluated by using a multivariate approach to capture relationships among watersheds that included both instream and upslope variables. Multivariate statistical tools are helpful when examining a suite of outcome variables simultaneously. Two multivariate tools were implemented in this assessment: principal components analysis (PCA) and multi-response permutation procedure (MRPP) using PC-ORD (version 7.0) (Wild Blueberry Media LLC 2020). MRPP is a nonparametric, permutation-based significance test for comparing groups of samples based on within-group similarities using Euclidean distance (McCune and Grace 2002). Tests by MRPP provide both a likelihood p-value and an effect size (A) that measures the chance-corrected, within-group homogeneity: $-1 < A \leq 0$ when there is no difference between groups and $0 < A < 1$ when groups differ. It is important to note that statistical significance may result between groups even when the effect size is small ($A < 0.1$) if the sample size is large (e.g., >100 samples). In these cases, the ecological significance of the result should be interpreted with caution. As such, we focus our results on group differences that are statistically significant ($p < 0.01$) and have effect sizes (A) > 0.05.

Datasets for Multivariate Analysis

Instream metrics selected for inclusion in the multivariate assessment included a measure of benthic macroinvertebrate (BMI) community composition (observed to expected ratio), a measurement of instream habitat complexity (count of large wood), and a measure of substrate fines (mean percentage of fines from transect pebble counts, ≤2 mm) (table A4.1). These three measurements were selected because they represented important characteristics of instream habitat condition. Instream variables were collected by Aquatic and Riparian Effectiveness Monitoring Program (AREMP) field crews at sample locations. The number of sample locations varied among watersheds (app. 1). Instream data were summarized across surveys within the watersheds using hierarchical linear models (see “Methods” in app. 2).

Upslope metrics that were selected for inclusion in the multivariate assessment included a measurement of human disturbance (road density), the riparian upslope forest stand condition (mean percentage of canopy cover), and a measurement of trees with potential for wood recruitment into the stream channel (density of riparian large trees) (table A.4.1). For upslope data, which is comprehensive across the entire AREMP area, this required summarizing data from the broader landscape at the watershed scale (see "Methods" in app. 3).

Additional datasets that were included in analysis were important for grouping the data and did not change over time. Sampled watersheds were identified by management classifications of key and non-key watersheds, the dominant federal land use allocation (LUA: matrix, congressional reserve, late-successional reserve), and survey time period (2002–2009 or 2010–2017) (table A4.1).

Multivariate Analyses

Principal Components Analysis

Principal components analysis (PCA) uses multivariate ordination of samples by Euclidean distances to describe linear relationships among variables (McCune and Grace 2002). We conducted 12 different PCAs to look closely at the watershed conditions as described by six relevant instream or upslope variables. These PCAs were conducted from the entire dataset of 406 watersheds together, and then subsets of data grouped by aquatic province, key and non-key watersheds, and federal LUAs. For instream variables, the first site visit captures conditions sampled during 2002–2009 (time period 1), while the second sample event occurred during 2010–2017 (time period 2). For upslope variables, the GIS-derived geospatial data are from 1993 (time period 1) and 2017 (time period 2), representing the condition on the landscape at the onset of the Northwest Forest Plan and in the most recent year of available geospatial data, respectively. Habitat variables with Pearson

Table A4.1—Instream and upslope habitat variables used in multivariate analyses to compare watershed conditions across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) study area over time

Variable name	Variable description	Data source	Year
Time	Designates first or second time period	NA	First visit 2002–2009 Second visit 2010–2017
Key/non-key	Watershed designation of key or non-key based on their potential value as high-quality salmonid habitat and restoration potential	NA	NA
Land use	Dominant federal management type in the watershed (matrix, late-successional reserve, or congressional reserve)	Upslope GIS analysis	NA
Road density (RD)	Density of forest roads by length per watershed area	Upslope GIS analysis	1993, 2019
Percentage of canopy cover (CC)	Mean percentage of canopy cover of the RMA in the watershed	Upslope GIS analysis	1993, 2017
Large tree density (LT)	Mean trees per hectare with a diameter at breast height (d.b.h.) ≥50 cm, within NWFP variable-width riparian buffer	Upslope GIS analysis	1993, 2017
Benthic macroinvertebrate ratio (O/E)	Ratio of observed to expected (O/E) benthic macroinvertebrate diversity based on instream sampling	Instream sites	First visit 2002–2009 Second visit 2010–2017
Large wood pieces (LW)	Count of wood pieces with a diameter ≥61 cm	Instream sites	First visit 2002–2009 Second visit 2010–2017
Percentage of fines (%F)	Mean percentage of fine sediment (≤2 mm) from transect pebble counts	Instream sites	First visit 2002–2009 Second visit 2010–2017

NA = not applicable; NWFP = Northwest Forest Plan; RMA = riparian management area.

correlation coefficients of $r > |0.5|$ were included as black line vectors on PCA plots where their length is scaled to correlation strength.

Owing to the broad spatial distribution of sites in the AREMP, aquatic provinces were used to group sites for some analyses. Each aquatic province was analyzed separately. Only aquatic provinces with at least 10 watersheds were included in the analysis; therefore, the Franciscan province in northern California (with five watersheds and 10 total samples) was not included in the individual province multivariate analysis (but samples were included in the All data analyses). The Puget Sound aquatic province was also excluded because no field sampling was completed there. Thus, a total of six aquatic provinces were analyzed, each with varying numbers of watersheds available for analysis that compared data from the first and second time periods (table A4.2).

Multi-Response Permutation Procedure

Multi-response permutation procedure (MRPP) tests were completed on the overall dataset of 406 watershed samples from 219 watersheds, most with samples from both time 1 (2002–2009) and time 2 (2010–2017), although 4 watersheds had only a time 1 sample and 27 watersheds had only a time 2 sample. Watersheds from the Franciscan province were excluded from the MRPP tests by aquatic province because of the small sample size (n = 10 samples). MRPP was used to test for differences between groups of the (1) first and second time periods, (2) six provinces, (3) LUAs, and (4) key and non-key watersheds (sample sizes in table A4.2). MRPP is a nonparametric procedure designed for multivariate analyses comparing groups (McCune and Grace 2002). Euclidean distances were calculated to compare between groups in multivariate space.

Table A4.2—Number of samples within the study area that were included in analyses by aquatic province, first and second time periods, key and non-key watersheds, and dominant federal land use allocation

Aquatic Province	Total	Time 1	Time 2	Key	Non-key	Matrix	LSR	CR
Franciscan	10	5	5	4	6	0	8	2
High Cascades	54	24	30	14	40	27	11	16
Klamath/Siskiyou	110	52	58	40	70	54	36	20
North Cascades	42	19	23	19	23	2	18	22
Olympic Peninsula	17	7	10	8	9	2	5	10
WA/OR Coast Range	44	19	25	18	26	4	40	0
Western Cascades	129	65	64	48	81	54	46	29
Total	406	191	215	151	255	143	164	99

CR = Congressional reserve, LSR = late-successional reserve.

Appendix 5: Physical Processes and Watershed Condition

The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) is focused on tracking status and trends of instream conditions and how they relate to riparian and upslope watershed processes (Reeves et al. 2004) that include a host of physical, chemical, and biological influences (Beechie et al. 2010, Naiman et al. 2000). These shape key features of what is commonly referred to as "habitat" (Hall et al. 1997) for highly valued species such as Pacific salmon (Quinn 2018) in streams and a host of other ecosystem services such as provisioning of water (Luce et al. 2017) and many other functions (Martin-Ortega et al. 2015).

Watershed conditions interact to produce observable responses within stream channels (Hynes 1975). Field surveys of stream channels conducted by the AREMP track stream channel geometry, sediment, large wood, and temperature. In addition to these field-based measurements, trends in riparian and upslope conditions are tracked through remote sensing of land cover (e.g., forest structure and composition), climate (e.g., temperature and precipitation), and updating of regional databases relating to human influences on watershed processes, including watershed features such as forest roads and culverts.

Although instream and riparian/upslope responses are reasonably straightforward to quantify, the processes that link them are diverse and can vary strongly in time and space (Beechie et al. 2010). Consequently, establishing specific process or causal linkages between riparian and upslope conditions and what is observable in stream channels can be extremely challenging (Reeves et al. 2004). Our intent here is to provide a more detailed depiction of process linkages between riparian/upslope and instream conditions in watersheds to establish a foundation for interpreting the processes that link them. For the purpose of assessing watershed condition, the following describes our logic and approach.

Ultimately, desired watershed conditions are intended to support processes that address the aquatic conservation strategy objectives within the Northwest Forest Plan (NWFP) (see box 1.1). Here we focus on watershed processes influencing major instream conditions monitored by the AREMP: sediment, channel geometry, large wood, and temperature. These can be framed in terms of the interplay of riparian and upslope watershed processes that drive instream responses. For each general category (stream discharge, sediment, wood, and temperature), we provide background on key processes in play. With this, we describe our measures of responses in streams and how we expect available indicators of riparian and upslope processes should influence them.

Stream Discharge

The volume of running water over time (i.e., stream discharge), including the timing, magnitude, duration, and frequency of flows, is considered to be the master variable that drives nearly all processes in stream ecosystems (Poff et al. 1997). Although stream discharge is a critically important factor, it is labor-intensive to quantify (Turnipseed and Sauer 2010), and consequently most assessments of discharge are based on stream gages operated by the USDI Geological Survey (USGS 2021). Data from USGS stream gages are heavily used for countless applications, but discharge at these locations may not be representative of conditions upstream (Deweber et al. 2014, Falcone et al. 2010, Kovach et al. 2019), including nearly all of the streams monitored by the AREMP. Given the importance of stream discharge and the difficulty of quantifying it within streams sampled by the AREMP, we opted for a model-based assessment across the NWFP area.

We used the National Hydrologic Model for use with Precipitation-Runoff Modeling System (NHM-PRMS) (Regan et al. 2018) to estimate streamflows within subwatersheds monitored by the AREMP. The NHM-PRMS was developed by USGS and has been continually enhanced to support coordinated, comprehensive, and consistent hydrologic modeling at multiple scales (Regan et al. 2018). PRMS is a deterministic hydrologic model that represents hydrologic processes (box A5.1) at a daily time step.

