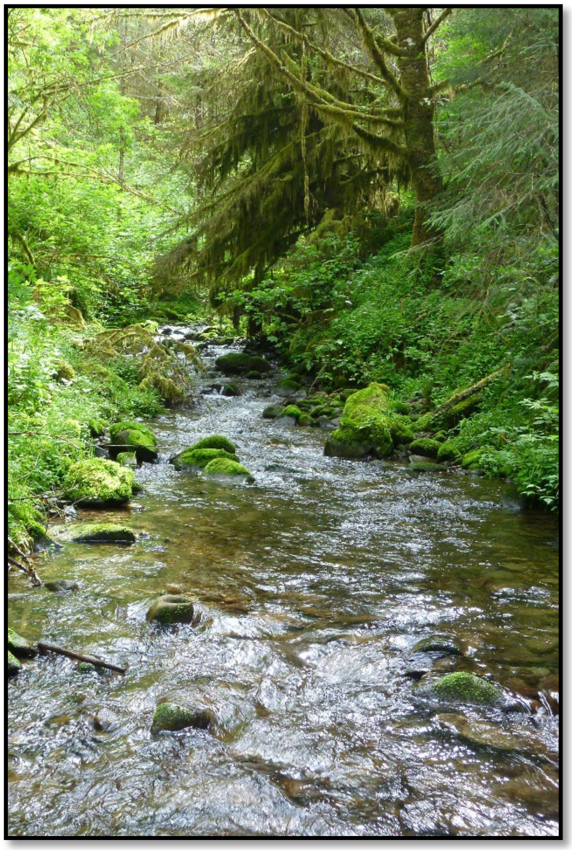
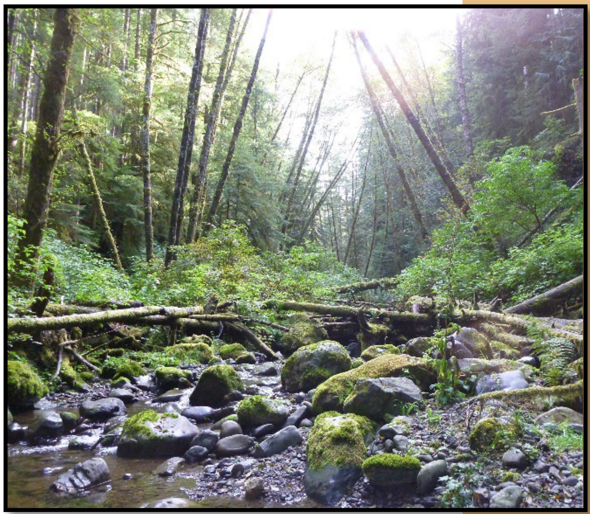
# Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest

## 2013-2020 Results

Published March 2022





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# Acknowledgements


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# Acronyms and Abbreviations

7-DADmax – 7-Day Average Daily Maximum Temperature

BA – Basal Area

DBH – Diameter at Breast Height

DNR – Washington Department of Natural Resources

GIS – Geographic Information System

GPS – Global Positioning System

HCP – Habitat Conservation Plan

LiDAR – Light Detection and Ranging (a remote sensing method)

NOAA – National Oceanic and Atmospheric Administration

OESF – Olympic Experimental State Forest

RCW – Revised Code of Washington

SD – Standard Deviation

TFW – Timber, Fish, and Wildlife

TPA – Trees Per Acre

USDA – United States Department of Agriculture

USFWS – United States Fish and Wildlife Service

USGS – United States Geological Survey

# Executive Summary

Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest (OESF) is conducted to document long-term change in habitat conditions in watersheds managed by Washington State Department of Natural Resources (DNR) for timber, fish and wildlife habitat, and other ecosystem values. The primary objectives are: (*i*) to provide empirical data to evaluate DNR's progress in meeting the 1997 State Trust Lands Habitat Conservation Plan (HCP) riparian conservation objectives, and (*ii*) to reduce uncertainties around the integration of habitat conservation and timber production. This monitoring also provides reliable information to inform DNR's adaptive management.

Status and Trends Monitoring is effectiveness monitoring for the Riparian Conservation Strategy implemented in the OESF. As an HCP commitment, the monitoring focuses on effectiveness of stream buffers and other conservation measures designed to protect physical and biological functions of riparian systems. The central hypothesis tested through this monitoring is that the HCP riparian conservation strategy, implemented through the OESF Forest Land Plan, allows natural processes of disturbance and succession to improve riparian and aquatic habitat conditions over time, relative to conditions prior to adoption of the HCP (DNR 1997).

This monitoring program is an extensive, long-term study of riparian and aquatic habitats of Type 3 streams (the smallest class of fish-bearing streams) in the OESF. Streams in 50 DNR-managed watersheds and in 12 reference watersheds are monitored to document habitat under managed and predominantly unharvested conditions, respectively. Nine aquatic and riparian habitat indicators are sampled in stream reaches near the outlet of each watershed: channel morphology, channel substrate, in-stream large wood, channel habitat units, stream shade, water temperature, stream discharge, riparian microclimate, and riparian forest vegetation. The monitoring program was initiated in 2012; field data collection began in 2013 and has been conducted each year since, sampling streams on a rotating basis.

This report presents habitat status and trend results, based on data collected between 2013 and 2020. The analysis focuses on a set of habitat indicator metrics selected for their relevance to forest management. The report also includes a habitat condition assessment that integrates a series of indicators to produce a habitat condition score for each monitored stream.

Monitoring has provided a clear picture of habitat conditions, though the trends observed so far must be interpreted with caution because they represent a short time interval in the context of ecological change. Results clearly show that the stream buffers in DNR-managed watersheds have produced multiple habitat benefits. All sampled streams were found to be well-shaded; stream temperatures and riparian microclimate remained cool during summer.

Findings show that riparian forests in DNR-managed watersheds vary widely as a result of different disturbance histories; 36% of the DNR-managed riparian forests sampled were similar in structure and composition to the riparian forest condition most prevalent in reference watersheds. In-stream wood conditions suggest an interruption to the long-term input of large-diameter pieces of wood to streams, apparently a result of intensive 20th-century harvesting. In-stream wood plays a key role in creating stream habitat for salmonids and other aquatic species. Though large pieces of wood are still present in streams, the majority are in later stages of decay, reflecting a lack of new, large pieces of wood from riparian forests.

Most of the habitat indicators monitored were found to differ by channel type, which is a function of stream gradient. This supports the idea that sensitivity of streams to management is also a function of stream gradient and management effects are likely to be expressed differently among channel types.

The stream habitat condition assessment ranked as highest-quality salmonid habitat those streams with many large pieces of in-stream wood, large stream size, and complex stream channels. Streams ranking low in salmonid habitat quality were those with a lack of fish cover (e.g., in-stream wood or boulders), small stream size, and simple, more uniform channels.

Anticipated changes in climate are expected to produce warmer stream temperatures, which, when combined with low summer streamflow, may be detrimental to salmonid populations. Deep pools, often created by large in-stream wood, are expected to be refuge habitat for juvenile salmonids in summer, which underscores the importance of large wood in streams.

Status and Trends Monitoring is meeting the OESF goal of developing, using, and distributing information on aquatic and riparian ecosystem processes and their maintenance in commercial forests. The program has thus far produced several peer-reviewed scientific publications, a series of reports, field tours, and public presentations. The project findings have informed new experimental research and have provided data on ecological conditions and relationships that inform HCP priorities. This report includes an analysis of statistical power to guide future monitoring as well as recommendations for adjusting field protocols to optimizing data collection. A budget analysis indicates high efficiency in the monitoring program, mainly due to in-house implementation and effective project management including planning and communication.

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# Introduction

## Status and Trends Monitoring

This report presents findings from Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest (hereafter called “Status and Trends Monitoring”) (Minkova et al. 2012). This monitoring program is an extensive, long-term study of riparian and aquatic habitats of small, fish-bearing streams in the Olympic Experimental State Forest (OESF). The program was initiated in 2012; field data collection began in 2013 and has been conducted each year since then. This report summarizes data collected from 2013 through 2020.

Status and Trends Monitoring is a State Trust Lands HCP effectiveness monitoring commitment, focusing on the Riparian Conservation Strategy implemented in the OESF (DNR 1997; DNR 2016a, p. 3-23). As one of three types of monitoring defined in the HCP (the other two are implementation and validation monitoring), effectiveness monitoring examines “habitat conditions developing over time after a management activity or series of activities” (DNR 2016a, p. 4-19). Its goal is to determine whether implementation of conservation strategies results in anticipated habitat conditions (DNR 1997, p. V.2). The HCP specifically calls for aquatic and riparian research in the OESF (DNR 1997, p. IV.107). This monitoring program is a key part of DNR’s adaptive management in the OESF: long-term monitoring (i.e., 10 or more years) provides data that will help make inferences about management effects on habitat, thus contributing to the adaptive management required by the HCP.

The primary objectives of Status and Trends Monitoring are: (*i*) to generate empirical data that can be used to evaluate whether DNR is meeting the 1997 State Trust Lands HCP riparian conservation objectives, and (*ii*) to reduce uncertainties around the integration of habitat conservation and timber production. The riparian conservation strategy, described in the HCP and implemented through the OESF Forest Land Plan (DNR 2016a), aims to provide habitat for viable salmonid populations and other aquatic and riparian-obligate species (DNR 1997). The strategy operates under the hypothesis that the natural processes of forest succession and disturbance will improve riparian and aquatic habitat conditions across the landscape over time, relative to conditions prior to adoption of the HCP in 1997 (DNR 1997).

A major component of the Riparian Conservation Strategy is the application of largely unmanaged buffers around most streams during timber harvest. The HCP states, “a principal working hypothesis of this approach is that buffers designed to minimize mass wasting and blowdown will be sufficient to protect other key physical and biological functions of riparian systems” (DNR 1997, p. IV.106). By applying buffers, the Forest Land Plan hypothesizes habitat will be maintained or restored as riparian forests supply large woody debris to the stream channel, provide shade, and stabilize stream banks (DNR 1997, p. IV.106; DNR 2016a, p. 3-24). Status and Trends Monitoring measures a series of nine habitat indicators that reflect habitat conditions that are directly or indirectly affected by DNR’s riparian management strategy.

Status and Trends Monitoring has produced a variety of reports, and data collected during monitoring were used to produce several publications in peer-reviewed journals (Table 1). In 2016, a Habitat Status Report was published, describing the initial habitat status after completion of the first round of sampling during 2013 to 2015 (Minkova and Devine 2016). The current report differs from the 2016 status report in that it not only analyzes habitat status (this time using additional data) but also analyzes trends from 2013 to 2020 and includes a habitat condition model.

**Table 1.** Publications describing or reporting results from Status and Trends Monitoring. This list does not include publications from the Riparian Validation Monitoring Program (Martens 2016).

| Publication | Year | Title |
|---|---|---|
| Study plan | 2012 | Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest: Study Plan |
| Establishment report | 2013 | Establishment Report: Field Reconnaissance and Delineation of Sample Sites |
| Establishment report | 2014 | Establishment Report: Field Installations and Development of Monitoring Protocols |
| Progress report | 2015 | Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest: 2014 Progress Report |
| Status and progress report | 2016 | Habitat Status Report and 2015 Project Progress Report |
| Data quality control report | 2016 | Quality Control Report for Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest |
| Status report | 2016 | 2015 Hydrology Status Report |
| Monitoring protocol | 2017 | Status and Trends Monitoring of Riparian and Aquatic Habitat in the Olympic Experimental State Forest: Monitoring Protocols |
| Journal publication | 2019 | Stream conditions after 18 years of passive riparian restoration in small fish-bearing watersheds. Environmental Management. 63(5):673-690. K.D. Martens, W.D. Devine, T.V. Minkova, and A.D. Foster |
| Master's thesis | 2019 | Ecological drivers of forested riparian microclimate on the Olympic Peninsula of Washington state. The Evergreen State College. K.R. Keleher |
| Journal publication | 2020 | Paired air-water annual temperature patterns reveal hydrogeological controls on stream thermal regimes at watershed to continental scales. Journal of Hydrology. 587:124929. Z.C. Johnson, B.G. Johnson, M.A. Briggs, W.D. Devine, C.D. Snyder, N.P. Hitt, D.K. Hare, and T.V. Minkova |
| Journal publication | 2021 | Heed the data gap: guidelines for using incomplete datasets in annual stream temperature analyses. Ecological Indicators. 122:107229. Z.C. Johnson, B.G. Johnson, M.A. Briggs, C.D. Snyder, N.P. Hitt, and W.D. Devine |
| Journal publication | 2021 | Watershed characteristics influence winter stream temperature in a forested landscape. Aquatic Sciences. 83(45):1-17. W.D. Devine, E.A. Steel. A.D. Foster, T.V. Minkova, and K.D. Martens |
| Journal publication | 2022 | Drivers of forested riparian microclimate on the Olympic Peninsula of Washington state. Northwest Science [*In publication*] K.R. Keleher, R.E. Bigley, and W.D. Devine |

# Contents of this report

## SCOPE

This report presents an analysis of the nine monitored aquatic and riparian habitat indicators, based on data collected from 2013 to 2020 under the Status and Trends Monitoring program. The analysis focuses on key habitat indicator metrics relevant to land management but reflects only the beginning of the data's full potential. For example, for the Status and Trends water temperature indicator, we present here an important regulatory temperature metric—average 7-day daily maximum temperature—but three journal articles have already been published using other temperature metrics from the same dataset. In this report, our goal is to present key findings in a comprehensive yet concise way.

## ORGANIZATION

Within this report are four main sections. First, the Introduction provides a background on the monitoring program, study area, and methodology. The second section presents an analysis of the nine habitat indicators. For each indicator, one to three key indicator metrics are reported. These metrics describe the condition of the indicator. For example, for the channel substrate indicator, two metrics are presented: median substrate particle size and percent fines.

The third section of this report is a habitat condition assessment of the monitored streams that calculates a composite score by using a process similar to that used in the OESF Forest Land Plan Environmental Impact Statement riparian analysis (DNR 2016b, p. G-81). This third section differs from the second section in that instead of assessing OESF-wide conditions for each habitat indicator, it integrates many indicators to produce a single habitat condition score for each monitored stream so that the streams can be ranked and evaluated.

The fourth and final section of the report contains discussion of the findings and implications for land management. A series of appendices are also provided; these contain data and details on analytical methods. The project budget appears in Appendix A.

## LIMITATIONS

It is important to remember that the trends analyzed in this report represent a short time period in the context of ecological change, which in forests may occur over decades or centuries. The currently monitored habitat conditions represent the cumulative effects of past management, current management, and natural processes including natural disturbances and climate. We expect to see short-term fluctuations in habitat indicators in addition to any potential long-term change; thus, in some cases the trends reported here may represent short-term variability rather than a long-term trend. This is considered when interpreting and discussing the data.

A second consideration when interpreting trends is that this monitoring began 15 years after the HCP riparian conservation measures were first implemented.[1] Therefore, it is likely that the most rapid phase of riparian habitat change resulting from the HCP had already occurred by the time monitoring

[1] Riparian management zones (RMZs) existed prior to the HCP but were much less restrictive than those of the HCP; additional information appears in the Study Area section below.

began. Given that caveat, Status and Trends Monitoring assesses whether habitat conditions are currently changing under DNR's OESF riparian conservation strategy.

## Study area

### GEOGRAPHY, GEOLOGY, AND CLIMATE

The study area is the Olympic Experimental State Forest (OESF), a planning unit established by DNR's HCP (Figure 1). Approximately 21% of the land within the OESF boundary, or 272,000 acres (110,000 ha), is state land managed by DNR (DNR 2016a).[2] The OESF includes Washington's water resource inventory area (WRIA) 20 and parts of WRIAs 19 and 21.

The OESF is bounded by the Pacific Ocean to the west, the Strait of Juan de Fuca to the north, the Olympic Range crest to the east, and the Quinault River watershed to the south. Elevation ranges from near sea level to 3,790 feet (1,155 m). Overall, the OESF is characterized by steep, erodible, mountainous terrain that transitions to wide river valleys toward the Pacific Ocean.

The geology of the OESF is composed mainly of marine sedimentary bedrock (sandstone and shale), overlain in many areas by glacial drift. The sedimentary rock was uplifted during subduction of the oceanic plate beneath the continental plate. The glacial drift was deposited following at least four Pleistocene glaciations that extended from the Olympic Mountains west through the major river valleys (Crandell 1964).

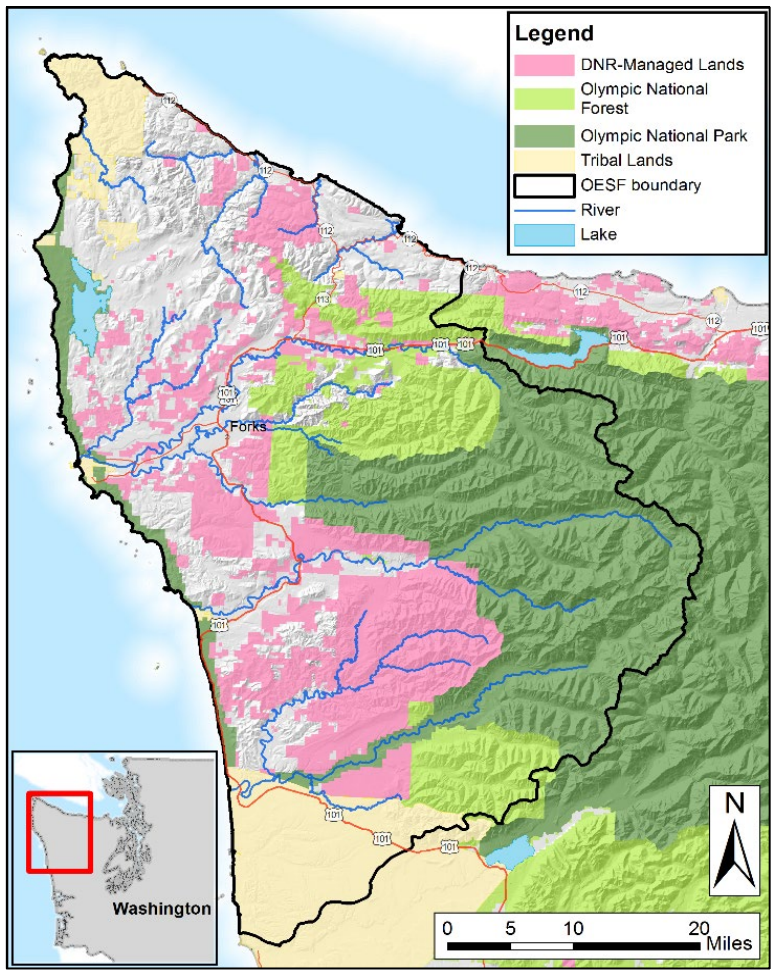


**Figure 1.** Map of the Olympic Experimental State Forest, showing public and tribal land ownership.

[2] The remaining 79% of the land in the OESF planning unit not managed by DNR consists of: private lands (30%), Olympic National Park (27%), Olympic National Forest (12%), and tribal lands (10%).

Climate in the OESF is heavily influenced by the Pacific Ocean. The climate is generally mild, with warm, dry summers and generally temperate, wet winters. Annual precipitation in the OESF ranges from approximately 80 to 180 inches (200 to 460 cm) per year. At lower elevations, where much of the DNR-managed land is located, most of the precipitation occurs as rain. Season-long snow accumulation generally occurs only above approximately 1,500 ft (460 m) elevation.

Climax vegetation zones (Franklin and Dyrness 1973) in the OESF are Sitka spruce (*Picea sitchensis* (Bong.) Carrière) from 0 to 500 ft (0 to 150 m) elevation, western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) from 500 to 1800 ft (150 to 550 m), and Pacific silver fir (*Abies amabilis* (Douglas ex Loudon) Douglas ex Forbes) above 1800 ft (550 m). On DNR-managed lands in the OESF, the most prevalent conifer species are western hemlock and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. menziesii); Sitka spruce and Pacific silver fir are also common. Western redcedar (*Thuja plicata* Donn ex D. Don) occurs at a somewhat lower frequency. Red alder (*Alnus rubra* Bong.) is the most common hardwood and is especially prevalent in riparian and disturbed areas.

Wind is the most frequent natural disturbance in forests of the OESF, with windthrow especially likely on exposed sites and non-glacial soils. Other significant sources of natural disturbance are landslides and debris flows. Large wildfires are less common than in most other parts of the state as a result of the moist climate (Agee 1993).

### RIPARIAN AND AQUATIC HABITATS

The OESF has a dense network of nearly 2,800 miles (4,500 km) of streams, many of which are small or headwater streams. The Status and Trends Monitoring program monitors Type 3 streams in the OESF; this stream type is the smallest class of fish-bearing streams (stream type definitions are in Appendix B). In the OESF, the valley-form topography produces streams with a wide range in gradient which influences many habitat features in Type 3 streams. In higher-gradient streams, cascade reaches with periodic plunge pools, or step-pool reaches with rapids, are common features. In lower-gradient stream reaches, alternating pools and riffles are common.

For many aquatic species, habitat quality is a reflection of stream channel geomorphic complexity, which is increased by the presence of in-stream wood and boulders. Many watersheds have evidence of past landslides and subsequent debris flows that altered in-stream and riparian habitats. Although these disturbance events occur naturally at a low frequency, they increased significantly as a result of intensive forest harvesting and road construction practices during the mid-20$^{th}$ century (Cederholm et al. 1981).

Riparian forests near Type 3 streams vary in age and structure. They are typically dominated by conifers, but red alder is often present in stream valley bottoms and along streambanks, as it is well-adapted to establishing after fluvial disturbances.

The OESF is home to nine native resident or anadromous salmonid species: Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), sockeye salmon (*O. nerka*), pink salmon (*O. gorbuscha*), steelhead/rainbow trout (*O. mykiss*), coastal cutthroat trout (*O. clarkii clarkii*), bull trout (*Salvelinus confluentus*), and mountain whitefish (*Prosopium williamsoni*). Whereas only two salmonids species in the OESF have been listed under the Endangered Species Act (Lake Ozette sockeye salmon and bull trout), most salmonid populations are largely reduced from historical levels (Weitkamp et al. 1995; NPCLE 2013; McMillan et al. 2021). Type 3 streams of the

OESF typically have some combination of juvenile coho salmon, steelhead/rainbow trout, coastal cutthroat trout, lampreys (*Lampetra spp.*), and/or sculpins (*Cottus spp.*), with coastal cutthroat trout being the most commonly found salmonid species (Martens 2016).

## LAND MANAGEMENT IN THE OESF

State trust lands in the OESF are managed under an integrated management approach intended to combine management for revenue production (primarily timber harvest) and ecological values (mainly habitat conservation) across the same landscape. The mosaic of successional forest stages is expected to shift over time, thus resembling the range of conditions resulting from natural disturbances. DNR implemented this experimental approach in the OESF in contrast to the more traditional management approach in which a landscape is separated into areas designated in the long-term for timber management or for habitat conservation. Despite the integrated management approach, there are significant portions of state lands in the OESF that have been effectively removed from timber management for habitat objectives (notably, the recently adopted marbled murrelet long-term conservation strategy (DNR 2019)).

Historically, the rate of timber harvest in the OESF increased during the early to mid-1900s but did not peak until the 1960s–1980s. A major shift in harvest occurred after the 1989 Commission on Old Growth Alternatives for Washington's Forest Trust Lands. Harvest of old growth forest on state lands ceased and the focus changed to managing for a long-term sustainable timber supply. The Commission also resulted in the establishment of the OESF. The current sustainable harvest level for the OESF was set at 739 million board feet for the fiscal year 2015-2024 planning decade by the Board of Natural Resources (Resolution no. 1560). Across all DNR-managed land in the OESF, the annual rate of harvest (regeneration harvest plus thinning) in recent years has averaged approximately 1% of the land base. However, for any given Type 3 watershed[3] in the OESF, many years may elapse between harvests due to the inherent spatial aggregation of timber harvest.

## RIPARIAN MANAGEMENT IN THE OESF

Extensive clearcutting that peaked in the 1960s-1980s significantly affected many streams and riparian zones in the OESF (Cederholm and Salo 1979; Cederholm et al. 1981). Timber harvesting and road construction practices of the era often produced large amounts of erosion and sediment delivery to streams, led to landslides and debris flows, and decreased stream shading (Reid and Dunne 1984; Cederholm and Reid 1987). Further, the installation of undersized culverts resulted in barriers to fish passage.

Riparian protections were initially implemented in Washington under the 1976 Forest Practices Rules as "streamside management zones" (Washington Forest Practice Board 1976). In Type 3 streams, the streamside management zones were 25 feet wide and were intended to minimize damage to the streambank and streambed during logging and to maintain at least 50% shade over streams that were designated as temperature sensitive (Washington Forest Practice Board 1976). Many equipment limitations were specified for the streamside management zone, but with the exception of the shade requirement, harvesting in streamside management zones was not restricted.

[3] The watershed surrounding a Type 3 stream.

The 1987 Timber, Fish, and Wildlife Agreement (TFW 1987) specified riparian management zones (RMZs) of 25 to 50 ft (8 to 15 m) in width on Washington's Type 3 streams that were greater than 5 feet in width (TFW 1987). In contrast to the earlier rules, the TFW agreement limited harvest in the riparian zone by specifying a minimum density and size of residual trees to be retained. In the OESF, it was observed in 1987 that old-growth buffer strips at least 33 feet (10 m) in width were often left along Type 3 streams (Cederholm and Reid 1987).

Beginning in 1990, stream buffers averaging 100 ft (30 m) wide were applied to Type 3 and Type 4 streams in the Olympic Region where necessary to protect unstable ground (this was estimated to be 55% of the OESF), replacing the Forest Practices RMZ widths from the TFW Agreement in those locations (DNR 1997, p. IV-110; DNR 1998, p. 4-253). The remaining 45% of the OESF that was not unstable ground was still managed under Forest Practices RMZ guidelines until 1997.

Implementation of the 1997 State Lands HCP changed riparian management policy in the OESF, through the OESF Riparian Conservation Strategy (DNR 1997). This strategy is specific to the OESF and differs from that of other state lands covered by the HCP. The OESF Riparian Conservation Strategy seeks to "protect, maintain, and restore habitat capable of supporting viable populations of salmonid and other species dependent on in-stream and riparian environments" (DNR 2016b). The strategy includes five types of activities: establishment of interior-core stream buffers, establishment of exterior wind buffers where needed, comprehensive road maintenance planning, protection of forested wetlands, and an integrated riparian research and monitoring program (DNR 1997).

To implement the OESF Riparian Conservation Strategy on a site-specific basis, the HCP established a 12-step watershed assessment procedure (DNR 1997). That procedure ensured that all timber management activities were meeting the objectives of the Riparian Conservation Strategy.

The 1997 HCP required that interior-core stream buffers be applied to Type 1-4 streams to minimize disturbance of channel banks and adjacent hillslopes (Figure 2). Regeneration harvest was not permitted within these buffers, but thinning was allowed for purposes such as enhancing riparian habitat, diversifying forest structure, and promoting wind-firmness of residual trees (DNR 1997). In practice, thinning of stream buffers typically only occurs in the OESF when the adjacent stand is thinned. Buffers are rarely thinned adjacent to a regeneration harvest.

The 12-step watershed assessment procedure determined the width of interior-core stream buffers based on local conditions including "channel size, valley confinement, and landform characteristics"; buffers extended from the 100-year floodplain on a site-specific basis to incorporate unstable slopes and landforms, wetlands, and other sensitive features (DNR 1997). The HCP expected that the interior-core buffer width determined by the 12-step assessment would average 100 feet (30 m) on Type 3 and Type 4 streams and 150 feet (45 m) on Type 1 and 2 streams (DNR 1997).[4] In addition to the interior-core buffers, the OESF Riparian Conservation Strategy specified that exterior buffers be applied in locations where there was risk of wind damage to the interior-core buffer (DNR 1997). These exterior buffers were 150 ft (45 m) wide on Type 3 and larger streams and 50 ft (15 m) wide

[4] On Type-5 streams, the need for stream buffers was determined through the 12-step watershed assessment.

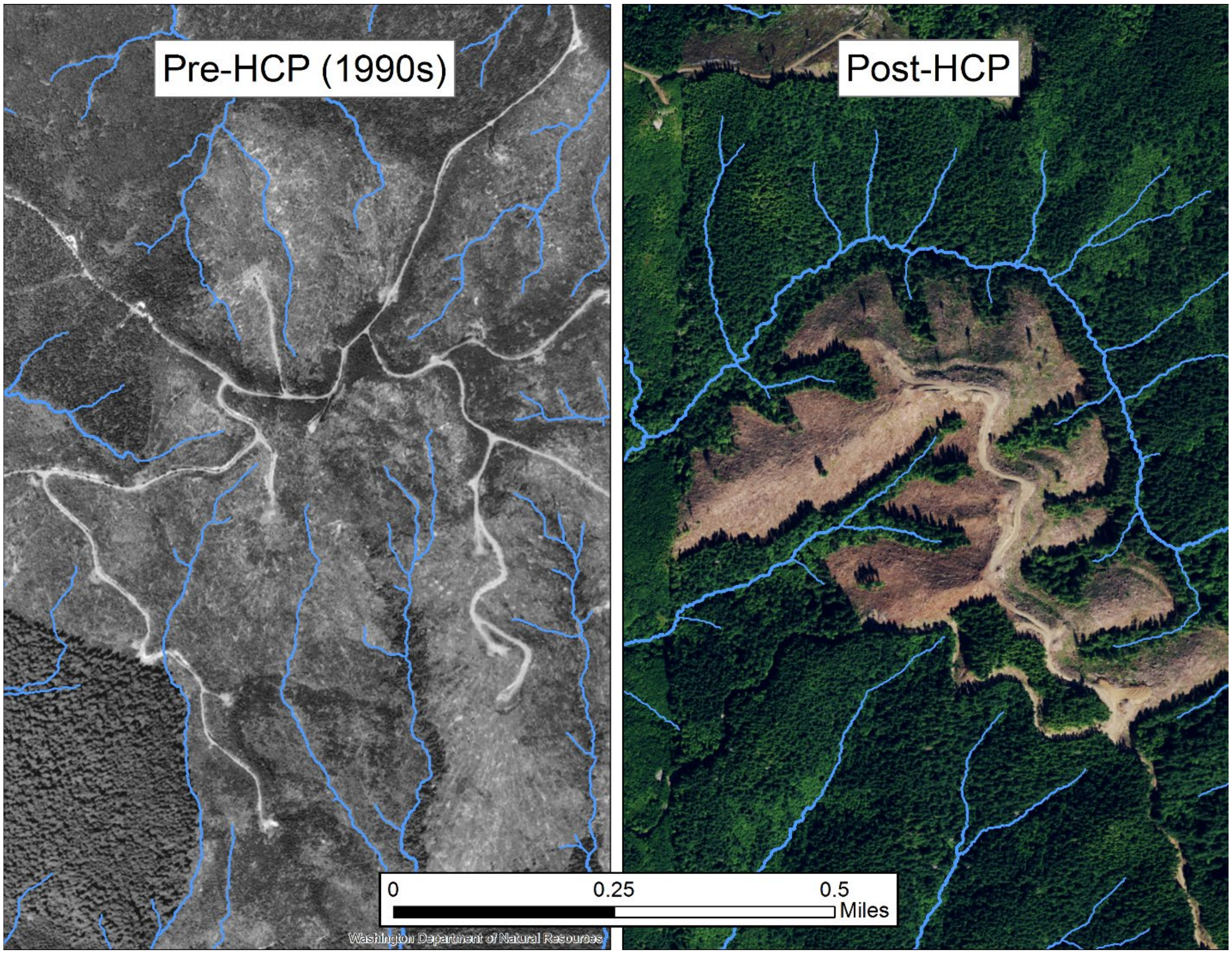


**Figure 2.** Regeneration harvests prior to and after implementation of the 1997 State Lands HCP, with an increase in stream buffer application apparent post-HCP. Streams are highlighted in blue.

on Type 4 and smaller streams. Thinning was permitted within exterior buffers, given various limitations, if it led to improved windfirmness of the residual trees (DNR 1997).

An implementation assessment in 2007 found that the average total buffer width (interior and exterior buffers were not differentiated) applied in the OESF under the HCP was 160 ft (49 m) for Type 3 streams, 89 ft (27 m) for Type 4 streams, and 50 feet (15 m) for Type 5 streams (Munzing 2008).

Procedures for implementing the OESF Riparian Conservation Strategy were updated under the 2016 OESF Forest Land Plan (DNR 2016a, p. 3-21). Among other revisions, the "expected" interior-core buffer widths (listed above) were set as default widths but were still increased on a site-specific basis to include unstable slopes and landforms, wetlands, and other features. Thus, 100 feet (30 m) became the minimum width of the interior-core buffer on Type 3 and 4 streams. The exterior buffer width was changed to 80 feet (24 m), to be applied in locations with severe endemic windthrow risk. An additional update to OESF riparian management was the implementation of "allotted acres", beginning in fiscal year 2018.[5] Allotted acres represent a small amount of regeneration harvest that is permitted within interior core buffers; the amount of allotted acres is determined by DNR through a watershed assessment and these acres are allotted to each Type 3 watershed for each 10-year period (DNR 2016a).[6] It should be noted that DNR is not required to harvest allotted acres.

## Monitoring sites

### WATERSHED SELECTION

The Status and Trends Monitoring program is designed to characterize the aquatic and riparian habitat conditions of Type 3 streams on DNR-managed lands across the OESF. This is accomplished by monitoring streams in selected watersheds in the OESF. These watersheds consist of:

- 50 Type 3 watersheds that are statistically representative of all 236 DNR-managed Type 3 watersheds in the OESF (Minkova et al. 2012, Minkova and Vorwerk 2014).[7] Throughout this report, these 50 sampled watersheds are referred to as **"DNR-managed watersheds"**. These 50 watersheds were selected through stratified random sampling to ensure that they represented the full range of physical and ecological conditions in the OESF. In the selection process, watershed median slope was used as the stratifying factor. Generally, watershed slope is correlated to stream gradient, and sensitivity to management is a function of gradient. Slope strata (0-9%, 10-19%, 20-29%, 30-39%, etc.) were defined, and the number of watersheds randomly selected from each slope stratum was proportional to the total number of OESF Type 3 watersheds in that slope stratum.

- 12 Type 3 **"reference watersheds"** in which at least 80% of watershed area has never been harvested,[8] and any harvest that has occurred was limited to the watershed periphery, not near any perennial streams. These watersheds are monitored to provide reference information on: (1) current habitat conditions in the absence or near-absence of past timber harvest, and (2) the amount of interannual variation in habitat conditions that occurs as a result of natural factors, including climate. Understanding natural variation in habitat conditions helps in interpreting habitat status and trends in the 50 DNR-managed watersheds.

  Unlike the 50 DNR-managed watersheds, which are statistically representative of a larger population, the 12 reference watersheds do not represent a larger population. This is because unharvested Type 3 watersheds in the OESF are rare,[9] and there were

[5] Fiscal year 2018 was when the 2016 OESF Forest Land Plan riparian procedures were implemented.
[6] Additional information on allotted acres is in the OESF Forest Land Plan, page 3-25 through 3-32 (DNR 2016a).
[7] Only Type 3 watersheds that contained at least 50% DNR-managed land are included in this count. An additional 210 Type 3 watersheds contain less than 50% DNR-managed land but these were not sampled because it would be more difficult to detect influences of DNR-management in such watersheds.
[8] See Table D-3 in Appendix D for the methodology used to estimate the area of past harvest.
[9] The portion of Olympic National Park that is within the boundary of the OESF contains many unharvested Type 3 watersheds, but these are restricted to the eastern portion of the OESF and most of these watersheds are characterized by higher-elevation, steeper valley slopes, and steeper stream gradients than are typical for watersheds on state lands in the OESF. Because of these ecological differences, direct comparison between these watersheds and DNR-managed watersheds is problematic.

few accessible alternatives to the 12 reference watersheds chosen. This sampling limitation affects our capacity to make direct statistical comparisons between the DNR-managed and reference watersheds. Of the 12 reference watersheds, 4 are in Olympic National Park, 6 are in Olympic National Forest, and 2 are on state land.

It should be noted that DNR's riparian and upland management practices in the 50 selected watersheds were not altered after the watersheds were selected for monitoring. Management in these watersheds remains representative of the larger population of DNR-managed watersheds in the OESF.

## PHYSICAL ATTRIBUTES OF WATERSHEDS

Among the 50 DNR-managed watersheds, 34 are in the rain-dominated zone, defined by DNR as below 1,500 ft elevation (460 m). Seven watersheds are partially in the rain-on-snow zone (defined as above 1,500 ft elevation (460 m)), and 9 watersheds are completely within the rain-on-snow zone. Among the 12 reference watersheds, 2 are in the rain-dominated zone and 10 are partially in the rain-on-snow zone. Additional data on each watershed can be found in Appendices C and D.

## WATERSHED-SCALE MANAGEMENT HISTORY

### *Pre-HCP*

The 50 monitored DNR-managed watersheds vary widely in harvest history. Figure 3 shows the harvest history of the 50 watersheds as the percentage of each watershed harvested pre-HCP, post-HCP, or never harvested. It is important to note that all post-HCP harvest in the OESF occurred on land that had been previously harvested prior to the HCP; thus, the percentage of pre-HCP harvest in Figure 3 is equivalent to the sum of all pre- and post-HCP harvest. The percentage of each watershed harvested pre-HCP[10] ranges from 32 to 100% (Figure 3).

Among the 12 reference watersheds, pre-HCP harvest averaged 4% of watershed area, and ranged from 0 to 20% (Table D-2 in Appendix D).

### *Post-HCP*

Since HCP implementation, harvests are classified as either variable-retention harvest (VRH; i.e., regeneration harvests) or variable-density thinnings (VDT) (DNR 2016a). Between implementation of the HCP and 2020, the proportion of the 50 watersheds harvested by VRH ranged from 0 to 40% and averaged 5% (Figure 3). During the same interval, the proportion of the watersheds harvested by VDT ranged from 0 to 53% and averaged 7%. As of 2020, the percentage of each of the 50 DNR-managed watersheds in young forest—defined here as ≤25 years of age—varies widely. Thirty-five of the 50 watersheds contain less than 20% young forest; 12 contain 20 to 40% young forest, and 3 contained more than 40% young forest.[11]

Among the 12 reference watersheds, no post-HCP harvest occurred.[12]

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[10] The 1997 State Lands HCP was effectively implemented beginning in 1999, due to an approximate 2-year lag due to the 2-year duration of most timber sale contracts.

[11] Forest age summaries were derived from DNR's RS-FRIS and include entire watersheds (upland and riparian).

[12] Ten of the twelve reference watersheds are on federal lands, so the 1997 State Lands HCP does not apply to these.

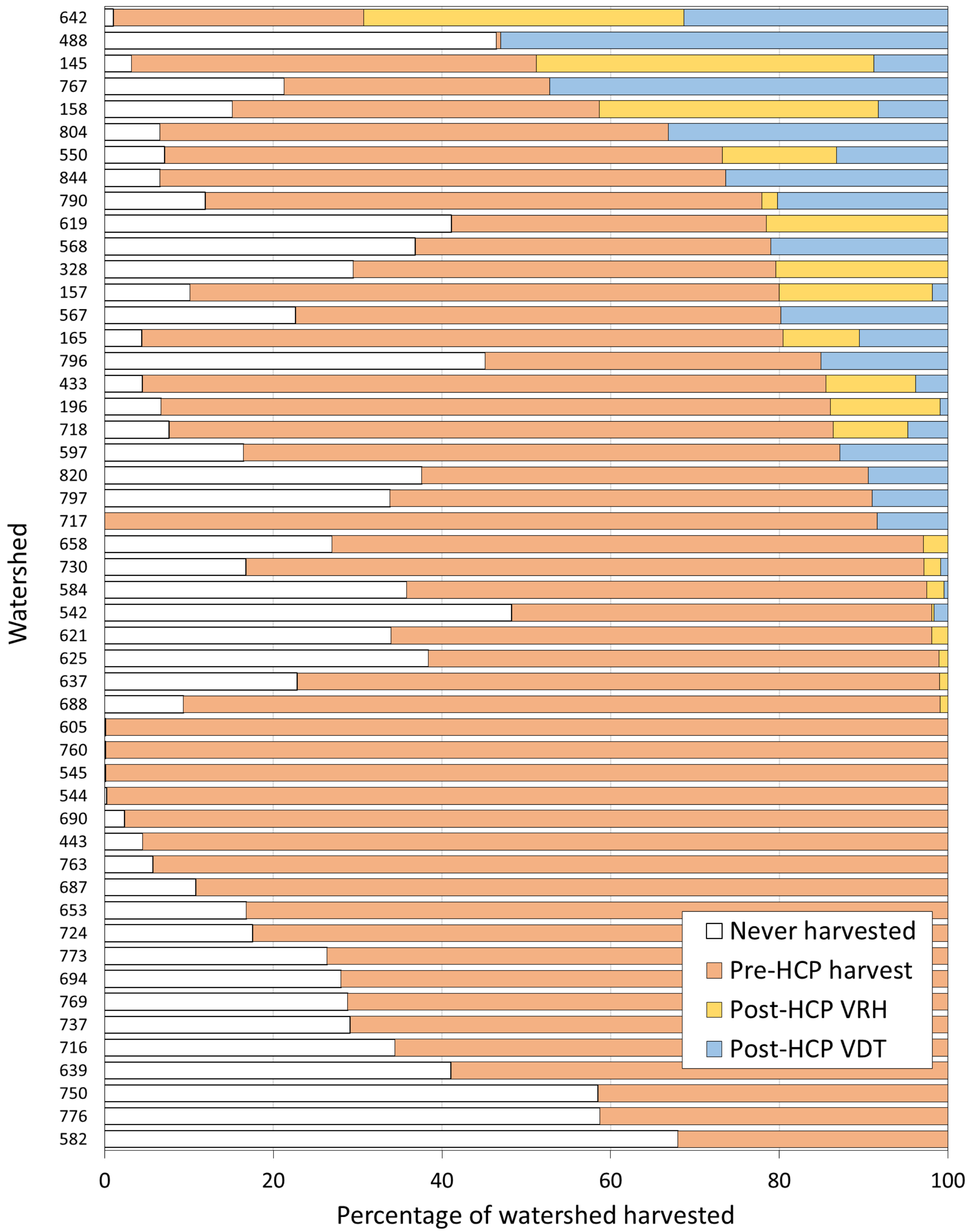


**Figure 3.** Percentage of each of 50 DNR-managed watersheds harvested prior to, and after, HCP implementation. Values are for the entire watershed, including upland and riparian areas. Note that all post-HCP harvest occurred on acreage that had already been harvested pre-HCP.

VRH = variable-retention harvest; VDT = variable-density thinning. Pre-HCP harvest was estimated using DNR's "Combined Origin Year" dataset and consists of all stands of origin year ≥1930; area never harvested is stands originating before 1930. Post-HCP harvest is timber sales from DNR's "Completed Harvests" dataset, 1999-2020.

### RIPARIAN ZONE MANAGEMENT HISTORY

#### *Pre-HCP*

Riparian zone management history in the 50 DNR-managed watersheds is more difficult to quantify than watershed-scale management history. The smaller scale of the riparian zone—for this discussion defined as the zone within 100 feet (30 m) of streams—requires more spatially precise data than a stand-scale estimate of harvest history. Modeled forest age estimates, used for the watershed-scale summaries, do not currently have sufficient spatial resolution to produce accurate riparian zone age summaries. Therefore, the watershed-scale pre-HCP harvest estimates presented above (Figure 3) are the best available approximation of pre-HCP riparian harvest in the 50 DNR-managed watersheds. For example, if 80% of a watershed was harvested pre-HCP, then it can be assumed that approximately 80% of that watershed's riparian forest was also harvested pre-HCP. This approximation is also based on an assumption that, in general, only a small number of riparian trees were left standing during pre-HCP harvests.

No pre-HCP riparian harvest occurred in the 12 reference watersheds; all harvest in those watersheds was located near the periphery of the watersheds, away from perennial streams.

#### *Post-HCP*

Since implementation of the 1997 State Lands HCP, no regeneration harvest (i.e., VRH) has occurred within the interior-core stream buffers of the 50 monitored DNR-managed watersheds. As of 2020, no allotted acres had been harvested in these watersheds.

In the 50 DNR-managed watersheds, an estimated 317 riparian acres, or 5% of the riparian area in those watersheds, has been thinned since implementation of the HCP.[13]

No post-HCP riparian harvest has occurred in the 12 reference watersheds.

## Monitoring methods

### SAMPLE REACHES

All in-stream and riparian monitoring is conducted at one Type 3 stream sample reach located near the outlet of each of the 62 monitored watersheds (Figure 4). The downstream end of the sample reach is located above the 100-year floodplain of the mainstem stream into which the Type 3 watershed drains. The sample reach therefore won't be impacted when the mainstem is at flood stage.

The length of the sample reach is 328 ft (100 m) or 20 times the bankfull width of the sample reach, whichever is longer (Minkova and Foster 2017). Bankfull width can be visualized as the width of the stream when streamflow is just high enough to fill the channel without overflowing onto adjacent floodplains.

In the 50 DNR-managed watersheds, sample reach bankfull width averaged 16.1 ft (4.9 m), and ranged from 6.2 ft (1.9 m) to 32.5 ft (9.9 m). Sample reach length averaged 374 ft (114 m). In the 12

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[13] For the purpose of this calculation, "riparian area" is defined as the area within 100 feet of Type 3 and Type 4 streams. Thinning data are from the Completed Harvest GIS layer.