Box A5.1

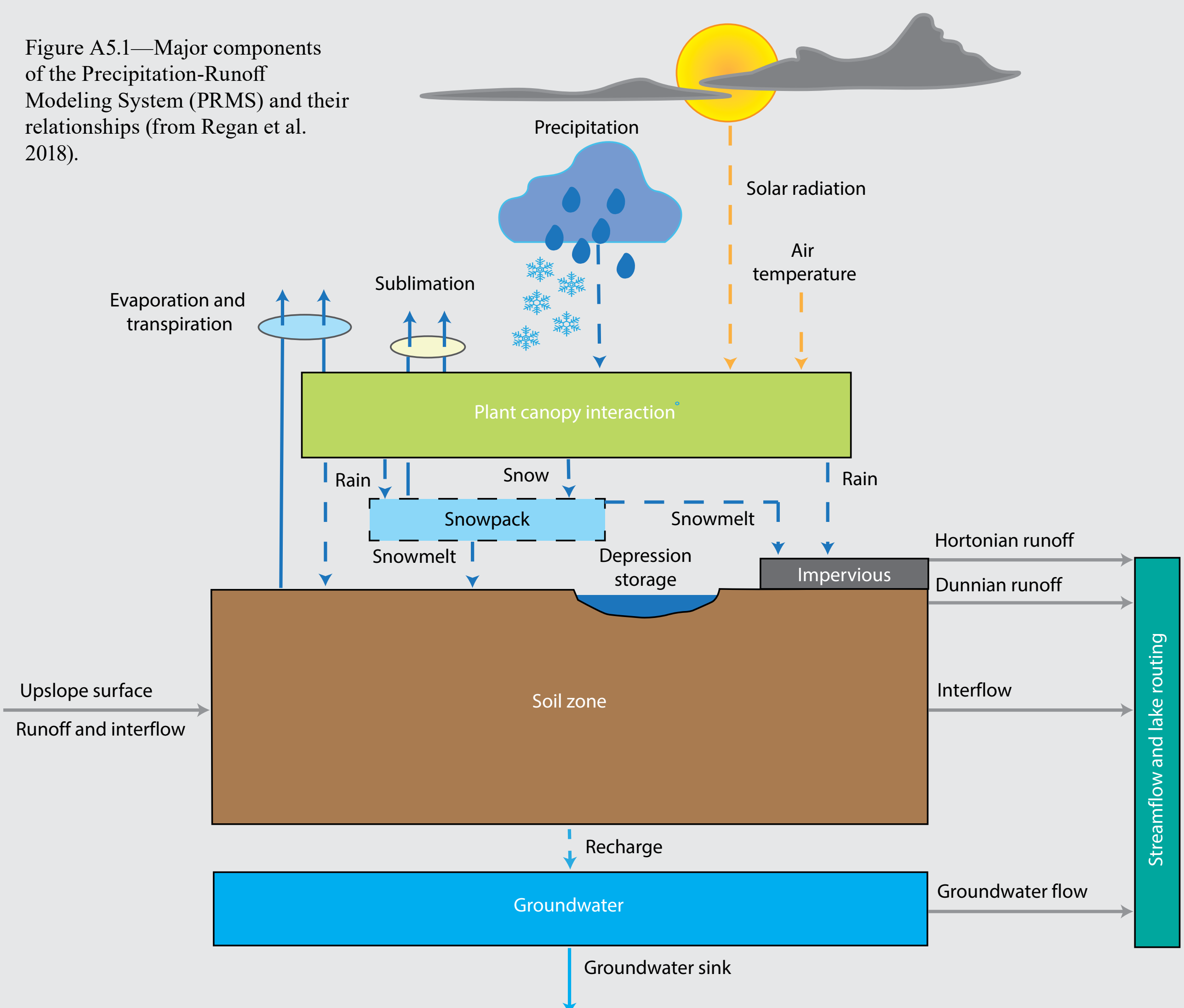
The Precipitation-Runoff Modeling System in a Nutshell

The Precipitation-Runoff Modeling System (PRMS) is a deterministic, distributed-parameter, physical process-based hydrologic model (Markstrom et al. 2015). Hydrological processes in PRMS are simulated by interacting modules, often at the daily time step. Current configuration of the PRMS (see Regan and LaFontaine 2017, Regan et al. 2018) includes modules for solar radiation, temperature, precipitation, evapotranspiration, canopy interception, snow, surface runoff, depression storage, lakes, soil zone, groundwater, streamflow, stream temperature, water use, and model summary output.

A PRMS model domain is discretized into interacting hydrologic response units (HRUs), subsurface reservoirs, groundwater reservoirs, and stream segments. The spatial distribution of HRUs and reservoirs can be grid-based or based on irregular polygons, e.g., the east and west aspects of a north-to-south running stream segment.

PRMS simulations are driven by meteorological forcing data, either gridded or station based, that are distributed across the model domain. Common inputs to the PRMS include daily minimum and maximum air temperature and daily precipitation amount.

Figure A5.1—Major components of the Precipitation-Runoff Modeling System (PRMS) and their relationships (from Regan et al. 2018).



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Model run parameters are distributed to each HRU, reservoir, and segment in the PRMS. These parameters control the function and interaction of modules and can be calibrated to fit observations, such as streamflow; remote-sensing products, such as snow-covered area; and other model output, such as evapotranspiration from a land-surface model. Some parameters have time-dynamic capabilities, e.g., forest cover density varying through time in response to urban development, forest health, or management.

Output variables representing the simulated hydrologic state of each module in PRMS are available at user-defined time steps and at varying spatial scales within the model domain. Common output includes streamflow, basin snowpack, groundwater recharge, and soil moisture, at segment, HRU, subbasin, and basin scales.

NHM-PRMS-based applications provide information to enable more effective water resources planning and management. Automated model subsetting and nesting facilitates regional to local applications (Hay and LaFontaine 2020) and fine resolution modeling where deemed necessary. At the CONUS scale, daily discharge is estimated within spatial units called hydrologic response units (HRUs).

Temperature

Like stream discharge, temperature is another factor that drives nearly every physical and biological process in stream ecosystems (Allan and Castillo 2007). Temperature is also a top-ranking cause of water quality impairment in the AREMP area (Poole et al. 2004) and is sensitive to the influences of forest management (Moore et al. 2005). A host of protective temperature standards have been applied across Western states for sensitive coldwater species, such as salmon and trout (summarized in Falke et al. 2016).

Empirical research on temperatures of small, forested streams indicates that shortwave solar radiation ($H_{shortwave}$) and surface-groundwater heat exchanges ($H_{convection}$) are often the most important components of the heat budget (Moore et al. 2005) (box A5.2). Shading by riparian and upslope vegetation can influence shortwave solar radiation effects on stream temperature (fig. A5.2). The effects of riparian and upslope vegetation on stream temperature are functions of distance from the stream, canopy or tree density, and tree heights (Dewalle 2010). The importance of these factors can depend on watershed aspect, presence of topographic shading, stream size, and flux of groundwater (Moore et al. 2005). Overall, theory and the existing body of empirical evidence indicate that shading by riparian or upslope vegetation is more important for smaller streams with southern-facing aspects, low availability of groundwater, and low topographic relief or shading. In other words, water temperatures in streams with these characteristics would be expected to be most sensitive to changes in watershed condition, as indicated by changes in forest cover and other activities associated with human uses of forests (Moore et al. 2005).

Large Wood

Large wood provides numerous functions in stream ecosystems, ranging from exerting controls on sediment and nutrient cycling to creation of habitat for aquatic biota, including fish (Dolloff and Warren 2003, Gurnell et al. 2002, Keller and Swanson 1979, Wohl et al. 2019). Currently, however, amounts of large wood in most streams within the NWFP area are believed to be far below historical levels. Losses of instream wood have occurred through active removals, splash damming, and log drives in streams, while forest harvest has reduced density of large trees available for recruitment (Miller 2010, Sedell et al. 1988). In the past 40 years, forest and stream management in the Pacific Northwest has focused on practices intended to increase the availability of large wood in streams (Everest and Reeves 2007, Reeves et al. 2018, Wohl et al. 2019). The aquatic conservation strategy of the NWFP specifically mentions instream wood as important to providing desired conditions in watersheds and streams (box 1.1).

Contemporary efforts to restore instream wood represent a major investment for enhancing salmonid populations (Jones et al. 2014, Roni et al. 2014) and to restore more general ecosystem processes influenced by large wood

Box A5.2

Temperature and Stream Heating in a Nutshell

Temperature is a measure of heat energy. In streams, temperature is the result of a balance between heat losses and gains. An overall heat budget for streams can be expressed as follows:

$$H_{net} = H_{shortwave} + H_{longwave} + H_{convection} + H_{latent} + H_{conduction}$$

(Boyd 1996, Boyd and Kasper 2003, Wondzell et al. 2019). In this equation, H_{net} is the net heat flux into or out of a stream reach. Net heat flux can be partitioned into several components. The first two components are shortwave and longwave radiation. Radiation is a means of heat transfer that does not require contact between a heat source and the heated object. Shortwave radiation, $H_{shortwave}$, is heat contributed by solar radiation, also called insolation. Longwave radiation is radiation emitted by the Earth's surface and atmosphere, also known as emittance. Streams receive and emit longwave radiation, and $H_{longwave}$ is the net gain or loss of heat that results from these. $H_{convection}$ is the transfer of heat through convection, or mixing, through the movement of liquids or gases. Mixing of groundwater and surface water can be an important source of heat gain or loss. H_{latent} is the loss or gain of heat that occurs when water changes in state, and without a change in temperature. For example, when liquid water changes to vapor (evaporation), the heat required for this process results in a loss of heat from the stream to the adjacent air: a process more commonly referred to as evaporative cooling. Sensible heat exchanges refer to changes in stream temperature that do not result in a change in phase (e.g., from liquid to vapor). Conduction ($H_{conduction}$) is the transfer of heat that occurs by direct contact of two bodies, without the need for active mixing (convection). In streams, conduction drives the exchange of heat between air and water at the air-water interface, and exchange of heat between the streambed and stream.

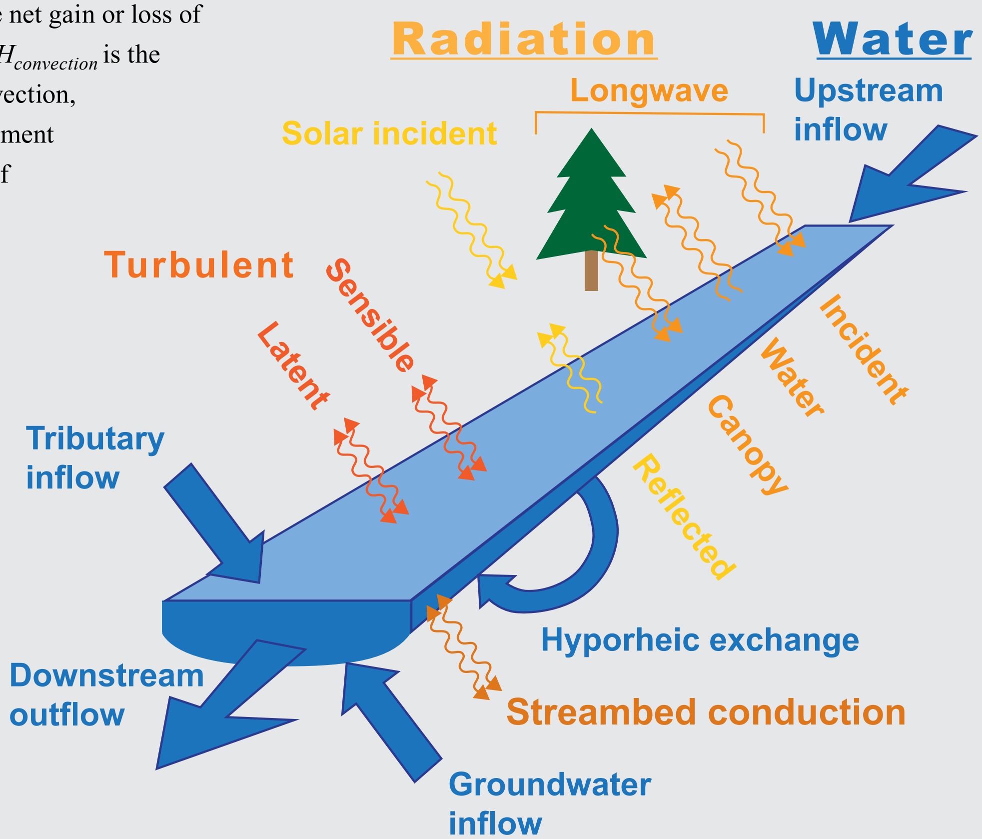

Figure A5.2—Heat fluxes that drive stream temperatures in forested streams not subject to flow modifications. Blue arrows indicate heat exchanges related to fluxes of surface or subsurface streamflows. Modified from Moore et al. (2005).

(Wohl et al. 2019). Although there is no question that active wood placements can provide benefits to specific stream reaches or small watersheds (Roni et al. 2014), this practice does not replace the need for restoring natural processes at broad extents (Beechie et al. 2010, Benda et al. 2016). For example, Jones et al. (2014) reviewed 91 instream wood placements in the central Oregon Coast Range and estimated they benefited <10 percent of the length of streams used by coho salmon, a major target of restoration. Ultimately, natural recruitment of wood through improved forest conditions is likely needed to restore streams to more desirable conditions in the NWFP area. An understanding of the full range of processes that influence recruitment and retention of wood in stream channels is needed to better inform management (box A5.3).

Fine Sediment

Bed forms of rivers are largely controlled by sediment transport and deposition (Wohl et al. 2015). The distribution of sediment or grain sizes in rivers can vary according to a host of geological (geological formations or rock types), geomorphic (landforms), and hydrological influences (O'Connor et al. 2014). The resulting distribution of grain sizes can have a host of physical and ecological influences on rivers. Larger or coarser gravels and cobbles, which tend to slide or roll along the channel bed as bed load, form the physical structure of gravel-bedded rivers and influence channel roughness and salmon spawning suitability (Sear and Devries 2008). In contrast, the smallest grain material (very fine silt and clay) typically remains suspended in the water column and moves downstream quickly as wash load, impacting water quality and clarity but having

Box A5.3

Instream Large Wood Dynamics in a Nutshell

Wood enters streams at multiple scales. Fine scale recruitment happens with individual tree mortality or localized bank erosion, and recruitment over large areas occurs with relatively infrequent, episodic processes involving major erosional events linked to large floods, or factors such as pathogens and wildfire that can kill large numbers of trees (Benda and Sias 2003, Keller and Swanson 1979, May and Gresswell 2003, Meleason et al. 2003, Wohl 2017). Time since harvest or other disturbance is a major influence in determining the size of trees available for recruitment.

To better understand expectations for status and trends of instream wood across the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) area, we begin with a simple model of the dynamics of wood in streams (Benda and Sias 2003, Wohl 2017):

$$\Delta S = [L_i - L_o + Q_i/\Delta X - Q_o/\Delta X - D]\ \Delta t,$$

where ΔS represents change in wood (e.g., volume of wood per m² of streambed) over time (Δt), L_o represents lateral wood outputs (see below for more details), Q_i represents inputs of wood from upstream delivery over a given distance (ΔX), Q_o represents fluvial exports of wood, and D represents losses of wood due to decay. A key parameter is L_i, lateral wood inputs, which can be further partitioned into the following:

$$L_i = l_m + l_f + l_{be} + l_s + l_e + l_{bv},$$

where l_m represents inputs from individual tree mortality, l_f represents inputs from mass tree mortality, l_{be} represents inputs from bank erosion, l_s represents inputs from hillslope instability, l_e represents inputs from exhumation of old wood within the stream channel, and l_{bv} represents wood inputs from beaver (which could also be lumped into individual tree mortality). Wood removals or inputs from active placement of wood by humans could also be included as terms in this model.

Studies of wood recruitment in streams often point to different processes as being important, and much of this variability can be explained by the study location within the river network, as well as the study timeframe considered (Miller and Burnett 2008, Wohl and Jaeger 2009). With respect to the types of streams, duration of Northwest Forest Plan protections (25 years), and duration

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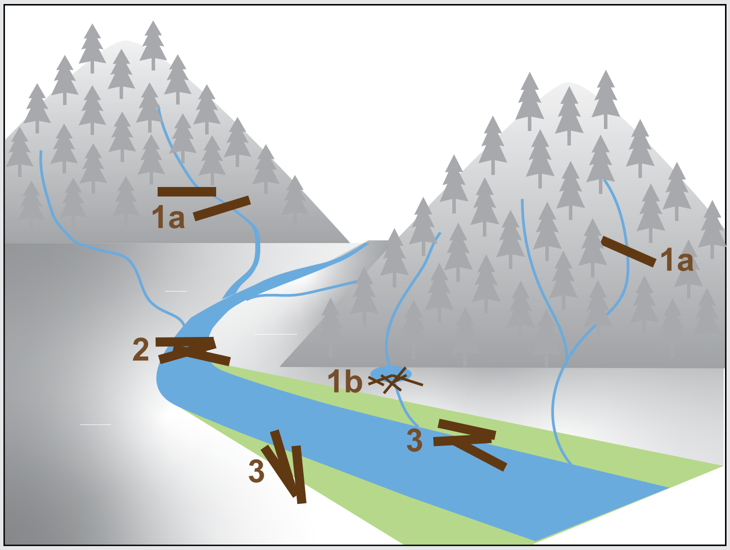
of AREMP instream wood monitoring (17 years), it is more likely that observable changes in instream wood are linked to shorter term processes, acting on annual to decadal time scales (e.g., bank erosion) (Benda and Bigelow 2014). There would likely be a limited number of cases involving infrequent but high-magnitude events (Naiman et al. 1992, Reeves et al. 1995), such as mass mortality of trees from wildfire or forest pathogens, or wood recruitment from hillslopes via debris flows or landslides (fig. A5.3).



Wood

1a Single pieces across channel

1b Beaver dams

2 Wood jams across or partially across channel

3 Stream bank, floodplain wood storage

Processes

Transport-limited
Forest dynamics
Hillslope dynamics

Network (tributary) dynamics

Recruitment-limited
Transport
Channel dynamics
Valley-bottom dynamics

Figure A5.3—Schematic stream network view of wood dynamics. Headwater streams are too small to move wood and are transport-limited. Steeper hillslopes in many upstream areas may contribute wood through erosional processes (e.g., landslides and debris flows). Beaver may contribute instream wood via dam construction activity in headwaters with lower stream slopes (Suzuki and McComb 1998). Mid-order streams are large enough to move large wood pieces but may form jams that cannot be completely washed out and form channel-spanning structures. Downstream, discharges are sufficient to move all wood, and wood is found along stream banks and within the floodplain. Modified from Wohl and Jaeger (2009) and Wohl et al. (2019).

little interaction with the bed. Intermediate-size particles (coarser silt and sand) are typically suspended in the water column during higher flows (suspended load) but may move as bed load during moderate flows and come to rest on the bed during low flows.

Management concerns often focus on fine sediment in streams (Waters 1995). The presence of some amount of fine sediment along the channel bed is normal and benefits some species, such as native lamprey (Gonzalez et al. 2017). However, excess fine sediment deposition can be detrimental to some species, for example, when it reduces salmon egg-to-fry survival by clogging spawning gravels, reducing dissolved oxygen flow and preventing emergence of fry (Jensen et al. 2009, Sear et al. 2008). Excess fine sediment in the water column, including the wash load component, can also negatively impact downstream water quality and clarity and cause a cascade of impacts in stream ecosystems (Wood and Armitage 1997) and human water supplies (Hallema et al. 2018). Here we identified particles of ≤2 mm (b-axis) as fine sediment, which includes particles ranging in size from fine gravel to sand, silt, or clay (in order of decreasing sizes) (Bunte and Abt 2001). An understanding of the full range of processes that influence recruitment and retention of sediment in stream channels is needed to better inform management (box A5.4).

Forest management can increase fine sediment delivery to streams in several ways. Most notably, poor road construction practices and slash burning have been associated with significant increases in sediment delivery from surface erosion, rills, landslides, and debris flows

Box A5.4

Instream Sediment Dynamics in a Nutshell

The fine sediment fraction on the streambed is a function of upstream watershed conditions (supply) and the reach-scale hydrology and channel characteristics that govern local transport capacity. However, there will typically be substantial short-term variability, due to the variable nature of upstream sediment delivery, stream discharge, and transient changes in channel conditions (e.g., the addition or removal of large wood) that influence sediment deposition. We will consider processes that (1) initially supply sediment to stream channels, (2) influence transport or retention within stream channels or reaches, and (3) influence downstream transport to describe important processes that govern sediment dynamics (Wohl et al. 2015).

Sediment is initially supplied to channels through a variety of hillslope erosion processes, including landslides, debris flows, rills, and sheetwashes (Wohl 2013, Wohl et al. 2015). Fine sediment may also enter the channel through the secondary remobilization of fine sediment stored in channel banks and bluffs, or through the abrasion of coarse gravels as they are transported downstream (O'Connor et al. 2014). The natural supply of fine sediment is a function of geology and climate, which dictate the rock strength, topography, vegetation cover, and hydrology of the watershed. Glaciers can also strongly influence fine sediment delivery, both through the contemporary erosive action of modern glaciers (e.g., Jaeger et al. 2017) and through the remobilization of alpine and continental glacial deposits that mantle many landscapes in the Pacific Northwest (e.g., Church and Slaymaker 1989).