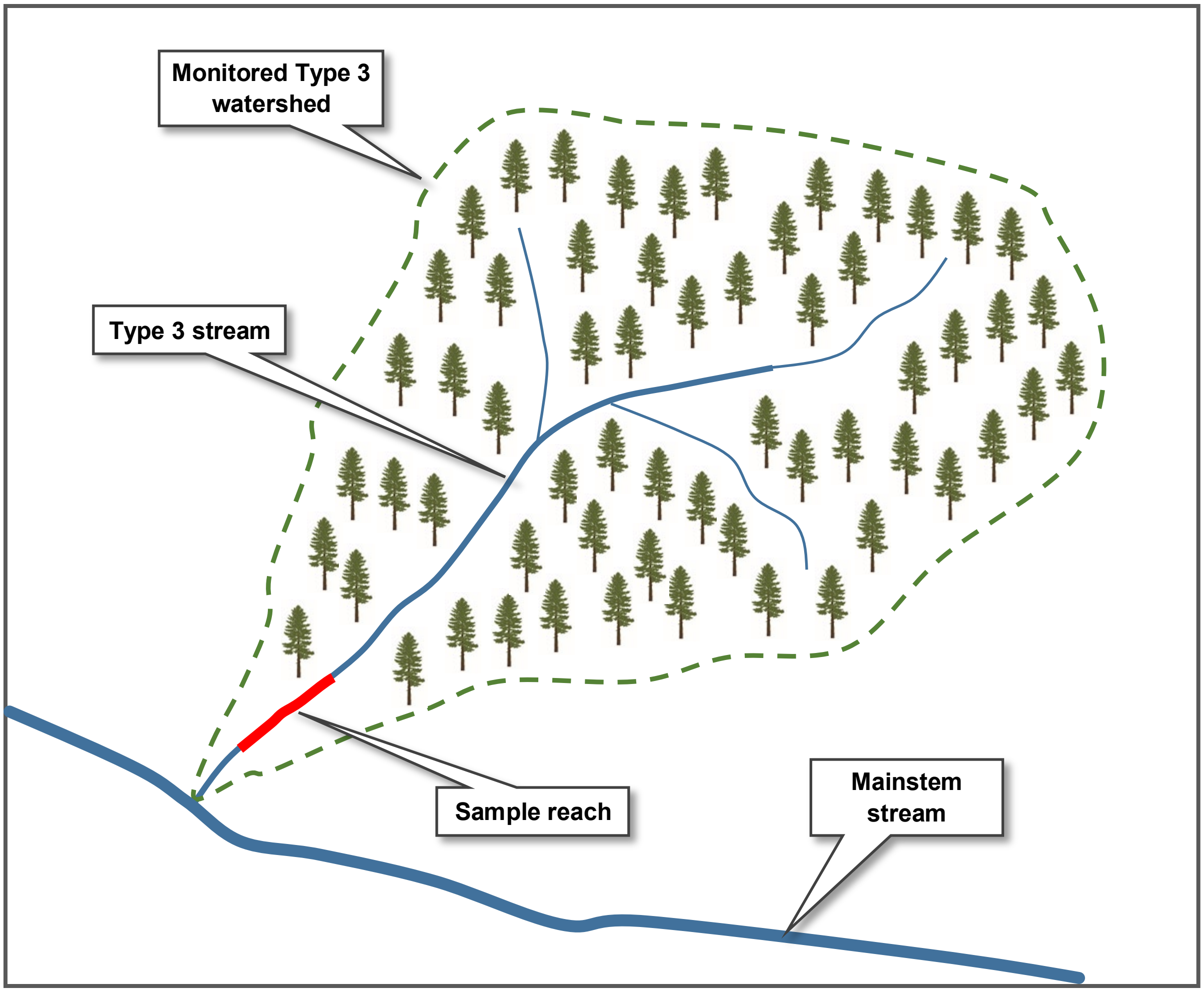



**Figure 4.** Schematic showing the location of a sample reach in a monitored watershed.

reference watersheds, sample reach bankfull width averaged 14.2 ft (4.3 m), and ranged from 5.3 to 14.2 ft (1.6 to 4.3 m). Sample reach length averaged 354 ft (108 m).

The monitored sample reaches varied widely in gradient, which is typically associated with different channel types, as classified by Bisson et al. (2006).[14] Channel type is key to understanding a stream's sensitivity to management impacts and is therefore a factor in our data analyses. The 62 reaches in this study represented three of the six channel types described by Bisson et al. (2006): pool-riffle, step-pool, and cascade (Figure 5). For DNR-managed watersheds, the pool-riffle channel type averaged 1.9% gradient; the step-pool type averaged 5.3% gradient, and the cascade type averaged 10.8% gradient (Table 2). For reference watersheds, the pool-riffle channel type averaged 2.4% gradient; the step-pool type averaged 4.8% gradient, and the cascade type averaged 16.3% gradient. The locations of all sample reaches, by channel type, are shown in Figure 6. Additional data on sample reaches in DNR-managed and reference watersheds are in Appendix E.

[14] These types were presented earlier in Montgomery and Buffington (1993).

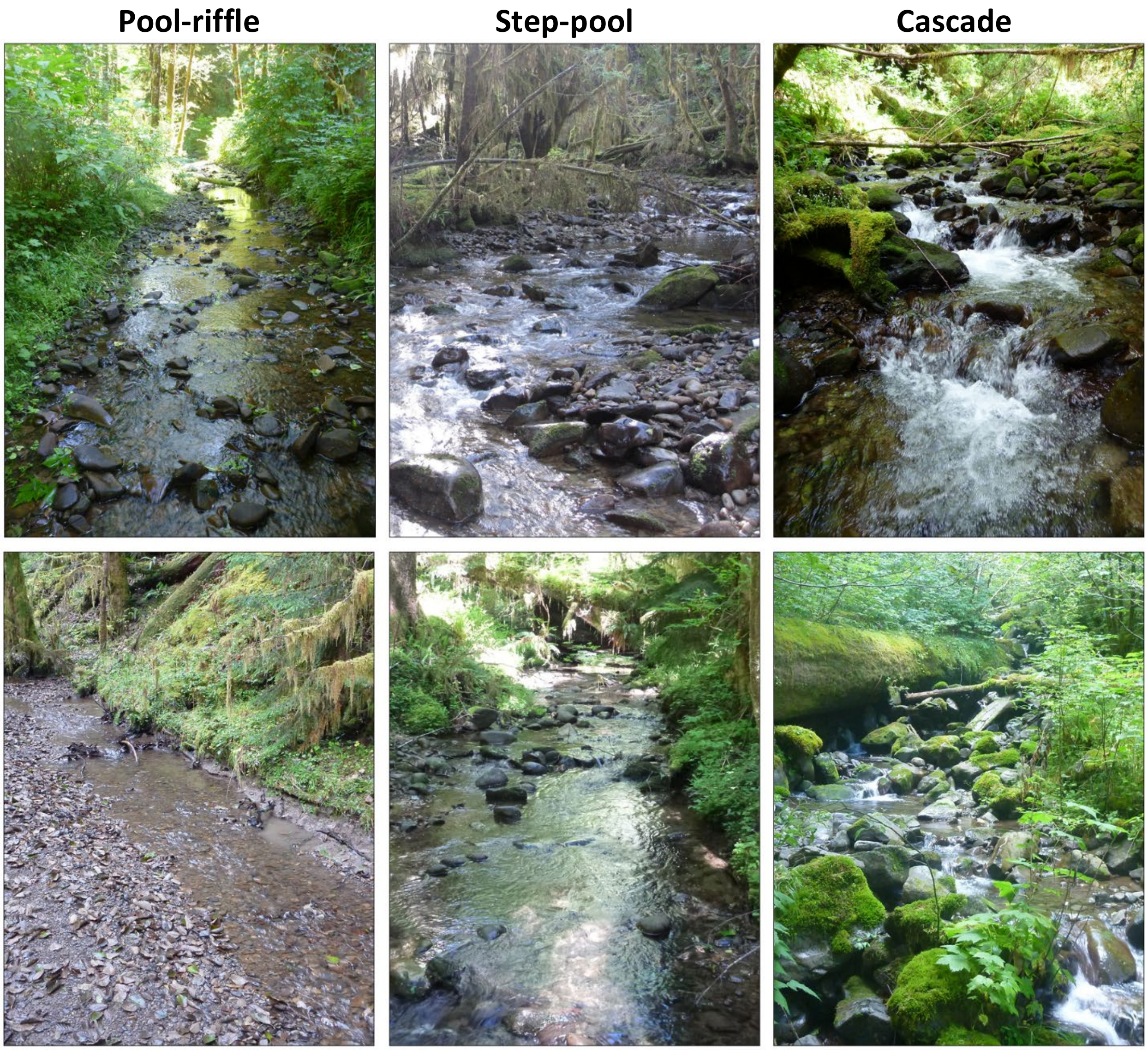



**Figure 5.** Examples of the three stream channel types observed during Status and Trends Monitoring: pool-riffle (left), step-pool (center), and cascade (right).

**Table 2.** Summary of sample reaches by channel type, with mean reach gradient and one standard deviation.

| | DNR-managed watersheds | | Reference watersheds | |
|---|---|---|---|---|
| **Channel type** | **No. sampled** | **Reach gradient (%)** | **No. sampled** | **Reach gradient (%)** |
| Pool-riffle | 15 | 1.9 ± 0.6 | 2 | 2.4 ± 0.9 |
| Step-pool | 24 | 5.3 ± 1.6 | 8 | 4.8 ± 2.6 |
| Cascade | 11 | 10.8 ± 4.3 | 2 | 16.3 ± 0.3 |

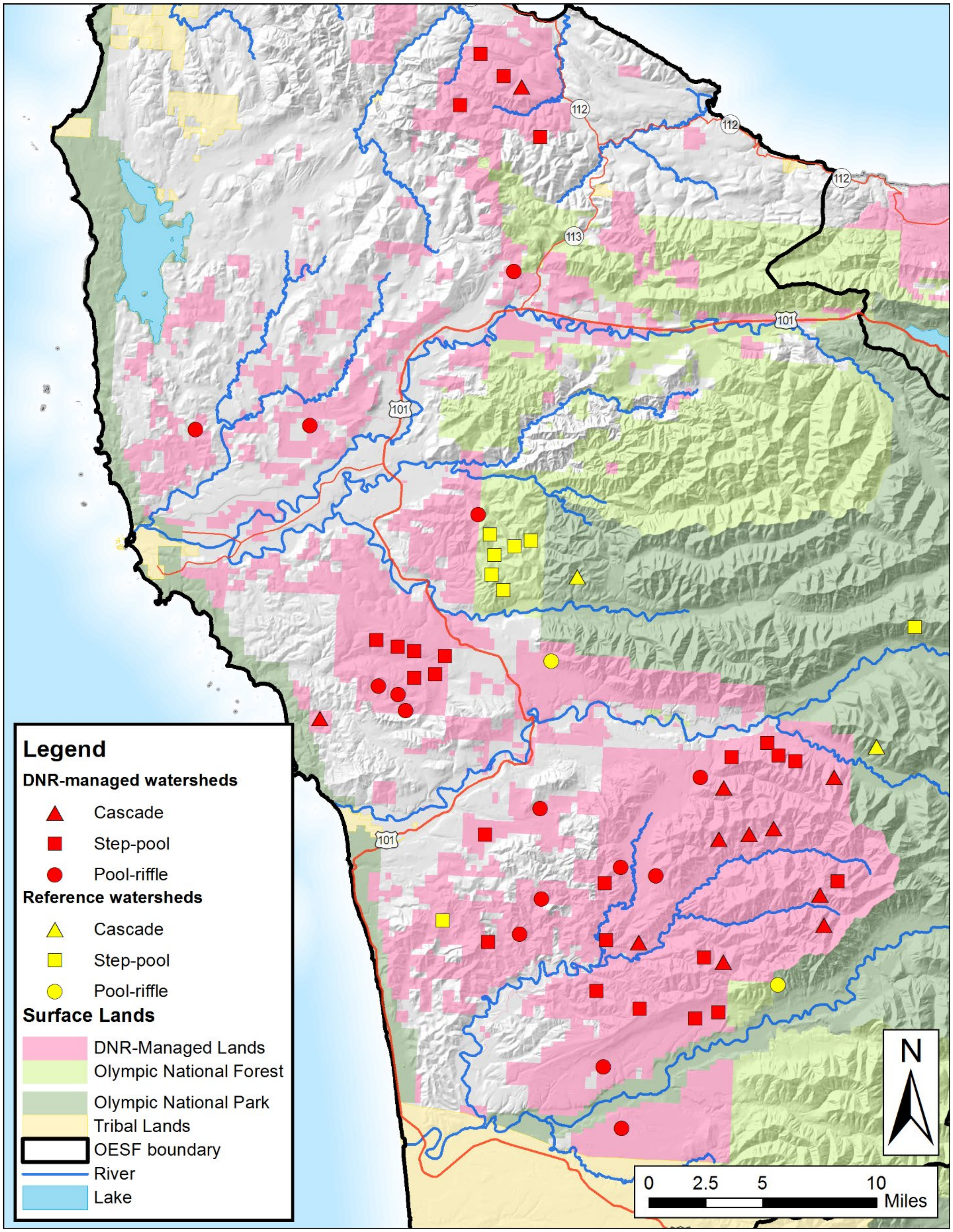



**Figure 6.** Map of the locations of monitored DNR-managed and reference watersheds, by stream channel type.

## HABITAT INDICATORS

Nine aquatic and riparian habitat indicators are measured at the stream sample reaches (Table 3), following the protocols in Minkova and Foster (2017). These indicators were chosen after extensive consultation with subject-matter experts and development of a series of process-based models for aquatic and riparian habitat of Type 3 streams (Minkova et al. 2012). The final set of indicators were ultimately selected based on factors that included their relevance to the study objectives, indicator sensitivity, the existence of established sampling protocols, and the feasibility of sampling.

Whereas seven of the nine habitat indicators are measured at all 62 monitored watersheds, two indicators—riparian microclimate and stream flow—are measured at subsets of these watersheds (at 10 and 14 watersheds, respectively). The reason for this is that these two indicators necessitate a significant monitoring cost per watershed, in terms of equipment and personnel time. The subsets of 10 and 14 watersheds are all DNR-managed and were selected to represent the full range of stream sizes and the geographical distribution of the 50 monitored watersheds. Thus, despite the smaller sample size, the data collected at these subsets of watersheds still provide estimates of the range of conditions in the OESF.

**Table 3.** Nine habitat indicators monitored in Status and Trends, number of reaches sampled, and sampling frequency.

| Indicator | Reaches sampled | Frequency of sampling |
|---|---|---|
| In-stream wood | 62 | 1-5 years[1] |
| Channel habitat units | 62 | |
| Channel substrate | 62 | |
| Channel morphology | 62 | |
| Stream shade | 62 | 5 years |
| Riparian vegetation | 62 | 10 years |
| Water temperature | 62 | Hourly, year-round |
| Riparian microclimate | 10 | Every 2 hours, for 3 years (2013-2015) |
| Steam flow | 14 | Every 15 minutes, year-round |

[1] See Table 4 for sampling schedule.

## MONITORING OVER TIME

The nine indicators are sampled at different time intervals according to the rate at which they are expected to change. The three indicators that change most rapidly—water temperature, riparian microclimate, and stream flow—are measured continuously. For these three indicators, measurements are made using automated sensors that record data and are then downloaded periodically by field personnel. Five other indicators—in-stream wood, channel habitat units, substrate, morphology, and stream shade—are measured during “stream surveys”. Riparian vegetation is measured at a 10-year interval, owing to its slower rate of change.

A steam survey is a 1- to 2-day process during which a seasonal field crew measures the five aforementioned indicators at a given sample reach. Based on a survey schedule (described below), the stream survey is repeated every 1 to 5 years at a reach. Stream surveys are conducted when streamflow is at or near its annual low (known as “base flow”), which is typically June through

September. This seasonal sampling approach improves consistency in the data but is also based on the fact that stream habitat conditions during base flow are particularly important to the juvenile salmonids present in the streams at that time (Woelfle-Erskine et al. 2017).

Stream surveys in each of the 50 DNR-managed watersheds were initially conducted between 2013 and 2015. Among the 12 reference watersheds, stream surveys in 4 were initiated in 2015; 2 more were initiated in 2017, and 6 were initiated in 2018, with funding from an agreement with the U.S. Forest Service under the Good Neighbor Authority.

Between 2013 and 2018, stream surveys were conducted with a goal of surveying each stream at least once every two years. This relatively high frequency of survey was used to develop an understanding of variation among sites and from year to year. Beginning in 2019, stream surveys were performed according to a schedule known as a "rotating panel with sentinel sites" design (hereafter referred to simply as "rotating panel") (Table 4). The schedule of the rotating panel is structured to optimize efficiency, given that there are too many watersheds to measure all in one year. The majority of watersheds (both DNR-managed and reference) are split into groups A and B which are each surveyed on a 5-year rotation. A third group, known as the sentinel sites (8 DNR-managed and 4 reference watersheds), is surveyed every year. The purpose of the sentinel group is to quantify the amount of year-to-year change in habitat indicators. This helps explain how much of the observed variation in habitat indicators is due to natural fluctuations, such as those related to weather conditions. Additionally, the annually surveyed sentinel group ensures that if an unusual weather event or major natural disturbance occurs, the 12 watersheds will have been surveyed in the years before and after the event, thereby documenting its effects.

Between 2013 and 2020—the study years analyzed for this report—a total of 239 stream surveys were conducted (Table 5; Appendix F).

**Table 4.** Sampling schedule for stream surveys (2019-2025). The actual number of surveys conducted in a given year (see Table 5) often differs slightly from this schedule due to logistical limitations.

| Group | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 | 2025 |
|---|---|---|---|---|---|---|---|
| A | 25 | - | - | - | - | 25 | - |
| B | - | 25 | - | - | - | - | 25 |
| Sentinel sites | 12 | 12 | 12 | 12 | 12 | 12 | 12 |

## Data analysis

### ASSESSING STATUS AND TRENDS

Status and Trends Monitoring was designed to assess the current condition (i.e., status) of the nine habitat indicators and to identify changes occurring over time (i.e., trends). The initial status of the indicators was described in the 2015 status report, produced after the first round of stream surveys (2013-2015) was completed (Minkova and Devine 2016). For the present report, which includes a trend analysis, all data collected from 2013 to 2020 were analyzed (i.e., all 239 stream surveys in Table 5). It should be noted that the rotating panel sampling design does not significantly affect the trend analysis reported here because that sampling design was not adopted until 2019. However, future trend analyses will utilize the rotating panel and sentinel site design.

**Table 5.** Number of stream surveys conducted, by year. A survey includes measurements of in-stream wood, channel habitat units (pool habitat), channel substrate, and channel morphology.

| Year | DNR-managed watersheds | Reference watersheds | Total |
|---|---|---|---|
| 2013 | 10 | 0 | 10 |
| 2014 | 32 | 0 | 32 |
| 2015 | 13 | 4 | 17 |
| 2016 | 43 | 3 | 46 |
| 2017 | 24 | 6 | 32 |
| 2018 | 19 | 11 | 30 |
| 2019 | 30 | 8 | 38 |
| 2020 | 26 | 10 | 36 |
| *Total* | 197 | 42 | 239 |

The trend analysis was conducted for the 50 DNR-managed watersheds, but not for the 12 reference watersheds. The reason for this is that the sample size of reference watersheds was only 4 until 2017 and only 6 until 2018. Thus, there were not enough reference watersheds sampled for long enough to produce a statistically reliable trend analysis. Instead, we present the status of the habitat indicators in reference watersheds, based on all data collected in those watersheds between 2013 and 2020.

The trend analysis in this report is based on changes in the 50 DNR-managed watersheds between 2013 and 2020. For each of these 50 watersheds, at least five years elapsed between the initial survey and the most recent survey. As there were too many watersheds to install sample reaches in all of them in a single year, the initial sampling of the 50 DNR-managed watersheds took place over three years: 2013 to 2015 (Appendix F). Thus, for some of these watersheds the five-year interval was 2013-2018, for some it was 2014-2019, and for others it was 2015-2020. Most watersheds, however, were sampled a total of three or four times from 2013 to 2020; in the trend analysis all intermediate samples are used, not just the earliest and latest samples.

### *Reference watersheds*

This monitoring program was not designed to make direct statistical comparisons between habitat conditions in DNR-managed and reference watersheds. The primary reason for this is that, as described above, it was not possible to select the reference watersheds in a way that represents a larger population.[15] Instead of a direct statistical comparison between the two types of sampled watersheds, results from reference watersheds are presented separately in tables and figures so that they can be interpreted on their own. Despite not serving as a statistical comparison, the reference watersheds still provide a valuable reference for baseline ecological conditions—and change over time—in the absence of harvest.

[15] Reasons for this include: (1) there were very few potential reference watersheds in the vicinity of the DNR-managed watersheds, and (2) most of the reference potential watersheds were not ecologically comparable to the 50 DNR-managed watersheds.

## ANALYSIS OF VARIANCE

Analysis of variance (ANOVA) was used to analyze data from the 50 DNR-managed watersheds. ANOVA was not used for the 12 reference watersheds because that sample size was too small, especially since 6 of the 12 watersheds were not sampled until 2018, as noted above. Instead, indicator metric values from the reference watersheds are summarized and distributions are shown graphically.

Trend over time was analyzed by including a "year" effect in the analysis of variance models. However, to more easily visualize potential change in the distribution of indicator values over time, data from the first measurement (2013-2015) and the most recent measurement (2018-2020) are also presented in tables and graphs.

For each of the nine habitat indicators, between one and three key indicator metrics were analyzed for this report. These metrics describe the condition of the habitat indicator. For example, pool habitat is an indicator, and pool frequency, pool area, and residual pool depth are indicator metrics. These indicator metrics are the dependent variables in the statistical models.

With a few exceptions, data from the 50 DNR-managed watersheds were analyzed by mixed-model type-III ANOVA. These models were used to detect whether various factors (or "effects") were influencing a dependent variable (i.e., a habitat indicator metric). An example of this model, expressed in a simplified format, is:

$$\text{Indicator metric} = \text{Channel type} + \text{Year} + (\text{Channel type} \times \text{Year}) + \text{Bankfull width} + \text{Watershed} \\ + \text{Other covariates}$$

In this model, each of the variables (i.e., the effects) to the right of the "=" potentially explain some of the observed variation in the dependent variable. Specifically:

**Channel type (three levels):** each stream was classified according to channel geomorphology, using the channel classification system described by Bisson et al. (2006), which reflects the quantity and type of sediment supply relative to the fluvial transport capacity of the channel. The stream reaches monitored in this study fell into three of the six channel types described by Bisson et al. (2006): pool-riffle, step-pool, and cascade. Channel type is included in the model because many aspects of stream habitat are influenced by channel morphology. If channel type is determined to have a significant effect on an indicator metric, then that habitat indicator metric is going to have different average values for the different channel types.

**Year (continuous variable):** year was included in the model to test whether the indicator metric was changing over time from 2013 to 2020. If year was found to have a significant effect, then the habitat metric was increasing or decreasing over time.

**Channel type × Year interaction:** this interaction term tests whether streams of the three different channel types changed differently over time. If it is significant, it indicates that channel type affects how streams are changing over time.

**Bankfull width (covariate; continuous variable):** this covariate was not of primary research interest but often explained variation in the indicator metric, and thus led to better overall model fit. Bankfull width was included as a covariate in most models because most of the stream habitat indicator metrics were expected to be influenced by stream size. Bankfull width is strongly correlated to watershed area ($r$=0.85), so statistically the two variables can be interpreted as nearly equivalent.

**Watershed (random effect):** the role of a random variable is to explain variation in the dependent variable (i.e., the indicator metric) in order to improve overall model fit, even though the random variable itself is not a focus of the research. In this case, watershed was included as a random variable because we expect more variation between watersheds than within the repeated measurements that take place at the same watershed over time.

**Other covariates:** for some indicator metrics, additional covariates were used to explain other sources of variation. These covariates are described in the sections that report results for each indicator. See Table D-3 of Appendix D for descriptions of how the covariates were measured.

In this report, the results of ANOVA are F-values and $p$-values, which are listed for each effect (other than the random effects). The F-value is the ratio of variation explained by an effect to unexplained variation; a large F-value indicates than an effect had a strong influence on the indicator metric. The $p$-values help in interpreting exactly how strong an effect is – specifically, they tell us the probability that an effect really was having an influence on the indicator metric. A small $p$-value indicates an effect was strong and was unlikely to have occurred by chance. A $p$-value smaller than 0.05 indicates a significant effect: in other words, that effect was 95% likely to have been real and not to have occurred by chance. We used the 95% confidence level for interpreting whether each effect was significant. However, when an effect fell just short of this level of significance (i.e., when the confidence level was 90 to 94%), this is noted.

All analyses were performed using R software (R Core Team 2021; Appendix G).

### *Quadratic effects*

For all ANOVA models, the distribution of residuals was examined after running the model for the first time. In some cases, the pattern of residuals suggested that the year effect was non-linear. For example, the value of the habitat indicator metric decreased over the first few years and then began to increase. When this type of pattern was observed, the quadratic year effect was added to the model (i.e., "Year × Year"). If this produced a residual pattern with a reasonably even distribution across years, then the quadratic year effect was retained in the model.

### *Data transformations*

Some metrics, such as those recorded as percentages, can occur in patterns that do not meet the assumptions necessary for ANOVA, such as the normal distribution and homoscedasticity of residuals. Such occurrences were evident when the model residuals were examined. In these cases, one of two types of data transformations were applied prior to re-running the ANOVA: the angular transformation or a logarithmic transformation. Where a data transformation was used, this is noted in the ANOVA results table. The data presented in the summary tables (tables with the "a" suffix to their number) and in the figures are not transformed data.

# Interpreting Results in Tables and Figures

For most of the indicator metrics in this report, results are shown in two Tables and four graphs. The top table (“a”; see example to the right) presents the average metric values for all DNR-managed watersheds, for the three channel types, and at the first and last measurements. The reference watersheds are not broken into groups. The sample sizes (“n”) refer to the total number of surveys, including repeat surveys of the same streams. SD is standard deviation, a measure of variability in the data. The population confidence interval is the range within which we are 95% sure that the mean for *all* OESF Type 3 streams falls (i.e., not just the ones sampled). The lower Table (“b”) presents ANOVA statistical results that indicate whether various factors had significant effects on the indicator metric (see Data Analysis section above).

**Table 9a.** Pool frequency (number of pools per 328 ft (100 m)) in 50 DNR-managed Type-3 streams and in 12 unharvested watersheds.

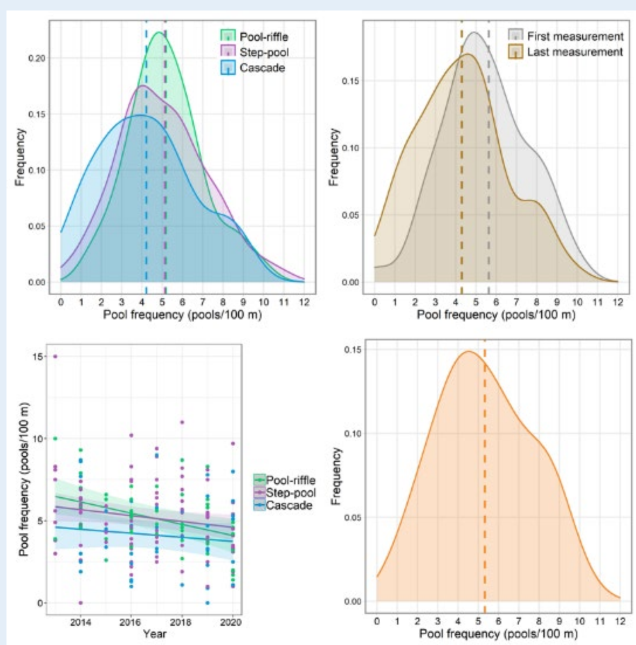
| Group | Sample summary | | | | Population 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 4.9 | 1.6 | 1.4 | 8.4 | 4.5 – 5.4 |
| | | | | | |
| Pool-riffle channel type (n=59) | 5.2[a] | 1.3 | 3.0 | 7.3 | 4.5 – 5.9 |
| Step-pool channel type (n=97) | 5.2[a] | 1.7 | 1.4 | 8.4 | 4.6 – 5.7 |
| Cascade channel type (n=41) | 4.1[a] | 1.8 | 2.4 | 7.4 | 3.3 – 5.0 |
| | | | | | |
| First measurement (n=50) | 5.6 | 2.5 | 0.0 | 15.0 | 4.9 – 6.3 |
| Last measurement (n=50) | 4.3 | 2.2 | 0.0 | 9.7 | 3.7 – 4.9 |
| *Unharvested watersheds* | | | | | |
| All unharvested watersheds (n=42) | 5.3 | 2.2 | 2.5 | 9.2 | 4.0 – 6.5 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 9b.** Analysis of pool frequency (number of pools per 328 ft (100 m)) in 50 Type-3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 1.7 | 0.18 | No effect |
| Year | 10.2 | <0.01 | Significant |
| Channel type x Year interaction | 0.9 | 0.42 | No effect |
| Bankfull width (covariate) | 19.8 | <0.01 | Significant (negative) |

A set of four graphs show the frequency distribution of the sampled watersheds across the values of each indicator. The **top left** graph compares the distributions of watersheds within each channel type for DNR-managed watersheds. The **top right** graph shows the distributions of all 50 watersheds at the first measurement and five years later at the last measurement. The **lower left** graph plots change over time for each channel type, as points and trendlines, illustrating potential channel type × year interaction (i.e., when an indicator follows a different trend over time for the different channel types). The **lower right** graph shows the distribution for the reference watersheds.



### *Comparing means*

Because the channel type variable had three levels, the test of significance in the ANOVA does not indicate where differences lie among those three levels. Thus, a mean separation test (the Scheffé test) was used to identify differences among means for the three channel types. A protected approach was used: if the ANOVA showed no difference among channel types at the 95% confidence level ($p \geq 0.05$), then a mean separation test was not used. If the $p$-value in the ANOVA was less than 0.05, then the Scheffé test was used to determine where differences occurred among channel types. The Scheffé test is an appropriate test to use when comparing groups of unequal sample sizes, as was the case with channel types. The Scheffé test comparisons were interpreted using a 95% confidence level.

## STATISTICAL POWER TO DETECT CHANGES IN INDICATORS

As part of this study, we performed an analysis of statistical power in the Status and Trends Monitoring program (Appendix H). Statistical power is the power of a study to detect treatment effects, or in this case, the power to detect change over time in habitat indicators. Statistical power is a key element of any research design: if a study has low statistical power, a real change may occur but the study is unlikely to detect it. We separately assessed statistical power for each of the indicator metrics included in this report, as power is unique to each. Our power analysis projects, for various hypothetical rates of change, how many years of monitoring are necessary before we will know with confidence whether or not long-term change is actually occurring. These results inform the Future Monitoring and Research section at the end of this report.

## MEASUREMENT ERROR

During any long-term monitoring project, consistency in interpreting and following the field protocol is of key importance. Failure to interpret and apply the protocol consistently over time will result in measurement error in the data collected. In Status and Trends Monitoring, a concerted effort was made to produce a detailed, comprehensive field protocol that covered all aspects of data collection, including the installation of sample sites, the field measurements, and downloading of electronically collected data (Minkova and Foster 2017). This field protocol was then revised as field methods were refined during the first several years of monitoring. Due to inevitable turnover in seasonal field crews, comprehensive field training took place at the beginning of each field season as a quality assurance measure.

To assess consistency in measurement and protocol interpretation, we conducted a protocol quality control assessment (Devine and Minkova 2016). This assessment measured where variability in the data (i.e., error) came from. We were particularly interested in two sources of error: (1) how much error occurred when the sampling protocols were repeated by the same field crew during the same season at the same streams, and (2) how much error occurred when two different field crews followed the same protocol in the same season at the same streams. By understanding these sources of measurement error, we learned how much variability to expect in the data when applying our field protocols. The results of the quality control assessment (Devine and Minkova 2016) informed our discussion of Future Monitoring and Research in this report.

# Status and Trends of Habitat Indicators

This section of the report focuses on the results of the status and trend analyses of the nine habitat indicators. Each indicator's subsection begins with a background on the relevance of the indicator, followed by a brief description of how the indicator was sampled, monitoring results, and implications for management.

## In-stream wood

### ECOLOGICAL CONTEXT

In-stream wood provides vital functions in shaping the stream channel and in providing stream habitat features necessary for fish and other aquatic organisms. Wood affects the formation of the stream channel in a variety of ways. Logs, root wads, and log jams trap and retain sediment, slow the velocity of the water, divert flow, and alter the channel's shape (Fetherston et al. 1995; Hassan et al. 2005). As a result, the channel becomes more complex, with a greater variety of habitat features (Naiman et al. 2002; Montgomery et al. 2003). Flow alterations created by in-stream wood can form pools, such as when the streambed is scoured by a pourover, or dammed by a woody debris jam. These slow-water areas provide important rearing habitat for juvenile salmonids (Swanson et al. 1976). Deep pools created by log jams serve as cold-water refugia for fish during summer. In-stream wood also provides cover for fish, both directly—shielding them from predation—and indirectly, as wood helps to form deep pools that are another form of cover. Finally, wood provides a nutrient source for organisms in streams, releasing nutrients as it decomposes, and storing nutrients within the fine sediments that it traps (Bisson et al. 1987, Cummins 1974).

The role of wood in shaping stream habitat has not always been understood; as recently as the 1970s, woody debris was intentionally removed, or "cleaned", from streams after logging (Ralph et al. 1994). Harvesting of streamside trees continued until the early 1980s (Bilby and Ward 1991). Research in recent decades indicates that wood is currently deficient in many streams of the Pacific Northwest; this has been blamed in part for declines in salmonid populations (Bisson et al. 1987). In this monitoring project, in-stream wood is surveyed to determine whether or not there is evidence of a deficit of wood owing to riparian management practices and to assess whether in-stream wood is increasing or decreasing over time. In the environmental impact analysis for the OESF Forest Land Plan, in-stream wood recruitment was recognized as an important indicator for fish habitat and riparian areas (DNR 2016b).

### SAMPLING

Within the sample reach, every piece of wood at least 4 in. (10 cm) diameter and at least 6.6 ft (2 m) long was tallied, measured for diameter and length, and assessed for attributes such as stage of decay, species class (i.e., conifer or hardwood), and contribution to pool formation and storage of sediment.[16] Only dead wood was sampled, and only pieces that were either within the stream channel

[16] In-stream wood survey protocols follow a slightly modified version of the Level II procedure described in the Timber, Fish and Wildlife Monitoring Program Method Manual (Schuett-Hames et al. 1999).

or suspended above it (for example, logs that span the stream but are not in the water). Across all monitored streams, both DNR-managed and reference, a total of 237 surveys of in-stream wood were conducted. For the 50 DNR-managed streams, each stream was surveyed an average of 3.9 times from 2013 to 2020.

Three in-stream wood metrics are analyzed in this report: the frequency of pieces, average piece diameter, and decay status. Additionally, functional attributes of in-stream wood are summarized.

## RESULTS

### *Frequency of in-stream wood*

The frequency of in-stream wood pieces was compared among streams based on the count of pieces per 328 ft (100 m) of stream.[17] In the 50-DNR managed watersheds, the overall average frequency of in-stream wood was 49.1 pieces per 328 ft (100 m) (Table 6a). This frequency of wood is somewhat lower than what was reported by other studies on the western Olympic Peninsula conducted in the 1980s and 1990s (Grette 1985, McHenry et al. 1998, Martens et al. 2019). However, direct comparisons are challenging because studies often sample streams of different sizes and use different piece size criteria when surveying in-stream wood. In Martens et al. (2019), this issue was addressed by comparing carefully matched datasets to assess differences.

Among the 50 DNR-managed streams, frequency of in-stream wood differed among channel types (Tables 6a, 6b, Figure 7a). Streams of the step-pool channel type had the highest frequency of wood at 57.0 pieces per 328 ft (100 m), and streams of the pool-riffle channel type had the lowest frequency at 33.5 pieces. The higher frequency of wood in the step-pool channel type may be at least partially a result of the depositional behavior of debris flows, a common channel-forming process in the OESF. The channel gradient of step-pool channels is 3.0 to 7.5 percent, which brackets the channel slope to which debris flows commonly deposit materials (Benda and Cundy 1990; Bisson et al. 2006).

During the 2013-2020 monitoring period, wood frequency in DNR-managed streams changed over time for all three channel types, following a curved trend that showed an initial increase followed by a slightly larger decrease (Figure 7c). Assuming this unexpected trend is not a result of measurement error, it indicates that short-term increases and decreases in wood frequency are occurring over time periods as brief as three to four years. Comparing the frequency distributions at the first and last measurement (Figure 7b), the overall shapes of the distributions were similar, with a slight shift toward fewer pieces. However, given the fluctuations in wood frequency observed, it is too soon to conclude that this rate of decrease will continue as a long-term trend.

Frequency of in-stream wood in the 12 reference watersheds averaged 48.8 pieces per 328 ft (100 m), and the frequency distribution (Figure 7d) was generally similar to that of the DNR-managed watersheds.

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[17] Because some sample reaches were longer than 328 ft (100 m), those counts were adjusted to a 328-ft basis.

**Table 6a.** Instream wood frequency (pieces per 328 ft (100 m)) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 49.1 | 24.0 | 7.5 | 119.7 | 42.5 – 55.8 |
| | | | | | |
| Pool-riffle channel type (n=59) | 33.5 [b] | 14.9 | 10.8 | 64.5 | 23.1 – 43.8 |
| Step-pool channel type (n=97) | 57.0 [a] | 25.1 | 25.3 | 119.7 | 48.8 – 65.2 |
| Cascade channel type (n=41) | 50.8 [ab] | 22.1 | 7.5 | 84.3 | 38.6 – 62.9 |
| | | | | | |
| First measurement (n=50) | 48.6 | 25.5 | 8.0 | 119.2 | 41.5 – 55.7 |
| Last measurement (n=50) | 42.0 | 22.7 | 7.0 | 118.3 | 35.7 – 48.3 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 48.8 | 22.4 | 20.7 | 107.9 | 36.2 – 61.5 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 6b.** Mixed-model analysis of variance results for instream wood (pieces per 328 ft (100 m)) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 5.7 | <0.01 | Significant |
| Year | 33.8 | <0.01 | Significant |
| Year x year | 39.3 | <0.01 | Significant |
| Channel type x Year interaction | 1.1 | 0.35 | No effect |
| Bankfull width (covariate) | 10.9 | <0.01 | Significant (positive) |
| Watershed median slope (covariate) | 0.7 | 0.41 | No effect |

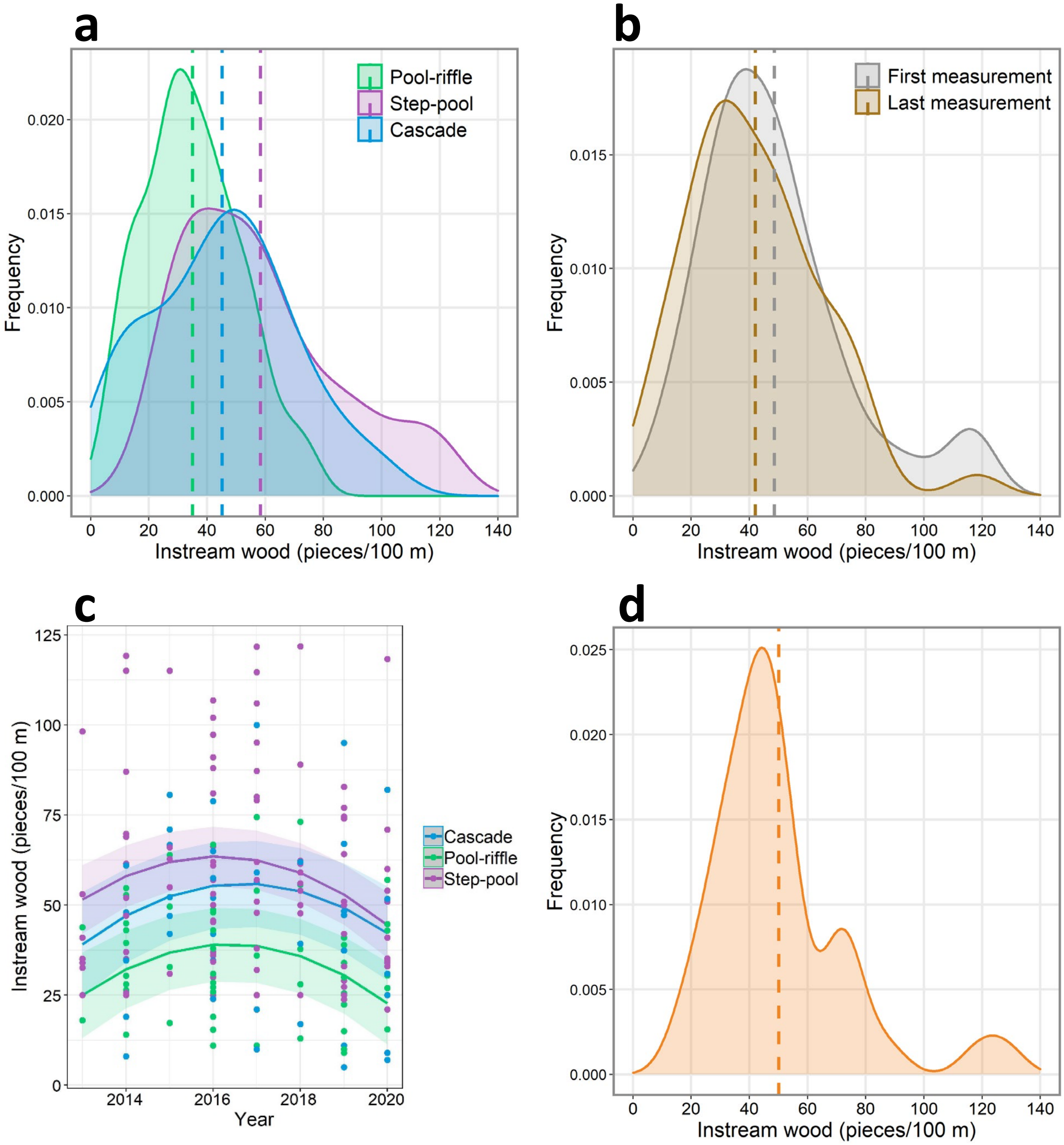



**Figure 7.** Distribution of DNR-managed watersheds according to the number of in-stream wood pieces per 328 ft (100 m), by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of in-stream wood for all reference watersheds (d). Vertical dashed lines indicate averages.

### *Diameter of in-stream wood*

Size of in-stream wood is important, as larger pieces—nearly always conifer in the OESF—have the greatest impact on channel morphology and provide the greatest habitat benefits, as well as persisting in streams for much longer than smaller pieces. Often the larger pieces are known as “key pieces”.

Average diameter[18] of in-stream wood pieces was calculated for each of the in-stream wood surveys. In the 50-DNR managed watersheds, the overall average diameter of in-stream logs was 15.3 in. (38.9 cm) (Table 7a). Among the three channel types, average diameter of in-stream wood ranged from 14.4 in. (36.6 cm) to 16.8 in. (42.6 cm), but these differences were not significant. Among the reference watersheds, the average diameter of in-stream wood was 15.2 in (38.5 cm).

In the DNR-managed watersheds, the average diameter of wood within a sample reach was strongly related to watershed median slope, with steeper watersheds having larger-diameter wood (Table 7b). Watershed slope is correlated with other variables, such as distance from the coast and time since the watershed was first harvested. Thus, harvest history is likely relevant to the size of in-stream wood, because harvest practices in riparian settings evolved over time. The diameter and frequency of legacy wood present in streams today is likely affected by the practices used last time the riparian zone was harvested. Additionally, economics at the time of harvest would have influenced the amount of wood left in the riparian zone. Linking harvest history of individual watersheds to in-stream wood will be further evaluated in future research.

After watershed slope, the second strongest effect in our analysis of average wood diameter was year. Between 2013 and 2020 there was a slight yet statistically significant decrease in average wood diameter of 0.6 in. (1.6 cm) (Figure 8). This decrease in average diameter is likely a result of the addition of new, small-diameter pieces of wood to streams in recent years, particularly red alder. It is less likely that the observed decrease in average diameter is caused by decay or loss of large-diameter pieces, as large conifer logs in streams decay relatively slowly. Long-term, the diameter of trees in the riparian forest will ultimately determine the range of log sizes in streams, as discussed in depth by Martens et al. (2020).

[18] Diameter was calculated as quadratic mean diameter, which is equivalent to averaging the cross-sectional area of logs or trees and therefore it is preferable to a simple diameter average when the volume of wood is important.

**Table 7a.** Diameter[1] of instream wood pieces (inches; centimeters in parentheses) in 50 DNR-managed Type 3 streams and in 12 reference streams. Minimum diameter of pieces was 4 in. (10 cm). SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 15.3 (38.9) | 2.5 (6.3) | 8.8 (22.4) | 20.9 (53.1) | 14.6 – 16.0 (37.2 – 40.7) |
| | | | | | |
| Pool-riffle channel type (n=59) | 15.6 [a] (39.5) | 1.7 (4.3) | 12.8 (32.5) | 16.7 (42.3) | 14.5 – 16.7 (36.8 – 42.3) |
| Step-pool channel type (n=97) | 14.4 [a] (36.6) | 2.6 (6.6) | 8.8 (22.4) | 20.9 (53.1) | 13.6 – 15.3 (34.5 – 38.8) |
| Cascade channel type (n=41) | 16.8 [a] (42.6) | 1.7 (4.3) | 14.3 (36.4) | 19.5 (49.6) | 15.4 – 18.2 (39.1 – 46.2) |
| | | | | | |
| First measurement (n=50) | 15.7 (39.8) | 3.3 (8.5) | 7.5 (19.1) | 26.4 (67.0) | 14.7 – 16.6 (37.4 – 42.1) |
| Last measurement (n=50) | 15.0 (38.2) | 2.9 (7.3) | 8.6 (21.9) | 21.4 (54.4) | 14.3 – 15.8 (36.2 – 40.2) |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 15.2 (38.5) | 1.9 (4.8) | 14.1 (31.3) | 16.2 (47.5) | 12.3 – 18.7 (35.8 – 41.2) |

Note: channel type means followed by the same letter do not differ significantly.

[1] Diameter calculated as quadratic mean diameter.

**Table 7b.** Mixed-model analysis of variance results for diameter[1] of instream wood pieces in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 2.0 | 0.14 | No effect |
| Year | 5.9 | 0.02 | Significant |
| Channel type x Year interaction | 0.0 | 0.99 | No effect |
| Bankfull width (covariate) | 3.5 | 0.07 | No effect |
| Watershed median slope (covariate) | 14.0 | <0.01 | Significant (positive) |

[1] Diameter calculated as quadratic mean diameter.