Reach-scale sediment transport capacity (or retention) is a function of discharge, channel geometry, and channel roughness. Because most of the sediment load of headwater streams is moved during short periods of high flows (Wohl et al. 2015), the intensity, frequency, and flashiness of those high flows are typically the most important elements of the discharge record. Channel geometry includes the reach-scale slope, which is unlikely to change over monitoring time

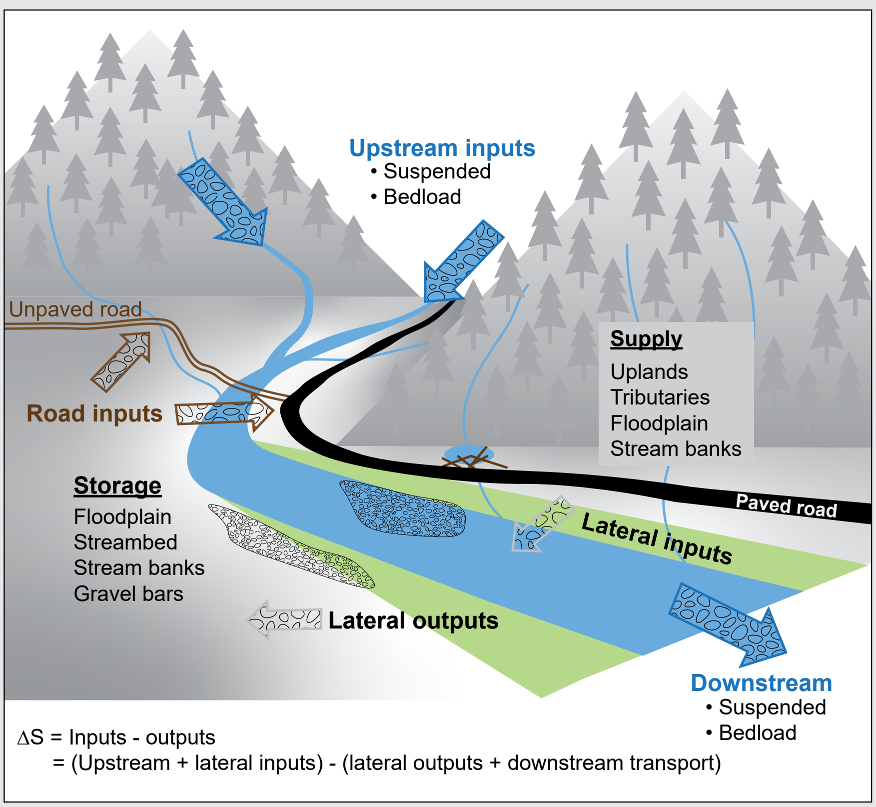


Figure A5.4—Simplified depiction of factors influencing sediment dynamics within a stream network. Overall dynamics are driven by climatic and associated hydrologic variability, landform, and geologic controls that operate on different temporal and spatial scales. Whereas many human influences can affect sediment fluxes, we are emphasizing roads in this analysis, as they are commonly addressed in forest management activities (Luce et al. 2001). Modified from Wohl et al. (2015). ΔS = Change in sediment.

continued on next page

scales, and channel width, which can adjust. Channel roughness reflects the combined influences of bank roughness, bed roughness, both grain scale and bed-form scale, and roughness from instream wood. All else being equal, rougher channels tend to have a reduced transport capacity (greater retention) and finer beds; complex, rough channels also tend to have patchy and spatially variable grain-size distributions (Buffington and Montgomery 1999).

Fine sediment tends to move downstream relatively quickly. Individual particles may travel dozens of kilometers during individual storm events (e.g., Bonniwell et al. 1999); rivers subjected to large pulses of sediment tend to pass much of the finer material through the basin over periods of months to years (East et al. 2015, Platts and Megahan 1975). The response time for channel sediment conditions following changes in reach or watershed conditions is then expected to be on the order of months to several years.

(Anderson and Potts 1987, Beschta 1978, Megahan and Kidd 1972). Clearcut areas may also experience a transient increase in landslides several years after harvest, as roots decay and soil cohesion decreases (Reid and Keppeler 2012). The degree to which these hillslope erosion processes impact streams will necessarily depend on how much of the eroded material ends up in streams, making location and landscape connectivity a key control on downstream impacts (Nelson et al. 2014). Forest harvest may also indirectly increase sediment loads through decreased bank cohesion associated with the removal of bank vegetation and increased streamflow associated with decreased evapotranspiration and increased surface run-off efficiency (Lewis and Keppeler 2007). Both reductions in bank cohesion and increases in streamflow can increase the rate at which channel bank sediment is eroded.

The sediment response to forest harvest in a given watershed will depend strongly on the geology and topography of the harvested area and the specific practices employed (Bywater-Reyes et al. 2017, Cristan et al. 2016, Kaufmann et al. 2009, Luce and Black 1999). The most significant sediment impacts have typically occurred in steep watersheds with weak lithology, subjected to high-impact activities, including large clearcut areas, poor road design, the absence of riparian buffers, and broadcast slash burning (Beschta 1978, Brown and Krygier 1971, Fredriksen 1970, Grant and Wolff 1991). Post-harvest sediment responses are often less obvious in watersheds with more coherent lithologies, lower gradients, or where contemporary forest practices have been applied (Cristan et al. 2016, Hatten et al. 2018, Lewis 1998, Megahan et al. 1992, Stednick 2008). However, older roads constructed to lower standards may continue to influence contemporary sediment delivery, creating a legacy impact that can overshadow the improvement of modern practices (Madej et al. 2012).

The time required to return to pre-disturbance sediment conditions following logging impacts has ranged from as little as a single year to more than a decade; the maximum response has most often occurred within the first 2 years (Bathurst and Iroumé 2014, Safeeq et al. 2020). Given the strong modulating roles of geology and forest harvest practices, no consistent relation between the spatial extent of logging and the degree of sediment response has been observed (Bathurst and Iroumé 2014).

Appendix 6: Supplementary Data

Here we present additional information that may be of interest to readers. Additional information includes figures of field-based instream trends (relative wetted width, instream wood, percentage of fine sediment, particle distribution, and macroinvertebrate ratio of observed to expected) by local administrative unit. Additional summaries for upslope subwatershed descriptions are included for GRAIP Lite maps of road density at the subwatershed scale. More information is included regarding the distribution of unknown but potential culverts.

Field-Based Instream Trends by Local Unit

Wetted Width

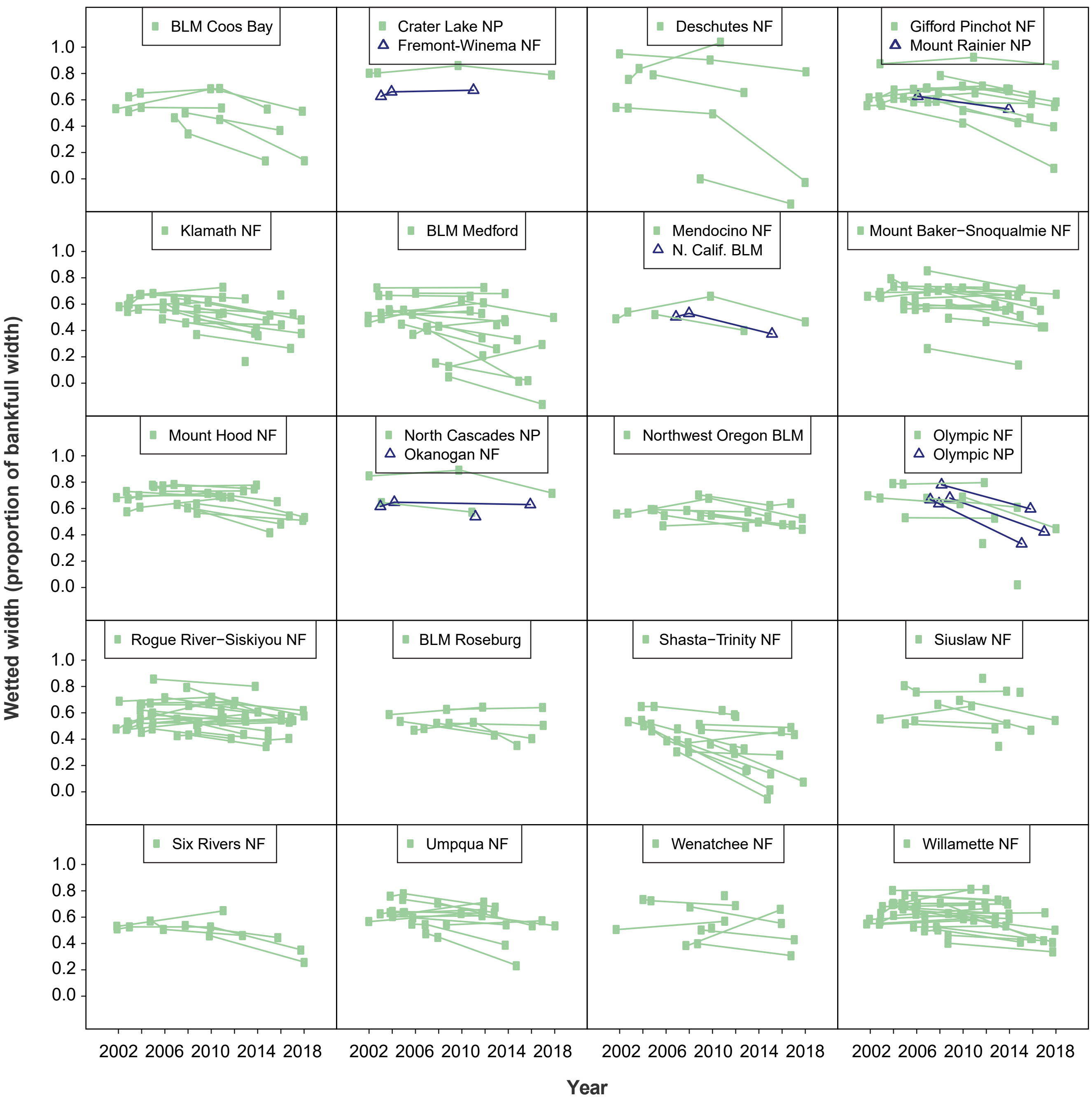


Figure A6.1—Subwatershed-level estimates of stream wetted width as a proportion of long-term average bankfull width by spatial administrative groupings. Negative values indicate dry streams. Fitted values apply to the average day of year in the sample, July 27. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

Instream Wood

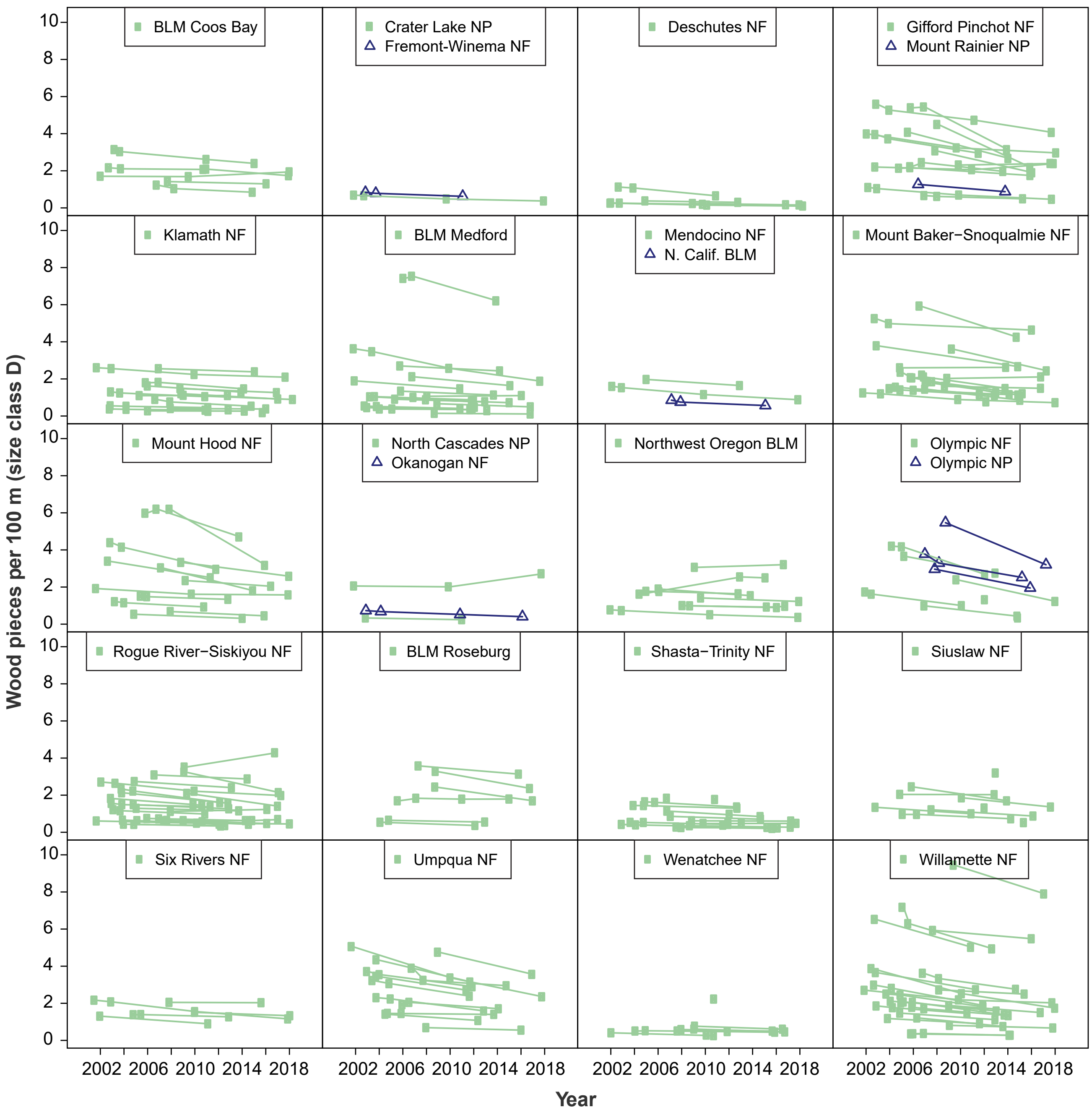


Figure A6.2— Subwatershed-level estimates of instream wood (size class D: ≥25 ft in length and ≥24 inches diameter) by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

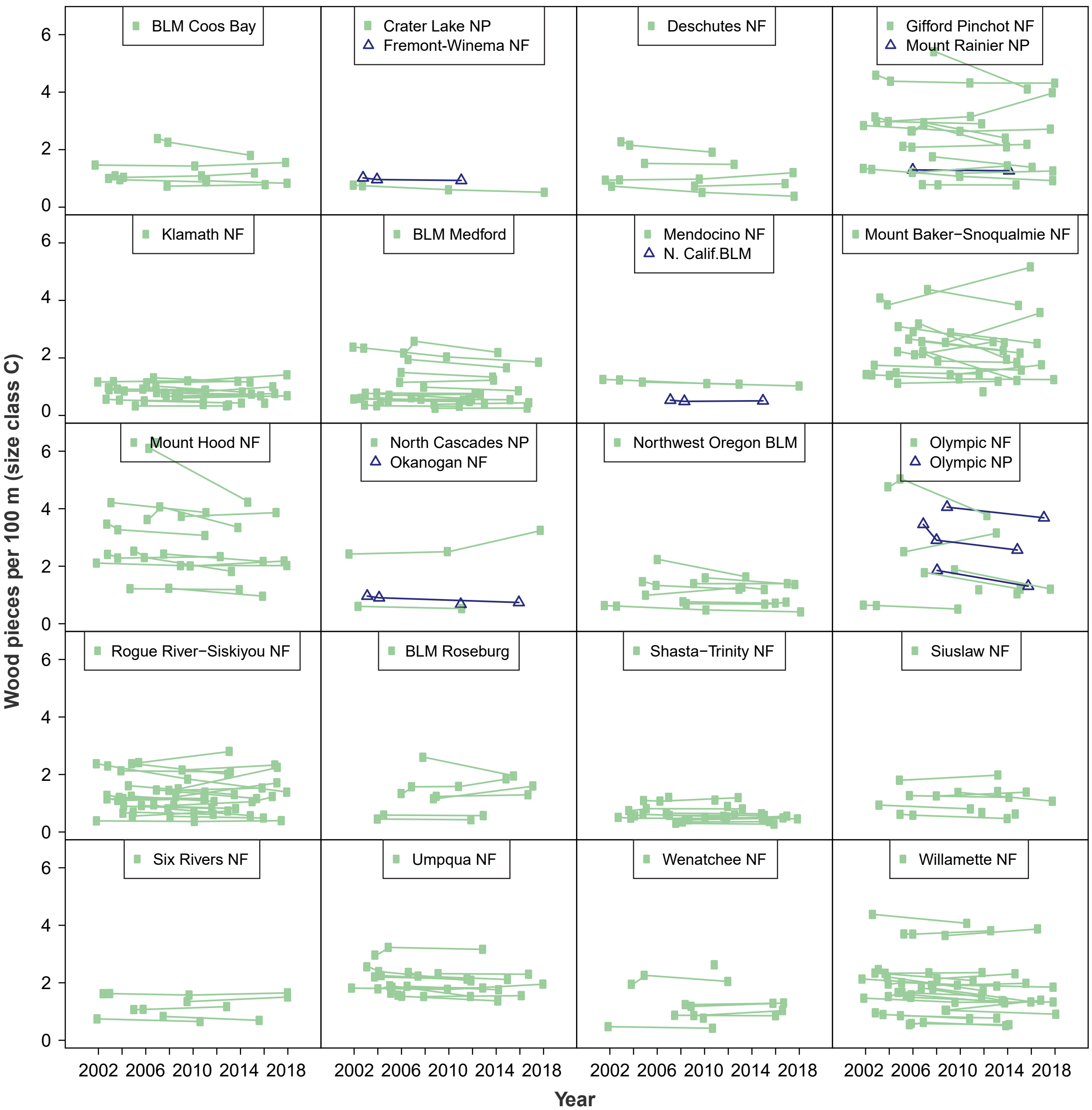


Figure A6.3—Subwatershed-level estimates of instream wood (size class C: ≥25 ft in length, <24 inches diameter and ≥18 inches diameter) by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

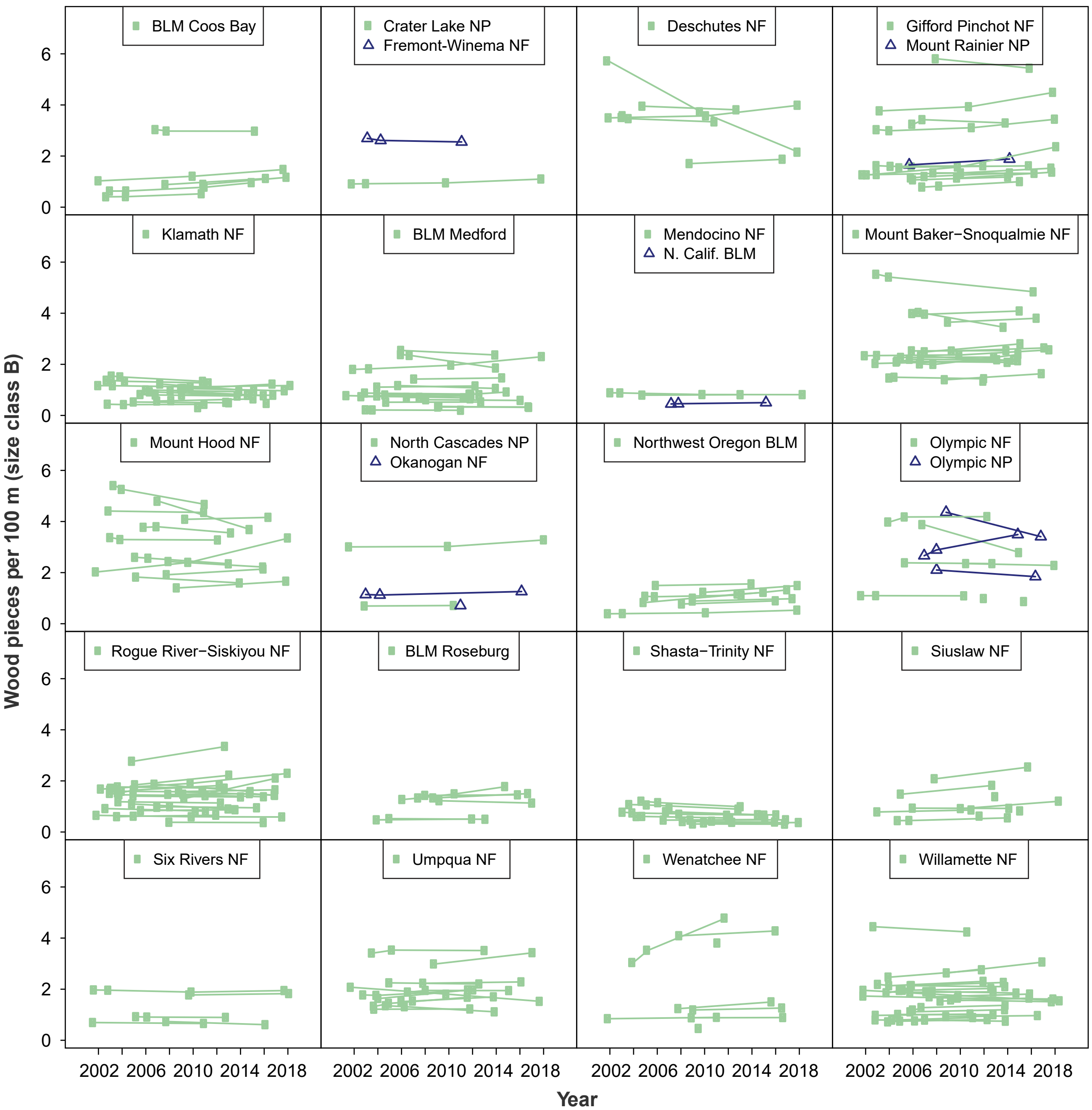


Figure A6.4—Subwatershed-level estimates of instream wood (size class B: ≥25 ft in length, <18 inches diameter and ≥12 inches diameter) by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