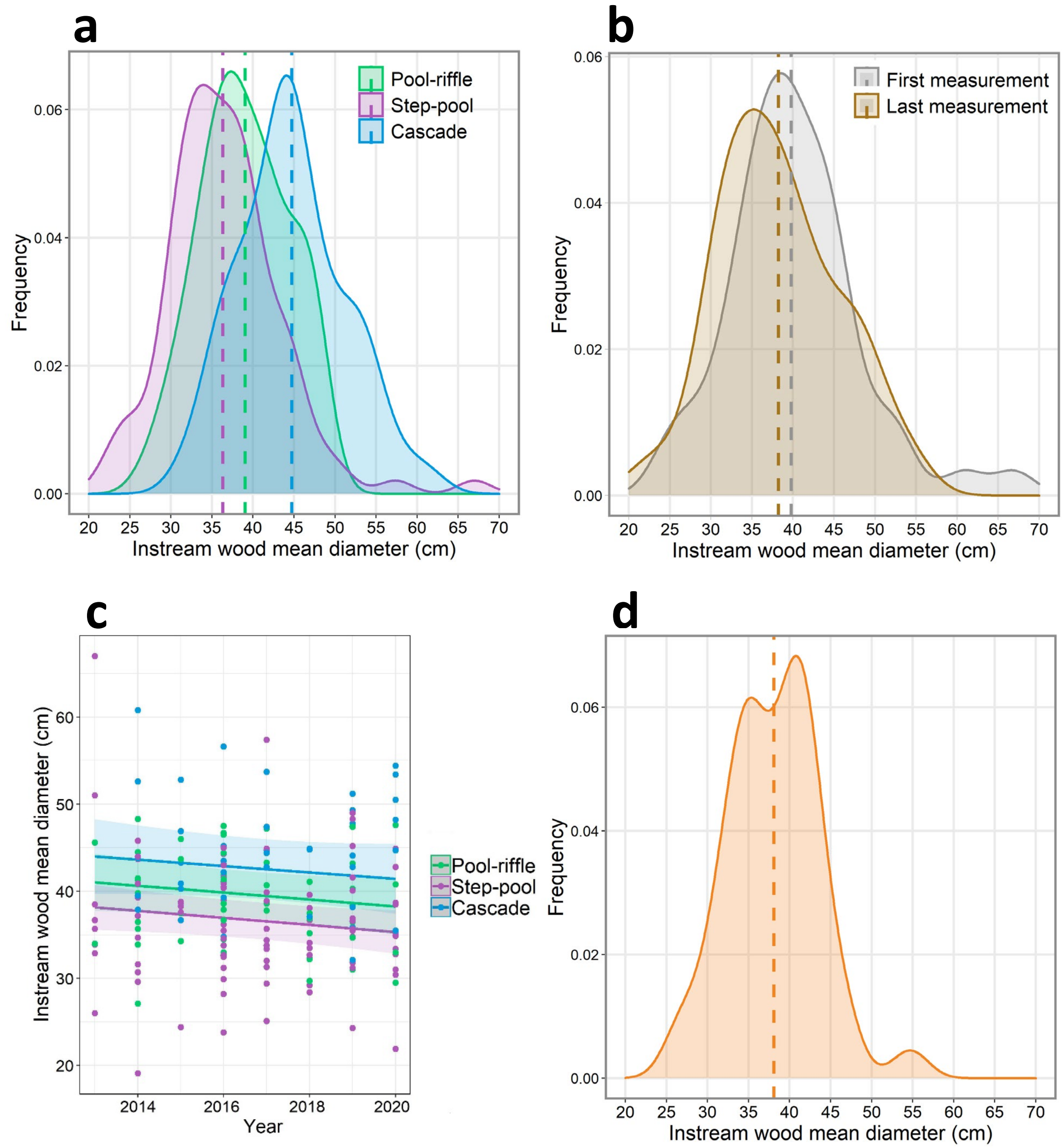



**Figure 8.** Distribution of DNR-managed watersheds according to mean diameter of in-stream wood, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of mean diameter of in-stream wood for all reference watersheds (d). Vertical dashed lines indicate averages.

### *Functional attributes of in-stream wood*

Two functional attributes important to stream channel formation were recorded for each piece of in-stream wood: pool formation and sediment storage. For each attribute, either "yes" or "no" was recorded for each piece of wood. Pool-forming pieces are defined as those pieces contributing directly to the formation of a pool. Pools are a key habitat feature examined in a separate section of this report. Sediment-storing pieces contribute directly to trapping sediments that would otherwise move downstream if the piece were not present.

In the DNR-managed watersheds, 10% of in-stream wood pieces formed pools (Table 8). The average diameter of pool-forming pieces was 4.3 in. (10.9 cm) larger than that of non-pool-forming pieces. Conifer pieces were more than twice as likely to form pools as hardwood pieces (13% vs. 6%), probably due to the fact that, overall, conifer pieces found in DNR-managed watersheds averaged more than twice the diameter of hardwood pieces, at 16.3 in. (41.4 cm) compared with 8.0 in. (20.2 cm).

Thirty-three percent of in-stream wood pieces in DNR-managed watersheds trapped and stored sediment (Table 8). The average diameter of sediment-storing pieces was larger than non-sediment-storing pieces by 2.9 in. (7.5 cm). As with the pool-forming function, conifer pieces were more than twice as likely to trap sediment as hardwood pieces.

**Table 8.** Percentage of pieces and average diameter (inches; centimeters in parentheses) of in-stream wood pieces performing pool-forming and sediment-storing functions in 50 DNR-managed watersheds.

| Group | Overall[1] | Conifer | Hardwood |
|---|---|---|---|
| | *Percentage of pieces* | | |
| Pool-forming pieces | 10% | 13% | 6% |
| | *Average piece diameter* | | |
| Pool-forming pieces | 17.1 (43.5) | 19.3 (48.9) | 11.9 (30.2) |
| Non-pool-forming pieces | 12.8 (32.6) | 15.9 (40.3) | 7.8 (19.7) |
| | | | |
| | *Percentage of pieces* | | |
| Sediment-storing pieces | 33% | 36% | 15% |
| | *Average piece diameter* | | |
| Sediment-storing pieces | 15.2 (38.7) | 17.7 (45.0) | 9.4 (23.8) |
| Non-sediment-storing pieces | 12.3 (31.2) | 15.4 (39.0) | 7.8 (19.9) |

[1] This category includes pieces for which species class could not be identified in the field due to their advanced state of decay.

### *Decay of in-stream wood*

To better understand the dynamics of in-stream wood, each piece of in-stream wood tallied during surveys is classified according to its stage of decay. Decay classes are:

> **Class 1**: Least decayed; wood is sound and most bark is still intact; branches may be present.
>
> **Class 2**: Heartwood sound; decay is shallow; some bark is missing.

**Class 3**: Heartwood sound; sapwood decaying, pieces can be pulled apart by hand. Bark absent.

**Class 4**: Decayed throughout, heartwood and sapwood can be pulled apart by hand.

Comparing the distribution of logs among decay classes for different log diameter classes can help to explain which diameter classes are made up of new wood inputs and which diameter classes contain older, more decayed logs. Because large-diameter logs tend to persist in streams for much longer than small logs and take longer to decay, we would expect to find more large logs in later stages of decay, compared to smaller logs. This pattern is apparent for the streams in DNR-managed watersheds (Figure 9a, c).

Comparing the percentage of in-stream wood pieces by decay class and diameter class for DNR-managed and reference watersheds (Figure 9c, d), the distribution is generally similar for diameter classes up to 32 in. (80 cm). However, for the largest diameter class (≥32 in. (80 cm); 80% of pieces in DNR-managed watersheds are in decay classes 3 and 4 whereas only 45% of pieces in reference watersheds are in decay classes 3 and 4. Although this is not a true statistical comparison, it does suggest that the largest pieces of in-stream wood in DNR-managed watersheds are in more advanced stages of decay than those in the reference watersheds. This implies that streams in the reference watersheds have received more recent inputs of large-diameter logs, as 55% of the pieces in that diameter class are only in decay classes 1 and 2.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

Although we observed short-term fluctuations of in-stream wood frequency in DNR-managed watersheds, the more advanced decay of wood in the largest diameter class—relative to that of unmanaged watersheds—suggests a reduced supply of large-diameter pieces in recent decades. The trend in decreasing piece diameter may also indicate the same phenomenon, but longer-term monitoring is needed to confirm that trend. A decline of large wood input to streams was reported in recent analyses (Martens et al. 2019, 2020), and in earlier work in western Washington (e.g., Grette 1985, Ralph et al. 1994, McHenry et al. 1998).

If these trends are indeed pointing to a longer-term decrease of in-stream wood, the simplest explanation for the decrease is that historical harvest of riparian forests—prior to the initiation of riparian forest protections in the 1980s and 1990s—interrupted an existing cycle of wood input to streams. In some cases, conifers were planted and successfully established up to the stream bank following harvest, but elsewhere post-harvest riparian forests were dominated by alder regeneration. In-stream wood in second-growth forests was previously found to have a substantially greater proportion of alder than in-stream wood of old-growth forests (Bilby and Ward 1991). This shift from older, conifer-dominated uneven-aged riparian forests to younger riparian forests with more alder appears to have significant implications for the amount and size of wood entering streams over time (Martens et al. 2020).

Other 20$^{th}$-century events affected in-stream wood as well: debris flows and sedimentation caused by intensive harvest and road construction often reconfigured stream channels. Some in-stream wood was certainly buried or recruited by these processes, although elsewhere logging debris from that period, including old growth logs, is still prominent in and near OESF Type 3 streams. Altogether, the current state of in-stream wood in the DNR-managed watersheds is a product of riparian forest

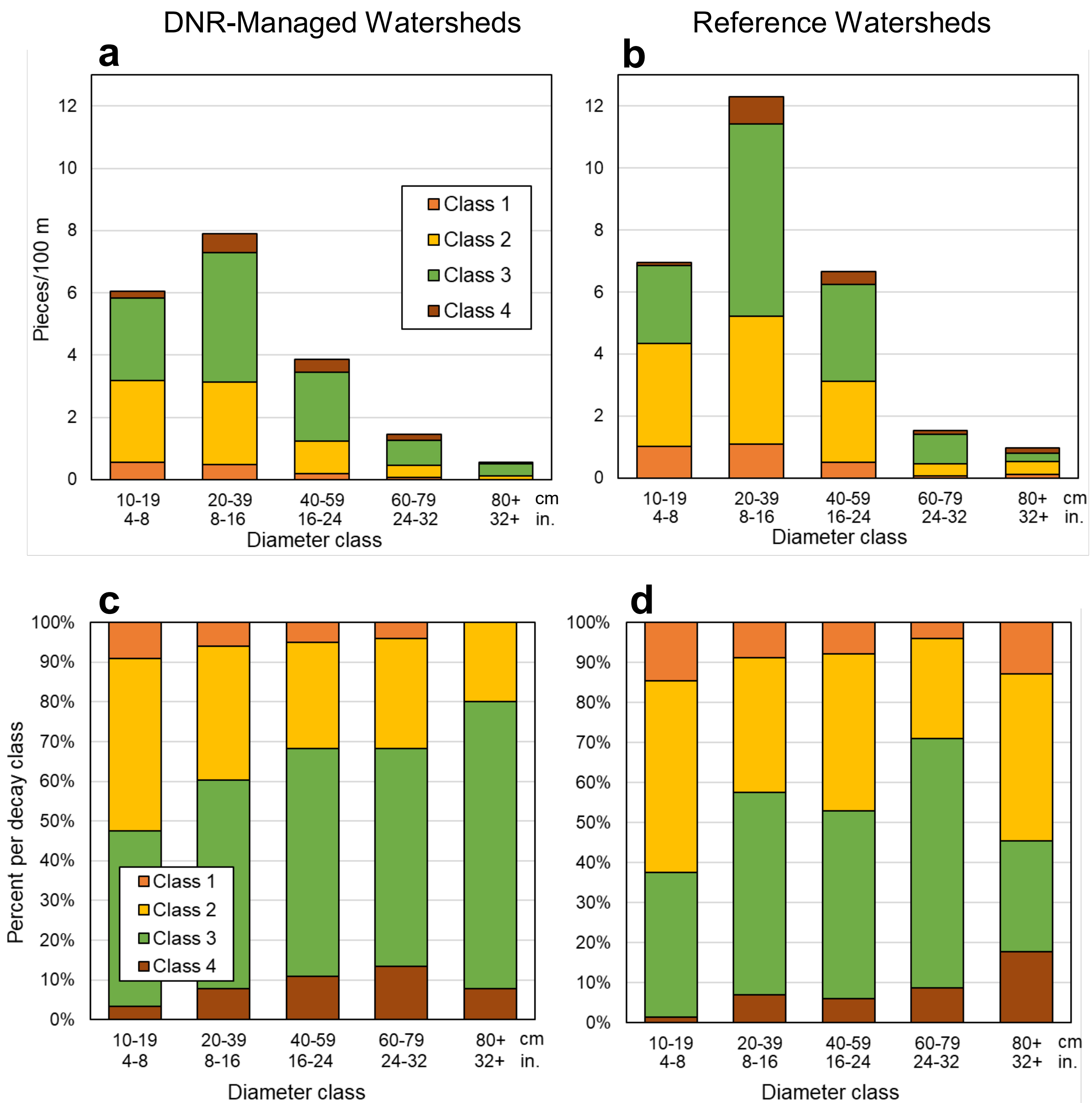



**Figure 9.** Distribution of in-stream wood by decay class for streams in 50 DNR-managed watersheds (left) and 12 reference watersheds (right). Graphs (a) and (b) show piece frequency and graphs (c) and (c) show percentage of pieces by decay class within each diameter class.

management history and local conditions, both of which are quite variable across the 50 sampled watersheds.

Given the residual impacts of 20th-century riparian forest harvest on current riparian forests and on in-stream wood in the OESF (Grette 1985; McHenry et al. 1998), DNR has taken the step of exploring riparian forest management techniques to accelerate the development of old-forest structure and ultimately to produce in-stream wood inputs of historical size and frequency. DNR's ongoing Riparian Forest Restoration Strategy (RFRS) Effectiveness Monitoring Study is an example of this research, as is the riparian portion of the T3 Watershed Experiment (Martens et al. 2021).[19]

[19] The T3 Watershed Experiment is a collaborative OESF study initiated by the University of Washington (Olympic Natural Resources Center) and DNR (Forest Resources Division and Olympic Region).

# Pool habitat

## ECOLOGICAL CONTEXT

A stream channel can be divided into segments of relatively homogenous conditions based on depth, stream velocity, gradient, and substrate. These segments are often called channel units or habitat units. Habitat units differ from channel type as channel type is identified at the scale of an entire reach, whereas each reach contains many habitat units. Each habitat unit is classified according to its characteristics. Under the classification system used in this monitoring program (Bisson et al. 2006), the most common habitat units in the monitored Type 3 streams were pools, rapids, cascades, and riffles (Figure 10). Of these, pools are the most important for salmonid habitat. Pools are relatively deep, slow-water habitat where fish can feed or find cover while expending less energy than in


**Figure 10.** Examples of common habitat units in the monitored sample reaches.

swifter waters. In Type 3 streams, pool depth is an important factor during summer, when stream flows are often at their lowest, and juvenile salmonids move to pools to seek refuge from shallow water and predators. Deep pools also serve as cold-water refugia when streams warm in summer.

Forest management within the riparian zone has the potential to affect the type, frequency, and size of channel habitat units, including pools; this can occur if the amount of wood entering streams is impacted or if sedimentation occurs (Ralph et al. 1994; WoodSmith and Buffington 1996).

## SAMPLING

During each stream habitat survey, the sample reach is broken down into individual habitat units. Each unit's length and width are measured and the unit is classified by type (Minkova and Foster 2017). For habitat units classified as pools (whether scour pools, dammed pools, or backwater pools), two depth measurements are recorded: maximum depth and depth at the tail-crest, which is the pool's outlet. The difference between these two measurements is known as residual pool depth. Residual pool depth is important because it describes how much water would remain in the pool if the stream stopped flowing, which is not unusual in dry summer months in the OESF. The residual pool depth calculation is independent of streamflow at the time of measurement (Lisle 1987).

Three pool habitat metrics are analyzed in this report: pool frequency, pool area, and residual pool depth.

## RESULTS

### *Pool frequency*

Pool frequency was calculated as the number of pools per 328 ft (100 m) of stream length.[20] A high frequency of pools indicates high stream complexity. Streams in DNR-managed watersheds averaged 4.9 pools per 328 ft (100 m), and this pool frequency did not vary significantly among channel types (Tables 9a, 9b, Figure 11a). The significant bankfull width covariate indicated that pool frequency decreased as stream width increased, a trend that was expected, because larger streams naturally have larger pools and fewer large pools fit within a 328-ft (100-m) sample reach, relative to the smaller pools of a smaller stream (Table 9b).

There was a decrease in pool frequency over the monitoring period for the DNR-managed watersheds (Figures 11b, c), at a rate of 0.2 pools per 328 ft (100 m) per year. This rate of decrease did not differ among the three channel types, and it is currently unclear whether the decrease in pool frequency was associated with the observed decrease in in-stream wood frequency.

Of all pools surveyed, 95% were scour pools, 4% were dammed pools, and 1% were backwater pools.

The 12 reference watersheds had a similar frequency of pools, 5.3 per 328 ft (100 m), as the DNR-managed watersheds (Figure 11d).

[20] For sample reaches longer than 328 ft (100 m), pool counts were adjusted to a 328-ft (100-m), basis.

**Table 9a.** Pool frequency (number of pools per 328 ft (100 m)) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 4.9 | 1.6 | 1.4 | 8.4 | 4.5 – 5.4 |
| | | | | | |
| Pool-riffle channel type (n=59) | 5.2[a] | 1.3 | 3.0 | 7.3 | 4.5 – 5.9 |
| Step-pool channel type (n=97) | 5.2[a] | 1.7 | 1.4 | 8.4 | 4.6 – 5.7 |
| Cascade channel type (n=41) | 4.1[a] | 1.8 | 2.4 | 7.4 | 3.3 – 5.0 |
| | | | | | |
| First measurement (n=50) | 5.6 | 2.5 | 0.0 | 15.0 | 4.9 – 6.3 |
| Last measurement (n=50) | 4.3 | 2.2 | 0.0 | 9.7 | 3.7 – 4.9 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 5.3 | 2.2 | 2.5 | 9.2 | 4.0 – 6.5 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 9b.** Analysis of pool frequency (number of pools per 328 ft (100 m)) in 50 Type 3 streams on DNR-managed land in the OESF.

| **Effect** | **F value** | ***p*-value** | **Interpretation** |
|---|---|---|---|
| Channel type | 1.7 | 0.18 | No effect |
| Year | 10.2 | <0.01 | Significant |
| Channel type x Year interaction | 0.9 | 0.42 | No effect |
| Bankfull width (covariate) | 19.8 | <0.01 | Significant (negative) |

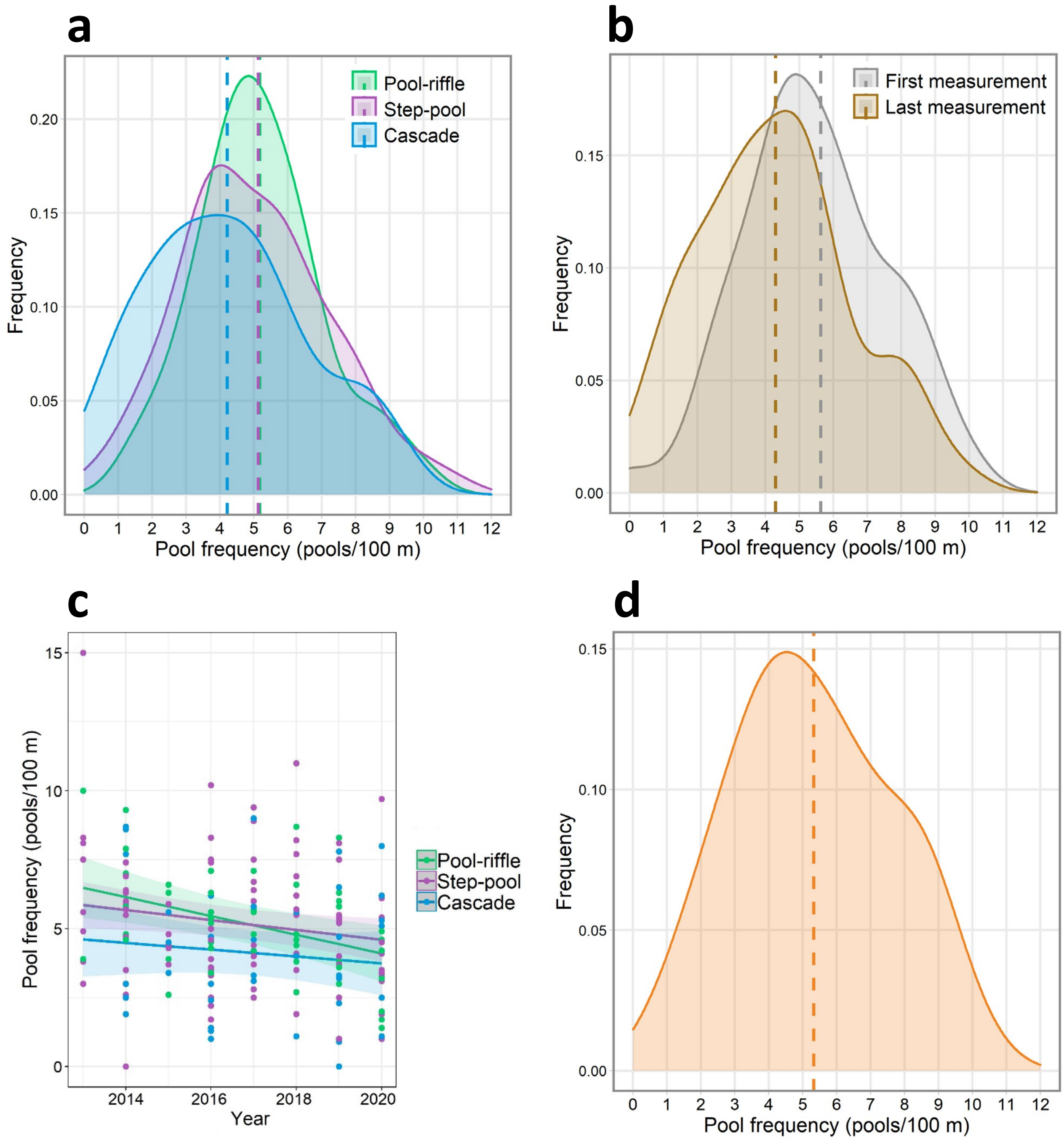



**Figure 11.** Distribution of DNR-managed watersheds according to pool frequency, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of pool frequency for all reference watersheds (d). Vertical dashed lines indicate averages.

### *Pool area*

The second pool metric analyzed was pool area, which was defined as total surface area of pools as a percentage of the total stream surface area in the sample reach. This metric differs from pool frequency because some streams may only have a few pools, but those pools may be very large and provide substantial habitat opportunities. Average pool area for DNR-managed watersheds was 30.9%, but pool area differed widely among channel types, with the pool-riffle type having the highest pool area at 47.7% (Tables 10a, 10b, Figure 12a). Because the three channel types naturally differ in gradient, this difference in pool area among types is expected, with the pool-riffle type having the lowest gradient and also the greatest pool area.

**Table 10a.** Pool area (% of total surface area) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | **Mean** | **SD** | **Min.** | **Max.** | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 30.9 | 16.2 | 5.2 | 65.4 | 26.4 – 35.4 |
| | | | | | |
| Pool-riffle channel type (n=59) | 47.7 [a] | 13.5 | 23.4 | 65.4 | 42.6 – 52.8 |
| Step-pool channel type (n=97) | 25.8 [b] | 9.6 | 10.7 | 46.7 | 21.8 – 29.9 |
| Cascade channel type (n=41) | 17.2 [b] | 7.7 | 5.2 | 29.0 | 11.2 – 23.1 |
| | | | | | |
| First measurement (n=50) | 31.6 | 17.2 | 0.0 | 76.6 | 26.8 – 36.3 |
| Last measurement (n=50) | 31.2 | 20.9 | 0.0 | 88.1 | 25.4 – 37.0 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 28.3 | 15.1 | 7.1 | 60.0 | 19.7 – 36.8 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 10b.** Mixed-model analysis of variance results for pool area (% of total surface area) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 14.6 | <0.01 | Significant |
| Year | 0.5 | 0.50 | No effect |
| Channel type x Year interaction | 1.3 | 0.27 | No effect |
| Bankfull width (covariate) | 8.5 | <0.01 | Significant (positive) |

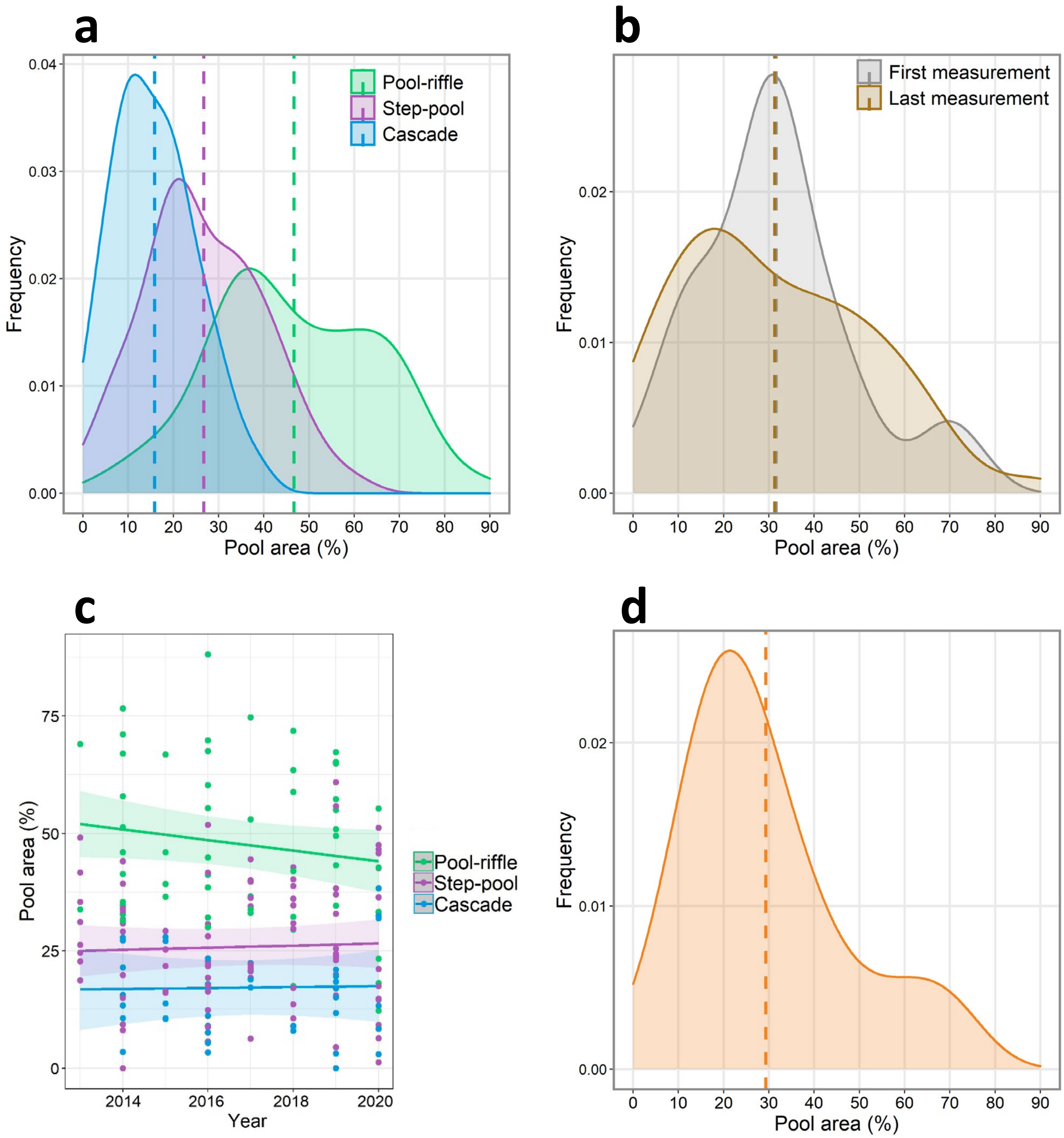



**Figure 12.** Distribution of DNR-managed watersheds according to percent pool area, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of percent pool area for all reference watersheds (d). Vertical dashed lines indicate averages.

Unlike pool frequency, pool area did not change significantly over time (Tables 10a, 10b, Figure 12c), though there was an apparent shift in the shape of the distribution between the first and last measurement (Figure 12b). Pool area in the 12 reference watersheds was similar to that of the DNR-managed watersheds (Figure 12d).

### *Residual pool depth*

Residual pool depth averaged 14.6 in. (37.1 cm) for the DNR-managed watersheds (Table 11a, Figure 13). This represents the average depth of pools (measured at the deepest point of each pool) if streamflow were to cease. Residual pool depth was strongly correlated to stream size, as indicated by the significant bankfull width covariate (Table 11b).

**Table 11a.** Residual pool depth (inches; centimeters in parentheses) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | **Mean** | **SD** | **Min.** | **Max.** | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 14.6 (37.1) | 4.5 (11.5) | 8.0 (20.2) | 27.5 (69.8) | 13.3 – 15.9 (33.9 – 40.3) |
| | | | | | |
| Pool-riffle channel type (n=59) | 16.5 [a] (42.0) | 5.9 (15.1) | 9.0 (22.9) | 27.5 (69.8) | 14.8 – 18.3 (37.6 – 46.4) |
| Step-pool channel type (n=97) | 13.2 [a] (33.6) | 3.6 (9.2) | 8.0 (20.2) | 22.1 (56.1) | 11.9 – 14.6 (30.2 – 37.1) |
| Cascade channel type (n=41) | 14.2 [a] (36.0) | 3.3 (8.4) | 9.1 (23.1) | 20.1 (51.0) | 12.2 – 16.2 (30.9 – 41.2) |
| | | | | | |
| First measurement (n=50) | 13.9 (35.3) | 6.3 (13.0) | 6.3 (15.9) | 27.9 (70.8) | 12.5 – 15.3 (31.7 – 38.9) |
| Last measurement (n=50) | 15.2 (38.5) | 7.5 (14.1) | 7.5 (19.0) | 34.8 (88.5) | 13.6 – 16.7 (34.6 – 42.4) |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 14.6 (37.2) | 6.6 (13.8) | 6.6 (16.7) | 25.4 (64.5) | 11.6 – 17.7 (29.4 – 45.0) |

Note: channel type means followed by the same letter do not differ significantly.

**Table 11b.** Mixed-model analysis of variance results for residual pool depth (cm) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 0.4 | 0.70 | No effect |
| Year | 8.6 | <0.01 | Significant |
| Channel type x Year interaction | 5.1 | <0.01 | Significant |
| Bankfull width (covariate) | 37.0 | <0.01 | Significant (positive) |

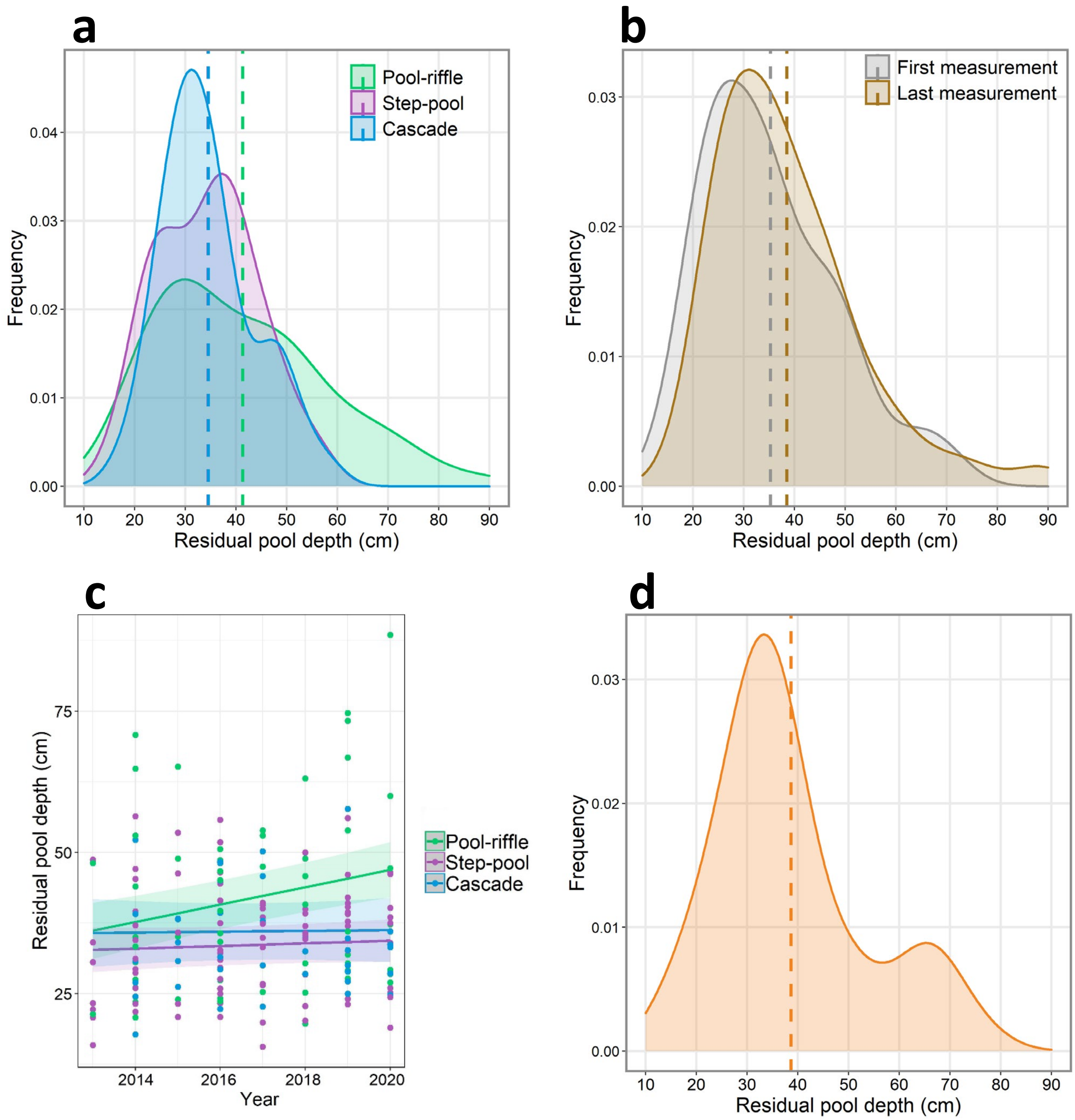



**Figure 13.** Distribution of DNR-managed watersheds according to residual pool depth, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of residual pool depth for all reference watersheds (d). Vertical dashed lines indicate averages.

Analysis of variance showed a significant interaction between the effects of channel type and year on residual pool depth. Thus, the change over time in residual pool depth was different for the three channel types. As shown in Figure 13c, pool depth for the pool-riffle channel type (the lowest gradient type) increased over time while the pool depth did not change for the other two types.

Residual pool depth in reference streams averaged 14.6 in. (37.2 cm) (Table 11a, Figure 13d).

Overall, residual pool depth in both DNR-managed and reference watersheds was very similar to the averages reported previously for partially harvested (13.8 in.; 35 cm) and reference (14.2 in.; 36 cm) watersheds across western Washington (Ralph et al. 1994).

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

Cumulatively, trends in the three metrics of pool habitat present a mixed result. Though pool frequency declined during the measurement period, pool area did not change. The pool-riffle channel type stood out as having the greatest pool area and showed an increase over time toward greater residual pool depth. The pool-riffle channel type also had the greatest variability in pool area (Figure 12a), suggesting that this channel type may have potential for a broader range of pool conditions than the other types. It should be stressed, however, that the short-term trends evident so far should not be extrapolated over long time periods. This is because we don't yet know what portion of these trends is a result of short-term variation versus long-term change.

In the small streams of rain-dominated westside forests, pool spacing, pool volume, and stream sediment storage are positively correlated to the amount of in-stream wood (Montgomery et al. 1995; Beechie and Sibley 1997). In coastal streams of southeastern Alaska, where channel conditions and channel types are similar to those of the coastal Olympics, 73% of all pools were formed by large organic debris (Heifetz et al. 1986). Future research in Status and Trends Monitoring will evaluate relationships between observed in-stream wood and pool habitat, at the level of individual streams.

## Channel substrate

### ECOLOGICAL CONTEXT

Channel substrate refers to the mineral and organic materials that compose the streambed, commonly gravel, cobbles, sand, and boulders. Substrate composition determines the roughness of the stream channel, which influences channel hydraulics (stream depth, width, velocity) and consequently stream habitat. Channel substrate provides the microhabitat conditions required by many aquatic species including periphyton, macroinvertebrates, amphibians, and fish (Cummins 1974; Hicks et al. 1991; Mellina and Hinch 2009; Roni et al. 2006). Substrate particle size, composition, and stability can be limiting factors in anadromous salmonid spawning and rearing habitats (Bain 1999; Kondolf 2000), as different species require different sizes and amounts of gravel substrate to build a nest, or redd. Specific particle sizes are needed for spawning because eggs adhere to gravel surfaces, and interstitial water must flow through the substrate to maintain high oxygen levels around buried eggs.

Historically, forest harvest and road management activities within and upslope of the riparian zone produced significant sediment delivery to streams; this was locally documented in the Clearwater drainage of the Olympic Peninsula during the peak rates of harvest in the 1970s (Cederholm and Salo 1979; Reid and Dunne 1984). The primary mechanisms of sediment delivery were the increased frequency and severity of mass-wasting events, hillslope erosion, and erosion from road surfaces during use (Beschta 1978, Wemple et al. 2001, Reid and Dunne 1984). Road construction methods used prior to the early 1970s were especially prone to causing landslides and erosion (Cederholm et al. 1981). Road use then and today still results in sediment delivery, though impacts are highly variable and most delivery occurs at specific “hotspots” (Al-Chokhachy et al. 2016). Large additions of sediments to stream channels, such as through landslides, may in some cases continue to be transported downstream over multiple decades (Tschaplinski and Pike 2017).

Between the 1970s and present, modern riparian best management practices (BMPs) have evolved. These BMPs usually consist of stream buffers where forest harvesting is limited or prohibited and where heavy machinery and road construction activities are regulated. Other common BMPs are cross drains to disperse water in ditches, silt fences, catch basins, rock berms, and culvert flumes. Road construction and harvest timing restrictions also are often used to prevent sediment delivery. BMPs have dramatically reduced sediment delivery to streams during forest harvest and road construction (Reiter et al. 2009; Anderson and Lockaby 2011). Recent field studies in the coastal Pacific Northwest found that when stream buffers and other BMPs were properly implemented, logging and road improvement activities produced no detectible increase in stream suspended solids or turbidity (Rashin et al. 2006; Clinton 2011; Arismendi et al. 2017; Hatten et al. 2018; Rachels et al. 2020).

High levels of fine sediments in streams are detrimental to salmon spawning habitat, as excessive amounts of fines fill the interstitial space among spawning gravel, decreasing the necessary flow of water and negatively affecting incubation and emergence (Cederholm et al. 1981; Jensen et al. 2009; Kondolf 2000). The stream channels most vulnerable to sedimentation are low-gradient response reaches (e.g., pool-riffle channels), which naturally have a slower water velocity and thus accumulate the suspended sediments transported downstream from higher-gradient parts of the stream network (Allan and Castillo 2007). Because of the potential for fine sediments to impact stream habitat, changes in the composition of channel substrate is often used as an indicator of management impact.

In the environmental impact analysis for the OESF Forest Land Plan, fine sediment delivery (small soil particles such as sand, silt, and clay) was recognized as an important indicator of management effects on fish habitat and riparian areas (DNR 2016b). Because empirical sediment delivery data were not available at the time of that analysis, they were modeled using road inventory data and traffic impact scores based on road surface type, road proximity to streams, and projected traffic levels. The present monitoring program provides directly measured data on the status and trends of stream channel substrate.

## SAMPLING

During each channel substrate survey, 21 random streambed substrate particles were sampled at equally spaced intervals across each of 6 channel cross sections for a total of 126 particles per sample reach (Minkova and Foster 2017). The size class of each substrate particle was determined using a gravel size template, or gravelometer. There are 16 diameter classes ranging from 0–0.08 in. (0–2 mm), a class commonly known as 'fines', to ≥10 in. (25 cm), a class known as 'boulders'.

Many different summary statistics can be calculated to describe the composition of channel substrate in a sample reach. Two statistics commonly reported in the scientific literature were selected for this report: median particle size (commonly called '$D_{50}$', the 50$^{th}$ percentile diameter) and percent fines. The dataset analyzed here consists of a total of 197 channel substrate surveys conducted in the DNR-managed watersheds and 42 surveys conducted in reference watersheds.

## RESULTS

### *Median particle size*

Median particle size ($D_{50}$) is the median size class of the 126 particle samples measured during each sample reach survey. Among the DNR-managed watersheds, $D_{50}$ values ranged widely, from 0.2 in. (5.6 mm) to 7.1 in. (180 mm). Our analysis showed distinct differences in $D_{50}$ among the three channel types (Tables 12a, 12b, Figure 14). The lowest-gradient channel type, pool-riffle, had the smallest $D_{50}$ and the highest-gradient type, cascade, had the largest $D_{50}$. The frequency distribution for the pool-riffle channel type also had a much narrower range for $D_{50}$ than for the other types. The differences in sediment size among channel types were expected, based on sediment production and hydrologic characteristics of a watershed that control the input and transport of sediment in channels. Steeper streams are closer to hillslopes and receive direct hillslope inputs that may include silt to boulder-sized particles. Flows large enough to mobilize the boulder- and cobble-sized particles seldom occur and consequently only the smaller grains are transported downstream. In contrast, sediment inputs to low gradient streams may be almost entirely derived from particles mobilized in the upstream, steeper channels. The hydraulic sorting that occurs in the upstream channels limits the range of grain sizes that enter and deposit in the low-gradient channels.

Given the $D_{50}$ differences observed among channel types, there also was a significant positive association between stream size (i.e., bankfull width) and $D_{50}$ (Table 12b). This indicates that within the range of Type 3 streams monitored, larger streams had larger $D_{50}$. This likely occurred because—all else held equal—larger streams are capable of transporting small substrate particles more easily than smaller streams; thus larger streams retain relatively more large particles than small particles.

No change in $D_{50}$ was observed over time.

**Table 12a.** Median particle size, or $D_{50}$ (inches; millimeters in parentheses), in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary Mean | SD | Min. | Max. | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 2.4 (60) | 1.2 (30) | 0.6 (16) | 5.4 (138) | 2.0 – 2.7 (51 – 68) |
| | | | | | |
| Pool-riffle channel type (n=59) | 1.5 [c] (37) | 0.6 (16) | 0.6 (16) | 3.1 (79) | 1.0 – 1.9 (25 – 49) |
| Step-pool channel type (n=97) | 2.4 [b] (61) | 1.1 (27) | 0.8 (20) | 4.5 (115) | 2.0 – 2.8 (52 – 71) |
| Cascade channel type (n=41) | 3.4 [a] (86) | 1.3 (33) | 1.9 (47) | 5.4 (138) | 2.8 – 3.9 (72 – 100) |
| | | | | | |
| First measurement (n=50) | 2.2 (57) | 1.3 (33) | 0.4 (11) | 7.1 (180) | 1.9 – 2.6 (48 – 66) |
| Last measurement (n=50) | 2.3 (59) | 1.2 (30) | 0.6 (16) | 5.0 (128) | 2.0 – 2.6 (51 – 67) |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 1.7 (43) | 1.0 (26) | 0.4 (11) | 4.1 (105) | 1.1 – 2.3 (28 – 58) |

Note: channel type means followed by the same letter do not differ significantly.

**Table 12b.** Analysis of median particle size, or $D_{50}$, in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 7.1 | <0.01 | Significant |
| Year | 0.4 | 0.53 | No effect |
| Channel type x Year interaction | 0.2 | 0.83 | No effect |
| Bankfull width (covariate) | 15.5 | <0.01 | Significant (positive) |

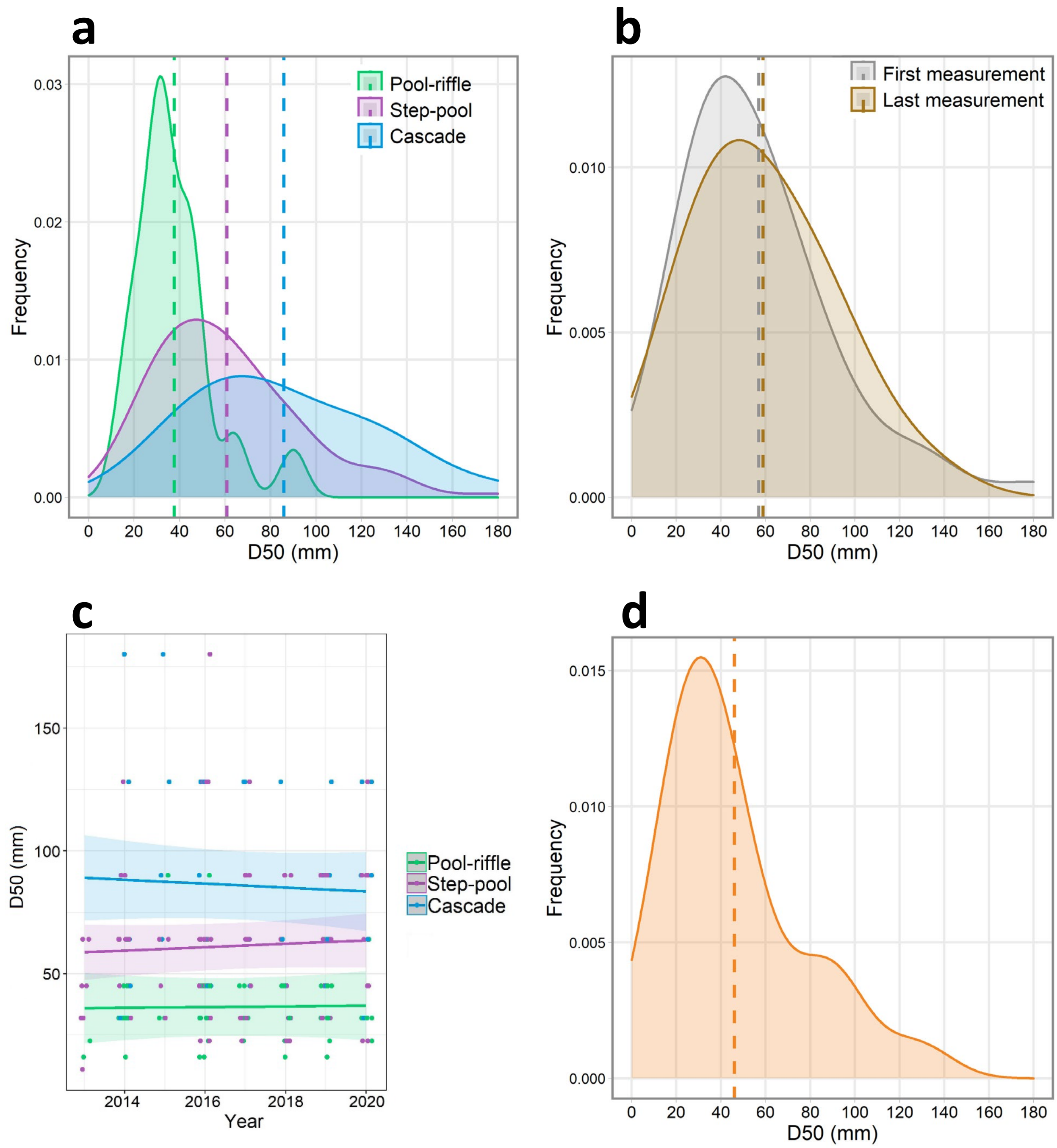



**Figure 14.** Distribution of DNR-managed watersheds according to median particle size ($D_{50}$), by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of median particle size for all reference watersheds (d). Vertical dashed lines indicate averages.