Substrate Sediment

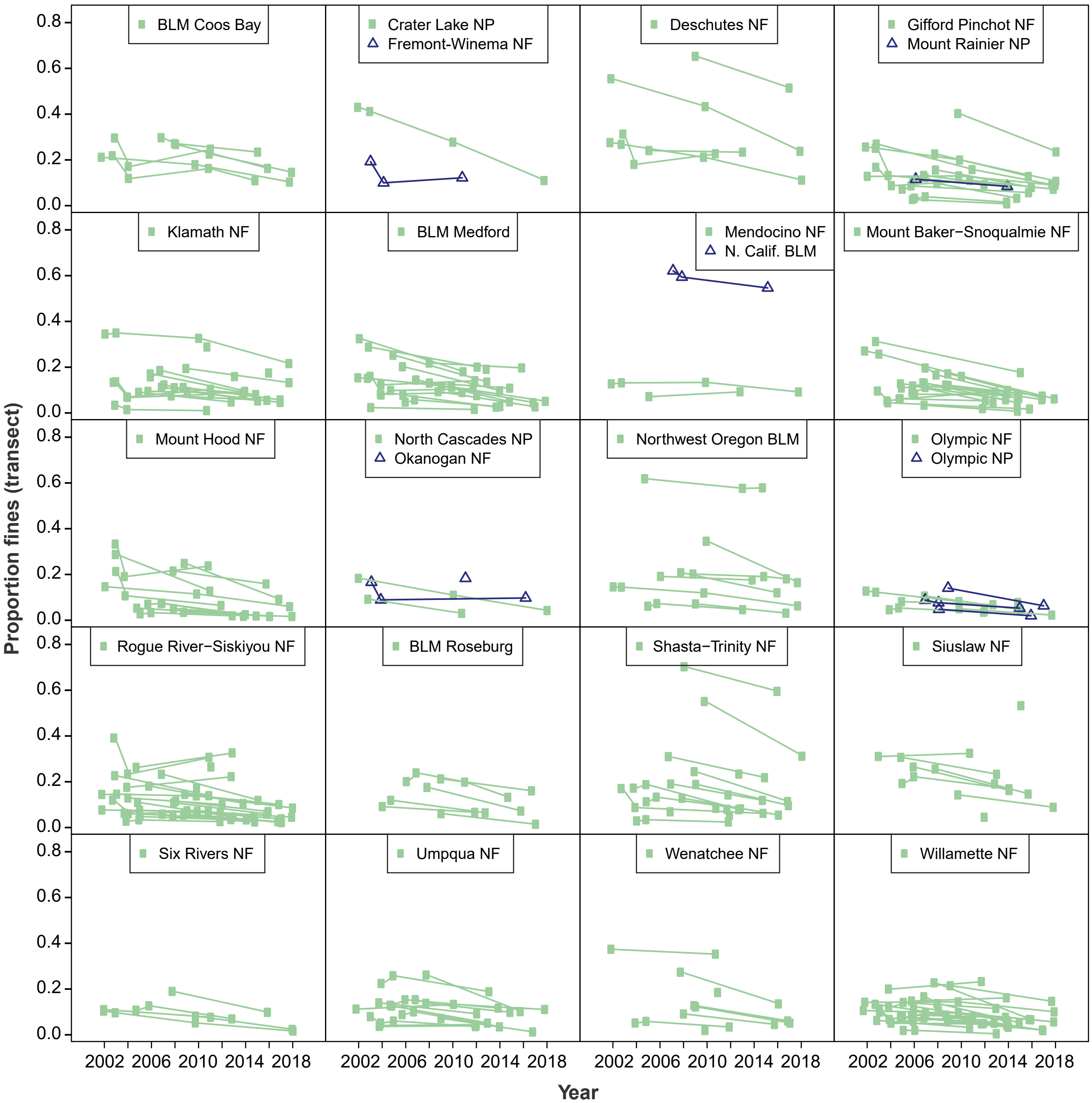


Figure A6.5—Subwatershed-level estimates of the proportion of substrate composed of fine material (≤2 mm) based on transect sampling by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

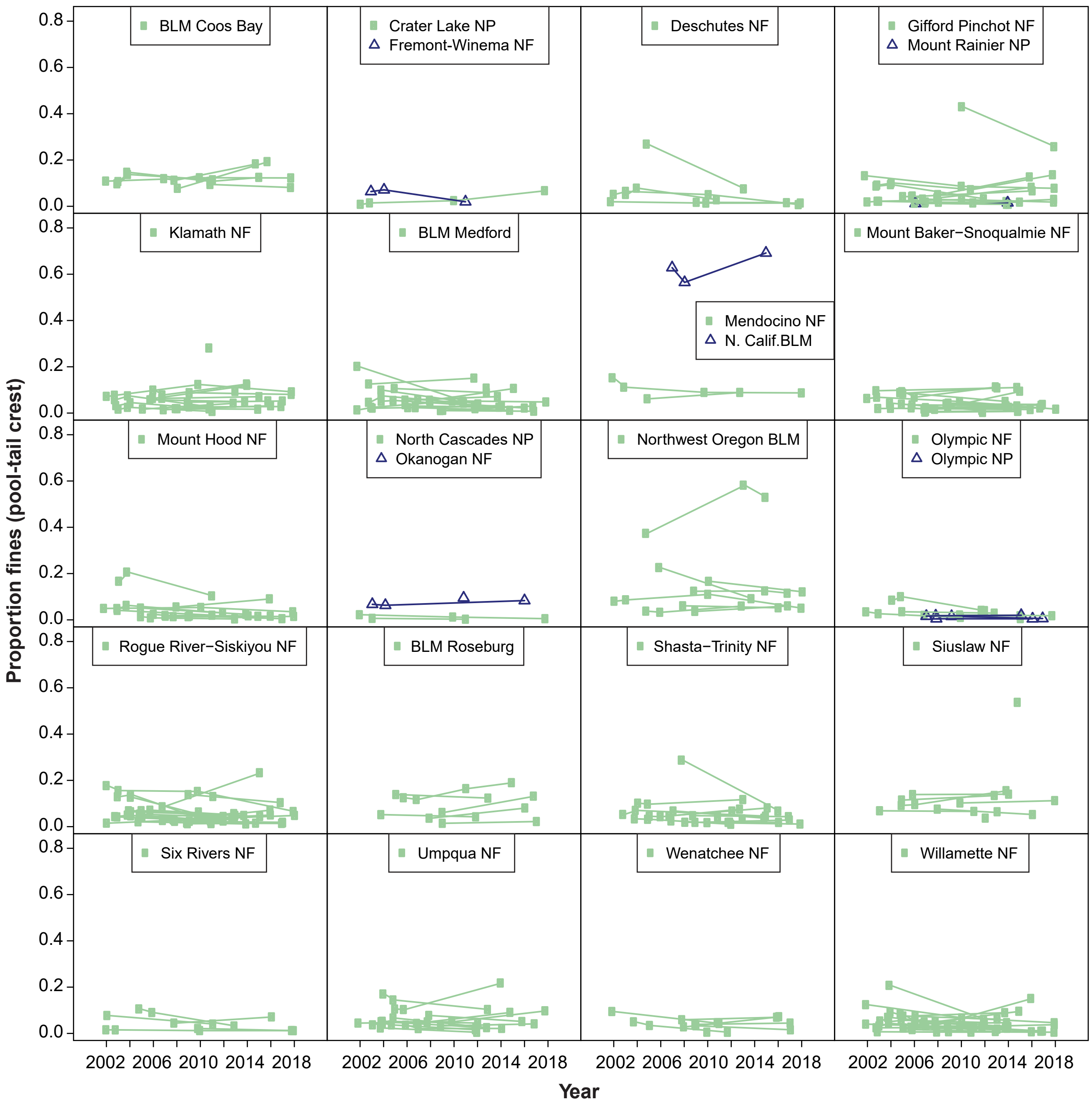


Figure A6.6—Subwatershed-level estimates of the proportion of substrate composed of fine material (≤2 mm) based on pool-tail fines sampling by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

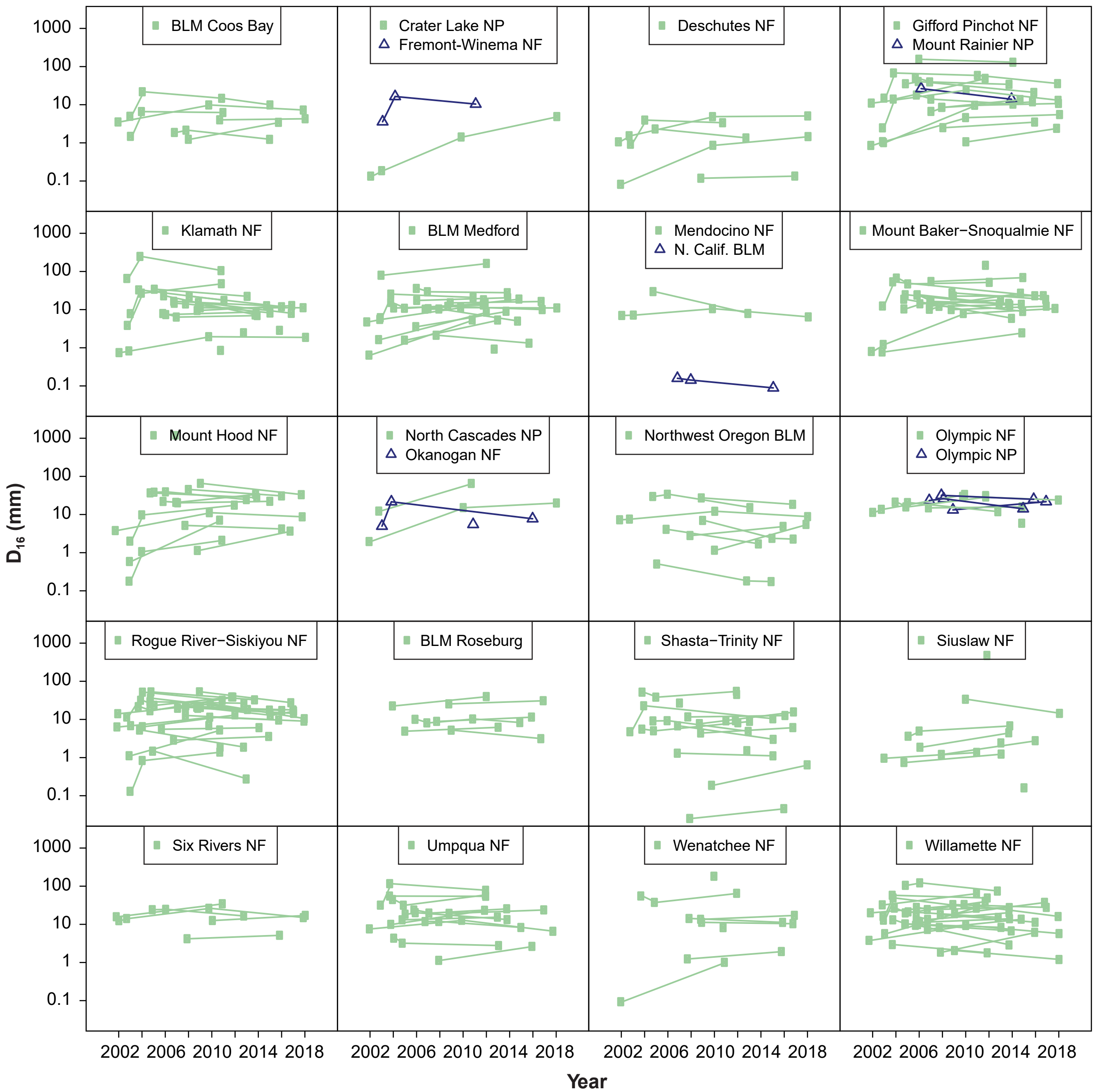


Figure A6.7—Subwatershed-level estimates of substrate particle size widths corresponding to the 16th percentile (D_{16}) of the particle size distribution by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