### *Percent fines*

The percent fines indicator metric is calculated as the percentage of the 126 particle samples in a sample reach that are 2 mm or smaller in size. The 50 DNR-managed watersheds averaged 8.0% fines, overall (Table 13a). Percent fines differed significantly among the three channel types, with the lowest-gradient type—pool-riffle—having the highest level of fines (12.1%) of the three channel types (Tables 13a, 13b, Figure 15). Percent fines was not correlated with stream bankfull width. Percent fines did not change significantly over time based on a 95% confidence threshold, though it should be noted that the *p*-value for the effect of year was 0.07, missing the 0.05 threshold of significance by only a narrow margin. In the reference watersheds, fines averaged 13.0%.

**Table 13a.** Percent fines (% of substrate ≤2 mm) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 8.0 | 5.0 | 0.2 | 20.5 | 6.6 – 9.4 |
| | | | | | |
| Pool-riffle channel type (n=59) | 12.1[a] | 4.7 | 4.6 | 20.5 | 9.8 – 14.4 |
| Step-pool channel type (n=97) | 6.7[b] | 4.1 | 1.4 | 17.8 | 4.9 – 8.5 |
| Cascade channel type (n=41) | 5.3[b] | 4.2 | 0.2 | 13.0 | 2.6 – 8.0 |
| | | | | | |
| First measurement (n=50) | 7.6 | 5.7 | 0.0 | 25.0 | 6.1 – 9.2 |
| Last measurement (n=50) | 9.4 | 6.2 | 0.0 | 32.3 | 7.7 – 11.1 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 13.0 | 7.8 | 3.5 | 32.6 | 8.5 – 17.4 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 13b.** Mixed-model analysis of variance results for percent fines (% of substrate ≤2 mm) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 4.2 | 0.02 | Significant |
| Year | 3.4 | 0.07 | No effect |
| Channel type x Year interaction | 0.5 | 0.58 | No effect |
| Bankfull width (covariate) | 0.0 | 0.91 | No effect |

Note: prior to analysis, data were transformed using the angular transformation.

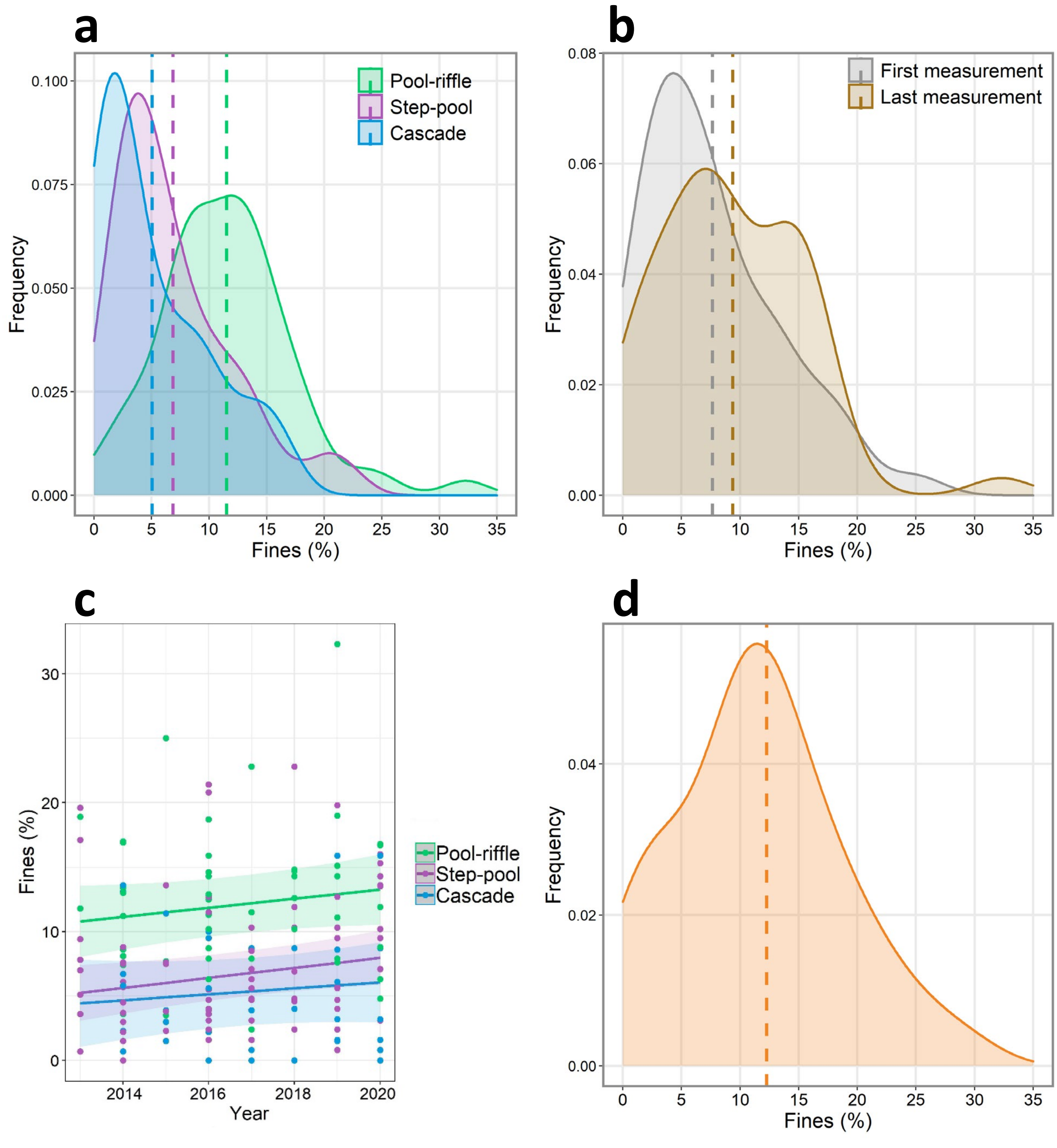



**Figure 15.** Distribution of DNR-managed watersheds according to percent fines, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of percent fines for all reference watersheds (d). Vertical dashed lines indicate averages.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

The channel substrate measurements showed no significant evidence of change for DNR-managed watersheds during the monitoring period, though it will be important to re-evaluate percent fines at the next monitoring interval (study year 10), to determine whether significant trends appear. It is possible that gradual changes in channel substrate are occurring, but at a rate too slow to detect over this relatively short monitoring period. Alternatively, the sampling methods used may not be sensitive enough to detect small changes in fine sediments, which are known to be difficult to detect (Kondolf 2000). In either case, the current channel substrate composition has been in a relatively stable condition since 2013.

Any changes in substrate condition—changes in percent fines or $D_{50}$—are likely to be most apparent in the low-gradient channels such as the pool-riffle type, as these accumulates sediments transported from upstream. The higher-gradient step-pool and cascade channel types, with greater water velocities and naturally coarser substrate, transport sediments but are much less likely to accumulate detectible changes in fine sediment (Montgomery and McDonald 2002).

Information from published field and laboratory studies can help in assessing the level of habitat quality provided by the current substrate conditions in DNR-managed watersheds. For percent fines, direct comparisons among studies are complicated by a historical lack of standardization in sampling methodology (Whitacre et al. 2007; Jensen et al. 2009). For example, extensive field observations that included the Olympic Peninsula indicated that approximately 12% fines (defined as particles <1 mm) is the maximum allowable for quality spawning gravel conditions (Kondolf 2000). But because this definition of fines differed from that of our sampling protocol (i.e., <2 mm), we cannot make direct interpretations of our data using the 12% threshold.

The pool-riffle channel type was, as expected due to its low gradient, significantly higher in fines than the other two channel types. Thus, low-gradient streams, such as the pool-riffle type, warrant close attention in future habitat monitoring. However, an important consideration in evaluating spawning habitat conditions is that different species of salmonids spawn in different reach types; for example, coho salmon are not expected to spawn in cascade reaches due to the high stream gradient (Martens and Dunham 2021). Future analysis will integrate fish population data from DNR's Riparian Validation Monitoring (Martens 2016) with the substrate surveys of the present monitoring program to determine whether streams with a high percentage of fines show evidence of reduced fish densities.

Although our substrate surveys sampled the entire stream channel and not specifically the spawning gravel, the calculated $D_{50}$ values provide a general picture of channel substrate size. For this purpose, $D_{50}$ values below 1.2 inches (30 mm) suggest gravel is smaller than optimal for spawning of the salmonid species present in the Type 3 streams of the OESF, whereas values of less than 0.6 in. (15 mm) suggest poor spawning habitat. However, it is important to note that there are significant differences in the size of spawning gravel among salmonid species, with differences generally related to size of adult fish (Kondolf and Wolman 1993). Mean values for $D_{50}$ were above the 1.2 in. (30 mm) threshold for all channel types. In future monitoring, we will consider including a sampling protocol to specifically target spawning gravel.

## Channel morphology

### ECOLOGICAL CONTEXT

Channel morphology describes the shape of the stream channel and is a result of stream-reach and watershed-level ecological process that affect sediment supply and transport (Montgomery and Buffington 1993). Channel morphology is influenced by valley slope, channel confinement, sediment inputs, the composition of stream banks, flow obstructions such as in-stream wood, and riparian vegetation (Allan and Castillo 2007; Montgomery and Buffington 1997). In turn, channel morphology influences the distribution and abundance of aquatic plants and animals by governing the characteristics of water flow and the capacity of streams to store sediment and transform organic matter (Bisson et al. 2006).

Forest management has the potential to affect stream channel morphology by altering sediment inputs and changing the amount of in-stream wood. Historically, intensive forest harvest without properly engineered roads or riparian conservation measures directly impacted stream channel morphology through sediment delivery to streams by landslides, debris flows, and road surface erosion (Bestcha 1978, Cederholm et al. 1981, Roberts and Church 1986, Wemple et al. 2001). Such occurrences were well-documented on the Olympic Peninsula during the 1960s to 1980s (Cederholm et al. 1981). Removal of wood from streams following harvest also had a long-term impact on channel morphology (Ralph et al. 1994).

A possible indirect effect of forest management on channel morphology is through the influence of forest harvest on watershed hydrology (e.g., higher peaks in streamflow). Streamflow influences channel morphology via processes such as channel scouring and streambank erosion. Streamflow responses to harvest have been documented in small watersheds after significant portions were clear cut (Fredriksen and Harr 1979), with road networks potentially making significant contributions to peak flows (Harr et al. 1975; Jones et al. 2000). A meta-analysis by Grant et al. (2008) investigated increases in peak flow using studies conducted in western Oregon and Washington. They found evidence that forest harvest increased peak flows by a detectable amount relative to reference watersheds but only in rain-dominated watersheds, during small to moderate peak flow events (those with a return interval of 6 years or less; i.e., a 6-year flood). Forest harvest effects were not detected during more extreme events, as flows then were very high in both the harvested and reference watersheds. Harvest effects on peak flow were detectable only when at least 29% of a watershed was clear cut in the rain-dominated zone, or at least 15% in the rain-on-snow zone. Surprisingly, the meta-analysis by Grant et al. (2008) found that no field studies to date had directly linked higher harvest-induced peak flows in rain-dominated watersheds to increased change in channel morphology, relative to reference watersheds.

Any potential changes in channel morphology caused by altered peak flows are most likely to be detected in low-gradient channels of less than approximately 2% slope, where the stream channel is composed of relatively fine materials, rather than larger cobbles and boulders (Grant et al. 2008). Under the channel classification system used here, these streams are classified as the pool-riffle channel type (Bisson et al. 2006). In these streams, high flows can transport fine sediments out of a reach or allow accumulation of sediments as flows recede.

Two metrics of channel morphology are presented here: the ratio of channel width:depth and bank erosion. Channel width:depth is affected by various factors that affect channel morphology, but it can be used specifically to assess how streams process sediment deposits originating from landslides or from various types of erosion (Ebersole et al. 2003; Platts 1991). When sediment inputs exceed the transport capacity of a stream, the channel can aggrade, becoming shallower and wider (Mallik et al. 2011), with a higher ratio of width:depth. Width:depth has also been used as an index of recovery from human-caused channel degradation (Ebersole et al. 2003). Bank erosion directly measures the most local source of sediment input to streams: the stream bank. Excessive sediment influx to streams can negatively affect habitat for fish and other aquatic organisms (Chamberlain et al. 1991).

## SAMPLING

Channel width:depth was calculated from measurements made at six permanent cross-sections within each sample reach (Minkova and Foster 2017). These cross-sections were located at equal intervals along the reach. At each cross-section, the stream bankfull width was measured. Bankfull width is the horizontal distance between bankfull stage indicators on each side of a stream, measured directly across the channel. The recurrence interval of these events varies among channels and regions, but is generally between 0.5 and 2.0 years (Williams 1978). Bankfull depth was measured as the vertical distance between bankfull stage and the streambed, regardless of the presence or depth of water. At each cross-section, bankfull depth was measured at 10 equally spaced intervals and these 10 measurements were then averaged. Finally, the ratio of bankfull width:depth was calculated for each cross section and then averaged for each sample reach.

Bank erosion was surveyed on both sides of the stream, along the entire sample reach. All areas of actively eroding bank that were above bankfull stage and greater than 6.6 ft (2 m) long and 1.6 ft (0.5 m) high, were measured. To qualify as actively eroding, an area had to have exposed soil, not covered by vegetation, that was crumbling or falling into the stream. Survey data from both banks of the stream were combined, and bank erosion was calculated as the percent of the total length occupied by active erosion. For example, if 20% of the length of one bank of a sample reach had active erosion and the other bank had none, active erosion for the reach would be 10%.

## RESULTS

### *Channel width:depth*

For the 50 DNR-managed watersheds, the ratio of channel width:depth averaged 24.4 and did not differ among the three channel types (Table 14a, 14b, Figure 16). The non-significant year effect indicated there was no significant change in width:depth over time.

In the width:depth analysis, only the bankfull width covariate effect was significant, indicating that among the 50 DNR-managed watersheds, larger watersheds tended to have streams with greater width:depth ratios. This result is expected among montane watersheds because as watershed size increases, the watershed outlet tends to have a lower-gradient channel where the transport capacity is often exceeded by sediment supply rates (Montgomery and Buffington 1997; Grant et al. 2008). As a result, larger watersheds are expected have wider streams at the outlet, with higher width:depth ratios than smaller watersheds.

Among the reference watersheds, width:depth averaged 22.1.

**Table 14a.** Bankfull width:depth ratio in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% confidence interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 24.4 | 5.5 | 13.0 | 38.1 | 22.5 – 25.5 |
| | | | | | |
| Pool-riffle channel type (n=59) | 24.4 [a] | 5.5 | 13.0 | 36.6 | 22.2 – 26.6 |
| Step-pool channel type (n=97) | 25.4 [a] | 5.3 | 17.1 | 38.1 | 23.7 – 27.1 |
| Cascade channel type (n=41) | 21.3 [a] | 4.4 | 14.5 | 27.8 | 18.7 – 23.9 |
| | | | | | |
| First measurement (n=50) | 23.6 | 5.8 | 11.4 | 38.6 | 22.0 – 25.2 |
| Last measurement (n=50) | 23.3 | 6.2 | 13.1 | 36.0 | 21.6 – 25.1 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 22.1 | 4.3 | 12.7 | 29.7 | 19.7 – 24.5 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 14b.** Analysis of bankfull width:depth ratio in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 1.7 | 0.19 | No effect |
| Year | 0.0 | 0.97 | No effect |
| Channel type x Year interaction | 0.3 | 0.72 | No effect |
| Watershed area (covariate) | 12.1 | <0.01 | Significant (positive) |

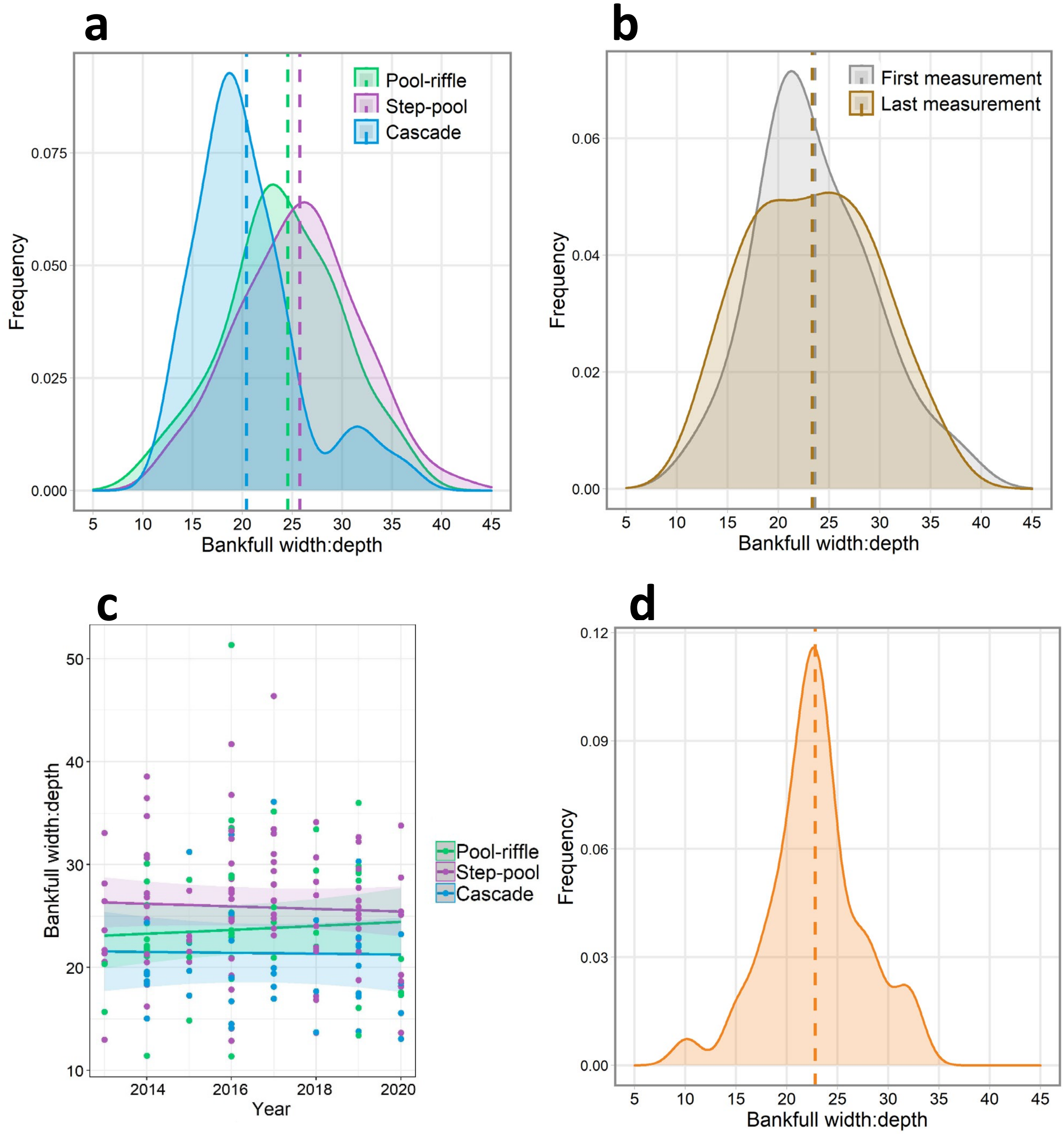



**Figure 16.** Distribution of DNR-managed watersheds according to bankfull width:depth ratio, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of width:depth ratio for all reference watersheds (d). Vertical dashed lines indicate averages.

### *Bank erosion*

Bank erosion in the 50 DNR-managed watersheds averaged 11.7 percent. Averaged over the 2013-2020 measurement period, the cascade channel type had less bank erosion than the pool-riffle channel type among the 50 DNR-managed watersheds (Table 15a). This pattern is expected because cascade channels are typically less sinuous than pool-riffle channels and less prone to bank erosion than lower-gradient channel types (Montgomery and MacDonald 2002). There was an interaction between year and channel type, with bank erosion declining over time for the pool-riffle and cascade channel types, but not for the step-pool channel type (Table 15b, Figure 17). The reason for these different trends is not clear, but bank erosion is a dynamic process and we anticipate that additional years of monitoring will reveal whether trends observed so far are short-term fluctuations or part of a longer trend.

Bank erosion in the 12 reference watersheds averaged 6.5 percent.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

The channel morphology data collected so far reflect differences in sediment transport and deposition that are expected to occur naturally as a result of differences in watershed size and associated stream gradient at the watershed outlet (Montgomery and MacDonald 2002). Nonetheless, channel morphology is quite variable among streams and within individual stream reaches, as a result of additional factors such as channel confinement, roughness of the channel associated with substrate, and in-stream wood. Because of this variability, it is difficult at this point to detect long-term trends. However, we observed no increases in width:depth or bank erosion, either of which would have suggested potential natural or human-caused increases in sediment within the stream.

Further evaluation of channel morphology will occur through additional years of monitoring. We also anticipate a more detailed technical analysis that is beyond the scope of this report. Such an analysis will evaluate various changes in the shape stream cross-sections, caused by scouring or aggradation.

**Table 15a.** Bank erosion (%) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=197) | 11.7 | 9.1 | 0.7 | 41.1 | 9.1 – 14.2 |
| | | | | | |
| Pool-riffle channel type (n=59) | 15.4 [a] | 10.9 | 1.3 | 41.1 | 10.7 – 20.0 |
| Step-pool channel type (n=97) | 10.8 [ab] | 8.6 | 0.7 | 34.3 | 7.1 – 14.5 |
| Cascade channel type (n=41) | 8.9 [b] | 5.8 | 1.4 | 18.8 | 3.5 – 14.4 |
| | | | | | |
| First measurement (n=50) | 12.6 | 12.5 | 0.0 | 48.8 | 9.1 – 16.1 |
| Last measurement (n=50) | 8.7 | 10.7 | 0.0 | 49.6 | 5.7 – 11.6 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=42) | 6.5 | 3.3 | 0.7 | 12.2 | 4.6 – 8.3 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 15b.** Mixed-model analysis of variance results for bank erosion (%) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 4.6 | 0.01 | Significant |
| Year | 9.1 | <0.01 | Significant |
| Channel type x Year interaction | 6.1 | <0.01 | Significant |
| Bankfull width (covariate) | 0.1 | 0.82 | No effect |

Note: For statistical analysis, bank erosion data were transformed using the angular transformation.

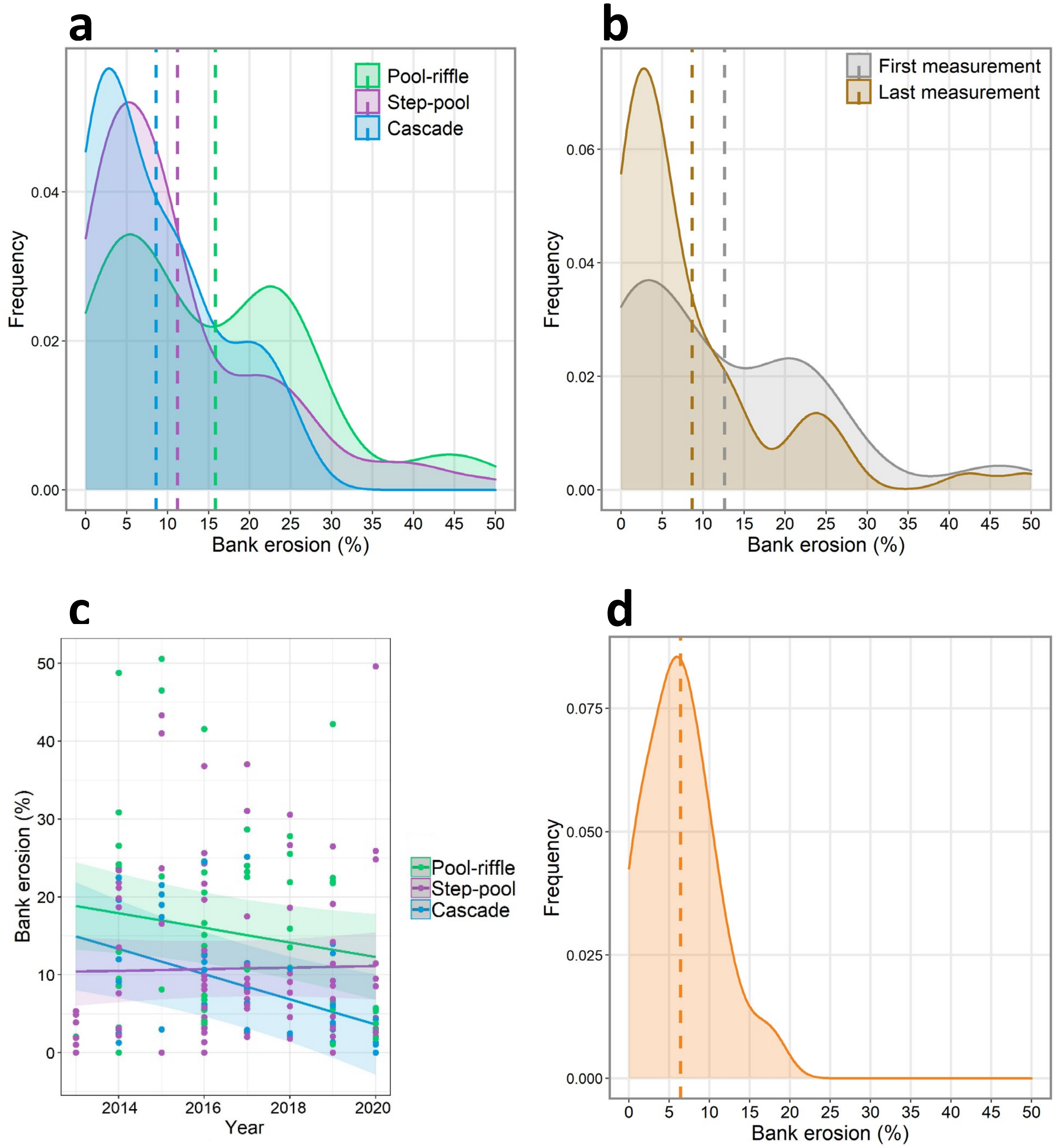



**Figure 17.** Distribution of DNR-managed watersheds according to percent bank erosion, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of percent bank erosion for all reference watersheds (d). Vertical dashed lines indicate averages.

## Riparian vegetation

### ECOLOGICAL CONTEXT

Understanding forest structure dynamics resulting from succession and past disturbances is key to assessing riparian forest influence on aquatic and riparian habitats. The riparian forest is an important component of these habitats, influencing stream temperature and riparian microclimate through its shade, supplying leaf litter and woody debris to streams and the forest floor, and stabilizing stream banks. Riparian buffer zones are expected to provide all of these riparian functions.

Prior to the implementation of stream buffers in the 1990s, many riparian zones in the OESF were clear cut to the stream edge or to the last row of trees bordering the stream. OESF riparian forests today are a mixture of natural regeneration, planted conifers, and residual trees that were not harvested. By assessing the current structure of these riparian forests, we can learn about their response to past disturbance and their capacity to provide current and future riparian functions. DNR's Riparian Conservation Strategy assumes that as riparian forests recover from stand-replacing disturbance, so too does riparian function. In the environmental impact analysis for the OESF Forest Land Plan, riparian vegetation height, composition, and mortality rate were key drivers in predicting improvement in stream habitat condition (DNR 2016a).

Under the 1997 HCP and subsequent agency guidance, management in the OESF uses riparian buffers to protect stream habitat by minimizing the disturbance of unstable channel banks and maintaining forest cover near streams (DNR 2016a, p. 3-26 to 3-43). As described in the Introduction of this report, riparian buffers implemented under the 2016 OESF Forest Land Plan consists of: (1) an interior-core buffer adjacent to the stream that is intended to protect and aid restoration of riparian processes and functions, and (2) an exterior wind buffer applied when the probability of windthrow in the interior-core buffer is high (DNR 2016a). Interior-core buffers have a default width of 100 feet on Type 3 and Type 4 streams but are often extended to incorporate potentially unstable slopes and landforms as well as wetlands. The exterior wind buffer is adjacent to the interior-core buffer and is intended to protect the integrity of the interior-core buffer from loss of riparian function.

The goal of our riparian forest monitoring is to document the current conditions in the riparian forests along Type 3 streams so that we can better understand the outcomes of DNR's Riparian Conservation Strategy in the OESF, in the context of past disturbance and ongoing forest succession.

### SAMPLING

At each sample reach in the DNR-managed and reference watersheds, two 0.44 ac (0.18-ha) rectangular plots were established for the purpose of assessing the riparian forest overstory (Minkova and Foster 2017).[21] Each overstory plot was 200 ft (60 m) long by 100 ft (30 m) wide, with one plot located on each side of the sample reach (Figure 18). The edge of the plot nearest the sample reach was located as close as possible to the channel without intersecting the stream itself. Within each plot, three zones were designated based on distance from the stream: 0-66 ft (0-20 m), 66-133 ft (20-40 m), and 133-200 ft (40-60 m). These are called zones 1, 2, and 3, respectively, and facilitate

[21] Some sample reaches could not accommodate two plots because of the orientation of the reach relative to other streams or rivers or to roads. Thus, the total number of plots was 116 instead of maximum possible 124.

comparison of the overstory composition according to distance from stream. Of the 116 overstory plots installed, 87 were on DNR-managed land and 29 were in reference watersheds.[22]

On each overstory plot, all trees at least 5.0 in (12.7 cm) diameter at breast height (DBH) were measured. For each tree, DBH, species, zone, and status (live or dead) were recorded. As of 2020, the overstory plots had been measured one time; therefore, change over time is not reported here.

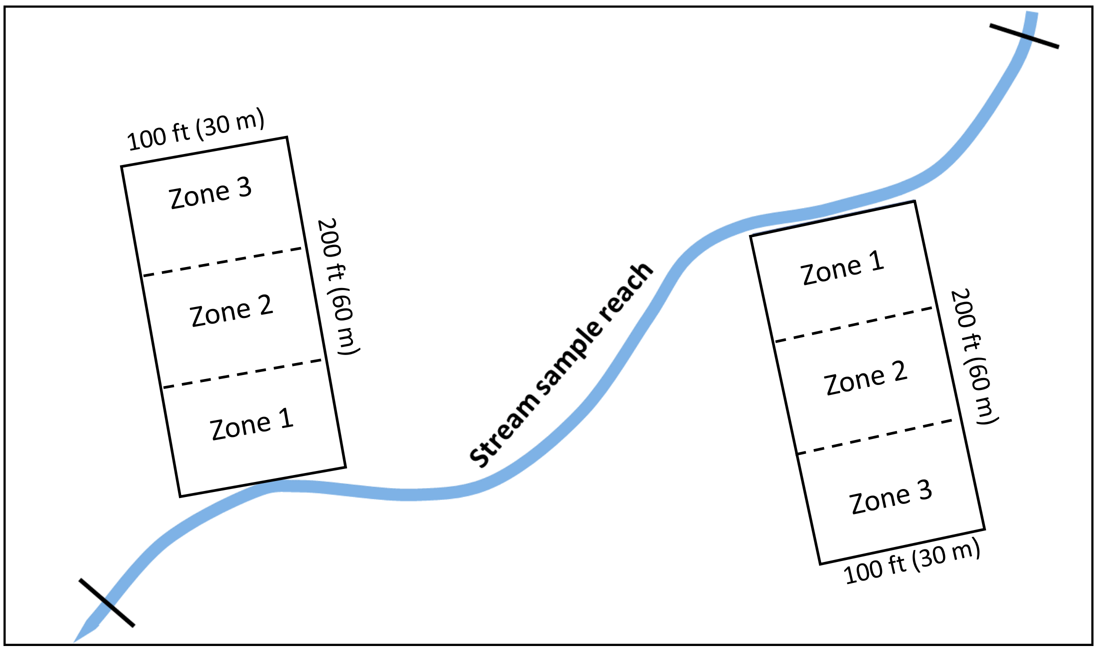



**Figure 18.** Layout of two riparian forest monitoring plots located along a stream sample reach.

## ANALYSIS

To understand the current condition and disturbance history across the 116 overstory plots, multivariate analysis was used to detect patterns in the structure and composition of the forest. Forest management history often differed between the two sides of a sample reach, so plots were analyzed individually instead of combining the two plots at each sample reach.

To perform multivariate analysis, stand summary metrics for each plot were first calculated: trees per acre (TPA), basal area (BA) ($ft^2/ac$), and mean diameter[23]. These metrics were calculated separately for conifers and for hardwoods. The multivariate analysis required that data be present for each metric for every plot; therefore, we did not include hardwood mean diameter, as some plots lacked

[22] In addition to the 12 reference watersheds, there were three DNR-managed watersheds in which reach outlet—and the sample reach—was just outside of state lands (watersheds 796, 797, 820), within Olympic National Park. Because the overstory plots for these reaches were located in the park, we group them with the reference watersheds for this riparian vegetation analysis.

[23] Calculated as quadratic mean diameter.

hardwoods which resulted in a missing value.[24] Our multivariate analysis instead used TPA, basal area, and mean diameter for conifers and TPA and basal area for hardwoods.

The multivariate analysis began with a principal components analysis, which allowed us to assess similarity among the 116 plots, based on the five metrics, and to identify the number of groups into which the plots should most logically be classified. In the second step of the analysis, plots were assigned to groups based on their similarities. Finally, summary statistics were calculated for the plots in each group, to characterize the groups.

## RESULTS

Across all 116 plots, the species composition on the riparian overstory plots was 57% western hemlock, 19% red alder, 12% Sitka spruce, 5% Douglas-fir, 4% Pacific silver fir, and 1% western redcedar, with the remaining 2% divided among various infrequently occurring species.

Through multivariate analysis using five stand metrics, we determined that the 116 plots were best classified into four distinct groups (Figure 19). This number of groups was based on a balance between similarities among plots *within* groups and differences *among* the groups. For example, if the 116 plots had been classified into only three groups, plots that were not very similar would have been lumped together. If the plots had been classified into five groups, plots that were not very different from one another would have been split into separate groups. Characteristics of the four groups are illustrated in Figure 20.

In this analysis, we present diameter distributions for the four groups but not for each of the 116 plots. Though beyond the scope of this report, we anticipate a more detailed future analysis of diameter distributions within plots. This will reveal more about stand histories, such as whether a plot is even-aged or not or whether selective riparian harvest may have occurred in the past.

### *Plot groupings*

We assigned names to the four groups of plots based on the unique stand characteristics of each group. The group names and group attributes are listed in Table 16.

The first group of plots, *Conifer*, comprises 21% of all plots and represents a conifer-dominated overstory in which the conifers average 15.0 inches (38.1 cm) diameter, intermediate in size (and apparent age) between the other two conifer groups. Conifers make up 85% of stand basal area, on average, with red alder making up most of the remaining 15%. This group is predominantly found on DNR-managed land, with only 2 of 24 plots occurring in reference watersheds (one a tributary of the South Fork Hoh River and one a Queets River tributary).

The second and largest group, *Conifer-large*, comprises 49% of all plots and represents the most advanced successional stage of the four groups. Mean diameter of conifers averages 21.4 inches (54.4 cm), and conifers make up 96% of stand basal area, on average. Twenty-four of the 29 plots located in reference watersheds fall into the Conifer-large group; therefore, this group represents the predominant condition of the riparian forest in reference watersheds. However, this group was also common in DNR-managed watersheds, making up 36% of all plots in DNR-managed watersheds.

[24] However, we could include TPA and BA for plots without hardwoods because those values were zero, rather than a missing value.

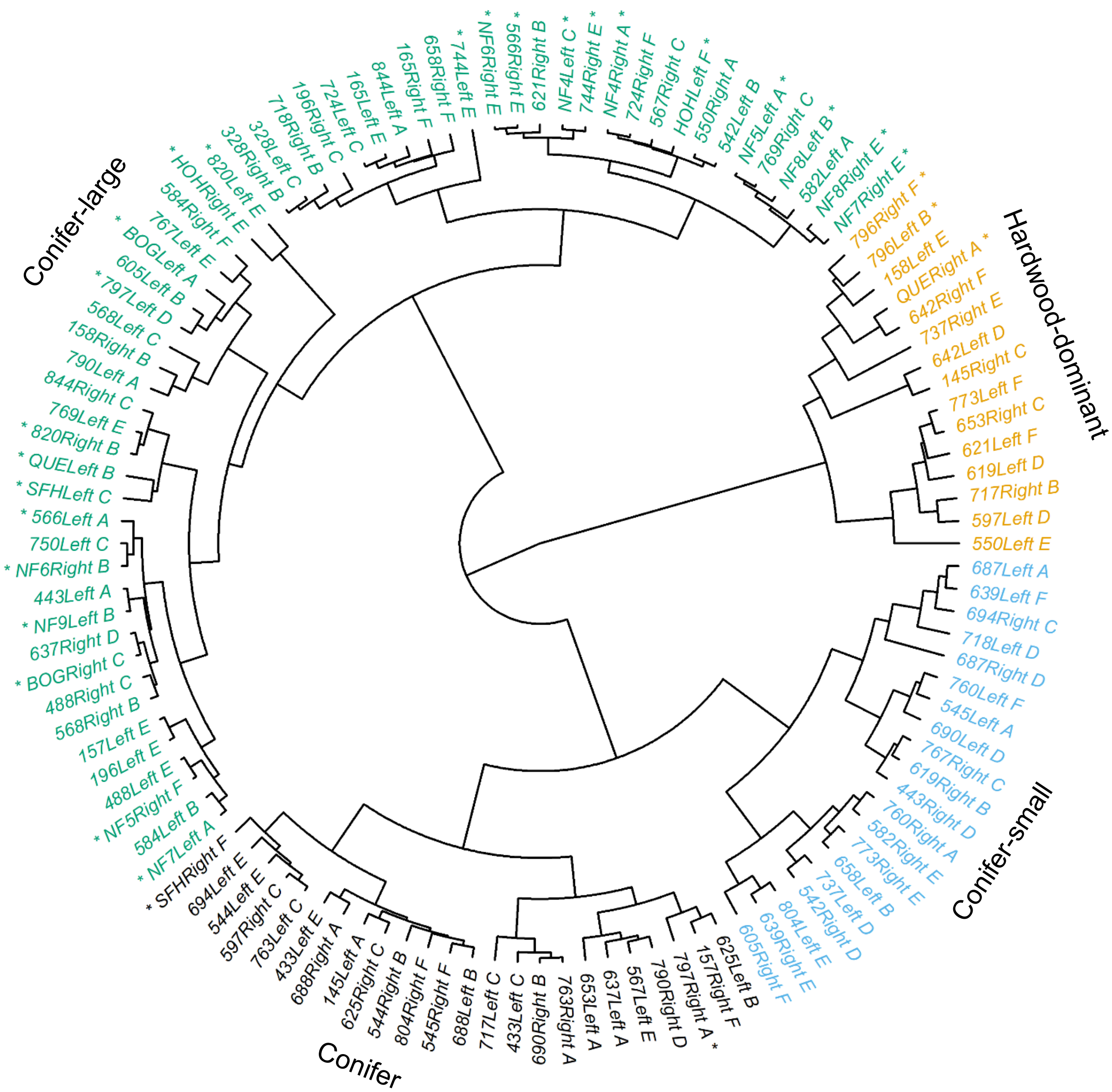



**Figure 19.** Diagram of results of multivariate analysis, showing the relative similarity of 116 overstory plots, classified into four groups that are color-coded and named. Reference plots are marked with an asterisk. The length of the lines radiating outward from the center indicates how similar any two plots are to one another. See Table 16 and Figure 20 for summaries of each group.

The third group, *Conifer-small*, comprises 17% of all plots and is characterized by a relatively high number of smaller, younger conifers per acre averaging 10.9 inches diameter (27.7 cm) and making up 93% of stand basal area. These conifer stands are found only in DNR-managed watersheds.

The fourth group, *Hardwood-dominant*, comprises 13% of all plots and is characterized by a much higher hardwood component than the other three groups (57% of basal area). Of the hardwoods in this group, 97% are red alder. Fourteen of the 17 plots in this group occur on DNR-managed watersheds, with three exceptions were reference plots all located on Queets River tributaries.

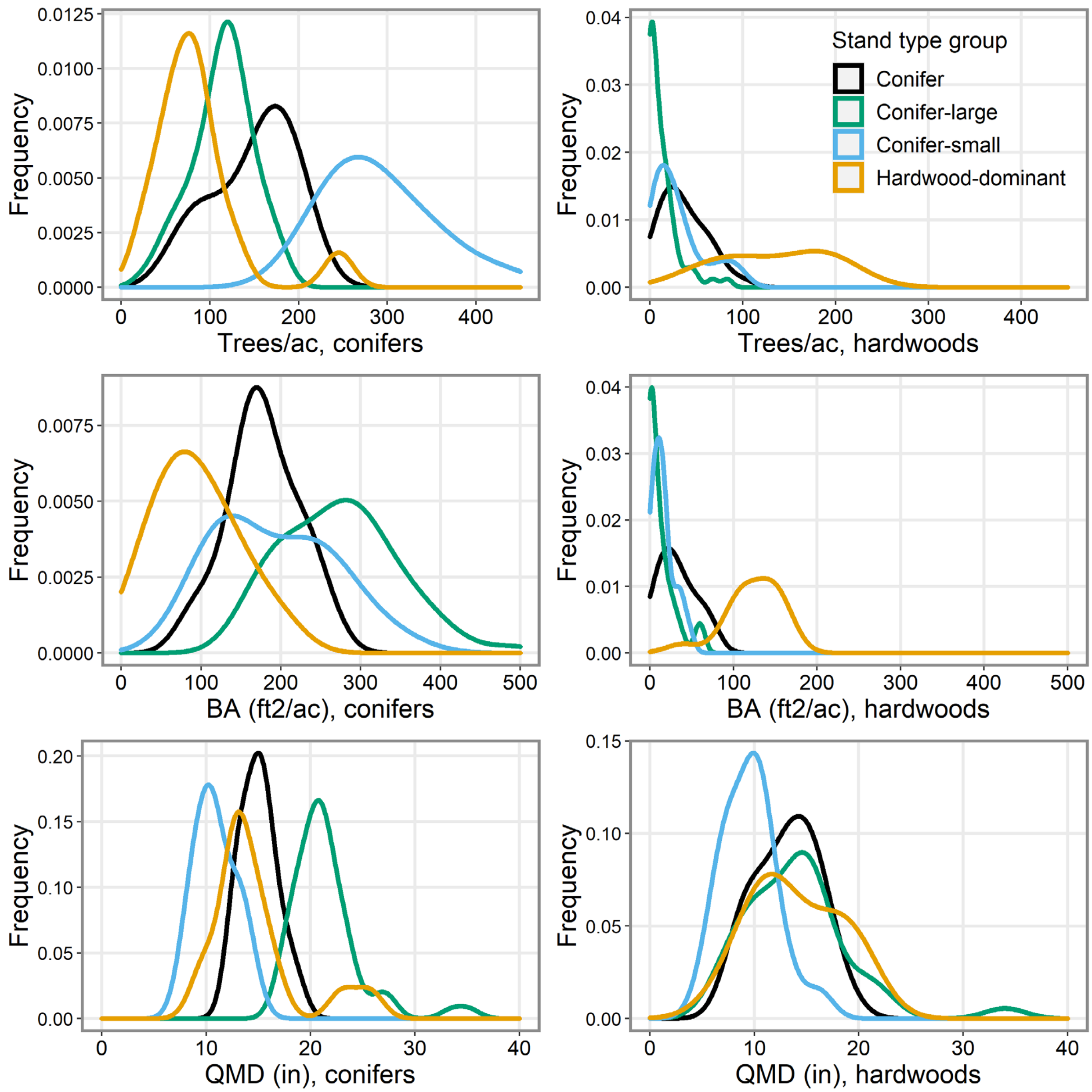



**Figure 20.** Frequency distributions of measurement plots in four stand type groups, for three metrics for conifers and hardwoods. These graphs illustrate differences among the four groups. For example, in the graph showing trees/ac for conifers, most plots in the hardwood-dominant group had fewer than 150 trees/ac, whereas in the conifer-small group, most plots had greater than 150 trees/ac. Note that the QMD graphs do not show diameter distributions within a plot but rather show the distribution of plot mean diameters.
BA=basal area; QMD=quadratic mean diameter.

**Table 16.** Characteristics of four forest overstory groups, derived from 116 riparian plots extending 200 ft (60 m) from the stream. Values are for live trees ≥5 in. (12.5 cm) diameter at breast height.

| Group name | Conifer mean diameter, in. (cm) | Trees/ac[1] (trees/ha) | | Basal area, ft²/ac (m²/ha) | | Number of plots | |
|---|---|---|---|---|---|---|---|
| | | Conifer | Hardwood | Conifer | Hardwood | DNR-managed | Reference[1] |
| Conifer | 15.0 (38.1) | 149 (368) | 35 (86) | 178 (40.9) | 32 (7.3) | 22 | 2 |
| Conifer-large | 21.4 (54.4) | 114 (282) | 12 (30) | 274 (62.9) | 12 (2.8) | 31 | 24 |
| Conifer-small | 10.9 (27.7) | 291 (719) | 30 (74) | 192 (44.1) | 15 (3.4) | 20 | 0 |
| Hardwood-dominant | 14.7 (37.3) | 85 (210) | 135 (334) | 95 (21.8) | 124 (28.5) | 14 | 3 |

[1] In addition to the 12 unharvested watersheds, there were three DNR-managed watersheds in which reach outlet—and the sample reach—was just outside of state lands (watersheds 796, 797, 820), within Olympic National Park. Because the overstory plots for these reaches were located in the park, we group them with the reference watersheds in the context of this riparian vegetation analysis.

### *Distance from stream*

With plots extending 200 feet (60 m) from the stream into the forest, we expected to capture a transition in overstory composition. We compared values for forest overstory metrics among the three 66-ft (20-m) zones to assess whether the forest composition changed with distance from stream.

Among the four groups, patterns emerged associated with species class (Figure 21). For all four groups, hardwood TPA and BA declined with distance from stream. This undoubtedly is a result of red alder's prevalence near streams where it is highly competitive in establishing following disturbance associated with a stream, whether the disturbance is a channel migration, debris flow, or flooding. By contrast, conifer TPA and BA increased or remained similar with distance from stream for all four groups.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

The riparian forests bordering Type 3 streams represent a wide range in forest overstory conditions resulting from differences in disturbance history and—though beyond the scope of this analysis—site conditions. The four groups identified in this analysis appear to represent different disturbances histories, or different ages since the last disturbance. Nearly 84% of plots on DNR-managed lands were in one of the three conifer groups. These groups represent three stages of stand development, ranging from closely spaced smaller trees in the *conifer-small* group to the largest conifers in the *conifer-large* group. These groups are characterized by the dominance of conifers and scarcity of red alder, except in zone 1 (i.e., close to the stream).

The *hardwood-dominant* group represents riparian forests that naturally regenerated in predominantly red alder following disturbance. This disturbance may have been a harvest followed by a failed conifer plantation, or it may have been a disturbance such as flooding, the migration of a stream channel, a landslide or a debris flow.

The *conifer-small* and *conifer* groups occur primarily in DNR-managed watersheds and together represent 61% of the plots in DNR-managed watersheds. These groups appear to reflect a history of

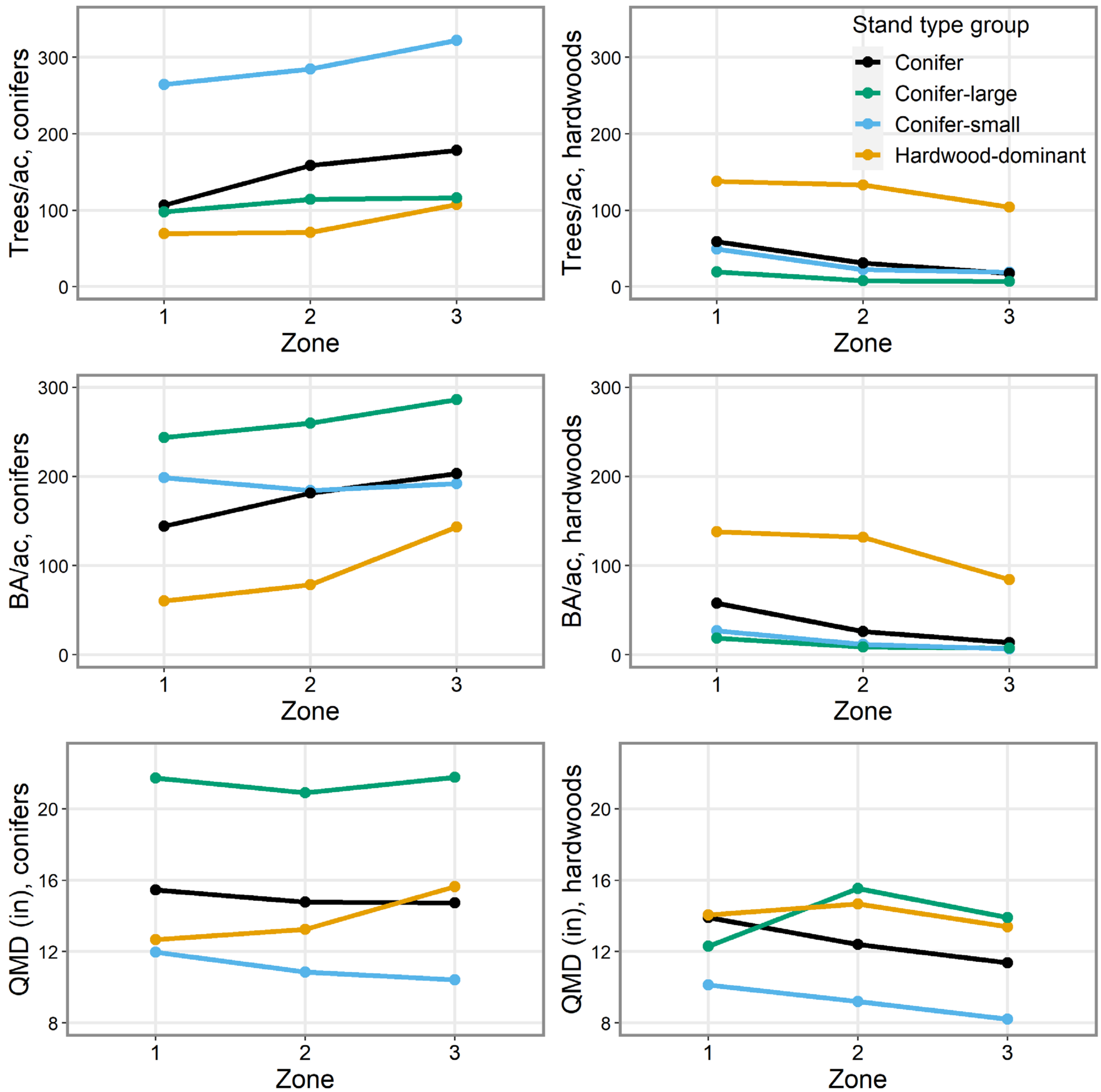



**Figure 21.** Mean values for three stand metrics for conifers (left) and hardwoods (right) by zone. Zone 1 is 0-66 ft (0-20 m) from the stream, zone 2 is 66-133 ft (20-40 m) from the stream, and zone 3 is 133-200 ft (40-60 m) from the stream.
BA=basal area; QMD=quadratic mean diameter.

forest management in which conifers were planted or naturally established within the riparian zone following harvest. In contrast, the *conifer-large* group is the predominant condition in the reference watersheds and therefore must reflect the outcome of natural disturbance and succession in the absence of harvest.

A key finding is that 36% of plots in DNR-managed watersheds are in the *conifer-large* group. This does not mean that harvest has been absent from all of these plots, but it does indicate that their stand structure, based on the metrics analyzed, is not different from that of the 83% of plots in reference watersheds that are also in the *conifer-large* group.

The three groups other than *conifer-large* make up a combined 64% of DNR-managed plots and represent comparatively younger forest conditions. A next step in this research is to use the riparian plot data to relate stand condition (and apparent history) to the riparian forest functionality targeted in the OESF Riparian Conservation Strategy. Important functions of the riparian forest include contribution of large woody debris to the forest floor and to streams as well as providing shade to streams (DNR 2016a). Such an analysis could indicate where different watersheds fall on the path toward the target range of conditions. It could also indicate which watersheds would benefit most from activities such as riparian thinning.

Information on OESF riparian forests collected through this monitoring program characterizes the condition of the riparian forest managed under the 1997 HCP. Future, more-detailed analyses of the riparian and stream datasets will assist in characterizing the riparian functions of these forests. The management of riparian forests is at the heart of the OESF Riparian Conservation Strategy (DNR 2016a, p. 3-22). Therefore, documenting the diversity of conditions, the developmental trajectories of riparian forest stands, and the relationship to the stream conditions is necessary to gauge the effectiveness of the strategy.

## Stream shade

### ECOLOGICAL CONTEXT

In a forested landscape, the riparian forest canopy is the most important factor driving riparian microclimate and stream water temperature, which in turn affect riparian and aquatic habitat conditions (Brown 1969; Brown and Krygier 1970; Beschta et al. 1987). The canopy directly affects the amount of sunlight reaching the stream; sunlight provides energy for the stream food web and influences stream productivity (Hill et al. 1995; Kiffney et al. 2003). In forested areas, aquatic and riparian species are adapted to the range of stream conditions that result from the level of riparian canopy shade characteristic of that ecosystem (Warren et al. 2016; Kaylor and Warren 2017). Removal of the riparian forest canopy through timber harvest without stream buffers can lead to large increases in stream temperatures, which are potentially harmful to locally adapted species, including salmonids (Brown and Krygier 1970; Beschta et al. 1987; Caissie 2006). For this and other reasons, maintaining stream shade is one of the measurable objectives outlined in the OESF Forest Land Plan (DNR 2016a, p. 3-22) to meet the conservation objectives of the OESF Riparian Conservation Strategy (DNR 1997, p. IV-106).

Among the nine indicators in Status and Trends Monitoring, stream shade is one of the indicators most directly affected by riparian forest management. The links between forest harvest, stream shade, and stream temperature have been the subject of a significant amount of research since the 1960s (Moore et al. 2005). By the 1980s, unharvested stream buffers had shown potential to alleviate much of the stream warming previously associated with clearcutting without riparian conservation measures (Beschta et al. 1987).

As a result of the stream buffers implemented under the HCP (DNR 1997), and to a lesser extent those implemented prior to the HCP,[25] the riparian forests within the buffers have developed over time, and even the youngest of these have entered the stem-exclusion phase of stand development (Oliver 1981). At this stage, the overstory trees are competing intensely for light, and as a result, little direct sunlight reaches the forest floor. Stream shade is assessed as part of Status and Trends Monitoring to better understand the shading that has resulted from past riparian management.

### SAMPLING

Shade beneath a forest canopy is typically measured in one of two ways: canopy closure or canopy cover (Jennings et al. 1999). Canopy closure is measured from individual points on the ground, looking up. Canopy cover is a vertical projection of the shade from the forest canopy onto the ground, and can be measured from above using LiDAR. In the present study, we chose to use canopy closure because we were interested in the amount of shading experienced by the stream (i.e., looking up).

Canopy closure was measured by hemispherical photography at six locations along each stream sample reach during summer when deciduous trees had leaves (Minkova and Foster 2017). At each location, the camera was mounted on a tripod at a height of 4.5 ft (1.37 m) above the center of the stream bed, with the lens oriented directly upward. Using a fish-eye lens, a photo was taken of the

[25] Stream buffer policies are described above in the Introduction section.

forest canopy at each location. This photo was then processed using Hemisfer software[26], and each pixel was identified as either shaded or unshaded. The sums of shaded and total pixel counts were used to calculate a percent canopy closure value for each photo (i.e., shaded pixels divided by total pixels, multiplied by 100). Finally, this value was converted to a percent shade value equivalent to the value one would get using a spherical densiometer (as described in Martens et al. 2019), which is a commonly used device for rapid canopy closure measurements.[27] Though hemispherical photography is less often used than densiometers, we chose to use photography because of its greater accuracy and repeatability and because the photos can be analyzed in a variety of ways for different objectives.

## RESULTS

Above-stream canopy closure averaged 93.7% for the DNR-managed watersheds (Table 17a). There was a small difference in canopy closure among channel types, with the pool-riffle type averaging higher canopy closure (95.5%) than the cascade type (91.7%) (Figure 22).

The significant year effect (Table 17b) indicated that canopy closure increased over time, also evident in the small increase between the first and last measurements (93.2% to 94.1%; Figure 22b). There was a negative effect of bankfull width on stream canopy closure: canopy closure tended to be higher for streams with smaller bankfull widths. Canopy closure in reference watersheds averaged 91.9%.

To measure the variability of canopy closure within a given reach, the standard deviation of the six canopy closure measurements from each sample reach was calculated for each year's photos. For the DNR-managed watersheds, the average standard deviation was 3.9%. For the reference watersheds the standard deviation averaged 5.8%. Among the 300 photo locations in the DNR-managed watersheds (50 watersheds with 6 photo locations each), only 7 had less than 70% canopy closure. Of the 72 photo locations in the reference watersheds, only one had less than 70% canopy closure.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

The consistently high degree of stream shading in the DNR-managed watersheds is not surprising, considering the current age of the riparian forests, which have not been completely harvested along Type 3 streams since the 1980s or earlier. Thus, even the youngest of these riparian buffers contain stands that were, at a minimum, 25 years of age when sampling began and are now in the stem-exclusion phase of stand development. During this phase, competition among trees for light is intense; as a result, very little light passes through the forest canopy reaching the forest floor or small streams. Because we observed a small but significant increase in canopy closure over time (from 2013 to 2020), competition for light among trees in the riparian overstory may still be increasing in at least some of the watersheds.

[26] Patrick Schleppi, Swiss Federal Institute for Forest, Snow and Landscape Research
[27] The field of view used in our analysis was 82.7° (equivalent to the field of view of a densiometer) rather than the full 180° of an uncropped hemispherical photo.

**Table 17a.** Canopy closure (%) for 50 DNR-managed Type 3 streams and for 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=107) | 93.8 | 5.0 | 67.1 | 99.5 | 92.3 – 95.1 |
| | | | | | |
| Pool-riffle channel type (n=31) | 95.5 [a] | 2.3 | 91.7 | 99.5 | 93.2 – 97.8 |
| Step-pool channel type (n=54) | 93.5 [ab] | 3.3 | 85.3 | 99.4 | 91.7 – 95.3 |
| Cascade channel type (n=22) | 91.7 [b] | 9.1 | 67.1 | 98.6 | 89.0 – 94.4 |
| | | | | | |
| First measurement (n=50) | 93.2 | 5.6 | 66.5 | 100.0 | 91.6 – 94.8 |
| Last measurement (n=50) | 94.1 | 5.2 | 67.7 | 100.0 | 92.6 – 95.6 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=15) | 91.9 | 3.9 | 87.0 | 97.7 | 89.7 – 94.1 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 17b.** Analysis of canopy closure (%) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 4.3 | 0.02 | Significant |
| Year | 12.2 | <0.01 | Significant |
| Channel type x Year interaction | 2.1 | 0.13 | No effect |
| Bankfull width (covariate) | 12.2 | <0.01 | Significant (negative) |

The difference in shade among channel types was influenced by the fact that one stream of the cascade type (watershed 690) was an outlier with much lower shade than the other streams (67%). Watershed 690 contained one of the largest monitored streams and there were few trees established close to its banks other that alder saplings. When we re-analyzed the data without this stream, there was no longer a significant difference in shade among channel types. The observed effect of bankfull width on stream shade is easier to interpret than channel type: wider streams had a greater distance between the trees on each streambank and thus the canopy over the stream was more sparse.

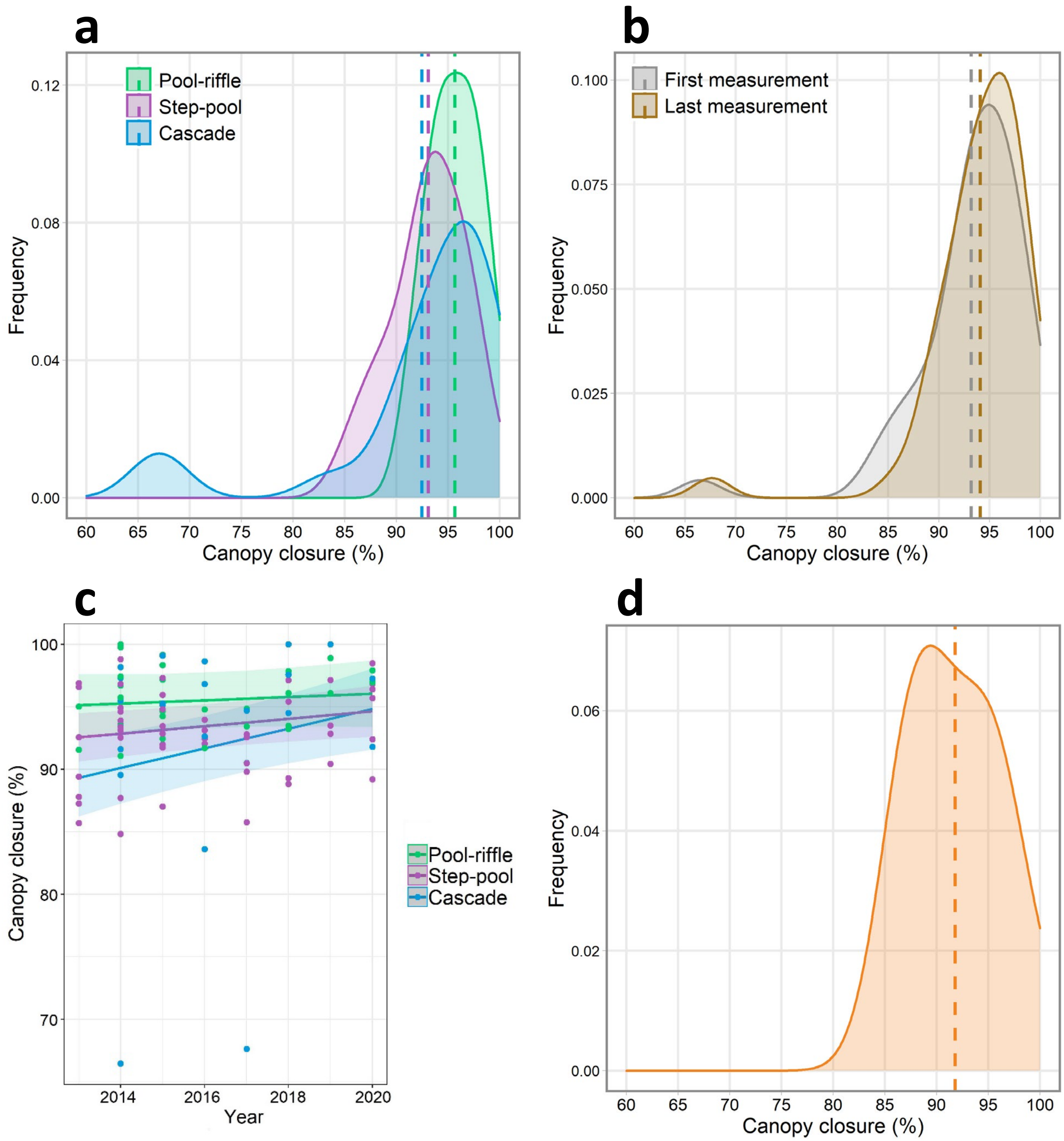



**Figure 22.** Distribution of DNR-managed watersheds according to percent canopy closure, by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type, with trendlines and shaded 95% confidence intervals (c); distribution of percent canopy closure for all reference watersheds (d). Vertical dashed lines indicate averages.

Our results agree with an earlier analysis of some of these same watersheds that concluded that the riparian forests in the stream buffers had reached a high level of canopy closure characteristic of the stem-exclusion phase (Martens et al. 2019, 2020). A meta-analysis of small forest streams across the Pacific Northwest showed that second-growth riparian forests as old as 100 years had greater shade than old growth riparian forests (age >300 years), owing to the more uniform canopy structure of the second-growth (Kaylor et al. 2017). Old growth forests were characterized by greater structural diversity which allowed more sunlight to reach streams, though streams in the old growth forests still averaged 82% canopy closure (Kaylor et al. 2017). Under the current riparian management strategy, we expect a high level of stream shade to persist for decades to come in the Type 3 streams of the OESF.

The level of canopy closure reported here strongly suggests that the stream buffers implemented as part of the OESF Riparian Management Strategy (DNR 2016a, p. 3-24) have been effective in providing stream shade. However, it should be noted that a direct comparison of this strategy's effects is not possible because that would require comparison of two groups of watersheds: one managed under HCP policies and one group still managed under pre-HCP policies.

The OESF Riparian Management Strategy addresses the problem of maintaining a well-shaded riparian environment that would keep streams cool in summer, in addition to many other habitat benefits. However, it is now apparent that many of the stream reaches are in a phase of maximum canopy closure and minimum light, which typically occurs between 20 and 60 years after stand initiation (Warren et al. 2016). When streams are so heavily shaded, the low amount of solar energy reaching the streams results in low overall biological productivity because growth of algae and other plants at the base of the aquatic food web is reduced (Hill et al. 1995; Kiffney et al. 2003).

Riparian thinning and gap creation have been proposed and tested as means of accelerating the development of riparian forest structure toward late-seral conditions (Berg 1995; Pollock and Beechie 2014; Benda et al. 2016), but riparian thinning can also be used to reduce stream shade by a desired amount (Roon et al. 2021). Exploring this effect on shade is a component of the active riparian restoration treatment in the T3 Watershed Experiment (Martens et al. 2021).

## Water temperature

### ECOLOGICAL CONTEXT

Stream temperature is of key importance for aquatic life, as it influences productivity, behavior, and life history of organisms. Water temperature affects plant life in addition to invertebrates and vertebrates such as fish and amphibians. Among salmonids, temperature can influence presence, health, emergence from eggs, juvenile growth, and migration timing. Each species has an optimal temperature range to which it is adapted, and in many cases water temperature has different influences on each life history stage.

Because land management activities, particularly in the vicinity of streams and other water bodies, have the potential to increase water temperatures, much research has been conducted on the causes of these increases and the effects that elevated water temperatures have on fish and other organisms (Caissie 2006). The single greatest forest management effect on stream temperature is when streamside vegetation is removed, causing an increase in direct sunlight reaching the stream (Beschta 1987). This effect has been one of the primary motivations for developing BMPs that include unharvested stream buffers.

Federal and state agencies including the U.S. Environmental Protection Agency (EPA) and Washington State Department of Ecology (WDOE) have established regulatory thresholds for maximum stream temperatures where human actions are involved in increasing water temperatures (see WAC 173-201A-200 for salmonid thresholds). The stream temperature metric most widely used for regulatory thresholds in the U.S. is the maximum 7-day average of the daily maximum temperature (7-DADmax). This is the average of peak daily temperatures during the 7-day period of the year when those peak temperatures have the highest average. The threshold 7-DADmax temperature relevant to the 50 monitored DNR-watersheds is the 16.0 °C (61 °F)[28] threshold applied to core summer salmonid habitat (Table 200 (1)(c) in WAC 173-201A-200). Although there are many other stream temperature metrics that can be calculated from the Status and Trends Monitoring data (e.g., six were reported in Devine et al. 2021), we report 7-DADmax here because it is the most commonly used metric when human impacts on streams are of concern.

### SAMPLING

Stream water temperature is measured year-round in each of the sample reaches using small dataloggers (TidbitT® v2 UTBI-001, Onset Computer Corp., Bourne, MA) that are anchored to tree roots, boulders or large, stable pieces of wood (Minkova and Foster 2017). These dataloggers measure and record temperature each hour year-round, and the dataloggers are then downloaded by field personnel once or twice per year. Because water levels drop in summer and the stream channel may change shape as a result of high-flow events, it is vital to ensure the loggers stay submerged in the water and only data from submerged loggers are analyzed. To verify that the logger anchored in the stream was continually submerged, a second datalogger is installed at each site, on a nearby tree, recording air temperature year-round. The temperature records from the paired air and water dataloggers are then compared, and for any period of time when temperatures are alike, it is probable

[28] Temperature is expressed in Celsius rather than Fahrenheit in this report because the state of Washington's habitat thresholds use Celsius.

that the water datalogger was out of the water. Any such data are excluded from analysis as part of a thorough quality control process performed on all data (Minkova and Foster 2017). The number of watersheds per year with a complete summer water temperature data record having passed the quality control inspection is listed in Table 18. The 7-DADmax metric was calculated for each of these watershed × year combinations.

**Table 18.** Sample size for stream temperature analysis: number of watersheds with complete summer data.

| Year | DNR-managed watersheds | Reference watersheds |
|---|---|---|
| 2013 | 38 | 1 |
| 2014 | 37 | 3 |
| 2015 | 37 | 4 |
| 2016 | 40 | 3 |
| 2017 | 46 | 3 |
| 2018 | 45 | 5 |
| 2019 | 42 | 9 |
| 2020 | 44 | 10 |
| *Total* | 329 | 38 |

### *Analysis*

Analysis of stream temperature in a managed forest landscape must include both the natural and human-caused factors that are likely to influence stream temperature. Because a substantial amount of research on stream temperature has been published over the past century, we combined that knowledge with our own experience in the OESF to select predictors that we believe are most likely to explain variation in stream temperature across the sampled streams.

In analyzing stream temperature, change over time was treated differently from the other analyses in this report. This is because stream temperature is known to be heavily influenced by the weather in any given year, especially air temperature. In turn, weather in a particular year is influenced by cyclical global climate patterns, such as the El Niño-Southern Oscillation (ENSO). Because the dataset is relatively short in duration (2013-2020), any trend in temperature during that period is likely to reflect these short-term climate patterns rather than a long-term (i.e., multi-decade) trend.

In the analysis of stream temperature (i.e., 7-DADmax), we tested several predictors selected to reflect both natural processes and potential effects of forest management (data in Appendices C, D, and E). The following predictors were used:

<u>Channel type:</u> this variable accounts for differences in stream temperature associated with the three different channel types: pool-riffle, step-pool, and cascade.

<u>Year:</u> this variable accounts for variation in temperature among years. In contrast to all of the other models in this report that use year to look for a positive or negative long-term trend, here the year variable accounts for natural year-to-year variation in temperature, though not necessarily as a trend in one direction.

<u>Bankfull width</u>: this variable was added to explain temperature variation associated with stream size. Stream temperature in summer generally warms with increasing stream size (Caissie 2006), though we observed the reverse pattern during winter (Devine et al. 2021). Bankfull width is strongly correlated to watershed area ($r$=0.85), so an influence of bankfull width is statistically similar to an influence of watershed area.

Elevation of sample reach: this variable was included based on the hypothesis that higher-elevation streams would be cooler than lower-elevation streams.

Stream shade: this variable explains variation in stream temperature that is associated with the amount of shade over the sample reach. Stream shade was measured by taking hemispherical photos in summer and then converting the results to a percentage that is equivalent to a densiometer measurement. Stream shade reflects forest management history: in some of the 50 DNR-managed watersheds, some or all of the riparian forest was harvested prior to when riparian buffers began to be used in the late 1980s. The amount of shade now present at these streams reflects the age of the riparian forest that established following those pre-buffer harvests.

Watershed solar exposure: this variable is a measure of the degree to which a watershed is exposed to direct sunlight during summer, calculated from a digital elevation model in GIS (Appendix Table D-3). Watershed solar exposure is strongly affected by aspect, but also by slope and shade from nearby topography.

Bedrock in the streambed: previous studies have shown that where the streambed consists of bedrock instead of alluvial deposits (gravel, cobbles, sand, etc.), stream temperature is more variable and sensitive to changes in weather (Johnson 2004; Hunter and Quinn 2009). Among the sampled watersheds, the percentage of the streambed composed of bedrock—measured during our channel substrate surveys—ranged from 0 to 37%.

Percent unharvested forest: this is a measure of what proportion of the watershed has never been harvested (see Appendix D for data and calculation method). Unharvested stands can vary in origin date and structure resulting in stand development stages ranging from simple to complex. This variable was analyzed because a previous study reported that watersheds with a higher proportion of unharvested forest had cooler summer stream temperatures (Pollock et al. 2009).

## RESULTS

### *Average 7-DADmax*

Overall, 7-DADmax averaged 14.4 °C (57.9 °F) across the DNR-managed watersheds (Table 19a; Figure 23). Among the three channel types, mean 7-DADmax varied by only 0.5°C (0.9 °F). Among the reference watersheds, 7-DADmax averaged 15.0 °C (59.0 °F). This slightly warmer temperatures in the reference watersheds may have been influenced by higher average solar exposure of those watersheds (discussed below). That higher level of solar exposure is a random occurrence, as the small sample size of 12 reference watersheds happened to include a high percentage of south-facing watersheds.

For the DNR-managed watersheds, there were 329 observations of 7-DADmax in the analysis; of these, 16, or slightly less than 5% of observations exceeded the 16.0 °C (60.8 °F) regulatory threshold for core summer salmonid habitat under impaired conditions (WAC 173-201A-200). Fewer data points were available for the reference watersheds because monitoring did not begin in many of them until 2017 or 2018. Thus, that dataset had only 38 values for 7-DADmax. Of these, 12, or 32%, exceeded the 16.0 °C (60.8 °F) regulatory threshold.

**Table 19a.** Maximum 7-day average daily maximum water temperature (°C) (7-DADmax) in 50 DNR-managed Type 3 streams and in 12 reference streams. SD = standard deviation.

| Group | Sample summary | | | | Population mean 95% conf. interval |
|---|---|---|---|---|---|
| | Mean | SD | Min. | Max. | |
| *DNR-managed watersheds* | | | | | |
| All DNR-managed watersheds (n=329) | 14.4 | 1.0 | 11.6 | 16.4 | 14.1 – 14.6 |
| | | | | | |
| Pool-riffle channel type (n=99) | 14.2 [a] | 0.9 | 11.6 | 15.0 | 13.7 – 14.7 |
| Step-pool channel type (n=160) | 14.6 [a] | 1.0 | 11.7 | 16.4 | 14.2 – 15.0 |
| Cascade channel type (n=70) | 14.1 [a] | 1.0 | 13.0 | 16.0 | 13.5 – 14.7 |
| | | | | | |
| First measurement (n=50) | 14.2 | 1.0 | 10.9 | 16.4 | 13.9 – 14.5 |
| Last measurement (n=50) | 14.1 | 1.0 | 11.5 | 16.5 | 13.8 – 14.4 |
| *Reference watersheds* | | | | | |
| All reference watersheds (n=38) | 15.0 | 1.1 | 13.6 | 16.7 | 14.3 – 15.6 |

Note: channel type means followed by the same letter do not differ significantly.

**Table 19b.** Analysis of maximum 7-day average daily maximum water temperature (°C) in 50 Type 3 streams on DNR-managed land in the OESF.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 0.3 | 0.71 | No effect |
| Year | 23.9 | <0.01 | Significant |
| Bankfull width (covariate) | 2.2 | 0.15 | No effect |
| Elevation of sample reach (covariate) | 4.0 | 0.05 | Marginally significant (negative) |
| Stream shade (covariate) | 3.1 | 0.09 | No effect |
| Watershed solar exposure (covariate) | 10.2 | <0.01 | Significant (positive) |
| Bedrock substrate (covariate) | 11.6 | <0.01 | Significant (positive) |
| Percent unharvested forest (covariate) | 1.0 | 0.32 | No effect |

### *Factors influencing 7-DADmax in DNR-managed watersheds*

Among the natural factors tested as predictors of stream temperature, year, watershed solar exposure, and bedrock substrate were all found to influence 7-DADmax (Table 19b). Average 7-DADmax varied among years, ranging from a low of 14.1 °C (57.4 °F) in 2013 and 2019 to a high of 14.9 °C (58.8 °F) in 2015.

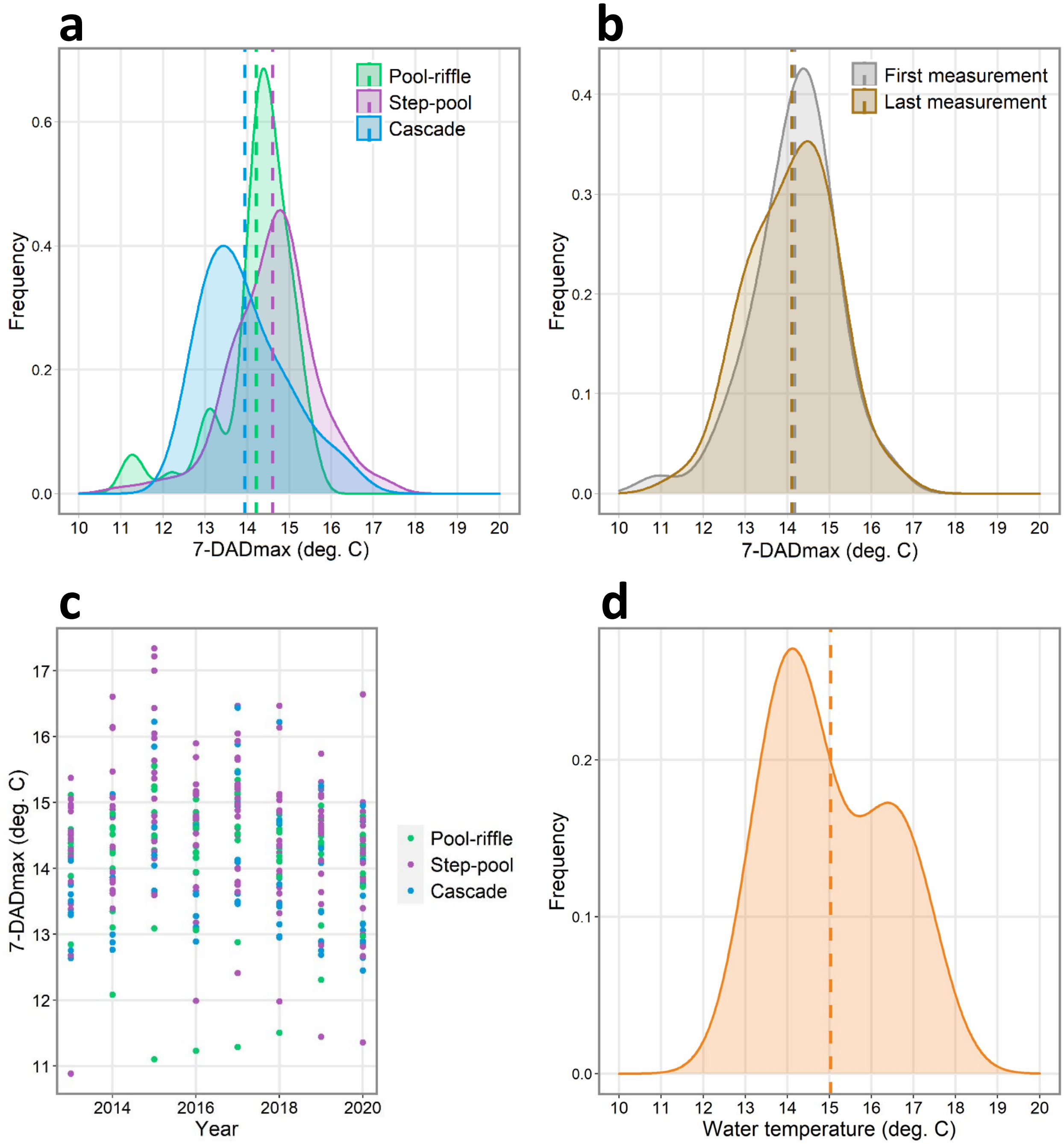



**Figure 23.** Distribution of DNR-managed watersheds according to maximum 7-day average daily maximum water temperature (7-DADmax), by channel type (a), and at the first and last measurement (b); values for each watershed plotted over time by channel type (c); distribution of 7-DADmax for all reference watersheds (d). Vertical dashed lines indicate averages.

Watershed solar exposure was positively correlated with 7-DADmax. This positive correlation was expected, as the amount of solar exposure was previously shown to be correlated with warmer riparian air temperatures (Keleher 2019). The 12 reference watersheds had an average watershed solar exposure that was higher than the DNR-managed watersheds (4.90 ± 0.22 vs. 4.79 ± 0.32

kWh/m$^2$); this may have contributed to their slightly warmer water temperatures and to their exceeding the regulatory threshold more frequently. The greater solar exposure of the reference watersheds was a random occurrence. With a relatively small sample size it is more likely that a sample's average will deviate from the true population average; thus, compared to the sample of 12 reference watersheds, the 50 DNR-managed watersheds are expected to provide a better representation of the average solar exposure of Type 3 watersheds across the OESF.

The percentage of bedrock in the streambed had a positive correlation with 7-DADmax. Although exposed bedrock in the streambed was relatively infrequent, averaging only 3% of the streambed across the monitored streams (range = 0 to 37%), its presence was clearly related to higher 7-DADmax temperatures. Previous studies in the Pacific Northwest, one on the Olympic Peninsula, documented the link between streambed substrate and stream temperature (Johnson 2004; Hunter and Quinn 2009). These studies showed that streams flowing over bedrock were much more sensitive to changes in weather than streams with streambed substrate composed of alluvial deposits such as gravel, sand, and cobles. This difference is apparently because streams flowing over alluvial deposits had subsurface flow as well as surface slow. The subsurface flow mixes with the surface flow and acts as a buffer, reducing the stream's sensitivity to the atmosphere. Streams flowing over bedrock do not have this buffer and are thus more likely to become hotter on hot summer days (Johnson 2004; Dent et al. 2008).

Elevation had a marginally significant negative correlation with 7-DADmax, indicating that stream temperatures were cooler at higher elevations, an anticipated pattern. The remaining predictors in the model—channel type, stream shade, and percent unharvested forest—did not significantly affect 7-DADmax. The lack of a stream shade effect on 7-DADmax was somewhat surprising, but we attribute it to the fact that stream shade was consistently high across the 50 DNR-managed sample reaches. With very little variation in the level of stream shade, we were unlikely to observe a relationship between stream shade and stream temperature. The same phenomenon occurred during analysis of Oregon headwater streams (Dent et al. 2008). With the exception of one reach that had 67% shade, all of the reaches in DNR-managed watersheds fell within a range of 85% to 100% shade. The reach with 67% shade was one of the larger Type 3 streams sampled (watershed 690), with a wide channel scoured by winter flows powerful enough to transport old-growth logs. Without enough examples of streams with low levels of shade, it is not possible to determine whether such streams have higher water temperatures.

### *Harvest history and 7-DADmax*

Percent unharvested forest in a watershed was included in our model because an earlier study of OESF streams concluded that percent unharvested forest was a strong predictor of summer stream temperatures (Pollock et al. 2009). That study reported that watersheds all or mostly unharvested had the coolest maximum daily stream temperatures in summer, whereas watersheds primarily in second growth had the warmest streams. But in our sample of 50 DNR-managed watersheds, we found no evidence for this trend, as percent unharvested forest did not predict 7-DADmax (Table 19b). One possible explanation for the difference between our findings and those Pollock et al. (2009) is that the latter collected data in 2004, whereas we collected data from 2013-2020. Thus, the riparian forests in DNR-managed watersheds were 9-16 years older during our monitoring period and may have provided a greater degree of shading to streams, thus reducing summer stream temperatures.

To further investigate this difference between the present study and that of Pollock et al. (2009), we addressed potential differences in sample site selection between the studies. Though the streams in both studies were similar in size and generally similar in geographic location, Pollock et al. (2009) analyzed 40 watersheds of which 18% had never been harvested. Because our analysis includes 50 DNR-managed watersheds, all of which had at least some history of harvest, we decided to re-run our stream temperature analysis using a dataset that combined our 50 DNR-managed watersheds with the 12 reference watersheds to create a sample of 62, 19% of which had never been harvested.[29] The resulting range of harvest histories in our combined sample of 62 watersheds is very similar to that of the dataset used by Pollock et al. (2009).[30] We analyzed this combined sample of 62 watersheds using the same statistical model that we used for the 50 watersheds. Because the combined sample was larger and included a wider range of watershed conditions, we anticipated that the results might differ slightly from the analysis of 50 DNR-managed watersheds (Table 20).

**Table 20.** Analysis of maximum 7-day average daily maximum water temperature (°C) in 62 Type 3 streams (50 on DNR-managed land in the OESF and 12 reference watersheds). Data are from 2013-2020.

| Effect | F value | *p*-value | Interpretation |
|---|---|---|---|
| Channel type | 0.1 | 0.93 | No effect |
| Year | 28.0 | <0.01 | Significant |
| Bankfull width (covariate) | 2.0 | 0.16 | No effect |
| Elevation of sample reach (covariate) | 6.1 | 0.02 | Significant (negative)[1] |
| Stream shade (covariate) | 2.7 | 0.11 | No effect |
| Watershed solar exposure (covariate) | 11.5 | <0.01 | Significant (positive) |
| Bedrock substrate (covariate) | 14.2 | <0.01 | Significant (positive) |
| Percent unharvested forest (covariate) | 6.4 | 0.01 | Significant (positive)[1] |

[1] The significance of this effect—with this combined dataset—is different from our analysis of the 50 DNR-managed watersheds presented in Table 19b.

In analyzing the combined dataset of 62 watersheds, we found that two predictor variables not significant in the earlier analysis were now statistically significant (compare Tables 19b, 20). The first of these variables was sample reach elevation: higher elevation was correlated with cooler 7-DADmax (Table 20). This relationship is not surprising and also was found to occur during winter for these streams (Devine et al. 2021). But it is worth noting that the 5 highest-elevation streams among the sample of 62 were all DNR-managed watersheds rather than reference watersheds.

[29] In this report, we normally do not combine data from the 12 reference watersheds with data from the 50 DNR-managed watersheds because our primary goal is to understand conditions in DNR-managed watersheds. Here, the goal differed somewhat, and an exception was made to create a dataset comparable with that of Pollock et al. (2009).
[30] Although the 12 unharvested Status and Trends watersheds were not selected randomly from a larger pool of watersheds, it should be noted that the 40 watersheds in Pollock et al. (2009) also were not selected randomly from a larger pool.

The second predictor variable now significant using the combined dataset was percent unharvested forest. However, in contrast to the results of Pollock et al. (2009), percent unharvested forest was *positively* correlated with 7-DADmax. In other words, summer stream temperatures tended to be warmer in watersheds with higher percentages of unharvested forest. The reason for this relationship is not obvious, given that the relationship was present after accounting for variation in stream shade, elevation, solar exposure, and bankfull width. Therefore, the reason for the unexpected trend must be another factor that was not in our model. At present it is difficult to explain the pattern with any certainty. What is clear is that, in the OESF, streams draining watersheds dominated by unharvested forest were not cooler than streams draining DNR-managed watersheds containing a mosaic of forest age cohorts. Anecdotally, two of the monitored watersheds with the least human disturbance, tributaries of the Bogachiel and Queets Rivers in Olympic National Park, contain two of the streams that have been among the warmest in our monitoring since 2013. But to put this into context, these streams only reached 7-DADmax temperatures between 16.5 and 17.5 °C (61.7 to 63.5 °F) in summer, temperatures unlikely to have negative effects on their fish populations (Carter 2005). It should also be noted that both of these watersheds have southerly aspects, which likely is at least partially responsible for their relatively warm stream temperatures.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

This analysis was performed to evaluate how selected variables—reported by other studies to have affected stream temperature—influence summer maximum stream temperatures in the OESF. After accounting for significant year-to-year temperature variation, natural landscape attributes clearly had the strongest influence on stream temperature. Watershed solar exposure and bedrock substrate were the strongest predictors of maximum temperature, though there also was evidence showing temperature was influenced by elevation, with higher-elevation streams having lower temperatures. No link was found between past watershed harvest and warmer temperatures; rather, we observed the opposite trend when all 62 watersheds were considered.

These findings indicate that three factors—watershed solar exposure, stream bedrock substrate, and elevation—can be used in the future to identify if a stream is likely to experience relatively high summer water temperatures. Of these, watershed solar exposure and bedrock substrate were the strongest predictors of summer high temperatures.

The consistently high degree of stream shading in DNR-managed watersheds is apparently a result of the stream buffers. The stream buffers, combined with the maritime climate, are likely the key reasons why the monitored streams remained relatively cool in summer. In landscapes where stream shade is more variable than the narrow range observed here—whether due to natural variation or the removal of riparian vegetation—reductions in stream shade are typically linked to warmer summer stream temperatures (Brown 1969, Johnson and Jones 2000, Johnson 2004, Roon et al. 2021).

# Riparian microclimate

## ECOLOGICAL CONTEXT

Riparian zones in temperate, forested ecosystems typically have a unique set of local climate conditions different from those of upland forests; these conditions are known as the riparian microclimate. Riparian microclimate—here we specifically focus on air temperature and humidity—affects many biological processes in plants, animals, fungi, and microorganisms. The Riparian Conservation Strategy in the state lands HCP (DNR 1997) aims to provide habitat not only for salmonids but also for riparian obligated species such as amphibians. Riparian microclimate is one of the indicators of functioning riparian habitat used in the OESF Forest Land Plan environmental impact analysis (DNR 2016b).

Riparian microclimate conditions are most distinct in summer, when the air in the riparian zone is typically cooler and more humid than in the uplands. Soils are characteristically wetter in the riparian zone as well. Many plant and animal species in the riparian forest are adapted to these cooler, moister conditions. For example, lungless salamanders in the family *Plethodontidae*, such as Western Red-backed and Van Dyke's salamanders, are common species in riparian areas of the OESF. They respire through their skin, and thus need cool and moist habitats to persist.

The influence of a stream on microclimate decreases with distance from the stream; this is known as a riparian microclimate gradient. In summer, air generally becomes warmer and drier with increasing distance from the stream. During wetter months, we don't expect a strong riparian microclimate gradient due to the cooler, wetter conditions across the landscape.

There are no regulatory standards or recommended targets for riparian air temperature and humidity, probably because microclimate conditions are highly site-specific. Riparian microclimate is understood to be affected by the forest canopy (Moore et al. 2005), so riparian buffers are expected to reduce the effect of adjacent regeneration harvests on the riparian microclimate. One of the few studies to measure the microclimate gradient within stream buffers found that, in western Oregon, the microclimate gradient from the stream to the edge of the 100-ft (30-m) forested buffers was similar to the gradient in unharvested stands (Rykken et al. 2007). The authors proposed that the lack of "edge effect" from the adjacent clearcut was a result of the stream's influence on microclimate in the riparian zone. Although the microclimate gradient within the buffer was not measured, a study in western Washington found that air temperature and relative humidity increased above streams in the summer immediately following harvest, despite the use of forested buffers ranging from 56 to 236 ft (17 to 72 m) in width (Brosofske et al. 1997; Dong et al. 1998).

By monitoring the riparian microclimate gradient, we are able to learn about the width of the riparian microclimate zone, the strength of the gradient, and the factors that influence riparian microclimate along Type 3 streams in DNR-managed watersheds. By understanding these factors, we can, in the future, create better models to predict how far the riparian microclimate extends from the stream.

Analysis of summer microclimate was conducted by Katrina Keleher, an Evergreen State College master's student, as a thesis research project (Keleher 2019). These findings, summarized here, are expected to be published soon in the journal Northwest Science (Keleher et al. 2022).

## SAMPLING AND ANALYSIS

Air temperature and humidity were monitored in 10 of the 50 DNR-managed watersheds in the OESF during calendar years 2014-2016.[31] In each of these watersheds, two sampling transects were installed on opposite banks of the sample reach, oriented perpendicular to the stream (Figure 24) (Minkova and Foster 2017). Each transect extended 200 feet (60 m) horizontal distance from the stream bank into the adjacent forest. Along each transect, five datalogger stations were situated at distances of 0, 32, 65, 130, and 200 feet (0, 10, 20, 40, and 60 m) from the transect origin, for a total of 10 stations per stream. Data loggers (2-channel HOBO® Pro v2, Onset Computer Corp., Bourne, MA), mounted 4.5 ft (1.3 m) above ground and protected from direct sunlight with a white plastic shield (Figure 25), recorded air temperature and relative humidity every two hours throughout the year.

Instead of analyzing relative humidity, air moisture data were analyzed as vapor pressure deficit (VPD). VPD measures the "drying power" of the air and therefore has a more direct biological relevance than relative humidity. As a frame of reference, VPD at night and in winter is usually close to 0 kilopascal (kPa) when the air is saturated with water vapor, but VPD reaches 1 kPa or higher during the afternoon of a warm summer day.