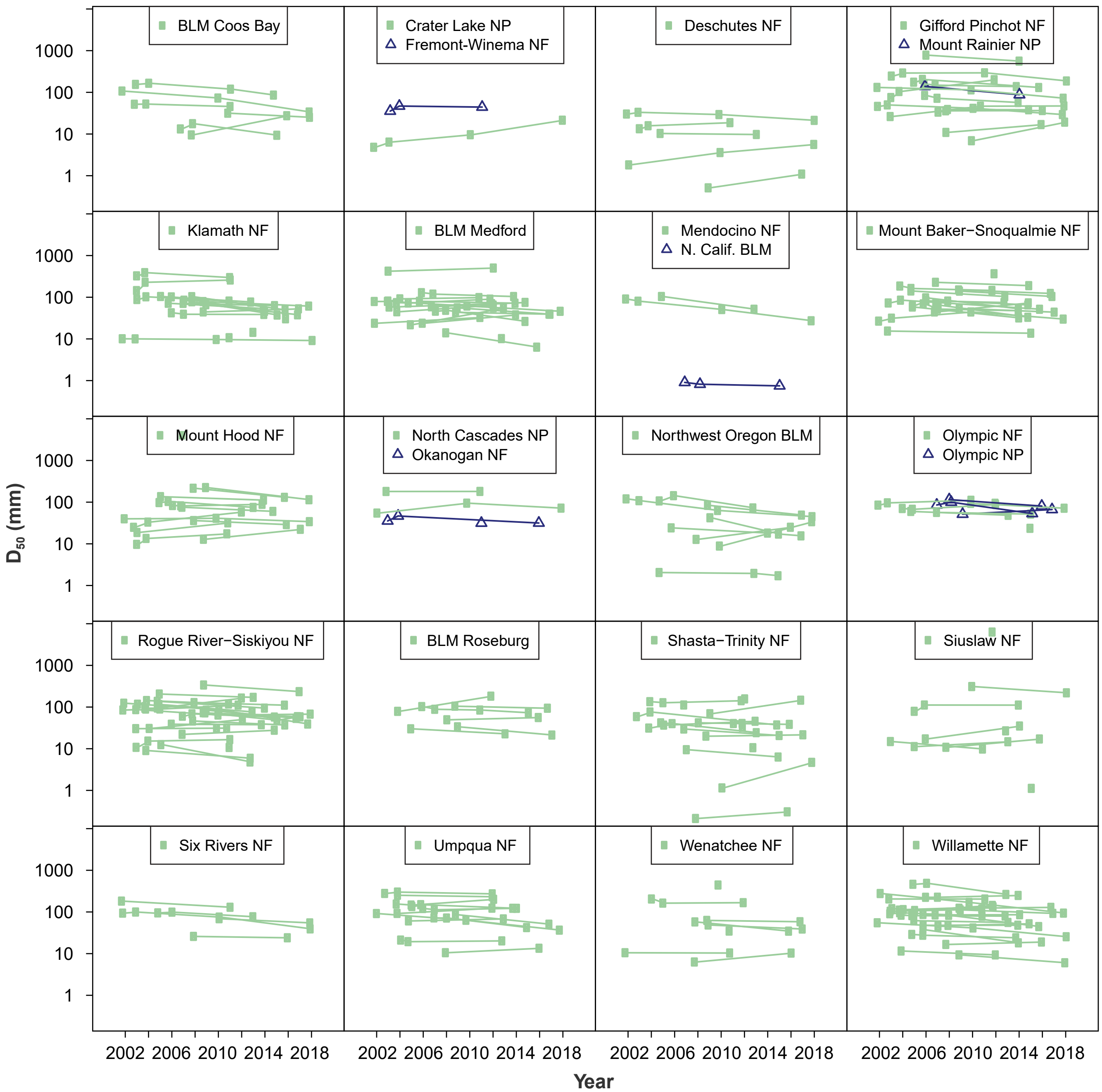


Figure A6.8—Subwatershed-level estimates of substrate particle size widths corresponding to the 50th percentile (D_{50}) of the particle size distribution by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

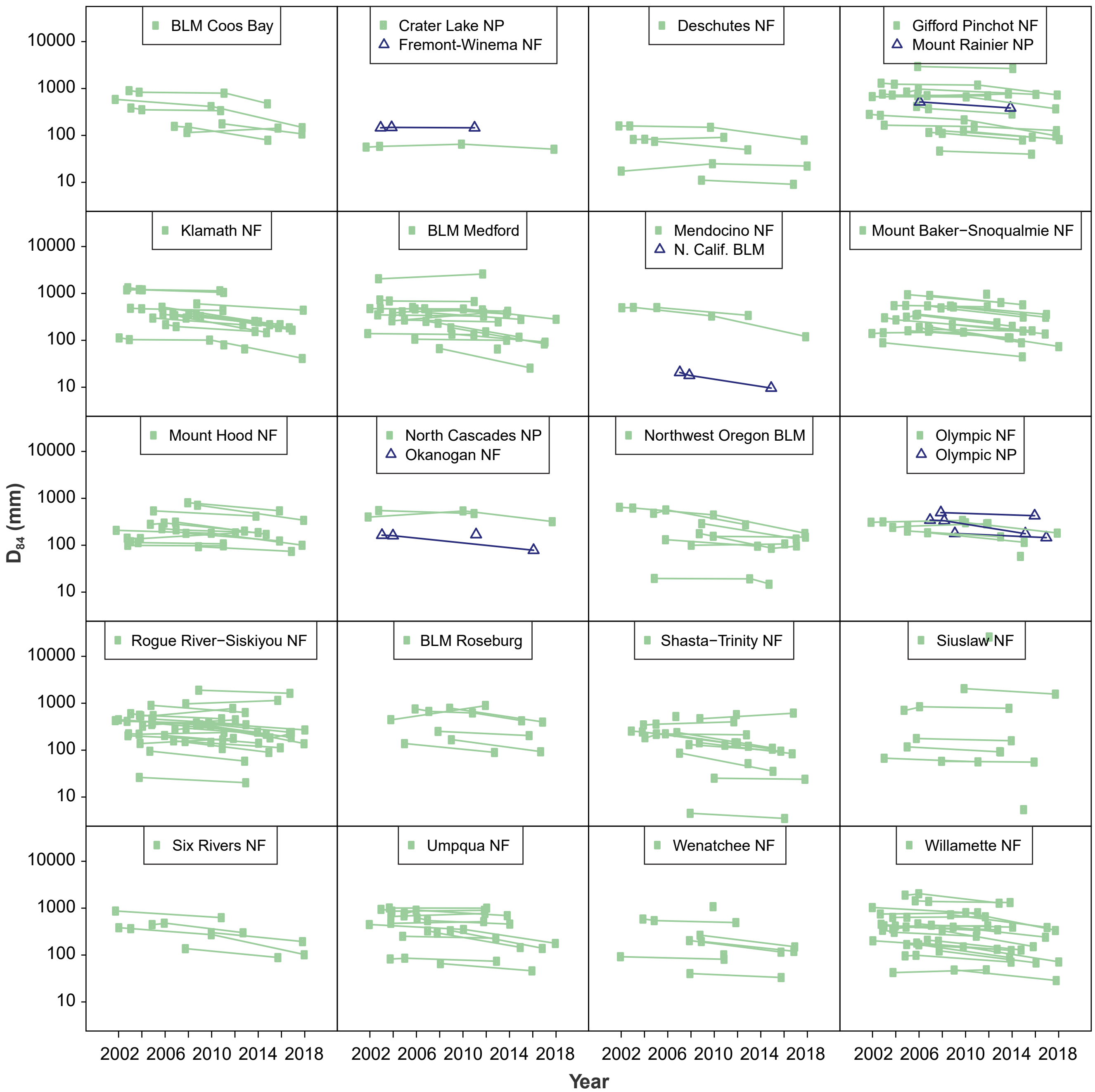


Figure A6.9—Subwatershed-level estimates of substrate particle size widths corresponding to the 84th percentile (D_{84}) of the particle size distribution by spatial administrative groupings. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