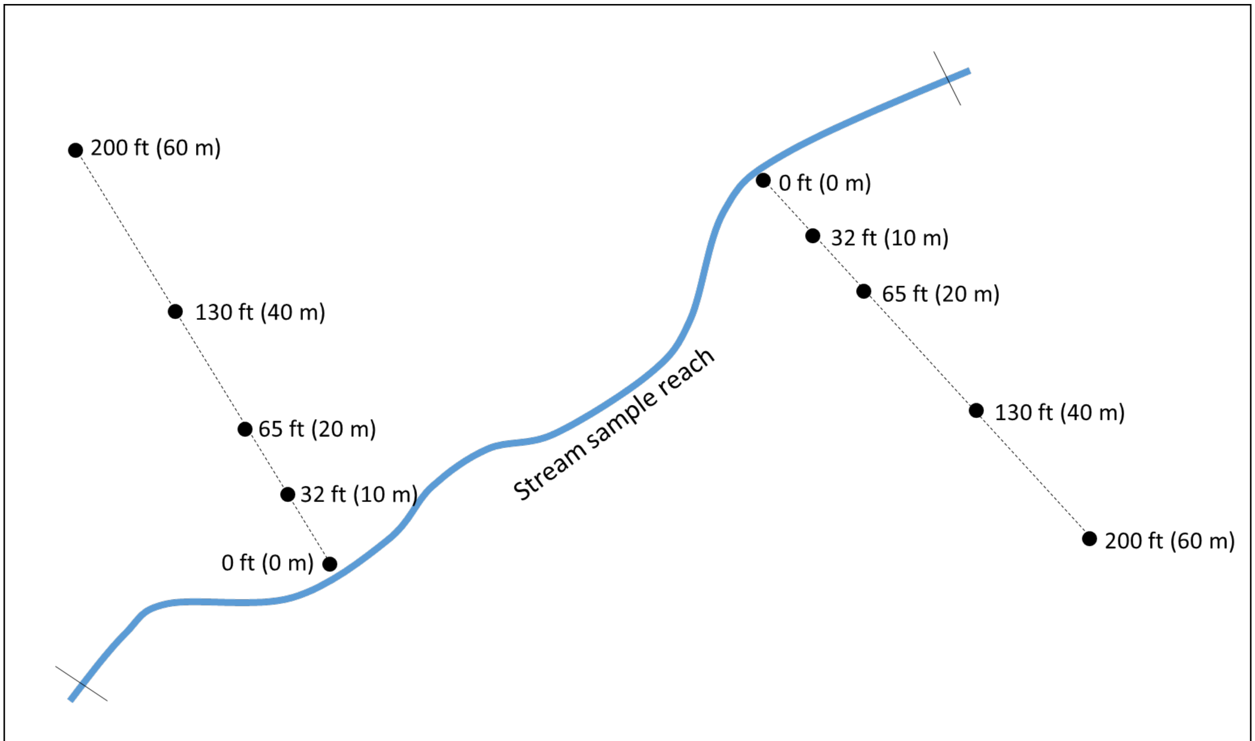



**Figure 24.** Layout of two microclimate transects along a sample reach; black dots are stations where air temperature and humidity are recorded.

[31] Because each sampled watershed required installation and maintenance of 10 data loggers, it was only feasible to sample in 10 of the 50 watersheds. The 10 sampled watersheds were selected to represent the full range of environmental conditions in the full group of 50.

**Figure 25.** A microclimate monitoring station, consisting of a data logger mounted on a fence post and shielded from direct sunlight (left), and three microclimate stations installed along a transect (right).

The microclimate variables selected for this analysis were summer (June – August) average daily maximum air temperature ($T_{max}$) and summer average daily maximum VPD ($VPD_{max}$).

The data analysis was conducted to identify which factors (here called "predictors") had significant influences on riparian microclimate. Based on the scientific literature, we selected six predictors that we hypothesized had an influence on riparian microclimate conditions. We did not test the influence of nearby forest harvest on microclimate because there were no recent or ongoing harvests in the vicinity of any of the 20 monitoring transects. The six predictors tested were:

<u>Distance from stream</u>: when designing the study, we established five datalogger stations per transect to assess the effect of distance from stream on the riparian microclimate gradient.

<u>Slope from stream</u>: among the ten stream valleys monitored, some had gradually sloping sides and some were steep-sided. Thus, some of our microclimate monitoring stations, particularly the ones at the ends of the transects farthest from the stream, were situated well above the stream. Using a hypsometer, we measured the height above the stream of each of the monitoring stations. Note that the height above the ground surface was constant for all monitoring stations, and so differences in height above stream are solely due to natural contours of the stream valleys where the stations were located. These variations in valley shape allowed us to analyze how slope from the stream up to the monitoring stations affected microclimate at the stations.

<u>Canopy shade</u>: at each of the 100 microclimate stations, we took a hemispherical photo with the camera oriented vertically. We analyzed the photos following the same method previously described for stream shade. The result was a canopy closure value (%) for each of the

microclimate stations. We used this canopy closure data to evaluate whether canopy closure directly above the microclimate monitoring stations influenced their microclimate.

Topographic solar exposure: in ArcGIS, we used the Area Solar Radiation tool to calculate solar radiation intensity at each of the study watersheds (Figure 26; methodology in Appendix Table D-3). We then extracted the radiation intensity (i.e., solar exposure) at each the locations of the 100 microclimate monitoring stations. Knowing the solar exposure at each of the monitoring stations, we then were able to test how local solar exposure influenced microclimate. The solar exposure calculation was strongly influenced by aspect but also by slope and by shading from nearby ridges or other topographic features. This calculated solar exposure value did not consider vegetation, only topography.

Elevation: higher elevation is generally expected to be associated with cooler air temperatures. Elevation was not of primary interest in the study, but it is a factor that should be considered when analyzing microclimate. The range of elevations for the 10 watersheds in which microclimate was monitored was 76 to 1,188 ft (28 to 362 m).

Distance from coast: due to the marine climate influence coming from the Pacific Ocean, we expected that watersheds located closer to the coast would have a cooler, moister microclimate. Distance from coast ranged from 6.6 to 19.9 miles (10.7 to 32.0 km) among the 10 watersheds. For these 10 watersheds, elevation and distance from coast were *not* correlated.

We evaluated whether each of these predictors influenced $T_{max}$ and $VPD_{max}$ over the course of three summers. This was done by creating a series of statistical models containing different predictors. These models were then compared to see which models best explained the observed variation in $T_{max}$ and $VPD_{max}$.

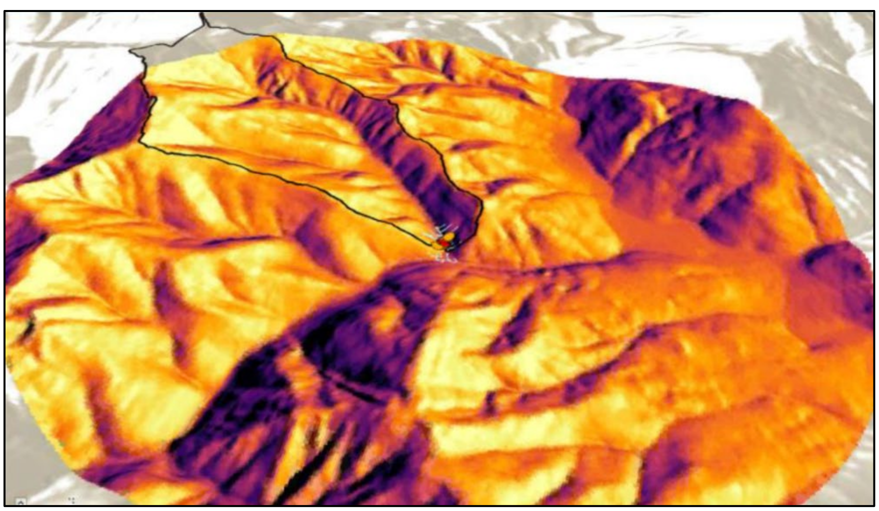


**Figure 26.** Example of calculated solar exposure across the landscape; the lightest colors indicate the highest intensity of solar radiation and the darkest colors indicate the lowest intensity. The black outline is the boundary of a study watershed.

## RESULTS

The results of the microclimate analysis tell us, for a given location on the ground within 200 ft (60 m) of a stream, which of the tested factors were significant predictors of the microclimate *at that location*. Four of the six predictors tested were statistically significant in explaining microclimate; two were not. The following four factors contributed to explaining riparian microclimate ($T_{max}$ and $VPD_{max}$) (Table 21):

Distance from stream: Air became both warmer and drier (increased $T_{max}$ and $VPD_{max}$) at greater distances from the stream (Figure 27). The largest increases in $T_{max}$ were between 0 and 32 ft from the stream and again between 130 and 200 feet. The largest increases in $VPD_{max}$ were between 0 and 65 feet from the stream.

Slope from stream: At any given horizontal distance from a stream (for example, 32 feet), the slope between that point and the stream affected microclimate. Where there was a steep slope up from the stream, air was warmer and drier than where the slope from the stream was gradual.

Solar exposure: Within the riparian zone, greater solar exposure was associated with warmer, drier air (increased $T_{max}$ and $VPD_{max}$). For example, on south facing valley slopes, microclimate was naturally warmer and drier than on north-facing slopes.

Elevation: Air was both warmer and drier (greater $T_{max}$ and $VPD_{max}$) at higher elevations.

**Table 21.** Influence of four significant predictors on summer average daily maximum air temperature ($T_{max}$) and vapor pressure deficit ($VPD_{max}$).

| Predictor | Effect on $T_{max}$ | Effect on $VPD_{max}$ |
|---|---|---|
| Distance from stream | $T_{max}$ increased with distance from stream at a rate of 0.4 °F (0.2 °C) per 100 ft (30.5 m). | $VPD_{max}$ increased with distance from stream at a rate of 0.004 kPa per 100 ft (30.5 m). |
| Slope from stream | $T_{max}$ increased as slope from stream increased, at a rate of 0.2 °F (0.1 °C) per 10% slope. | $VPD_{max}$ increased as slope from stream increased, at a rate of 0.04 kPa per 10% slope. |
| Solar exposure | $T_{max}$ increased with greater solar exposure. $T_{max}$ increased 1.1 °F (0.6 °C) from the least exposed to the most exposed monitoring station. | $VPD_{max}$ increased with greater solar exposure. $VPD_{max}$ increased by 0.14 kPa from the least exposed to the most exposed monitoring station. |
| Elevation | $T_{max}$ increased with elevation among the 10 sampled watersheds, rising 2.7 °F per 1,000 ft elevation (1.5 °C per 305 m). | $VPD_{max}$ increased with elevation among the 10 sampled watersheds, rising 0.3 kPa per 1,000 ft (305 m) elevation. |

The following factors did not significantly explain patterns in microclimate:

Distance from coast: The distance between a watershed and the coast was not correlated with its microclimate. This lack of effect may be at least partially due to a statistical phenomenon: the 10 watersheds were not selected to represent the distance-from-coast gradient and thus were not evenly distributed between the coast and the interior boundary of the OESF. It is important to note that, among the 10 watersheds in which microclimate was monitored, elevation and distance from coast were not correlated.

Canopy shade: As a predictor, canopy shade was not significant. Although the forest canopy undoubtedly had a strong influence microclimate, that influence was relatively uniform across the 100 microclimate monitoring stations because canopy closure had such a narrow range (87% to 98%). With such a narrow range in canopy conditions it is difficult to detect a statistical relationship with microclimate.

Distance from stream was expected to be a strong predictor of microclimate gradient. Distance from stream is typically used to delineate RMZs, both because it is convenient but also because riparian influences are understood to diminish within increased distances from a stream (Moore et al. 2005; Rykken et al. 2007). The distance-related patterns that we observed showed the steepest microclimate gradient close to the stream, but the gradient did not completely flatten at any distance along the transect (Figure 27). This indicates that the riparian gradient for $T_{max}$ and $VPD_{max}$ extended for at least 200 ft (60 m) from the stream during June-August. Because we accounted for factors such as

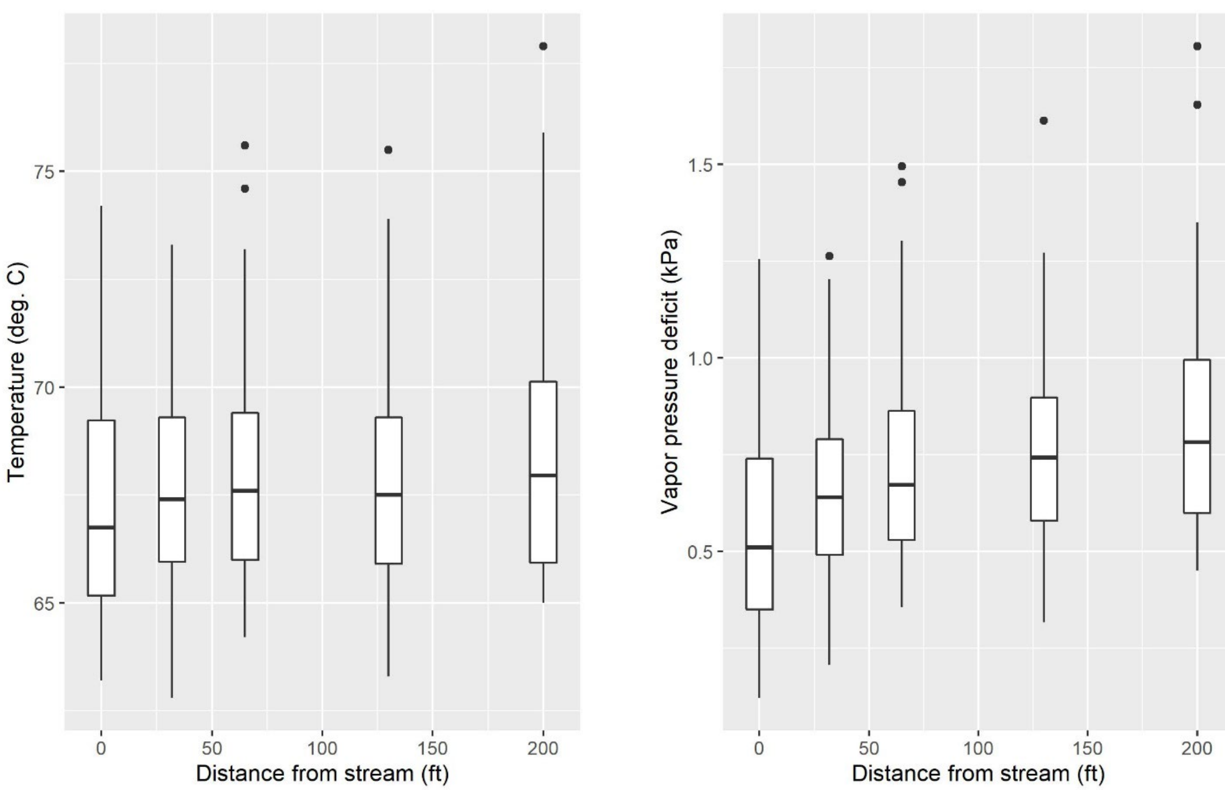



**Figure 27.** Boxplots of summer maximum daily air temperature (left) and maximum daily vapor pressure deficit (right), by distance from stream. The horizontal line at the center of the box is the median; the box represents the interquartile range; the whiskers extend from the interquartile range to the most extreme data point falling within a distance of 1.5 times the interquartile range beyond the box (i.e., beyond the interquartile range); plotted points are values that fell outside of the whisker.

slope, solar exposure, and shade, this distance effect may be largely a result of cooling from the moist conditions near the stream (i.e., the “stream effect”, see Rykken et al. 2007).

An important finding here was that stream valley slope also had a strong effect of microclimate. Therefore, in a steep-sided stream valley, we would expect the riparian microclimate influence to decrease more rapidly with distance (in a direction perpendicular to the stream) than it would on flatter topography. Streams in flatter topography are expected to have a microclimate gradient that is not as clearly defined.

The influence of topographic solar exposure on riparian microclimate was not surprising, as aspect—a major determinant of solar exposure—has a significant influence on a site’s vegetation and soil moisture. The amount of solar exposure can differ greatly between the two banks of a stream, particularly if that stream is running east-west, with the northern bank having a more southerly exposure. For this reason, the riparian microclimate gradient can differ between two sides of the same stream.

An effect of elevation on microclimate was expected, but the direction of the effect was not expected. Among the 10 sampled watersheds, the highest in elevation had the warmest and driest riparian microclimates. This is likely because these watersheds are high enough to receive less of the marine climate influence than lower-elevation watersheds. Elevation appears to be a better predictor of this influence than distance from coast. Though not intuitive, elevation and distance from coast were not correlated for the 10 sampled watersheds. This is because some watersheds nearer the coast happened to be at higher elevations and some further from the coast were at lower elevations.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

As a result of this microclimate monitoring effort, uncertainties raised during the OESF EIS analysis regarding riparian microclimate gradients in the OESF have been significantly reduced (DNR 2016b). Future modeling of riparian microclimate in the OESF can now incorporate the data and findings from this Status and Trends Monitoring work. For example, riparian microclimate gradient models in the OESF Forest Land Plan EIS analysis relied on distance from stream as the only predictor (DNR 2016b). We now know that slope from stream is also an important predictor and that solar exposure is a similarly strong predictor that should be incorporated into future models of riparian microclimate in the OESF. Slope above stream, elevation, and solar exposure are advantageous in that they predict microclimate gradients based on local topography rather than only on distance from a stream. As a result, landscape-level models of microclimate that incorporate these variables are expected to offer improved accuracy.

## Flow extremes

### ECOLOGICAL CONTEXT

Over a period of hours, stream flow in the OESF can vary from a trickle between cobbles to a torrent that moves the cobbles. This extreme flow variability is a natural phenomenon typical of small, mountainous stream channels during heavy rainfall. Aquatic life has evolved to survive these extremes. Yet small, persistent changes in hydrology such as the frequency of the highest and lowest flows, can alter channel morphology and the riparian ecosystem. In particular, human activities, such as urbanization and forestry, are well known causes of such persistent hydrologic change (Jones and Grant 1996; Burges et al. 1998).

Extreme low flows can restrict fish mobility when flow depths become so small that the stream channel is no longer passible by fish (Bradford and Heinonen 2008; Barnard et al. 2013). If flow is less than the sub-surface runoff component of the channel, stream flow may be entirely sub-terrain, causing the surface flow component of the channel to break up into isolated pools (Ward et al. 2018). Fish in these environments face an uncertain outcome, either moving out of a stream or becoming stranded in isolated pools. Pools and areas of slow, shallow flows tend to be warmer and can leave fish vulnerable to predation (Rosenfeld and Ptolemy 2012). However, if fish are stranded in deep pools, they can experience increased growth and survival (Martens and Connolly 2014).

Fish movement can also be restricted by extreme high flows. This is particularly likely in constricted stream channel reaches, including artificial channel reaches like culverts (Barnard et al. 2013), where flow resistance is low and flow velocity tends to be high. In more natural settings it is likely that these conditions would only be temporary and would delay rather than prevent upstream movement. Additionally, when flows exceed the threshold flow rate at which the channel bed material (e.g. sand, gravel and cobbles) mobilizes, aquatic life can be buried or scoured. This can be especially problematic for salmonids creating redds (i.e., nests) in the spring and fall. The effect of extreme high flow rates on the channel bed can be conceptualized with a simple empirical model of transport rate of gravel, sand and cobbles on the channel bed ($q_s$) as a function of the flow rate (Gaeuman et al. 2018):

$$q_s = a\left(Q - Q_c\right)^B \tag{1}$$

The variables $Q$ and $Q_c$ correspond to the flow rate and the critical flow rate. The critical flow rate is some threshold flow rate above which the channel bed moves. The coefficient $a$ scales Equation 1 and the exponent $B$ determines how rapidly the channel bed is disturbed as a function of flow. The exponent $B$ is typically larger than unity and any changes in flow rate have a disproportionate effect on the rate at which the bed is mobilized/disturbed.

Understanding trends in extreme flows in managed landscapes is thus important for evaluating land use effects on aquatic habitat in the context of channel bed stability and summer low flows. This section describes a preliminary assessment of the magnitude and trends in extreme low and high flows in 14 streams presently monitored as part of the OESF Status and Trends Monitoring project.

These results are preliminary because the hydrology record is at most only 7 years long, which is shorter than the minimum 10-year duration typically used for extreme value analysis of hydrologic data (England et al. 2019). Furthermore, the methodology used to establish rating curves (i.e., the empirical function between flow depth and flow rate at a specific stream reach) in unstable, mountainous stream channels like those of the OESF is new and still being refined as part of the OESF hydrology monitoring program. We have roughly 7 years of hydrologic and hydraulic field data at 10 DNR-managed watersheds and 2 to 3 years of data at 4 reference watersheds.

Stream flow gages are located at the outlets of 14 Type 3 watersheds, so that stream flow measured at each gage reflects the hydrology of the full watershed. This subset of 14 OESF watersheds was initially selected to represent the full range of hydrologic conditions of Type 3 watersheds within the OESF, without exceeding the work capacity of a single crew to visit repeatedly throughout the year to take stream flow and channel stability measurements. The 14 selected watersheds represent a range of Type 3 watershed sizes, the elevational range of the study area, the 2 dominant precipitation zones (rain and rain-on-snow), and the north-sound gradient, which roughly aligns with the rainfall intensity (Minkova and Vorwerk 2014, p. 23). However, the subset of 14 watersheds has changed with time. During the first several years of monitoring, deep deposition and scour occurred at 4 of the initial 14 stream gages and these 4 gage sites were abandoned. The sensors from these abandoned sites were subsequently installed in four reference watersheds with the intent of eventually providing a set of long-term data from these watersheds.

Hydrologic data collected at each gage consists of a record of water and atmospheric pressure automatically recorded every 15 minutes (a time series). From the time series of water and atmospheric pressure, a time series of flow depth can be determined. Hydraulic data collected at each site includes repeat flow, flow depth, channel geometry, slope and flow resistance measurements. The quantity of hydraulic data is limited by the number of field visits to each gage. From the hydraulic data, a hydraulics based empirical model and a simple statistical model that defines flow rate as function of flow depth can be created and combined to define a single rating curve following Le Coz et al. (2014). Because the hydraulic characteristics of OESF stream channels frequently change, each stream gage requires a new rating curve each time the stream channel changes. Once these rating curves are completed, they can be used to convert the time series of flow depth to a time series of stream flow (i.e., a hydrograph).

Relative to other flow monitoring programs (USGS and other state programs; Norris et al. 2008), the OESF Status and Trends flow monitoring program operates on a very limited budget and staffing level. As such, hydrologic and hydraulic data collection for all 14 gages is complete, but data analysis to date includes only 5 of the 14 monitored watersheds. These five watersheds fall within two geographic zones (the Clallam Bay area and the Goodman Creek area). As the flow records near 10 years and data collection and processing methods are refined, the analysis may be expanded to include all 14 watersheds. Nonetheless, this report represents a first look at the frequency and magnitude of extreme high and low flow events, defined here as flow rates that tend to exceed $Q_c$ or cause the stream channel to go dry.

Future reports may track the seasonality and duration of the extreme high and low flows. The hydrograph can also be used to compute the frequency of specific flow rates such as the bankfull and 100-year flow, two flow metrics used for stream crossing design. Finally, observed hydrographs are critical to hydrologic modeling and a well-calibrated hydrologic model is a powerful tool that can be used to answer management questions related to harvest planning and climate change. These potential uses of the hydrograph are described in detail at the end of this section.

## SAMPLING AND ANALYSIS

In this report, the hydrograph is expressed in units of depth per unit time (mm/hr) so as to permit comparison across watersheds and to precipitation (also recorded in mm/hr). Flow rates and cumulative totals were compared to daily average precipitation. Daily average precipitation is from the PRISM climate group (PRISM Climate Group 2021). The PRISM climate group develops spatial climate datasets from climate observations that reveal short and long-term climate patterns. A one-month segment of the hydrographs for the 5 watersheds relative to the daily average precipitation from the PRISM climate group is shown in Figure 28.

Annual extreme low and high flow statistics were tallied for each water year. A water year is defined as October 1st to September 30th of the following year. Following flow metrics commonly used to

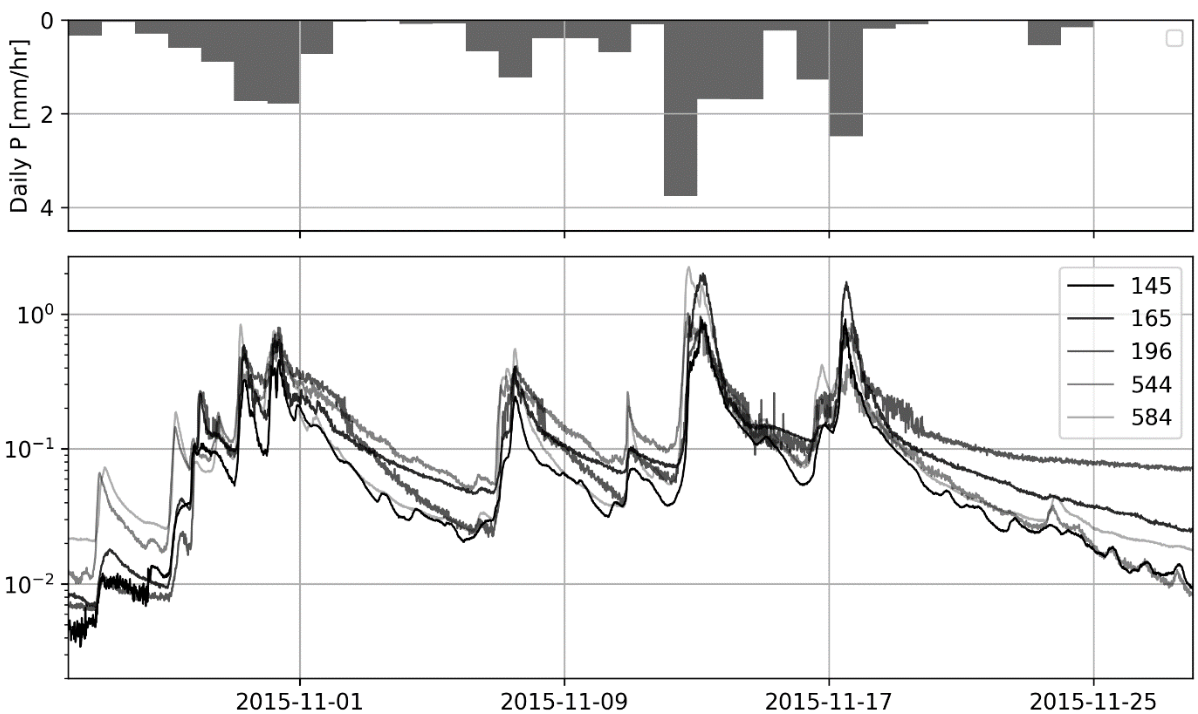



**Figure 28.** Daily average precipitation rates from PRISM (2021) and observed flow response at the five watersheds.

design fish passable stream crossing structures (Barnard et al. 2013), we used the minimum 7-day mean flow ($Q_{min7}$) to track extreme low flows and the flow event maximum peak flow rate ($Q_{max}$) to track extreme high flows. A partial duration series of the metric $Q_{min7}$ was determined from all independent, consecutive 7-day periods of flow in the hydrograph. Independent consecutive 7-day periods of low flow are defined as any 7-day period of low flow separated by at least 60 days. Any single year may have 5 to 6 independent low flow events. A partial duration series of the metric $Q_{max}$ was determined from all independent peak flow rates in the hydrograph. Independent $Q_{max}$ were separated by at least 30 days, and a single year may have roughly 11 to 12 peak flow events. Annual minimum $Q_{min7}$ and maximum $Q_{max}$ were tracked for each water year to visualize potential trends with time. The partial duration series of $Q_{min7}$ and $Q_{max}$ were used to convert magnitude to percentile values.

### *Individual flow events*

Both the magnitude and duration of a high flow event above the critical flow rate control the total impact of the flood on the channel bed. To infer magnitude-duration characteristics of the high flow events, for each watershed, we divided the hydrograph into individual flow events. An individual flow event was extracted from the hydrograph using a rules-based, automated flow extraction algorithm similar to that applied in Jones and Grant (1996) and Tang and Carey (2017). For each flow event in the hydrograph, both the hydrograph and the time-rate-change (derivative) of the hydrograph was used to identify the begin, peak and end of the flow event (Figure 29). The parameters of the extraction algorithm were selected so that performance of the automatically extracted flow events matched manual extraction results. Empirical cumulative distribution functions (CDF) of flow metrics, including peak flow rate, flow event duration and runoff ratio were tallied from all flow events. The runoff ratio is the ratio of the cumulative runoff depth to the cumulative precipitation depth of the coinciding precipitation event and is a useful metric for inferring how precipitation is routed through a watershed (e.g., high runoff ratio indicates little precipitation was lost to evapotranspiration or soil/snow storage).

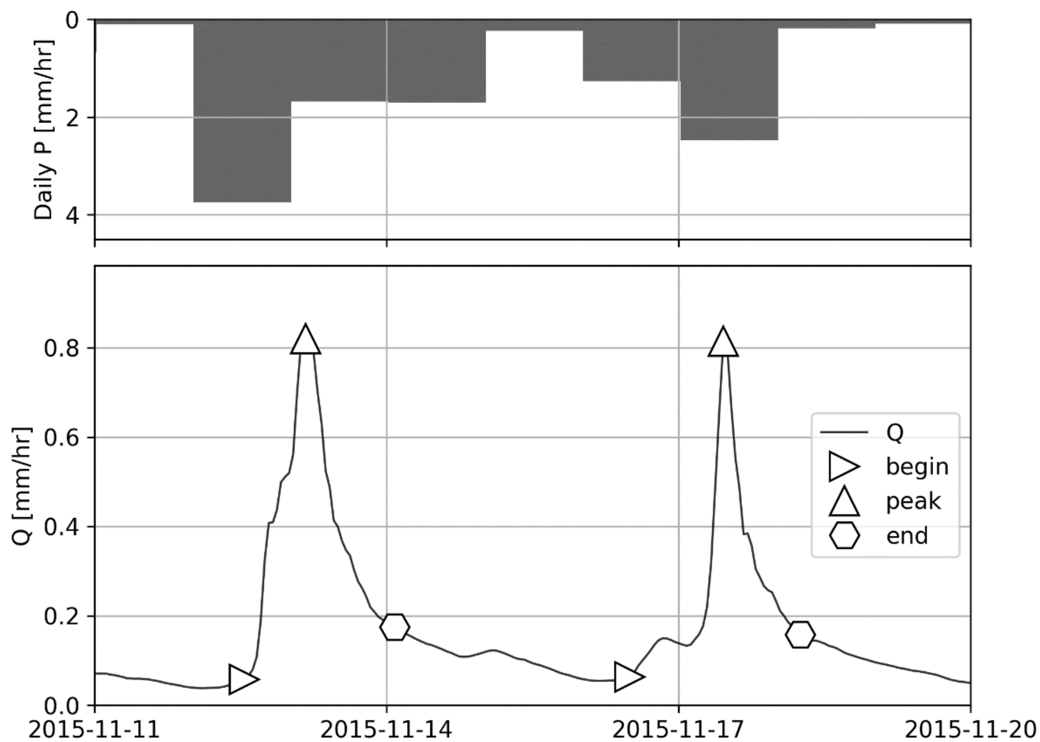



**Figure 29.** Example of how flow events are automatically extracted from the hydrograph and used to determine flow event statsitics such as flow event duration, peak flow rate and runoff ratio.

## RESULTS

### *Event precipitation and runoff*

Maximum daily precipitation rates and precipitation event cumulative totals relative to season are illustrated in Figure 30. Over the 7-year monitoring period, daily average precipitation rates were highest in the fall and winter. The highest daily average precipitation rate was 4.7 mm/h and occurred in the winter. The cumulative precipitation rate follows a similar pattern but winter cumulative rates are considerably higher than the fall season rates. During the summer, the maximum precipitation rate and precipitation totals drop to only a fraction of the fall and winter rates.

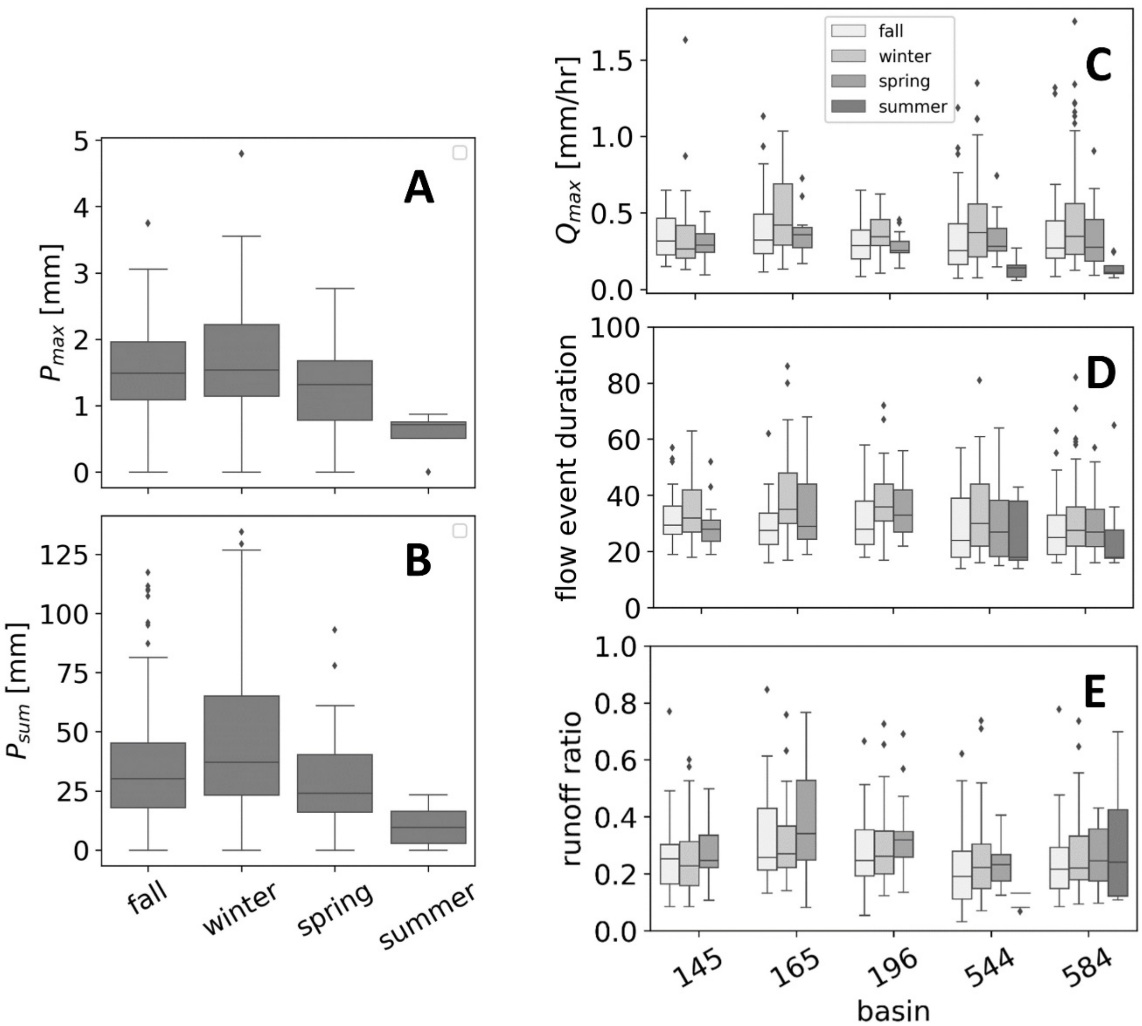



**Figure 30**. Boxplots of precipitation and flow metrics. Box shows the 25th percentile, median and 75th percentile (interquartile range-IQR). Whiskers extend 1.5*IQR past the edges of the box. Points beyond the whiskers are outliers. (a) Event peak precipitation rate, and (b) cumulative precipitation relative to season. (c) Peak flow rate, (d) flow event duration [hours], and (e) flow event runoff ratio relative to season. At watersheds 145, 165 and 196, flow response in the summer is too small to be considered a flow event.

Except for Watershed 145, $Q_{max}$ and flow duration at each of the five watersheds generally follow seasonal trends in maximum precipitation rates and cumulative precipitation totals. Peak flow rates—and thus the potential to mobilize and disturb the stream channel bed (high $Q$-$Qc$)—are highest during the winter months. Runoff ratio steadily increases through the water year. A likely explanation for this phenomenon is that during the late fall, winter and early spring months, vegetation growth rates slow and as the watershed becomes saturated with each rainfall event, less precipitation is sent to storage, leaving more to flow down the channels. Very little precipitation falls in the summer, so runoff depths often exceed precipitation depths. Note that during the summer, flow response at Watersheds 145, 165 and 196 did not meet the automated storm extraction thresholds but flow response at Watersheds 544 and 548 did, suggesting either a difference in precipitation or runoff rates between the watershed locations.

Figure 31b shows cumulative annual precipitation and Figure 31c shows $Q_{min7}$ versus water year. During the first four years of the monitoring program, $Q_{min7}$ was relatively consistent across all five watersheds. A period of zero surface stream flow appears to occur at Watershed 544 during all but the last year in the record. In 2018, all gages went dry at some point, but note that $Q_{min7}$ does not seem to track with annual precipitation. Other factors, such as the timing and cumulative amount of precipitation in the early summer may control the magnitude of $Q_{min7}$ and may be determined in future reports. During the last two years, $Q_{min7}$ was higher than in earlier years.

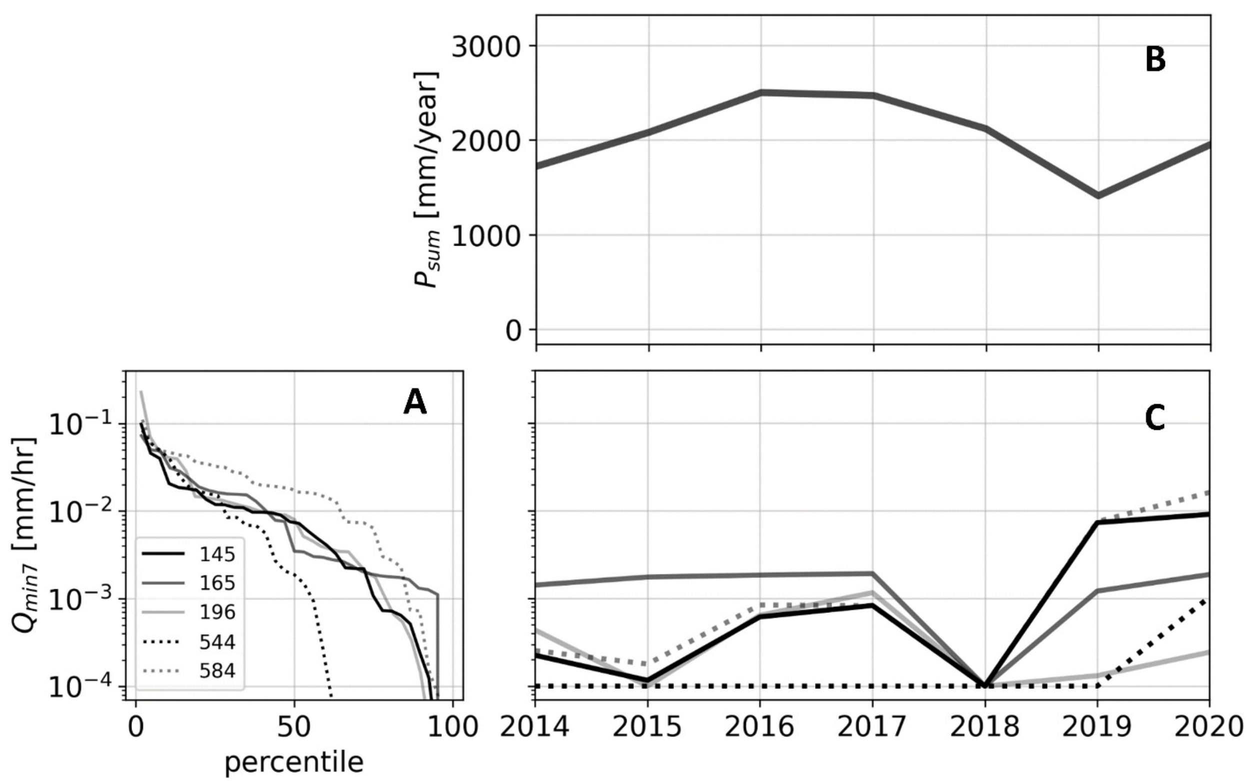



**Figure 31.** (a) Empirical cumulative distribution functions of $Q_{min7}$ for each watershed; (b) annual cumulative precipitation; (c) annual minimum $Q_{min7}$ over the observation period.
Note: (a) and (c) y-axis are the same.

### *Annual extreme high flows*

Figure 32a shows a CDF of $Q_{max}$ for each watershed. Note that the domain of the CDF has been adjusted to highlight the upper percentile flows. Surprisingly, the high flow behavior of Watersheds 544 and 584 are very similar while the high flow in Watersheds 145, 165 and 196 are more variable. $Q_{max}$ is generally higher at Watersheds 544 and 584, suggesting flashier flows and that channel disturbance is more likely (high $Q$-$Qc$ likely) at Watershed 544. This is also evident in the time annual series of $Q_{max}$. Notably, the time series of $Q_{max}$ is much less variable than the time series of $Q_{min7}$, and perhaps may indicate that $Q_{min7}$ is more sensitive to environmental factors than $Q_{max}$.

Annual maximum peak flow rates are shown in Figure 32c. Watershed 196 appears to be somewhat different from the other watersheds. Annual maximum peak flow rates at Watershed 196 are low and nearly constant with time. High and low values in the $P_{max}$ time series (Figure 32b) generally coincide with high and low $Q_{max}$ in the watersheds; however, the highest precipitation rate does not coincide with an equivalent peak in $Q_{max}$ at 4 of the 5 watersheds. Since we are comparing daily average precipitation to instantaneous flow rates, the peak in $P_{max}$ may not coincide with the instantaneous $Q_{max}$ values. Furthermore, the PRISM precipitation values are interpolated values and may not match the precipitation rates that actually fell over the watersheds. Finally, some of the flow variability (or lack of) in Figure 32c may be due to uncertainty in the rating curves.

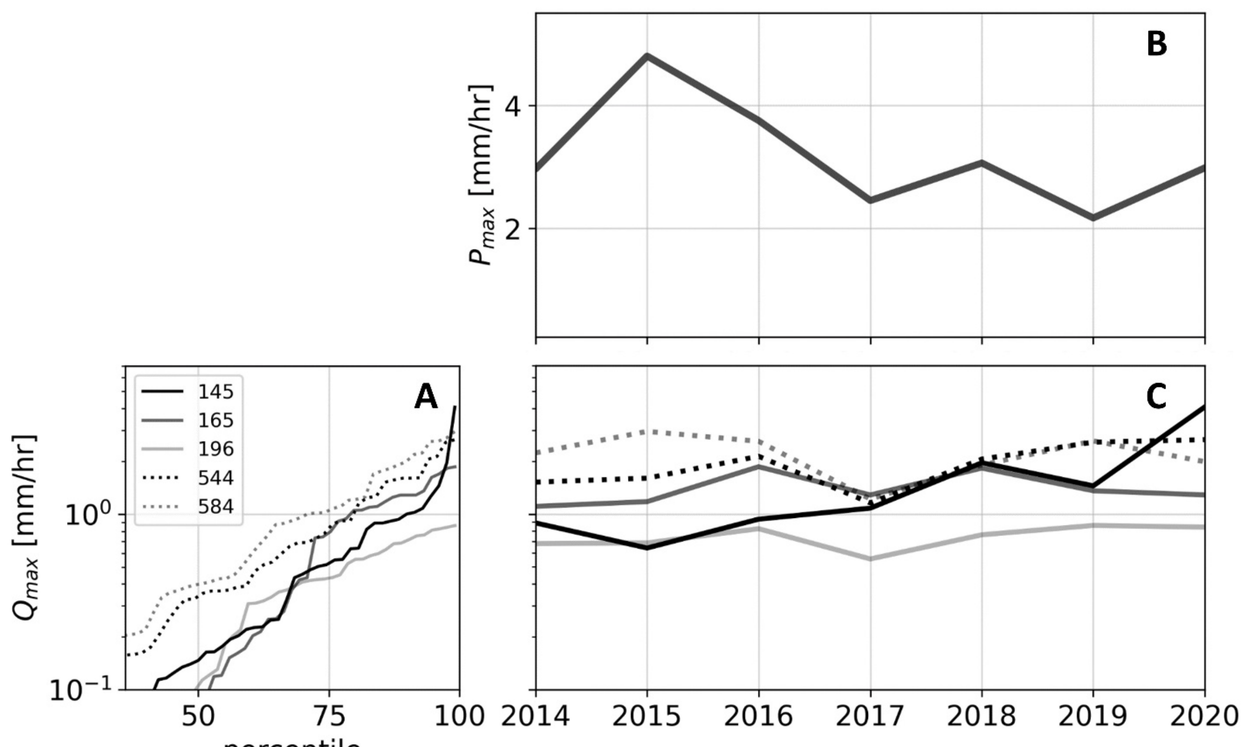



**Figure 32**. (a) Empirical cumulative distribution functions of $Q_{max}$ for each watershed; (b) annual maximum precipitation; (c) annual maximum $Q_{max}$ over the observation period.
Note: (a) and (c) y-axis are the same.

## IMPLICATIONS FOR DNR-MANAGED WATERSHEDS

In this analysis, we reported temporal trends in extreme maximum and minimum flows at 5 of the 14 gaged watersheds in the OESF. Initial results indicate that either precipitation rates or runoff processes at Watersheds 145, 165 and 196 may be more attenuated and less intense than precipitation rates/runoff response at Watersheds 544 and 548. Furthermore, $Q_{max}$ is generally higher at Watersheds 544 and 584, suggesting flashier flows and that channel disturbance (high $Q$-$Qc$) is more likely.

The minimum flow rate metric $Q_{min7}$ does not seem to track with annual cumulative precipitation. In contrast, peak flow rates ($Q_{max}$) do appear to track with annual maximum peak precipitation rates. The fact that $Q_{min7}$ does not track with annual changes in cumulative precipitation may indicate that $Q_{min7}$ is sensitive to environmental factors such as land use or that the timing and cumulative amount of precipitation in the early summer, rather than annual cumulative precipitation. Additionally, annual variability in extreme flows (or lack of) may also be due to uncertainty in the rating curves. Future reports will try to quantify this uncertainty and differentiate between hydrologic- versus measurement-caused variability in the hydrographs.

As the duration of the flow record nears 10 years and our data collection and processing methods are refined, the analysis may be expanded to include all 14 monitored watersheds. Potential applications of the OESF hydrographs for DNR-managed watersheds are listed in the Future Monitoring and Research section at the end of this report.

# Habitat Condition Assessment

## Introduction

This section presents the second of two approaches used to evaluate the habitat indicators measured in Status and Trends Monitoring. Both approaches share the overall goal of assessing the effectiveness of DNR's habitat conservation measures in producing the desired habitat conditions. In the previous section of this report, the focus was individual habitat indicators. Here, in the habitat condition assessment, the focus shifts to individual streams as the unit of analysis.

The goal of this habitat modeling effort is to compare the different streams by combining multiple habitat indicators to produce overall habitat condition scores. Thus, multiple measures of habitat quality are integrated to produce a single habitat score per stream to predict which streams are providing high- or low-quality habitat. It will then be possible to investigate what factors correlate with the stream habitat scores.

The modeling approach in this habitat condition assessment follows the general form that DNR used in the OESF Forest Land Plan EIS to calculate composite watershed scores (DNR 2016b, p. G-80). In that analysis, scores were calculated by using a hierarchical model that incorporated a series of habitat indicators (Figure 33). That model format was adapted from a similar modeling approach, often called Ecosystem Management Decision Support modeling, used in the Northwest Forest Plan Aquatic and Riparian Effectiveness Monitoring Plan (AREMP) (Reeves et al. 2004; Gallo et al. 2005) and by DNR on dispersal habitat for the northern spotted owl (*Strix occidentalis caurina*) (Gordon et al. 2014). In this type of model, expert opinion and scientific literature are used to develop a scoring system for a series of ecological indicators. Usually these are indicators that describe habitat condition or ecological processes at the watershed scale. Values for the indicators are then scored and scores are combined to produce a composite score for each watershed.