Macroinvertebrates

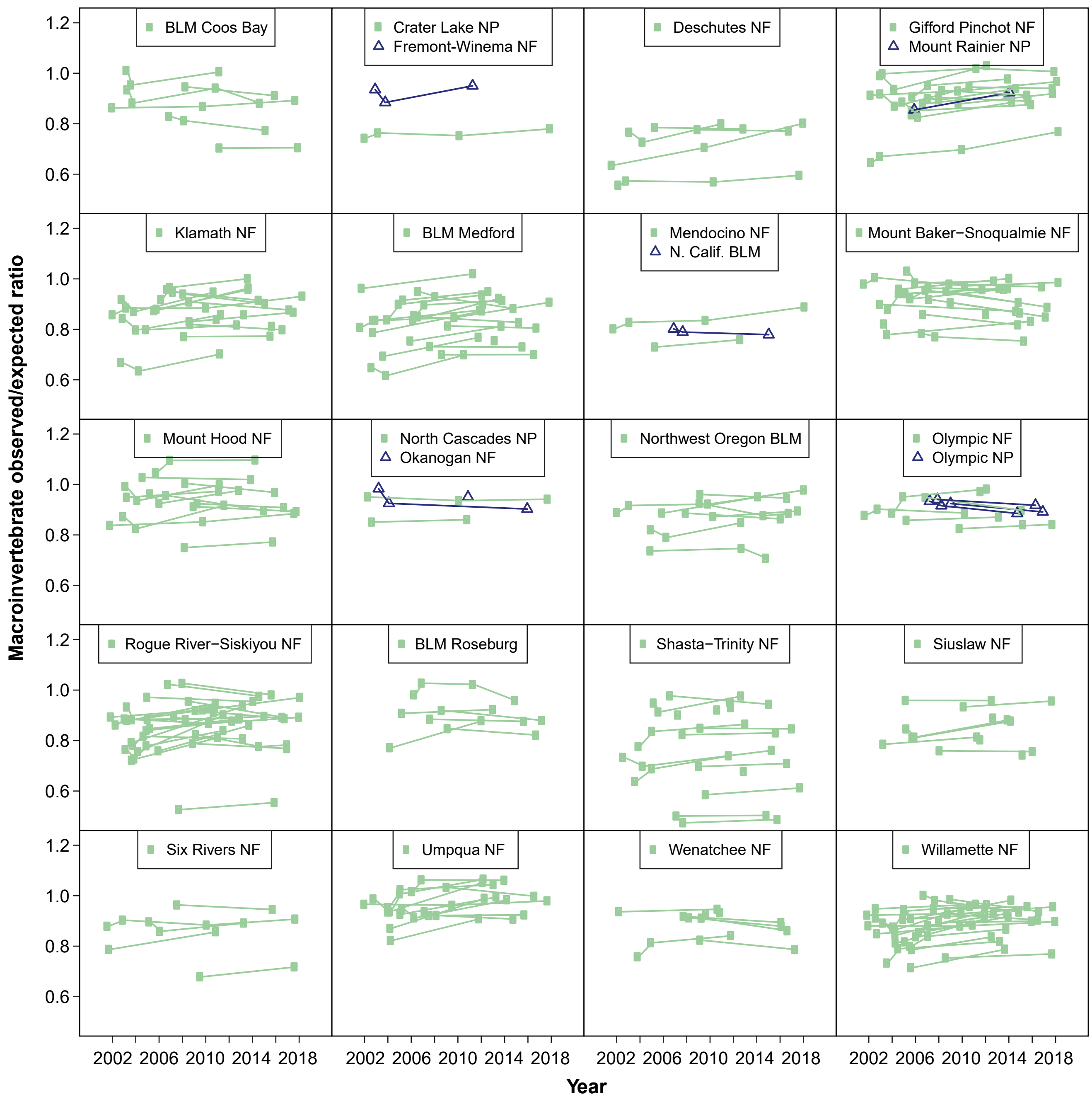


Figure A6.10—Estimated observed to expected ratios of macroinvertebrate assemblages sampled from Aquatic and Riparian Effectiveness Monitoring Program sites across the Northwest Forest Plan area from 2002 through 2018 by administrative grouping. BLM = Bureau of Land Management, NP = National Park, NF = National Forest.

Upslope Additional Information

GRAIP Lite

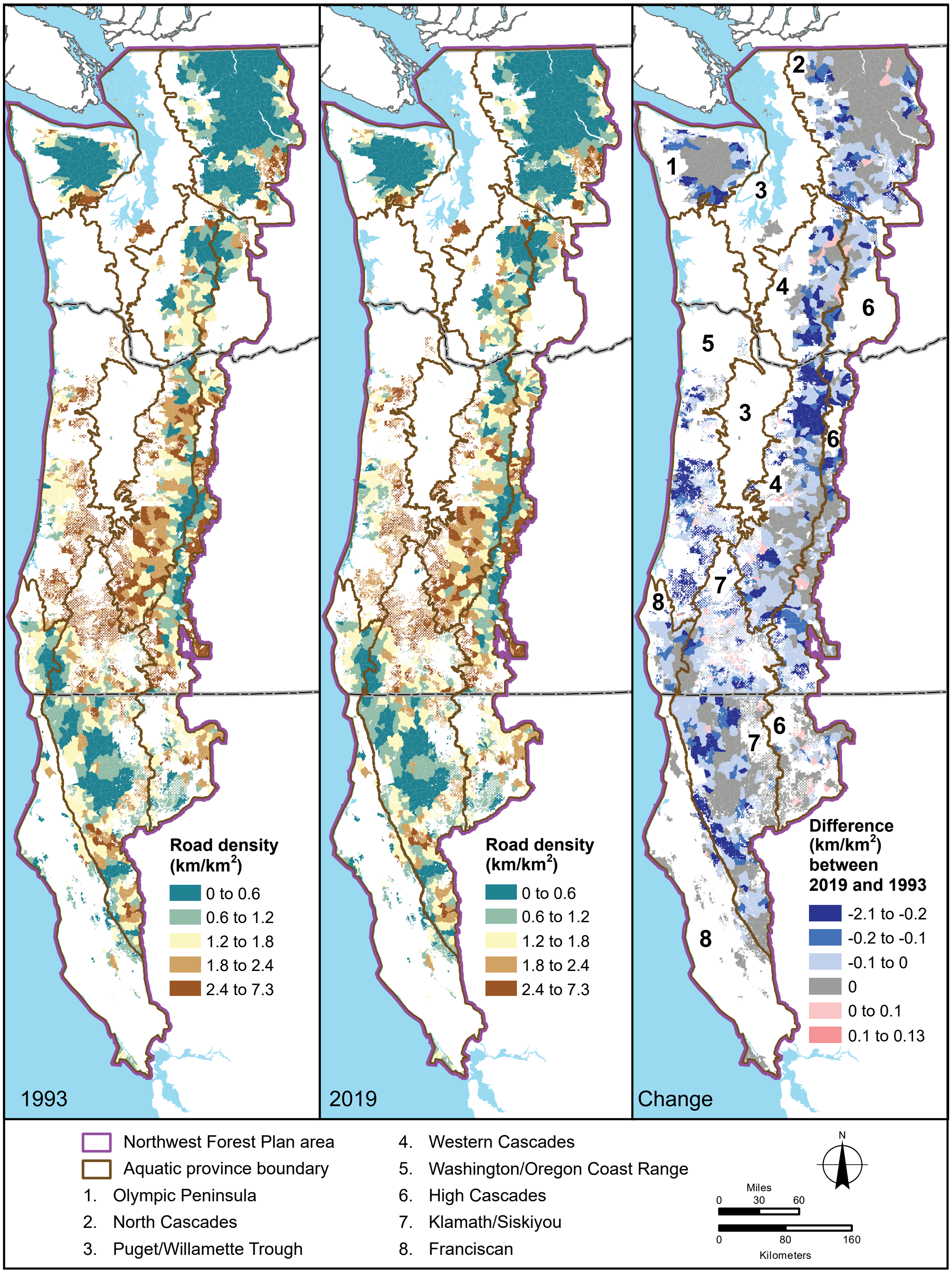


Figure A6.11—Road density values by subwatershed (hydrologic unit code 12) for roads on federal land for 1993, 2019, and the difference between the two time periods.

Culverts

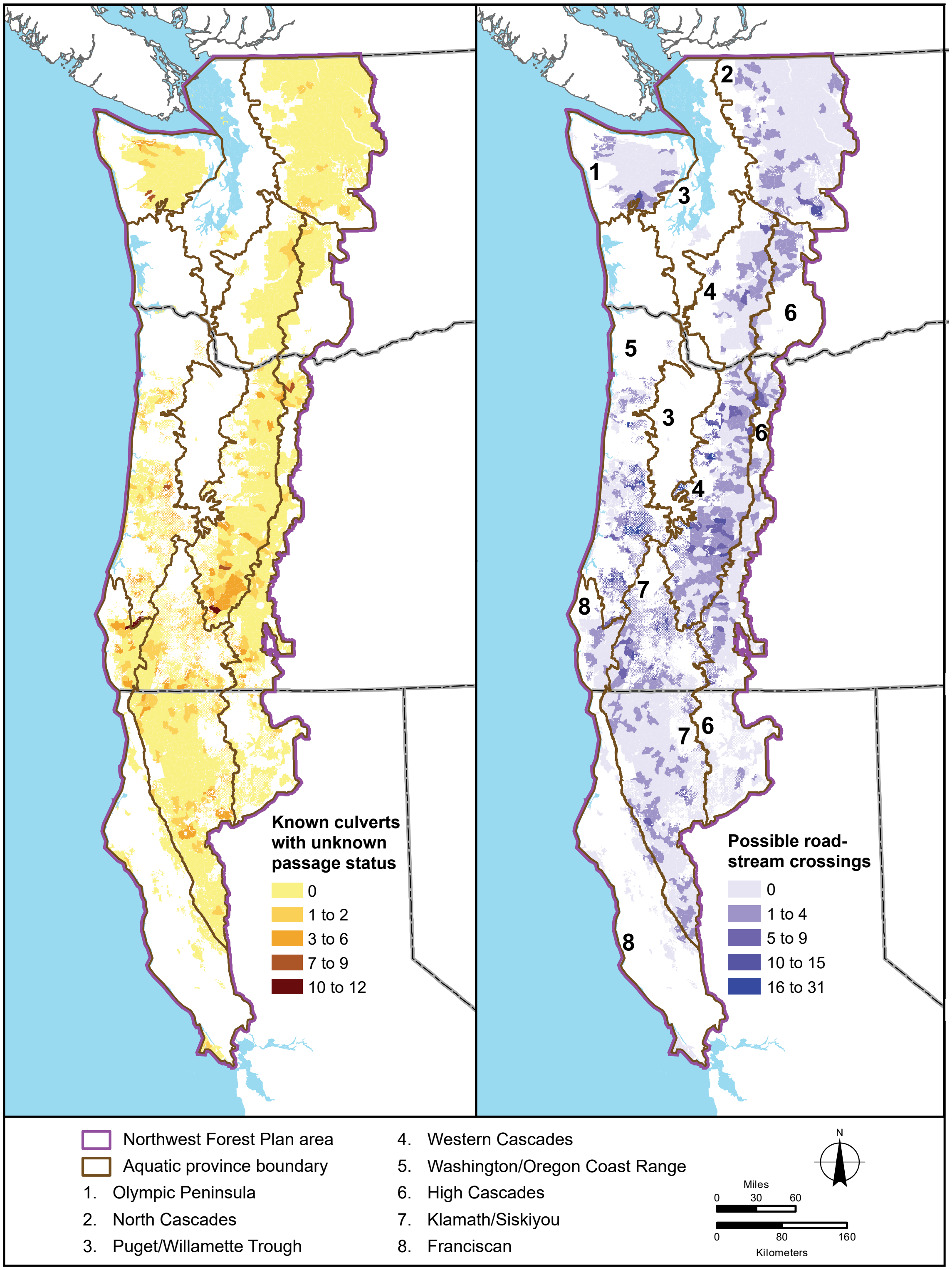


Figure A6.12—Count of culverts with unknown status and count of possible road-stream crossings not identified in an existing culvert database, both by hydrologic unit code 12 subwatershed.

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