The main differences between the OESF Forest Land Plan EIS model and the habitat condition model presented here is that this model is based on reach-scale data collected in the field, whereas the EIS model was based on watershed-scale habitat data derived from GIS analyses. The benefit of our approach is that the habitat indicators have been directly measured rather than predicted. Despite the different data sources, the objectives of the current analysis and the one in the OESF Forest Land Plan EIS are similar—a ranking of Type 3 watersheds by habitat condition.

It is important to remember that this habitat condition assessment is built, in part, upon expert opinion due to our incomplete understanding of the complex, interacting components of stream habitat. Results of this model are therefore a product of our current understanding of stream habitat quality, an understanding that is certain to evolve over time. Thus, the model presented here may be refined in the future as our knowledge of the system grows.

## The habitat model

This habitat model focuses on habitat of salmonids, specifically steelhead/rainbow trout, cutthroat trout, and coho salmon. These three species of salmonids were selected for the model because they are the most prevalent salmonid species in the monitoring area. Numerous aquatic and terrestrial species are present in the streams and adjacent riparian areas of the OESF, and similar models could

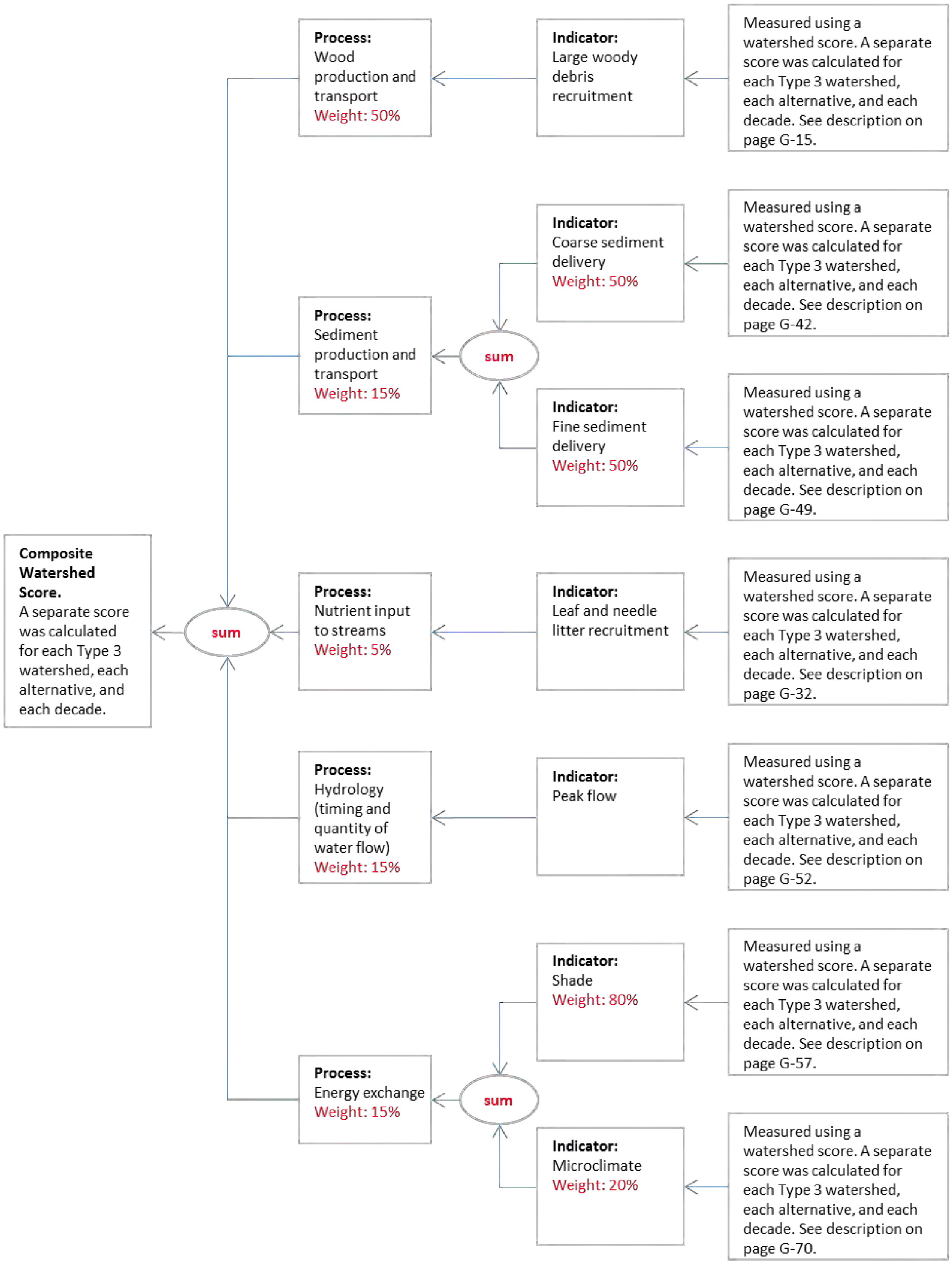



**Figure 33.** The framework of the model used to calculate the composite watershed score in the OESF Forest Land Plan EIS; weighting factors and the operators used to combine variables are shown in red. Figure is reproduced from DNR 2016b.

be developed for other species by using different combinations of the habitat indicators in Status and Trends Monitoring. This habitat model was developed recognizing that the three selected salmonid species differ in the ranges of habitat conditions that they occupy; the model focuses on habitat elements and conditions that are common to all three species. This generalized approach may be narrowed in future work to focus at the individual-species level.

In choosing habitat indicators for this model, we selected only those for which directly measured, reach-scale data were available, as opposed to watershed-scale variables predicted to influence stream habitat. Other natural resource decision-support models have used watershed-scale variables such as road density, stream crossings, and land use (e.g., Walker et al. 2007, Lanigan et al. 2012) or have combined these watershed-scale variables with reach-scale habitat variables in the same model (e.g., Gordon and Gallo 2011). We chose to use only reach-scale indicators after drawing a distinction between the two types of variables: reach-scale habitat indicators represent the in-stream and riparian *effects* of processes at the watershed scale, whereas watershed-scale variables are possible *causes* of change in the reach-scale habitat indicators.

The habitat model is designed to produce composite habitat scores for individual streams so that streams can be compared and ranked based on these scores. The model uses average habitat indicator values for each of the 62 streams for the 2013-2020 monitoring period. With this initial version of the model, we are not attempting to assess change in habitat condition over time, so we are using indicator values averaged over the whole monitoring period. However, after an additional five years of monitoring, and potential refinements to the model, we will evaluate temporal change in habitat condition.

## MODEL STRUCTURE

### *Habitat indicators*

The habitat model is designed to produce an overall habitat condition score for each of the 62 monitored sample reaches. This overall score is derived from 11 selected habitat indicators that are grouped into four habitat categories (Figure 34). Each of the 11 indicators was selected for its role in contributing to specific aspects of salmonid habitat (Table 22), and each indicator uses a specific metric calculated from the Status and Trends Monitoring data for every monitored stream (Table 23). Some of the indicator metrics were reported in the previous section of this report (e.g., water temperature, canopy closure), whereas others were not (e.g., % boulders in substrate, habitat unit frequency).

Each indicator metric is used to calculate an indicator habitat score, ranging from -1 to 1, for each stream, with -1 indicating the lowest quality and 1 indicating the highest quality. Indicator habitat scores are calculated using a spreadsheet, according to where a stream's score falls along an indicator curve. For example, for the in-stream wood indicator (Table 23), the curve shape shows a flat line, followed by an upward slanted line, followed by another flat line. The two corners, or nodes, on this curve represent values of 4 and 11 key pieces of wood. Streams with 4 or fewer pieces will receive a score of -1, whereas streams with 11 or more pieces will receive a score of 1. For streams with

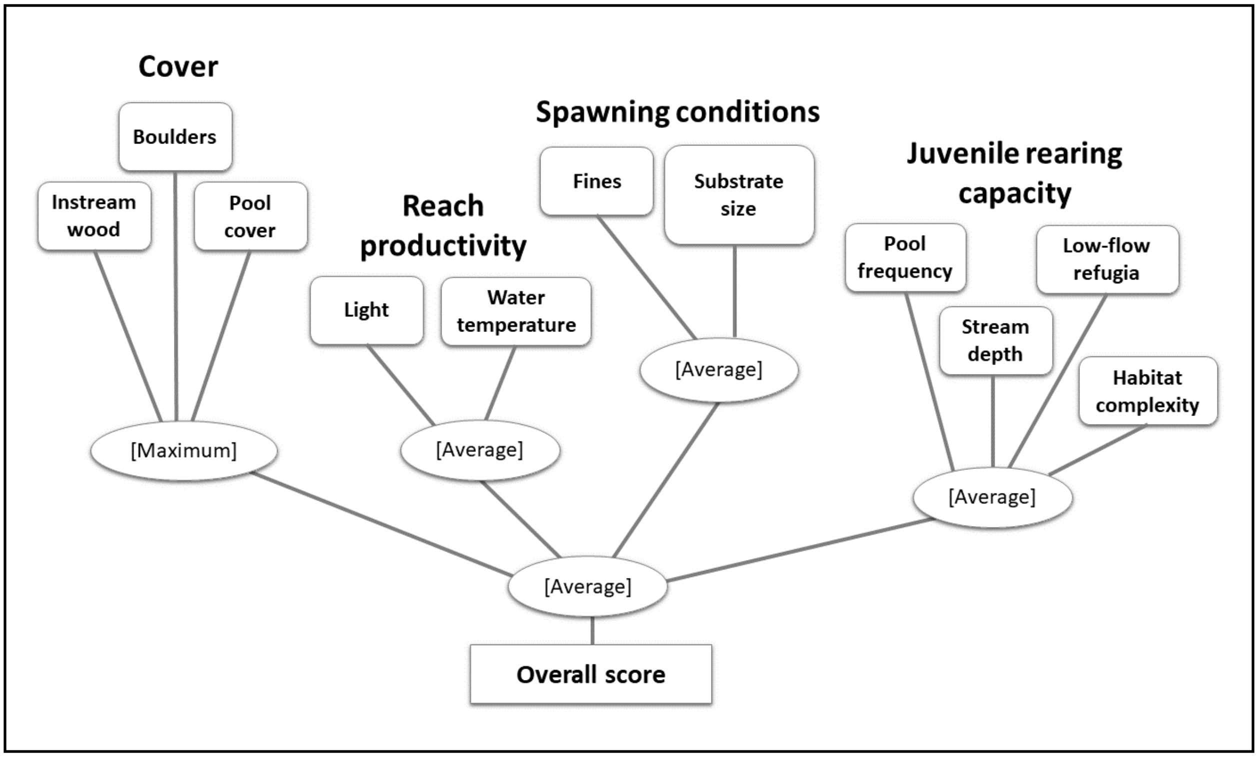



**Figure 34.** Structure of the habitat condition model; 11 habitat indicators are aggregated in four habitat categories.

between 4 and 11 pieces of wood, the calculated score will be intermediate (calculated by the spreadsheet). For example, 8 pieces produces an indicator score of 0.14.[32]

### *Habitat categories*

The habitat condition model includes four habitat categories: cover, reach productivity, spawning conditions, and juvenile rearing capacity. Each of the categories contains two to four indicators that are used to calculate category scores, which also have a potential range of -1 to 1.[33] Finally, the four category scores are averaged to create an overall habitat condition score for each sample reach, which again has a potential range of -1 and 1.

The score for the first habitat category, cover, is calculated as the maximum score of three indicators representing three potential sources of cover for fish: in-stream wood, boulders, and pools. If any of these cover types is abundant, the cover category will have a relatively high score. The other three habitat categories are calculated as averages of the indicators in each category. The second category, reach productivity, is based on the amount of available sunlight, which affects stream primary productivity (growth of plants at the base of the food web), and temperature, which influences the rate at which fish grow (Weatherly and Gill 1995).

[32] The calculation in this example is: ((8-4)/(11-4)*(1-(-1)))+(-1) = 0.1429

[33] Within each of the four categories, scores for the 62 watersheds are scaled so that they will range from -1 to 1. This ensures that when the four categories are averaged to produce an overall habitat condition score, each category will have the same influence on the overall score.

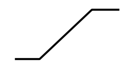
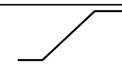
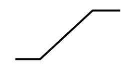
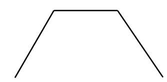
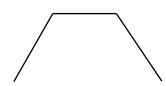
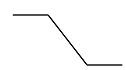
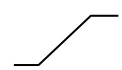
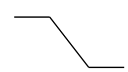
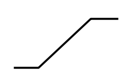
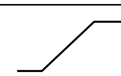
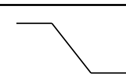
**Table 22.** Habitat condition model indicators, and rationale for selection, grouped by habitat category.

| Indicator | Rationale for selection |
| --- | --- |
| - - - - - - - - - - - - - - - - *Cover* | - - - - - - - - - - - - - - - - |
| Instream wood | Wood in streams is important in providing habitat for salmonids (Fox and Bolton 2007). |
| Boulders | Juvenile fish often use boulders for cover (Dambacher and Jones 1997; Martens 2019). |
| Pool cover | Pools are a source of cover for fish (Lanigan et al. 2012). |
| - - - - - - - - - - - - - - - - *Reach productivity* | - - - - - - - - - - - - - - - - |
| Light | The degree of stream shading is influenced by riparian forest management; stream primary productivity[1] is in turn influenced by the amount of light reaching a stream (Boston and Hill 1991; Warren et al. 2013). |
| Water temperature | Salmonids are adapted to a specific range of temperatures; growth is reduced if temperatures are too cold; if temperatures are too warm, growth is reduced and fish become more susceptible to disease (Carter 2005). |
| - - - - - - - - - - - - - - - - *Spawning habitat* | - - - - - - - - - - - - - - - - |
| Fines | Excessive fines reduces emergence of salmonids (Kondolf 2000). |
| Substrate size | There is an optimal particle size range for spawning of salmonids (Kondolf and Wolman 1993, Kondolf 2000; Jensen et al. 2009). |
| - - - - - - - - - - - - - - - - *Rearing habitat* | - - - - - - - - - - - - - - - - |
| Pool frequency | Pools are an important habitat feature for juvenile salmonids (Lanigan et al. 2012). |
| Stream depth | Density of juvenile fish increases with non-pool stream depth in the OESF (Martens et al. 2019). |
| Low flow refugia | Deep pools provide refuge for juvenile salmonids during periods of low stream flow in summer. |
| Habitat unit frequency | Habitat complexity has been found to be positively correlated with salmonid survival (Quinn and Peterson 1996). |

[1] The rate at which energy is converted to organic substances, mostly through photosynthesis by algae.

The third category, spawning conditions, includes two indicators of spawning substrate condition: the amount of fine particles present in the substrate and the median size of streambed substrate particles. The fourth habitat category is juvenile rearing capacity. Its pool frequency and habitat unit frequency indicators describe the amount of pool habitat in a stream and the frequency at which one habitat type transitions to another. Stream depth is correlated with the number of fish a stream has the capacity to support. Low-flow refugia is the frequency of deep pools (≥20 in.; ≥50 cm) which are important to survival of juvenile fish in summer when stream flow reaches its annual minimum. We were not able to use the hydrology data for the habitat model because only 14 of the 62 streams were monitored for hydrology.

**Table 23.** Habitat condition model indicator metrics, indicator curve shapes, and threshold values.

| Indicator | Metric | Source of threshold values | Curve shape | Threshold value and node x-value | Score at node y-value |
|---|---|---|---|---|---|
| *Cover* | | | | | |
| Instream wood | No. key pieces per 330 ft (100 m) | Fox and Bolton (2007) |  | 4 | -1 |
| | | | | 11 | 1 |
| Boulders | % boulders in substrate | Dambacher and Jones (1997) |  | 0% | -1 |
| | | | | 17% | 1 |
| Pool cover | Mean residual pool depth (in.) | Lanigan et al. (2012) |  | 13.8 in. (35 cm) | -1 |
| | | | | 29.5 in. (75 cm) | 1 |
| *Reach productivity* | | | | | |
| Light | % canopy closure above stream | Warren et al. (2013): thresholds based on 1 and 2 std. deviations around mean for streams in old growth[1] |  | 59.3% | -1 |
| | | | | 75.8% | 1 |
| | | | | 92.4% | 1 |
| | | | | 100% | -1 |
| Water temperature | 7-DADmax (°C) | Carter (2005) |  | 9 °C (48 °F) | -1 |
| | | | | 12 °C (54 °F) | 1 |
| | | | | 14 °C (57 °F) | 1 |
| | | | | 17.5 °C (64 °F) | -1 |
| *Spawning conditions* | | | | | |
| Fines | % fines in substrate | Kondolf (2000) |  | 2% | 1 |
| | | | | 12% | -1 |
| Substrate size | $D_{50}$ (median particle diameter) | Kondolf and Wolman (1993), Kondolf (2000), Jensen et al. (2009) |  | 0.6 in. (15 mm) | -1 |
| | | | | 1.2 in. (30 mm) | 1 |
| *Juvenile rearing capacity* | | | | | |
| Pool frequency | Bankfull widths per pool | Gallo et al. (2005), p. 105 (westside) |  | 5 | 1 |
| | | | | 14 | -1 |
| Stream depth | Bankfull depth | Martens (2019), thresholds derived from validation monitoring |  | 7.5 in. (19 cm) | -1 |
| | | | | 11.0 in. (28 cm) | 1 |
| Low-flow refugia | No. deep pools per 330 ft (100 m) | Professional judgement and OESF data |  | 0 | -1 |
| | | | | 2 | 1 |
| Habitat unit frequency | No. bankfull widths per habitat unit | Professional judgement and OESF data, Quinn and Peterson (1996) |  | 1.3 | 1 |
| | | | | 2.1 | -1 |

[1] Given the historical predominance of late-seral forest in the region, it was assumed that salmonids are adapted to late-seral canopy closure and light conditions.

### INTERPRETING HABITAT CONDITION SCORES

Independent of human influence, stream habitat conditions vary in response to natural variation across the landscape. Some of this natural variation affects the stream habitat scores produced by the model and should be considered when interpreting the habitat model results. In the monitored Type 3 streams, there are significant differences in habitat associated with differences in stream gradient. Differences in stream gradient are represented by our three stream channel types: pool-riffle, step-pool, and cascade. Certain habitat attributes, such as channel substrate, in-stream wood, and bank erosion, have already been shown in the previous section of this report to differ among channel types. Because of these inherent differences, we group streams by channel type within the habitat model and compare the scores of individual streams only to other streams within the same channel type.

## Results of the habitat assessment

Habitat condition scores for each monitored stream appear in Table 24, grouped by channel type. The ranges in overall habitat condition scores were 0.96, 1.26, and 0.91 for the pool-riffle, step-pool, and cascade channel types, respectively. The larger range for the step-pool type may have been a result of the fact that that group was much larger than the other two, with 32 streams compared to 17 in the pool-riffle type and 13 in the cascade type. Maps of the habitat condition scores appear in Appendix I. It should be noted that due to small sample sizes, especially for the pool-riffle and cascade channel types, apparent spatial patterns of scores in Appendix I are do not necessarily represent true geographic trends.

### CHANNEL TYPE DIFFERENCES

The distribution of overall scores within each channel type are shown in Figure 35. The shapes of the distributions of overall habitat condition scores were generally similar among the three channel types, with the exception of several streams at the low end of the distribution in the pool-riffle type. To better understand the influence of channel type of habitat condition scores, the scores within individual categories must be examined (Table 24).

For two of the four indicator categories—productivity and rearing—average category scores did not differ widely among the three channel types. However, average scores for the cover and spawning categories varied more widely among channel types. In the cover category, average scores for the three channel types increased as channel type gradient increased, averaging -0.01, 0.41, and 0.90, for pool-riffle, step-pool, and cascade channel types. Cover scores were high for the cascade type because boulders were usually present in streams to provide cover for fish. Boulders were less common in step-pool and rare in the pool-riffle types, where cover was instead associated with key pieces of in-stream wood and pools.

The second category in which habitat scores varied by channel type was the spawning category. Average scores in the spawning category were -0.06, 0.44, and 0.67, for pool-riffle, step-pool, and cascade channel types. The low average habitat score for the pool-riffle channel type was a result of a higher percentage of fines in the streambed substrate of the pool-riffle streams. A high amount of fines results in poorer spawning conditions because it inhibits the flow of water among eggs. A logical question is then: was this high level of fines in pool-riffle streams a result of past management

**Table 24.** Habitat condition model output for each of 62 monitored Type 3 watersheds; the overall score and the four category scores should be compared to other watersheds only within the same channel type group. Scores are color-coded, with the highest (1.00) green, intermediate (0) yellow, and the lowest (-1.00) red.

| Watershed ID | Management | Overall score | Category score | | | |
|---|---|---|---|---|---|---|
| | | | Cover | Productivity | Spawning | Rearing |
| *- - - Pool-riffle channel type - - -* | | | | | | |
| 597 | DNR-managed | 0.45 | 0.74 | 0.58 | 0.00 | 0.50 |
| 584 | DNR-managed | 0.44 | 1.00 | -0.24 | 0.00 | 1.00 |
| 844 | DNR-managed | 0.36 | 0.02 | 1.00 | 0.08 | 0.35 |
| 718 | DNR-managed | 0.33 | 1.00 | -0.40 | 0.29 | 0.44 |
| 730 | DNR-managed | 0.32 | -0.20 | 0.22 | 0.32 | 0.93 |
| 820 | DNR-managed | 0.32 | 0.05 | 1.00 | 0.00 | 0.24 |
| 760 | DNR-managed | 0.21 | 1.00 | -0.20 | -0.58 | 0.62 |
| 433 | DNR-managed | 0.19 | 0.19 | 0.18 | -0.60 | 1.00 |
| 658 | DNR-managed | 0.19 | -0.03 | 0.17 | 0.74 | -0.10 |
| 488 | DNR-managed | 0.12 | -0.63 | 0.78 | 0.36 | -0.04 |
| 582 | DNR-managed | -0.03 | 1.00 | -0.73 | 0.00 | -0.38 |
| 328 | DNR-managed | -0.18 | -0.94 | 0.39 | 0.46 | -0.64 |
| 443 | DNR-managed | -0.22 | -0.62 | 0.01 | 0.23 | -0.50 |
| 717 | DNR-managed | -0.41 | -1.00 | 0.84 | -0.93 | -0.54 |
| 566 | Reference | -0.45 | -0.34 | 0.06 | -1.00 | -0.50 |
| QUE | Reference | -0.49 | -0.45 | -1.00 | -0.71 | 0.20 |
| 642 | DNR-managed | -0.51 | -1.00 | -0.69 | 0.31 | -0.67 |
| | *Average for pool-riffle* | *0.04* | *-0.01* | *0.12* | *-0.06* | *0.11* |
| *- - - Step-pool channel type - - -* | | | | | | |
| 165 | DNR-managed | 0.84 | 1.00 | 0.54 | 1.00 | 0.84 |
| 716 | DNR-managed | 0.84 | 1.00 | 0.90 | 0.73 | 0.75 |
| 773 | DNR-managed | 0.80 | 1.00 | 0.74 | 0.90 | 0.55 |
| 625 | DNR-managed | 0.69 | 1.00 | 0.01 | 1.00 | 0.76 |
| NF4 | Reference | 0.68 | 1.00 | 1.00 | -0.08 | 0.78 |
| 145 | DNR-managed | 0.66 | 1.00 | 1.00 | 0.38 | 0.26 |
| 790 | DNR-managed | 0.66 | 0.55 | 0.96 | 0.63 | 0.50 |
| 196 | DNR-managed | 0.62 | 0.80 | 0.32 | 0.64 | 0.71 |
| 542 | DNR-managed | 0.60 | 1.00 | 0.06 | 0.76 | 0.58 |
| 797 | DNR-managed | 0.58 | 0.76 | 0.65 | 0.24 | 0.66 |
| NF9 | Reference | 0.56 | 0.38 | 1.00 | 0.00 | 0.87 |
| 724 | DNR-managed | 0.52 | 1.00 | 0.36 | 0.78 | -0.06 |
| 550 | DNR-managed | 0.51 | 1.00 | 0.13 | 0.54 | 0.35 |

| Watershed ID | Management | Overall score | Category score | | | |
|---|---|---|---|---|---|---|
| | | | Cover | Productivity | Spawning | Rearing |
| 804 | DNR-managed | 0.49 | 1.00 | 0.44 | 0.92 | -0.42 |
| 688 | DNR-managed | 0.45 | 0.16 | 0.99 | 0.85 | -0.20 |
| NF7 | Reference | 0.40 | 0.53 | 1.00 | 0.00 | 0.05 |
| NF5 | Reference | 0.38 | 1.00 | 1.00 | 0.05 | -0.53 |
| 796 | DNR-managed | 0.35 | 0.26 | 1.00 | 0.24 | -0.12 |
| 763 | DNR-managed | 0.35 | -0.36 | 0.98 | 0.28 | 0.51 |
| 568 | DNR-managed | 0.32 | -0.29 | 0.36 | 0.61 | 0.61 |
| 621 | DNR-managed | 0.28 | -0.09 | 0.37 | 1.00 | -0.17 |
| 157 | DNR-managed | 0.26 | 0.96 | 0.06 | 0.67 | -0.65 |
| 637 | DNR-managed | 0.25 | 1.00 | -0.73 | 0.65 | 0.07 |
| 567 | DNR-managed | 0.25 | -0.33 | 0.32 | 0.42 | 0.57 |
| 744 | Reference | 0.23 | 0.67 | 0.15 | 0.00 | 0.12 |
| 619 | DNR-managed | 0.09 | 0.14 | 0.22 | 0.47 | -0.48 |
| HOH | Reference | 0.05 | 0.24 | 0.21 | 0.60 | -0.83 |
| 544 | DNR-managed | -0.02 | -0.14 | 0.32 | 0.00 | -0.27 |
| NF6 | Reference | -0.05 | -0.69 | 0.33 | 0.13 | 0.05 |
| NF8 | Reference | -0.22 | -0.95 | 0.52 | 0.36 | -0.81 |
| 545 | DNR-managed | -0.33 | -0.64 | -0.08 | -0.01 | -0.58 |
| 769 | DNR-managed | -0.42 | -0.75 | 0.33 | -0.69 | -0.57 |
| | *Average for step-pool* | *0.36* | *0.41* | *0.48* | *0.44* | *0.12* |
| *- - - - - - - - - - - - - - - - - - - - - - - - - - - - Cascade channel type - - - - - - - - - - - - - - - - - - - - - - - - - - - -* | | | | | | |
| 750 | DNR-managed | 0.88 | 1.00 | 1.00 | 0.91 | 0.60 |
| SFH | Reference | 0.75 | 1.00 | 1.00 | 0.85 | 0.15 |
| 639 | DNR-managed | 0.70 | 1.00 | 1.00 | 0.79 | 0.01 |
| 653 | DNR-managed | 0.60 | 1.00 | 1.00 | 1.00 | -0.58 |
| 687 | DNR-managed | 0.58 | 1.00 | 0.13 | 0.20 | 1.00 |
| BOG | Reference | 0.55 | 0.47 | 0.11 | 0.60 | 1.00 |
| 776 | DNR-managed | 0.49 | 1.00 | -0.02 | 0.82 | 0.15 |
| 690 | DNR-managed | 0.48 | 1.00 | -0.22 | 0.72 | 0.43 |
| 694 | DNR-managed | 0.42 | 1.00 | -0.05 | 1.00 | -0.25 |
| 158 | DNR-managed | 0.25 | 1.00 | -0.10 | 0.44 | -0.33 |
| 737 | DNR-managed | 0.20 | 1.00 | -0.65 | 1.00 | -0.55 |
| 605 | DNR-managed | 0.10 | 0.34 | -0.01 | 0.00 | 0.07 |
| 767 | DNR-managed | -0.03 | 0.86 | -0.37 | 0.39 | -1.00 |
| | *Average for cascade* | *0.46* | *0.90* | *0.22* | *0.67* | *0.05* |

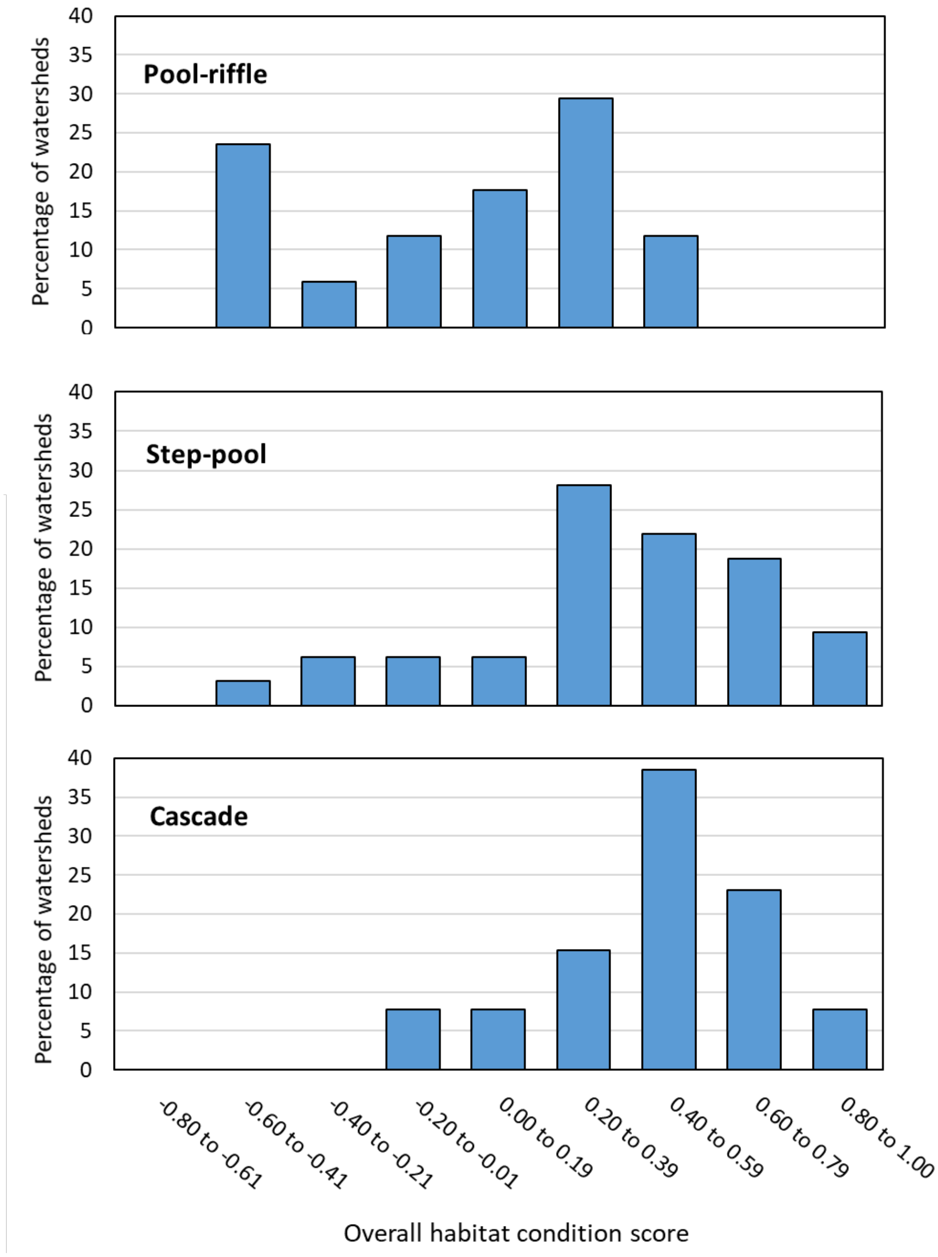



**Figure 35.** Distribution of overall habitat condition scores for watersheds in three channel type groups. Scores should not be compared between groups because naturally occurring habitat differences result in different ranges of scores for different groups.

or a natural feature associated with the low gradient of the pool-riffle channel type? Though this study cannot provide a conclusive answer to this, reviewing the scores of the unharvest watersheds can provide a clue. Only two of the reference watersheds had sample reaches of the pool-riffle channel type; both of these received the lowest possible habitat score (i.e., -1) for the percent fines habitat indicator, as both had greater than 12 percent fines.[34] Furthermore, after reviewing the scores of the mid-gradient, step-pool habitat type, four of the eight streams in reference watersheds also had habitat scores of -1 for the percent fines habitat indicator. Although this is a small sample size, the amount of fines in streams of reference watersheds suggests that, in OESF Type 3 streams, fines can be high in the absence of past harvest, for low-gradient streams.

## CHARACTERISTICS OF THE HIGHEST- AND LOWEST-SCORING STREAMS

### *What are the characteristics of the highest-scoring streams?*

To identify common characteristics of the streams with the highest-scored habitat, we created a group of streams that had the top 25% overall habitat scores from each channel type. We then compared the average habitat indicator scores from this group of 15 streams to the average indicator scores from all of the other streams to see which indicators were most different for the highest-scoring group.

The highest-scoring group of streams had four habitat indicators that had much different average scores than the rest of the streams: in-stream wood (0.60 vs. -0.16), stream depth (0.45 vs. -0.28), low-flow refugia (0.24 vs. -0.41), and habitat unit frequency (0.53 vs. -0.15). Therefore, the habitat characteristics that set the highest-scoring streams apart are:

(1) frequent large, or "key" pieces of in-stream wood,

(2) large stream size, characterized by deep streams with deep pools serving as low-flow refugia, and

(3) complex stream habitat, characterized by a large number of habitat units per sample reach.

### *What are the characteristics of the lowest-scoring streams?*

Following the same approach that we took with the highest-scoring streams, we created a group of streams that were in the lowest 25$^{th}$ percentile of overall habitat scores from each channel type. We compared the average habitat indicator scores from this group of 15 streams to the average indicator scores from all of the other streams to see which habitat indicators were responsible for the low overall scores in the lowest-scoring group.

The lowest-scoring group had a much lower average cover category score compared with the other streams (-0.18 vs. 0.58). Additionally, three of the juvenile rearing category scores were much lower for this group: stream depth (-0.83 vs. 0.13), low-flow refugia (-0.83 vs. -0.07), and habitat unit frequency (-0.51 vs. 0.18). Therefore, the habitat characteristics that were responsible for the low overall scores of this group were:

(1) a general lack of cover for fish, whether from in-stream wood, boulders, or deep pools,

(2) small stream size, characterized by shallow streams lacking deep pools that could serve as low-flow refugia, and

(3) a lack of stream habitat complexity, characterized by longer, uniform habitat units in the sample reach.

[34] Streams that have 12% or greater fines receive the lowest score of -1 (Table 23).

### *Differences between highest- and lowest-scoring streams*

Aside from stream size, which is a function of watershed size and is not expected to be affected by management, the two factors that differed most between the highest- and lowest-scoring streams were cover/in-stream wood and habitat unit frequency. These two indicators are often linked with one another. Among the 62 sample streams, frequency of in-stream wood key pieces and habitat unit frequency were moderately correlated ($r$=0.45). Frequency of all pieces of in-stream wood (not just key pieces) and habitat unit frequency had a stronger correlation ($r$=0.58). This implies that habitat unit frequency is increased by an increased frequency of in-stream wood.[35]

From these patterns we can infer that the two indicators, other than stream size, that most affected overall habitat score are in-stream wood frequency and habitat unit frequency. Because habitat unit frequency is understood to be influenced by in-stream wood frequency, in-stream wood is therefore an important factor affecting the overall habitat score.

## FUTURE HABITAT MODELING

It is important to note that the model presented here is the initial attempt at habitat quality modeling. We expect that, over time, greater insight into habitat indicators and modeling methodology will lead to improvements in habitat modeling. Studying the habitat model in the context of fish data from Riparian Validation Monitoring will be a key next step in model refinement. For example, indicator categories may need to be weighted differently or calculated differently. Some indicators may need to be added or dropped. Also, there are likely yet-undiscovered stream characteristics that would further improve the model.

[35] Although correlation does not imply causation, the scientific literature and field observations confirm this relationship.

# Discussion and Implications

## Key findings

Among the wide range of habitat indicators assessed in this report, several themes emerged that describe the observed patterns in riparian function.

### IN-STREAM WOOD DYNAMICS

Results from this analysis are consistent with the concept of an interruption in the long-term input of large pieces of wood to many of the streams sampled in DNR-managed watersheds (Martens et al. 2020). Though large pieces of wood are still present in most of these streams, the majority of these pieces are in later stages of decay, reflecting an interruption of the supply of new, large pieces of wood from riparian forests. This interruption is understood to have been caused by the harvest of many riparian forests during the mid-to-late 20$^{th}$ century. In addition to interrupting the supply of large pieces of wood, historical harvest practices are likely to have reduced in-stream wood in some channels by actively cleaning it from streams. It is also probable that some in-stream wood was buried as sediment accumulated in lower-gradient streams.

A wide range of riparian forest conditions was observed in DNR-managed watersheds. Not only did the DNR-managed riparian stands vary in structure and composition, but the overstory on greater than one third of the DNR-managed plots was structurally similar to the overstory on 83% of the reference plots. The wide range of riparian forest conditions in DNR-managed watersheds indicates that any management activities prescribed to enhance riparian forest structure or inputs of wood to streams must account for these existing variations in the riparian forest.

### STREAM SHADE, WATER TEMPERATURE, AND MICROCLIMATE

Shade, above the stream and in the riparian zone, was consistently high across the monitored watersheds. In fact, the monitored riparian areas were so consistently shaded that there were no examples of how aquatic and riparian conditions might differ without shade of the forest canopy. This uniformity of riparian shade was the reason shade was not an effective predictor of water temperature or microclimate.

The high level of stream shading observed is very likely attributable to the unharvested stream buffers applied under the State Lands HCP. The level of shade currently present is even greater than what has been reported for old-growth forest (Warren et al. 2013; Kaylor et al. 2017). As a young, even-aged forest stand matures, some individual trees die over time, creating canopy gaps that are eventually filled by new trees. This process increases structural diversity in the forest canopy and results in more instances of sunlight reaching the forest floor and streams. DNR has two ongoing studies evaluating the potential for active management to accelerate the development of riparian forest structural diversity: Riparian Forest Restoration Strategy Effectiveness Monitoring and the T3 Watershed Experiment (Martens et al. 2021).

### CHANNEL MORPHOLOGY

Channel morphology, including channel shape, erosion, and streambed substrate, varied widely by channel type but collectively the habitat indicators did not show a clear directional trend over the five-year monitoring period. Because channel-forming processes and streambed composition differ widely among channel types, long-term change in habitat condition should be assessed in that context. Specifically, the pool-riffle channel type represents a low-gradient response reach that accumulates sediment transported from upstream and is more prone to channel alteration than steeper stream channels. For this reason, these reaches are expected to have been most impacted by the intensive harvest practices of the 1960s-1980s. As a result, they may also have the greatest potential for improvement over time.

## Implications for management

### STREAM BUFFERS

Although this monitoring does not evaluate every potential effect of stream buffers, the habitat indicators that were linked to the riparian forest uniformly showed beneficial effects provided by stream buffers. Forested stream buffers shaded streams, resulting in cool stream and microclimate temperatures. Stream buffers also represent an ongoing source of woody debris which is key to creating productive stream habitat. Furthermore, buffers have been shown in prior studies to substantially reduce sedimentation and erosion in managed forest landscapes (Rashin et al. 2006; Hatten et al. 2018; Rachels et al. 2020).

Given the observed benefits of the stream buffers implemented under the HCP, there are potential opportunities for further improving riparian habitat through management. These focus on two habitat benefits: increasing the rate of wood input to streams and strategically and carefully increasing the amount of sunlight reaching streams. Increasing in-stream wood is intended to alleviate the interruption in the long-term supply of in-stream wood resulting from intensive harvesting practices during the 20th century (Martens et al. 2020). Increases in sunlight are intended to boost growth of algae and microscopic plant life at the base of the aquatic food chain; such increases can ultimately benefit fish and other vertebrates. To explore management options for achieving these habitat benefits, alternative buffer management approaches will be tested experimentally as part of the T3 Watershed Experiment (Martens et al. 2021). Specifically, an experimental treatment has been designed to combine buffer thinning, streamside canopy gap creation, and the addition of wood to streams to create log jams. The goal is to add large wood to streams immediately and, by thinning the riparian forest, accelerate the development of larger trees that will eventually provide key pieces of in-stream wood in the future. Both the thinning and gaps will also allow more light to reach the stream. The experiment will study the operational feasibility of the treatment as well as its effects on many aspects of aquatic habitat and fish populations.

### PLANNING AND ENVIRONMENTAL ANALYSIS

Status and Trends Monitoring is an ongoing source of data on riparian and aquatic habitat conditions that were previously either unknown in the OESF or described only sporadically in past studies, some of which pre-date the HCP. Because Status and Trends Monitoring followed a sampling design specifically intended to represent the OESF, it now provides a key data source for future planning and environmental analysis.

The current conditions documented by this monitoring will improve the reliability and accuracy of environmental impact analyses. For example, we now understand the range of stream temperature and riparian microclimate conditions in the OESF. Future analyses of environmental impacts will not have to rely on models based on data from elsewhere but can instead use local data. In the case of stream temperature, conditions in the OESF often differed from those reported for the Cascade Range or elsewhere in the Pacific Northwest.

This study is not only determining average habitat conditions in the OESF but also the range in conditions and the factors—such as stream size, stream gradient, and elevation—that can influence habitat conditions. This information will improve habitat models created for planning purposes beyond the OESF, including projections of management influence on habitat. For example, in this report we document that many of the habitat indicators vary significantly among channel types. These differences can now be considered when evaluating sensitivity to management impacts.

## Climate change and OESF stream habitat

Projections of regional change indicate a continuing increase in air temperature through the 21$^{st}$-century (Mote et al. 2013; Abatzoglou et al. 2014; Vose et al. 2017). Projections of future precipitation are far less certain than those of air temperature but overall suggest a slight increase in average annual precipitation by the end of the century (Mote and Salathé 2010; Mote et al. 2013; Janssen et al. 2014; Easterling et al. 2017). A summary of past and projected future OESF climate is in Appendix J.

On the western Olympic Peninsula, increases in temperature are expected to have a greater ecological impact on the forested landscape than changes in precipitation. In a broad sense, because forest growth in the area is limited by energy (i.e., solar radiation) rather than by moisture availability, it is possible that tree growth rates could increase (Littell et al. 2013). However, warming temperatures are predicted to increase the frequency of disturbances resulting from insects and diseases (Agne et al. 2018), wildfires (Halofsky et al. 2018, 2020), and stress from exacerbated heat waves such as that of summer 2021 (Overland 2021). Combinations of multiple climate-induced stressors may result in the greatest impacts on forests (Littell et al. 2010). Potential effects of a changing climate on the forests of the OESF are further discussed in the OESF Forest Land Plan EIS (DNR 2016b).

Increasing air temperatures are expected to have direct effects on aquatic habitat though increases in water temperature. The association between air temperature and stream water temperature is strong enough that air temperature is often used to make predictions of stream water temperature (Mohseni and Stefan 1999; Erickson and Stefan 2000; Mantua et al. 2010). This sensitivity of stream temperature to air temperature is reflected in the significant year-to-year changes in stream temperature reported here (i.e., the large Year effect in Tables 19b and 20). This effect is not limited to summer temperatures; winter stream temperature is also sensitive to air temperature (Devine et al. 2021).

Assuming that air temperatures increase uniformly across OESF watersheds during the 21st century,[36] we expect watersheds that currently have the warmest stream temperatures to remain the warmest in the future (Devine et al. 2021). Watershed solar exposure and bedrock substrate were the strongest predictors of summer high stream temperature in our analysis of DNR-managed watersheds; therefore, streams with exposed bedrock and streams with high solar exposure—typically those with southerly aspects—are expected to warm the most under projected climate scenarios.

Relative to streams across Washington state, the streams on the western Olympic Peninsula are less likely to be driven above regulatory temperature thresholds by a warming climate (Mantua et al. 2010). The 7-day maximum water temperature for streams in DNR-managed watersheds averaged only 57.9 °F (14.4 °C) study-wide, below the 60.8 °F (16.0 °C) threshold. The primary reasons for these cool temperatures are the mild coastal climate and the shading of streams. The impacts of climate change on aquatic organisms and food webs may not necessarily result from excessive peak summer temperatures but instead may come from altered temperature patterns that can occur during any season (Steel et al. 2012). With this in mind, it is important to monitor stream temperature on a year-round basis rather than only during summer. Doing so establishes the current range of conditions and allows better understanding of which aspects of temperature are changing over time (e.g., winter low temperatures or the rate of spring warming).

In addition to changes in stream temperature, a warming climate is also expected to affect riparian microclimate over the 21st century. As with water temperature, we found that one of the strongest influences on microclimate was solar exposure. Therefore, riparian zones in watersheds with southern exposure are expected to remain the warmest as the overall climate warms.

Future stream flow in the OESF is difficult to predict due to high uncertainty in precipitation projections and few studies conducted at a sufficiently fine spatial resolution. However, studies are in general agreement that there is potential for higher precipitation intensities in the Pacific Northwest by the middle of the 21st century (Janssen et al. 2014; Easterling et al. 2017). As additional climate studies are published, the accuracy of precipitation projections is expected to improve. Should precipitation intensities increase, stream peak flow rates are also likely to increase. The frequency of flow events that exceed the critical flow rate of the channel bed (*Qc*) could increase in fine-grained channel beds (i.e., channels consisting of fine gravel and sand rather than coarse gravel and cobbles).

## IMPLICATIONS

- Forested riparian buffers have been effective in providing shade in the riparian zones of Type 3 streams. These buffers are expected to remain the best management tool available for keeping streams cool as air temperatures warm over time. Any experimental manipulation of the riparian canopy that DNR conducts will include monitoring of stream temperature to ensure there are no adverse effects on temperature.
- Stream habitat restoration activities that increase channel complexity (for example, wood additions to streams) are expected to increase the frequency of deep pools that serve as thermal refuges for juvenile salmonids during dry or hot summer conditions when streams are at their warmest and shallowest.

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[36] This may not be the case, but finer-scale climate projections are not currently available.

- Any monitoring of water temperature or riparian air temperature should be year-round so as to identify a broad range of potential temperature changes that may impact habitat.

## HCP commitments and key uncertainties

The Status and Trends Monitoring program meets several commitments in the 1997 state HCP and reduces key uncertainties identified during the development of the 2016 OESF Forest Land Plan.

The State Lands HCP calls for aquatic and riparian research in the OESF: conservation objective 5 specifies that "DNR-managed lands within the OESF shall be managed to…develop, use, and distribute information about aquatic, riparian, and associated wetland-ecosystem process and on their maintenance and restoration in commercial forests" (DNR 1997, p. IV.107). Elaborating on this objective, the HCP specifies the need for "implementation of a structured and credible program of research, experimentation, and monitoring to aid forest management and the scientific understanding of riparian systems in managed landscapes".

The effectiveness of the HCP riparian conservation strategy, and especially the one implemented in the OESF, was highly uncertain when adopted in 1997. To mitigate that uncertainty, DNR committed to monitoring to "determine whether the implementation of the conservation strategies results in anticipated habitat conditions" (DNR 1997, p. V.2). As late as 2013, when the environmental impacts of the 2016 OESF Forest Land Plan were analyzed, the lack of empirical data on the state of streams and riparian areas in the OESF was still identified as a major knowledge gap. The findings from this Status and Trends Monitoring are the first long-term, extensive, and systematic effort to fill this gap.

Adaptive management – using new information and scientific developments to continuously improve forest management practices – is another HCP commitment (DNR 1997, p. B.10). The OESF adaptive management procedure, adopted in 2016, specified the process. The monitoring and research provisions of the HCP were in part designated to identify modifications of existing practices. A key objective of the Status and Trends Monitoring program is to make inferences about management effects on habitat, thus contributing to the adaptive management required by the HCP (Minkova et al. 2012). Although no specific management changes are recommended at this stage, the findings on in-stream wood and the high, uniform levels of stream shade informed the development of an experimental riparian treatment in the T3 Watershed Experiment (Martens et al. 2021). If that experiment provides evidence for ecological effectiveness and economic feasibility of a new riparian prescription, DNR managers may then consider it an additional tool for riparian management.

The analyses and findings in this report meet the HCP research objective to "assess and improve conservation strategies that are in place" (DNR 1997, p. V.6) and specifically:

- Research Priority 2 "to determine how to harvest timber and meet conservation objectives within riparian areas" (DNR 1997, p. V.7); and
- Research Priority 3 "Develop basic information on the relationship between forest management activities and riparian ecosystems in managed forests" and "Develop basic information on the relationship between forest management activities and hydrology in managed forests (DNR 1997, p. V.8).

In this context, the research findings from Status and Trends Monitoring inform riparian conservation on all state lands and beyond, to the broader scientific community. This occurs through reports and scientific publications (Table 1) and through professional meetings, consultations, presentations, and field tours.

## Meeting the goals for the OESF

The OESF Riparian Conservation Strategy has a series of five goals outlined in the OESF Forest Land Plan (DNR 2016a, 3-22). The first four goals seek to maintain or aid restoration of aquatic and riparian systems, including aquatic species, stream channels, water quality, flow rate, and sediment regimes. The fifth goal is to “Develop, use, and distribute information about aquatic, riparian, and associated wetland-ecosystem processes and their maintenance and restoration in commercial forests.” The Riparian Conservation Strategy seeks to achieve these goals through several management strategies including stream buffers, protections of unstable slopes and landforms, road maintenance and abandonment plans, and wetland protection (DNR 2016a, 3-23). Status and Trends Monitoring assesses the cumulative results of these management strategies.

As already described in this report, benefits of interior-core stream buffers such as shade and cool stream and microclimate temperatures are already evident in the monitoring results. Other anticipated results of buffers, such as in-stream wood supply and the stream channel attributes that depend on in-stream wood are more nuanced and will require continued monitoring and other research to understand more fully.

Status and Trends Monitoring is also meeting the OESF goal of developing, using, and distributing information on aquatic and riparian ecosystem processes and their maintenance in commercial forests. In addition to publications produced so far, there is significant potential for additional analysis of data already collected (see Future Monitoring and Research below). Results from this monitoring program have been presented at the annual OESF Science Conference and at other meetings. Importantly, we are continuing to work on reducing uncertainties identified in the HCP and OESF Forest Land Plan. We now have directly measured OESF habitat information that can be used as a basis for planning and environmental assessments, whereas only a few years ago such work relied primarily on scientific work conducted elsewhere. Beyond its internal use, this information on ecological conditions and management effects also will inform conversations with stakeholders. Ultimately, this monitoring program increases the visibility of DNR and the OESF within the scientific community and demonstrates DNR’s efforts to continually improve the understanding of ecologically sound management.

## Future monitoring and research

This program, which initiated field monitoring in 2013, was originally designed to assess habitat status and then monitor trends in habitat conditions over 10 years (see Appendix A for the project budget). Habitat status is reported in the Habitat Status report (Minkova and Devine 2016) and in this report, and we are currently in the middle of the 10-year trends monitoring period. Below, we discuss the future of the monitoring program in light of what we have learned to date, in terms of methodology and findings.

## DURATION OF MONITORING

To estimate how long it would be necessary to monitor an indicator metric before we can confidently state whether or not that metric is changing, we must first know the temporal variability of the metric. Using the data collected during the first five-year monitoring interval, we were able to perform a statistical power analysis to make such estimates (Appendix H). The analysis shows, for each indicator metric, how many years must pass at a specified rate of change, before the cumulative change is likely to be detected by Status and Trends Monitoring. Here, we define "likely" as an 80% probability, which is commonly used in ecological research and monitoring.

Among the different indicator metrics, there were large differences in the time until change could be detected. These differences arose because the capacity to detect change over time in a metric is a result of both the magnitude of the expected change and of the variability of that indicator over time. Variability can result from year-to-year, watershed-specific fluctuations in an indicator or it can result from measurement error.

Using a constant, hypothetical rate of change of one percent per year, we can compare the indicator metrics to evaluate how many years of monitoring would be necessary to detect that change (Table H-1 in Appendix H). The indicator metrics for which change can most rapidly be detected—in three or fewer years—are canopy closure and stream temperature. We attribute this to the high level of precision with which these two metrics are measured (digital canopy photos and electronic temperature data loggers, respectively). It is important to note that although changes in stream temperature can be detected after a short period of monitoring, large year-to-year variation in stream temperature is natural due to climate variability and does not necessarily reflect long-term trends.

Still assuming a one percent annual rate of change, five indicator metrics would need to be monitored for more than two decades before reaching the 80% threshold for probability of detecting change. These indicator metrics are: frequency of in-stream wood pieces, frequency of pools, pool area, median substrate diameter, percent fines in substrate, and bank erosion. All of these indicator metrics were highly variable over the first five-year monitoring interval, which is the reason that a long monitoring period would be necessary to detect potential changes.

Interestingly, three of the five indicator metrics with the longest projected monitoring requirement —frequency of in-stream wood pieces, frequency of pools, and bank erosion—showed a significant change after only five years (see Table H-1 in Appendix H and the Status and Trends of Habitat Indicators section of this report). In all three cases, the observed rate of change thus far was well above the hypothetical 1%. What we do not know is whether change in these metrics will continue in the long term in the same direction or whether the observed five-year change is part of a shorter-term fluctuation.

The results of this statistical power analysis were a factor in determining whether to make modifications to the existing monitoring protocol (see next section).

## PROTOCOL CHANGES

This section documents changes to be made in the monitoring protocol (Minkova and Foster 2017) for future Status and Trends Monitoring. These recommended changes are based on the results of our power analysis (Appendix H), our quality control analysis (Devine and Minkova 2016), and other practical factors such as the expense of measuring each indicator. Due to the long-term nature of this monitoring program, any changes in the protocol must be carefully considered. Changing the way that an indicator is measured means that data collected before and after the change may not be comparable.

### *In-stream wood*

Recommendation: maintain current monitoring of in-stream wood.

The quality control and power analyses found relatively high variability in this indicator among survey crews (i.e., measurement error) and over time, respectively. The in-stream wood survey protocol has undergone several minor revisions since the monitoring program was initiated. However, it has been optimized to the extent possible, and there are no additional recommendations for protocol changes at this time.

### *Channel habitat units/pool habitat*

Recommendation: maintain current protocol.

### *Channel substrate*

Recommendation: prior to the 2022 field season, researchers will meet to discuss objectives for future substrate sampling and select the most appropriate sampling protocol to meet those objectives. Two potential monitoring approaches are: monitoring to detect substrate change due to land use impacts (i.e., the current approach) or monitoring to determined substrate suitability for spawning salmonids (Kondolf 2000; Sutherland et al. 2010).

The existing channel substrate protocol is designed to characterize the particle size distribution of channel substrate at the sample reach level. The protocol appears useful in this regard but less so for detecting change over time. The quality control analysis indicated that the measurement is not very repeatable: different field crews are likely to get different results. The power analysis indicates that median particle diameter is not likely to change to a detectable degree over time (Appendix H). Additionally, this protocol is relatively time-consuming to implement.

An alternative sampling approach to consider is one that focuses on spawning gravel within the sample reach. However, if the protocol were changed to spawning gravel, then results before and after the change would not be comparable. Also, monitoring long-term change in spawning gravel conditions may not be feasible, as a new protocol would be implemented in 2022, already the tenth year of habitat monitoring. Thus, data collected under a spawning gravel protocol would not be used for detecting change over time. Instead, its habitat value would be interpreted by comparing our results with the results of other studies. For this reason, we should only adopt a spawning gravel protocol if it is well standardized and there are sufficient examples in the scientific literature with which we could compare our results.

### *Channel morphology*

Recommendation: maintain current monitoring protocol; consider refining erosion survey; emphasize bankfull identification during training.

Channel morphology data collection consists primarily of cross-section surveys and erosion surveys. Cross-section surveys are relatively repeatable, with the exception of bankfull identification which is prone to variation among field crews. Thus, an increased emphasis should be placed on bankfull identification during training. Erosion surveys are perennially challenging to implement consistently, but there may be room for improvement, for example with photographic field references of what is, and is not, active erosion.

***Riparian vegetation***

Recommendation: when riparian vegetation is surveyed again, add down wood sampling to the protocol.

Riparian vegetation was sampled to characterize the riparian forest at each of the sample watersheds. Because in-stream wood dynamics and the contribution of wood from the riparian forest are of key interest, future sampling of riparian vegetation should include down wood surveys.

***Stream shade***

Recommendation: continue current protocol.

The power analysis indicates that stream shade can be measured relatively precisely. No changes are recommended to the current protocol. We will continue to explore the use of remotely sensed data (e.g., LiDAR) for measuring riparian canopy cover.

***Stream temperature***

Recommendation: maintain current monitoring of stream temperature.

Stream temperature can be measured very precisely, and it is a key variable for monitoring effects of management and climate change on stream habitat.

***Riparian microclimate***

Recommendation: at this time, further riparian microclimate sampling is not planned.

Riparian microclimate was monitored continuously for three years. No additional sampling is planned at this time.

***Stream flow***

Recommendation: reduce number of gages to three or four gages, or hire a field technician based in Forks to help collect hydraulic data.

Typical hydrology monitoring programs employ one full time technician for every 11 to 17 gages and employ a data analyst for data processing and analysis (Norris et al. 2008). The OESF program employs less than one full-time employee for hydrology monitoring (hydrology program maintained by roughly half of the State Land Hydrologist position). The OESF flow monitoring protocol has evolved over time to utilize techniques that require fewer field visits and field time. It now differs significantly from the initial protocol draft for the monitoring program (Minkova and Foster 2017); however, if additional resources cannot be allotted to the hydrology monitoring program, it may be necessary to abandon all but three to four gages. If additional resources can be allotted to the hydrology monitoring program (e.g., a field technician stationed in Forks collects hydrologic and hydraulics data), operation of most of the current gages can continue.

## FUTURE MONITORING

In 2025, the 10th-year habitat surveys will be completed. At that time, an analysis of all habitat indicators will be conducted and a 10-year report will be published. Based on the results of our statistical power analysis, we anticipate that after 10 years of monitoring we will have much higher confidence in the existence or absence of indicator trends. Plans for continued monitoring in these same watersheds beyond 2025 will be discussed at that time.

## FUTURE ANALYSES

Using data collected to date in Status and Trends Monitoring, there is significant potential for additional analyses, at a level of detail that is beyond the scope of this five-year report, which only included a subset of possible indicator metrics. Potential topics identified for future analyses are listed in Table 25.

**Table 25.** Anticipated topics of future research, based on data collected during Status and Trends Monitoring; this is not a comprehensive list.

| Subject | Key concept | Application |
|---|---|---|
| - - - - - - - - - - - - - - - - - - - - *Interactions among habitat indicators and fish* - - - - - - - - - - - - - - - - - - - - | | |
| Clarifying relationships among habitat indicators. | Relationships among habitat indicators such as in-stream wood, pools, and the riparian forest are understood in a broad sense, but much still remains to be learned about how these habitat features interact in the OESF. | Riparian management will be most effective when the cause-and-effect relationships among various aspects of habitat are better quantified. |
| Linking habitat conditions to fish populations. | Status and Trends Monitoring and Riparian Validation Monitoring (Martens 2016) provide complimentary information on stream habitat conditions and on utilization of that habitat by fish. | Identifying the relationships between fish populations and stream habitat conditions will inform DNR's riparian management. |
| - - - - - - - - - - - - - - - - - - - - - *Management influences on stream habitat* - - - - - - - - - - - - - - - - - - - - - | | |
| Monitoring habitat during management activities | Status and Trends Monitoring is intended to identify long-term changes in habitat conditions resulting from management under the HCP; today, habitat conditions reflect the residual effects of pre-HCP management plus any potential effects of current management. | The Status and Trends Monitoring program supplements the standard sampling schedule with surveys of streams in watersheds where harvest is planned or has recently been completed. This provides documentation of conditions before and after harvest. |

| Subject | Key concept | Application |
|---|---|---|
| Understanding ongoing changes in channel morphology. | Natural processes cause changes in channel morphology over time; however, it is not clear how these natural processes are interacting with the residual effects of past management to produce current channel morphology. | By understanding how natural processes interact with the residual impacts of historical management, we can better gauge the current condition of streams.. |
| - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - *Hydrology* - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - | | |
| Clarify if the frequency of channel-disrupting peak flows will change in the future | While changes in peak flow rates caused by changes in temperature are not expected to be large in the OESF (Safeeq et al. 2015) changes due to increasing precipitation intensity are possible and could increase the frequency of extreme high flows. | The hydrograph is used to calibrate a hydrology model. The hydrology model is run with projected climate data to predict future flow rates. From the predicted flow rates and existing OESF streambed particle size data, temporal trends in the frequency of bed-disrupting flows can be assessed and management activities (e.g., harvest schedule, number of roads and road crossing, road runoff design) can be tailored to mediate those effects if necessary. Insight from the OESF models can be used to inform harvest-planning elsewhere in the state. |
| Clarify if the frequency and nature of the summer-low flow will change in the future | Evapotranspiration rates may increase and thus reduce the water available to sustain summer low flows. | The hydrograph is used to calibrate a hydrology model. The hydrology model is run with projected climate data (including future solar radiation levels) to predict evapotranspiration rates and summer low flows for different land use scenarios. Management activities (timing and location of harvests) can be tailored to mediate harvest impacts on summer-low lows if necessary. Insight from the OESF models can be used to inform harvest-planning elsewhere in the state. |
| Flood frequency in the OESF | Precipitation rates and runoff rates may vary across the OESF | Estimates of flood frequency are needed to properly designing stream crossings. Flood-frequency curves can be created for the observed OESF hydrology watersheds (with uncertainty limits) and used by engineers in Olympic Region. |

| Subject | Key concept | Application |
|---|---|---|
| Clarify how sediment inputs (landslides, surface erosion) impact future aquatic habitat. | Sediment generated by landslides and surface runoff impacts aquatic habitat but the transfer of that sediment through a watershed may take decades to centuries. Logging practices in the 1970s and 80s introduced massive amounts of sediment into OESF streams. Little is known about the effects of past sediment inputs on present day aquatic habitat or how mistakes we make today will affect aquatic habitat in the future. | The hydrograph is used to calibrate a hydrology model of each watershed. The calibrated hydrology model is then used to drive a geomorphology model. The geomorphology model estimates how sediment production by roads and landslides is routed through a watershed and impact the channel bed in lowland streams. Different harvest scenarios, under different climate scenarios, can be assessed. Methodology developed on sediment routing studies in the OESF could be applied to studies of other sensitive and high-interest watersheds in the state. |
| Clarify trends in the seasonality, duration and magnitude of extreme flow rates. | Extreme high and low flows can be used as a proxy for understanding trends in aquatic habitat. | Storm and low-flow events are extracted from the observed hydrographs and used to quantify trends in the seasonality (timing), duration and magnitude of the events. Alternatively, a hydrology model can be calibrated to the observed hydrograph and the same analysis can be applied to modeled hydrographs of much longer lengths from different climate and land-use scenarios. |

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# Appendix

## Appendix A. Project budget

This monitoring project is funded and implemented primarily by DNR, specifically by the Forest Resources Division, Habitat Conservation Plan and Scientific Consultation section. The funding varied over the 9 years of the project, averaging $163,000 per calendar year. Additional funding of $68,731 was provided from the Good Neighbor Authority agreement with Olympic National Forest in 2018, 2019 and 2020 to monitor six reference watersheds on the national forest.

The USDA Forest Service Pacific Northwest Research Station has been a research partner since the beginning of the study in 2012. It contributed $18,000 for equipment in 2012, $1,000 per year since then, and in-kind support in the form of research and field staff time in each year of the project. The scientific expertise and field support are estimated at about 600 hours per year including development and refinement of field methods, review of reports, presentations and publications, training of field technicians, and fieldwork.

The project expenses are presented in Table A-1. A majority of the expenses covered DNR researchers' and field technicians' time (40% and 36% respectively), followed by data management (20%), and field equipment and supplies (20%).

**A-1.** Project budget for calendar years 2012 through 2020.

| Item | Annual cost ($) | | | | | | | | | Total cost ($) |
|---|---|---|---|---|---|---|---|---|---|---|
| | **2012** | **2013** | **2014** | **2015** | **2016** | **2017** | **2018** | **2019** | **2020** | |
| DNR researchers | 83,058 | 73,829 | 64,601 | 74,525 | 75,096 | 57,802 | 57,802 | 60,670 | 60,670 | 608,051 |
| Field technicians and interns | 46,743 | 70,115 | 70,115 | 82,556 | 70,043 | 58,994 | 51,216 | 51,216 | 51,216 | 552,215 |
| Data manager | | | 36,131 | 74,496 | 74,496 | 48,000 | 28,800 | 20,223 | 20,223 | 302,369 |
| Field equipment and supplies | 4,000 | 24,000 | 4,000 | 3,500 | 2,000 | 2,000 | 8,000 | 4,700 | 2,400 | 54,600 |
| Study reviews and consultations | 6,000 | 7,000 | | | | | | | | 13,000 |
| Conferences/ publication fees | | | | 1,000 | | | | | | 1,000 |
| Total | 139,801 | 174,944 | 174,846 | 236,077 | 221,635 | 166,796 | 145,818 | 136,809 | 134,509 | 1,531,235 |

Direct cost comparisons with similar monitoring projects are difficult, owning to the lack of readily available data, differences in the geographic scope and field protocols, and the differences in accounting for direct and in-kind contributions. Nevertheless, a review of the long-term status and trends stream monitoring projects conducted by Washington Department of Ecology, Washington Forest Practices Adaptive Management Program, Aquatic and Riparian Effectiveness Monitoring

Program for the Northwest Forest Plan, and USFS Pacfish/Infish Monitoring Program, as well as consultations with colleagues working on these projects, indicate high efficiency of our monitoring program. The main factors for this efficiency are the in-house implementation of the project, the fact that most of the monitored watersheds are managed by DNR, and effective project management including planning, implementation, and communication.

# Appendix B. Stream type definitions

The stream type definitions presented here are reproduced from the Glossary of the OESF Forest Land Plan (DNR 2016a).

**Stream type:** On state trust lands in western Washington, DNR State Lands uses a numerical system (one through five) to categorize streams based on their physical characteristics such as stream width, steepness, and whether or not fish are present. Type 1 streams are the largest, Type 5 streams are the smallest. DNR and the Federal Services (NOAA Fisheries and USFWS) have agreed that the Washington Forest Practices Board Emergency Rules (stream typing), November 1996 meet the intent of DNR's HCP. Following are the emergency rules.

"Type 1 Water" means all waters, within their ordinary high-water mark, inventoried as "shorelines of the state" under Chapter 90.58 RCW and the rules promulgated pursuant to Chapter 90.58 RCW, but not including those waters' associated wetlands as defined in Chapter 90.58 RCW.

"Type 2 Water" shall mean segments of natural waters that are not classified as Type 1 Water and have a high fish, wildlife, or human use. These are segments of natural waters and periodically inundated areas of their associated wetlands, which:

a. Are diverted for domestic use by more than 100 residential or camping units or by a public accommodation facility licensed to serve more than 100 persons, where such diversion is determined by the Department to be a valid appropriation of water and the only practical water source for such users. Such waters shall be considered to be Type 2 Water upstream from the point of such diversion for 1,500 feet or until the drainage area is reduced by 50 percent, whichever is less;

b. Are diverted for use by federal, state, tribal or private fish hatcheries. Such waters shall be considered Type 2 Water upstream from the point of diversion for 1,500 feet including tributaries if highly significant for protection of downstream water quality. The Department may allow additional harvest beyond the requirements of Type 2 Water designation, provided the Department determines after a landowner-requested on-site assessment by the Department of Fish and Wildlife, Department of Ecology, the affected tribes, and the interested parties that:

    (i) The management practices proposed by the landowner will adequately protect water quality for the fish hatchery; and

    (ii) Such additional harvest meets the requirements of the water type designation that would apply in the absence of the hatchery;

c. Are within a federal, state, local, or private campground having more than 30 camping units: Provided that the water shall not be considered to enter a campground until it reaches the boundary of the park lands available for public use and comes within 100 feet of a camping unit, trail or other park improvement;

d. Are used by substantial numbers of anadromous or resident game fish for spawning, rearing or migration. Waters having the following characteristics are presumed to have highly significant fish populations:

(i) Stream segments having a defined channel 20 feet or greater in width between the ordinary high-water marks and having a gradient of less than 4 percent.

(ii) Lakes, ponds, or impoundments having a surface area of 1 acre or greater at seasonal low water.

e. Are used by salmonids for off-channel habitat. These areas are critical to the maintenance of optimum survival of juvenile salmonids. This habitat shall be identified based on the following criteria:

(i) The site must be connected to a stream bearing salmonids and accessible during some period of the year; and

(ii) The off-channel water must be accessible to juvenile salmonids through a drainage with less than a 5% gradient.

"Type 3 Water" shall mean segments of natural waters that are not classified as Type 1 or 2 Water and have a moderate to slight fish, wildlife, and human use. These are segments of natural waters and periodically inundated areas of their associated wetlands which:

a. Are diverted for domestic use by more than 10 residential or camping units or by a public accommodation facility licensed to serve more than 10 persons, which such diversion is determined by the Department to be a valid appropriation of water and the only practical water source for such users. Such waters shall be considered to be Type 3 Water upstream from the point of diversion for 1,500 feet or until the drainage area is reduced by 50 percent, whichever is less;

b. Are used by significant numbers of anadromous or resident game fish for spawning, rearing or migration. If fish use has not been determined:

(i) Waters having the following characteristics are presumed to have significant anadromous or resident game fish use:

(A) Stream segments having a defined channel of 2 feet or greater in width between the ordinary high-water marks in western Washington and having a gradient 16 percent or less;

(B) Stream segments having a defined channel of 2 feet or greater in width between the ordinary high-water marks in Western Washington and having a gradient greater than 16 percent and less than or equal to 20 percent; and having greater than 50 acres in contributing basin size in western Washington;

(ii) The Department shall waive or modify the characteristics in (i) above where:

(A) Waters are confirmed, long-term, naturally occurring water quality parameters incapable of supporting anadromous or resident game fish;

(B) Snowmelt streams have short flow cycles that do not support successful life history phases of anadromous or resident game fish. These streams typically have no flow in the winter months and discontinue flow by June 1; or

(C) Sufficient information about a geographic region is available to support a departure from the characteristics in (i), as determined in consultation with the Department of Fish and Wildlife, Department of Ecology, affected tribes, and interested parties.

(iii) Ponds or impoundments having a surface area of less than 1 acre at seasonal low water and having an outlet to an anadromous fish stream.

(iv) For resident game fish ponds or impoundments having a surface are greater than 0.5 acre at seasonal low water.

c. Are highly significant for protection of downstream water quality. Tributaries which contribute greater than 20 percent of the flow to a Type 1 or 2 Water are presumed to be significant for 1,500 feet from their confluence with the Type 1 or 2 Water or until their drainage area is less than 50 percent of their drainage area at the point of confluence, whichever is less.

"Type 4 Water" classification shall be applied to segments of natural waters which are not classified as Type 1, 2 or 3, and for the purpose of protecting water quality downstream are classified as Type 4 Water upstream until the channel width becomes less than 2 feet in width between the ordinary high-water marks. Their significance lies in their influence on water quality downstream in Type 1, 2, and 3 Waters. These may be perennial or intermittent.

"Type 5 Water" classification shall be applied to all natural waters not classified as Type 1, 2, 3, or 4; including streams with or without well-defined channels, areas of perennial or intermittent seepage, ponds, natural sinks and drainage ways having short periods of spring or storm runoff.

## Appendix C. Summary of watersheds

**Table C-1.** Summary data for 50 monitored DNR-managed watersheds.

| Watershed | Area (ac) | Median slope (%) | Minimum elevation (ft) | Maximum elevation (ft) | Median elevation (ft) | Managed by DNR (% of watershed) |
|---|---|---|---|---|---|---|
| 145 | 450 | 16 | 88 | 880 | 533 | 96.3 |
| 157 | 471 | 23 | 230 | 1449 | 814 | 100.0 |
| 158 | 522 | 27 | 223 | 1454 | 856 | 100.0 |
| 165 | 1653 | 38 | 247 | 1959 | 850 | 100.0 |
| 196 | 1121 | 38 | 239 | 1784 | 999 | 52.4 |
| 328 | 227 | 18 | 451 | 1252 | 662 | 94.1 |
| 433 | 1617 | 4 | 125 | 715 | 262 | 68.1 |
| 443 | 385 | 20 | 140 | 499 | 263 | 50.7 |
| 488 | 318 | 33 | 459 | 1289 | 854 | 55.7 |
| 542 | 369 | 17 | 223 | 1224 | 767 | 100.0 |
| 544 | 117 | 20 | 290 | 1225 | 766 | 100.0 |
| 545 | 77 | 23 | 330 | 1168 | 723 | 100.0 |
| 550 | 366 | 7 | 380 | 801 | 685 | 66.8 |
| 567 | 337 | 11 | 334 | 1058 | 561 | 100.0 |
| 568 | 443 | 12 | 297 | 1045 | 632 | 100.0 |
| 582 | 175 | 18 | 293 | 1080 | 527 | 100.0 |
| 584 | 982 | 20 | 298 | 1175 | 709 | 100.0 |
| 597 | 818 | 24 | 356 | 1179 | 629 | 68.3 |
| 605 | 86 | 14 | 101 | 551 | 360 | 100.0 |
| 619 | 248 | 25 | 492 | 2608 | 906 | 100.0 |
| 621 | 169 | 60 | 484 | 2822 | 1853 | 100.0 |
| 625 | 474 | 55 | 469 | 2938 | 1641 | 100.0 |
| 637 | 302 | 46 | 398 | 2291 | 1589 | 100.0 |
| 639 | 303 | 64 | 656 | 3012 | 1950 | 100.0 |
| 642 | 442 | 5 | 488 | 1897 | 642 | 100.0 |
| 653 | 144 | 65 | 705 | 2421 | 1521 | 100.0 |
| 658 | 670 | 8 | 433 | 1180 | 574 | 80.5 |
| 687 | 687 | 58 | 805 | 2940 | 2003 | 100.0 |
| 688 | 561 | 4 | 165 | 1055 | 377 | 70.7 |
| 690 | 1088 | 50 | 705 | 2937 | 1786 | 100.0 |

| Watershed | Area (ac) | Median slope (%) | Minimum elevation (ft) | Maximum elevation (ft) | Median elevation (ft) | Managed by DNR (% of watershed) |
|---|---|---|---|---|---|---|
| 694 | 529 | 54 | 860 | 2811 | 1765 | 100.0 |
| 716 | 897 | 57 | 1172 | 3406 | 2271 | 100.0 |
| 717 | 135 | 44 | 587 | 1659 | 1035 | 100.0 |
| 718 | 611 | 18 | 560 | 2446 | 889 | 100.0 |
| 724 | 164 | 44 | 550 | 1679 | 1172 | 100.0 |
| 730 | 904 | 39 | 389 | 1579 | 908 | 87.0 |
| 737 | 148 | 64 | 1184 | 2745 | 2094 | 100.0 |
| 750 | 328 | 57 | 1284 | 2965 | 2173 | 100.0 |
| 760 | 254 | 34 | 272 | 1050 | 551 | 100.0 |
| 763 | 457 | 30 | 273 | 1482 | 711 | 76.8 |
| 767 | 77 | 25 | 324 | 1274 | 753 | 100.0 |
| 769 | 38 | 37 | 309 | 1132 | 681 | 100.0 |
| 773 | 405 | 53 | 630 | 2175 | 1394 | 100.0 |
| 776 | 195 | 46 | 705 | 2143 | 1370 | 100.0 |
| 790 | 839 | 39 | 264 | 1270 | 778 | 100.0 |
| 796 | 1312 | 15 | 187 | 1994 | 600 | 97.0 |
| 797 | 1151 | 14 | 219 | 2269 | 671 | 73.6 |
| 804 | 427 | 22 | 644 | 1411 | 1150 | 100.0 |
| 820 | 1430 | 5 | 118 | 1289 | 412 | 88.5 |
| 844 | 918 | 4 | 131 | 483 | 374 | 98.0 |

**Table C-2.** Summary data for 12 monitored reference watersheds.

| Watershed | Area (ac) | Median slope (%) | Minimum elevation (ft) | Maximum elevation (ft) | Median elevation (ft) | Managed by DNR (% of watershed) |
|---|---|---|---|---|---|---|
| BOG[1] | 630 | 50 | 374 | 2183 | 1272 | 0.0 |
| HOH[1] | 219 | 68 | 689 | 4225 | 2805 | 0.0 |
| QUE[1] | 322 | 37 | 301 | 2323 | 1617 | 19.5 |
| SFH[1] | 291 | 69 | 754 | 3766 | 2572 | 0.0 |
| 566[2] | 65 | 39 | 538 | 1260 | 736 | 100.0 |
| 744[2] | 212 | 21 | 389 | 1081 | 743 | 100.0 |
| NF4[3] | 531 | 27 | 505 | 1292 | 827 | 0.0 |
| NF5[3] | 217 | 26 | 785 | 1432 | 1085 | 0.0 |
| NF6[3] | 153 | 27 | 566 | 1359 | 860 | 0.0 |
| NF7[3] | 243 | 38 | 617 | 1732 | 1061 | 0.0 |
| NF8[3] | 189 | 41 | 708 | 1765 | 1159 | 0.0 |
| NF9[3] | 591 | 31 | 633 | 1559 | 1004 | 0.0 |

[1] Located in Olympic National Park
[2] Located on DNR-managed land
[3] Located in Olympic National Forest

# Appendix D. Predictor data

**Table D-1.** Predictor data used in analysis of 50 monitored DNR-managed watersheds; other variables used as predictors are in Appendices C and E. See Table D-3 for the methods used to calculate these predictors.

| Watershed | Solar exposure (kWh/m$^2$) | Unharvested forest (%) | Canopy closure (%) | Bedrock substrate (%) |
|---|---|---|---|---|
| 145 | 4.94 | 3.20 | 89.3 | 0.1 |
| 157 | 4.72 | 10.14 | 95.5 | 24.4 |
| 158 | 4.78 | 15.16 | 95.4 | 0.6 |
| 165 | 4.67 | 4.41 | 88.0 | 37.2 |
| 196 | 4.63 | 6.71 | 90.1 | 23.3 |
| 328 | 4.85 | 29.47 | 95.1 | 0.8 |
| 433 | 4.97 | 4.47 | 95.8 | 0.0 |
| 443 | 4.89 | 4.50 | 95.9 | 4.7 |
| 488 | 4.66 | 46.46 | 93.1 | 2.6 |
| 542 | 5.12 | 48.27 | 95.7 | 0.0 |
| 544 | 5.24 | 0.26 | 94.9 | 0.0 |
| 545 | 5.21 | 0.13 | 97.0 | 0.0 |
| 550 | 5.11 | 7.13 | 95.1 | 0.0 |
| 567 | 4.89 | 22.66 | 94.5 | 0.0 |
| 568 | 4.92 | 36.85 | 93.1 | 0.0 |
| 582 | 5.10 | 68.00 | 100.0 | 5.6 |
| 584 | 4.97 | 35.84 | 97.1 | 0.1 |
| 597 | 4.85 | 16.48 | 94.3 | 0.0 |
| 605 | 4.81 | 0.12 | 96.8 | 0.0 |
| 619 | 4.36 | 41.15 | 93.9 | 0.0 |
| 621 | 4.07 | 33.97 | 95.2 | 0.0 |
| 625 | 4.18 | 38.39 | 96.0 | 1.5 |
| 637 | 4.46 | 22.85 | 100.0 | 0.0 |
| 639 | 4.06 | 41.07 | 87.7 | 0.2 |
| 642 | 4.62 | 1.03 | 97.2 | 0.0 |
| 653 | 3.89 | 16.78 | 92.0 | 2.6 |
| 658 | 5.09 | 26.96 | 96.0 | 0.0 |
| 687 | 4.87 | 10.83 | 96.2 | 0.3 |
| 688 | 4.92 | 9.33 | 92.4 | 0.0 |

| Watershed | Solar exposure (kWh/m$^2$) | Unharvested forest (%) | Canopy closure (%) | Bedrock substrate (%) |
|---|---|---|---|---|
| 690 | 5.01 | 2.38 | 67.1 | 1.8 |
| 694 | 4.96 | 28.04 | 97.0 | 8.8 |
| 716 | 4.42 | 34.43 | 92.8 | 2.6 |
| 717 | 4.69 | 0.00 | 93.1 | 0.0 |
| 718 | 5.10 | 7.66 | 97.9 | 0.0 |
| 724 | 4.92 | 17.56 | 94.6 | 14.6 |
| 730 | 4.74 | 16.75 | 95.7 | 7.3 |
| 737 | 4.93 | 29.12 | 99.7 | 0.0 |
| 750 | 4.39 | 58.50 | 91.6 | 3.9 |
| 760 | 4.85 | 0.12 | 97.7 | 0.0 |
| 763 | 4.84 | 5.75 | 92.4 | 0.0 |
| 767 | 5.16 | 21.28 | 97.9 | 0.0 |
| 769 | 5.10 | 28.80 | 94.3 | 0.0 |
| 773 | 4.37 | 26.38 | 93.6 | 1.7 |
| 776 | 4.37 | 58.74 | 96.9 | 6.1 |
| 790 | 4.62 | 11.94 | 89.4 | 4.1 |
| 796 | 5.20 | 45.12 | 89.1 | 0.0 |
| 797 | 5.15 | 33.83 | 85.3 | 0.0 |
| 804 | 4.89 | 6.55 | 94.8 | 0.3 |
| 820 | 5.08 | 37.61 | 91.7 | 0.0 |
| 844 | 4.97 | 6.56 | 92.3 | 0.0 |

**Table D-2.** Predictor data used in analysis of 12 reference watersheds; other variables used as predictors are in Appendices C and E. See Table D-3 for the methods used to calculate these predictors.

| Watershed ID | Solar exposure ($kWh/m^2$) | Unharvested forest (%) | Canopy closure (%) | Bedrock substrate (%) |
|---|---|---|---|---|
| BOG[1] | 4.84 | 100.00 | 87.1 | 15.0 |
| HOH[1] | 5.26 | 100.00 | 91.6 | 0.0 |
| QUE[1] | 5.04 | 98.60 | 97.7 | 0.0 |
| SFH[1] | 4.99 | 100.00 | 89.0 | 0.0 |
| 566[2] | 5.05 | 82.74 | 96.3 | 11.1 |
| 744[2] | 5.06 | 79.90 | 95.9 | 1.0 |
| NF4[3] | 4.90 | 100.00 | 88.3 | 2.1 |
| NF5[3] | 4.73 | 98.78 | 90.5 | 0.0 |
| NF6[3] | 5.05 | 100.00 | 95.4 | 0.5 |
| NF7[3] | 4.47 | 96.64 | 87.0 | 0.0 |
| NF8[3] | 4.58 | 100.00 | 94.5 | 0.0 |
| NF9[3] | 4.83 | 92.06 | 89.5 | 1.3 |

[1] Located in Olympic National Park
[2] Located on DNR-managed land
[3] Located in Olympic National Forest

**Table D-3.** Methodology for the variables describing watersheds and sample reaches.

| Variable | Explanation |
|---|---|
| Unharvested forest (estimated % of watershed) | Throughout this report, the percentage unharvested forest in each of the 50 DNR-managed watershed was calculated as the percentage of the watershed with forest originating prior to the year 1930, according to the Combined Origin Year dataset produced by DNR's Forest Resources Division Informatics section. Stands with an origin date prior to 1930 are assumed to have not been commercially harvested as evidenced by harvest history in the vicinity of these selected watersheds. Harvest did occurred in the OESF prior to 1930, but given the locations of the selected watersheds, we estimate that very little, if any, harvest occurred in these watersheds prior to 1930.<br><br>For the 12 reference watersheds, most of which were on federal lands that were not covered by the Combined Origin Year dataset, we manually measured the area that had been harvested by using a combination of aerial imagery and a LiDAR-derived vegetation height raster (produced by DNR's Forest Resources Division Informatics section). Areas previously harvested appeared distinctly different in terms of forest canopy structure, compared with those never harvested. |
| Bankfull width | Measured directly during every stream survey at six evenly spaced cross-sections within the sample reach. When used as a covariate in analysis, bankfull width was averaged across years for each stream. |
| Watershed median slope | Median slope for each watershed was derived from a USGS 10-m DEM raster grid using ArcGIS (version 10.5). |
| Watershed area | The area of each watershed was calculated using a pour-point analysis in ArcGIS (version 10.5) Spatial Analyst. A LiDAR-derived DEM with a 3-foot horizontal resolution was used. Vertical resolution was 1 foot. |
| Elevation of sample reach | Sample reach elevation was measured in ArcGIS (version 10.5), using a LiDAR-derived DEM with a 3-foot horizontal resolution. Vertical resolution was 1 foot. |
| Canopy closure ("stream shade") | Canopy closure was measured by hemispherical photography at six locations along each stream sample reach. At each location, the camera was mounted on a tripod at a height of 4.5 ft (1.37 m) above the center of the stream bed, with the lens oriented directly upward. Using a fish-eye lens, a photo was taken of the forest canopy at each location. This photo was then processed using Hemisfer software[37], and each pixel was identified as either shaded or unshaded. The sums of shaded and total pixel counts were used to calculate a percent canopy closure value for each photo (i.e., shaded pixels divided by total pixels, multiplied by 100). Finally, this value was converted to a percent shade value equivalent to the value one would get using a spherical densiometer (as described in Martens et al. 2019). |

[37] Patrick Schleppi, Swiss Federal Institute for Forest, Snow and Landscape Research.

| Variable | Explanation |
|---|---|
| Solar exposure | In ArcGIS (version 10.5), the Area Solar Radiation tool was used to model incoming solar radiation (kWh/$m^2$) across a USGS 10-m DEM raster grid of the OESF. The calculation uses aspect, slope, latitude, elevation, sun angle, atmospheric transmissivity, proportion of radiation that is diffuse, and topographic shading (Fu and Rich 2003). Solar radiation was modeled for three dates (the summer solstice and 30 and 60 days after the solstice) and then averaged across these dates. Finally, the 62 monitoring watersheds were clipped from the larger raster and solar radiation was averaged by watershed to produce the values in Tables D-1 and D-2. |
| Bedrock substrate (% of sample reach) | Bedrock substrate was sampled as part of the stream survey; see the Channel Substrate section for sampling methodology. When used as a covariate in analysis, percent bedrock substrate was averaged across years for each stream. |

# Appendix E. Summary data for sample reaches

**Table E-1.** Summary data for 50 DNR-managed sample reaches. For information on how the data were derived, see Minkova and Foster (2017).

| Watershed ID | Channel type | Elevation (ft) | Aspect | Gradient (%) | Mean bankfull width (ft) | Mean bankfull depth (in.) | Length (ft) |
|---|---|---|---|---|---|---|---|
| 145 | Step-pool | 93 | NW | 4.1 | 15.8 | 7.5 | 361 |
| 157 | Step-pool | 250 | E | 4.0 | 13.2 | 7.8 | 328 |
| 158 | Cascade | 245 | N | 8.1 | 13.7 | 7.7 | 328 |
| 165 | Step-pool | 268 | N | 2.8 | 32.6 | 13.3 | 623 |
| 196 | Step-pool | 280 | N | 4.6 | 24.1 | 10.8 | 486 |
| 328 | Pool-riffle | 470 | W | 2.6 | 9.7 | 5.8 | 328 |
| 433 | Pool-riffle | 120 | NW | 1.4 | 30.7 | 17.5 | 525 |
| 443 | Pool-riffle | 150 | SW | 1.7 | 12.7 | 6.4 | 328 |
| 488 | Pool-riffle | 460 | N | 4.1 | 12.1 | 6.5 | 354 |
| 542 | Step-pool | 225 | S | 7.1 | 18.9 | 13.4 | 344 |
| 544 | Step-pool | 297 | S | 6.3 | 8.8 | 9.0 | 328 |
| 545 | Step-pool | 331 | SW | 6.7 | 6.8 | 4.0 | 328 |
| 550 | Step-pool | 404 | SW | 7.1 | 22.0 | 10.1 | 394 |
| 567 | Step-pool | 338 | N | 5.5 | 19.8 | 9.6 | 328 |
| 568 | Step-pool | 298 | NW | 4.4 | 22.6 | 9.3 | 328 |
| 582 | Pool-riffle | 302 | W | 1.8 | 8.4 | 9.0 | 328 |
| 584 | Pool-riffle | 313 | W | 1.8 | 25.7 | 13.6 | 492 |
| 597 | Pool-riffle | 376 | W | 1.8 | 18.2 | 11.1 | 348 |
| 605 | Cascade | 108 | NW | 9.5 | 10.4 | 8.7 | 320 |
| 619 | Step-pool | 494 | N | 4.5 | 9.6 | 5.8 | 328 |
| 621 | Step-pool | 487 | NE | 6.6 | 11.1 | 6.3 | 328 |
| 625 | Step-pool | 471 | N | 6.6 | 17.5 | 11.0 | 427 |
| 637 | Step-pool | 415 | W | 8.6 | 11.3 | 8.9 | 328 |
| 639 | Cascade | 658 | N | 21.1 | 18.3 | 13.9 | 328 |
| 642 | Pool-riffle | 514 | N | 2.1 | 8.7 | 5.4 | 328 |
| 653 | Cascade | 712 | N | 13.1 | 9.1 | 6.3 | 328 |
| 658 | Pool-riffle | 450 | W | 2.0 | 17.5 | 8.0 | 443 |
| 687 | Cascade | 805 | S | 8.5 | 18.7 | 12.8 | 328 |
| 688 | Step-pool | 169 | N | 4.6 | 14.4 | 6.6 | 335 |

| Watershed ID | Channel type | Elevation (ft) | Aspect | Gradient (%) | Mean bankfull width (ft) | Mean bankfull depth (in.) | Length (ft) |
|---|---|---|---|---|---|---|---|
| 690 | Cascade | 749 | S | 6.4 | 22.1 | 11.5 | 673 |
| 694 | Cascade | 862 | SW | 4.5 | 14.1 | 9.5 | 328 |
| 716 | Step-pool | 1175 | NE | 6.1 | 22.1 | 11.9 | 328 |
| 717 | Pool-riffle | 596 | E | 2.1 | 7.0 | 7.5 | 328 |
| 718 | Pool-riffle | 562 | SW | 1.3 | 15.3 | 8.9 | 410 |
| 724 | Step-pool | 560 | SW | 5.8 | 10.9 | 5.0 | 328 |
| 730 | Pool-riffle | 411 | S | 1.5 | 20.2 | 9.2 | 382 |
| 737 | Cascade | 1188 | S | 11.6 | 7.6 | 5.4 | 328 |
| 750 | Cascade | 1287 | NE | 10.7 | 19.7 | 8.8 | 375 |
| 760 | Pool-riffle | 295 | SE | 2.4 | 17.5 | 9.6 | 328 |
| 763 | Step-pool | 292 | SE | 3.1 | 14.7 | 7.7 | 328 |
| 767 | Cascade | 323 | S | 13.8 | 8.7 | 6.5 | 351 |
| 769 | Step-pool | 312 | S | 5.4 | 6.3 | 3.8 | 328 |
| 773 | Step-pool | 658 | NE | 7.5 | 14.1 | 8.1 | 348 |
| 776 | Cascade | 744 | NE | 9.9 | 10.7 | 7.0 | 328 |
| 790 | Step-pool | 265 | N | 4.5 | 18.2 | 10.4 | 328 |
| 796 | Step-pool | 205 | S | 2.5 | 25.1 | 9.2 | 328 |
| 797 | Step-pool | 223 | SW | 3.3 | 25.6 | 10.8 | 669 |
| 804 | Step-pool | 649 | NW | 4.6 | 17.8 | 6.1 | 344 |
| 820 | Pool-riffle | 132 | S | 0.8 | 24.0 | 14.3 | 512 |
| 844 | Pool-riffle | 149 | N | 1.7 | 18.4 | 7.8 | 328 |

**Table E-2.** Summary data for 50 DNR-managed sample reaches. For information on how the data were derived, see Minkova and Foster (2017).

| Watershed ID | Channel type | Elevation (ft) | Aspect | Gradient (%) | Mean bankfull width (ft) | Mean bankfull depth (in.) | Length (ft) |
|---|---|---|---|---|---|---|---|
| BOG[1] | Cascade | 389 | S | 16.1 | 18.1 | 14.2 | 393 |
| HOH[1] | Step-pool | 689 | SE | 10.9 | 10.6 | 5.8 | 335 |
| QUE[1] | Pool-riffle | 320 | S | 1.7 | 13.8 | 6.3 | 344 |
| SFH[1] | Cascade | 780 | SW | 16.6 | 16.3 | 9.2 | 328 |
| 566[2] | Pool-riffle | 545 | S | 3.0 | 6.3 | 5.3 | 338 |
| 744[2] | Step-pool | 400 | SE | 3.8 | 15.8 | 10.3 | 358 |
| NF4[3] | Step-pool | 505 | SW | 4.6 | 15.6 | 11.6 | 348 |
| NF5[3] | Step-pool | 784 | N | 2.9 | 11.5 | 6.8 | 328 |
| NF6[3] | Step-pool | 564 | SW | 4.5 | 11.0 | 8.5 | 335 |
| NF7[3] | Step-pool | 624 | N | 5.3 | 16.6 | 7.1 | 338 |
| NF8[3] | Step-pool | 725 | NE | 3.7 | 11.7 | 7.2 | 328 |
| NF9[3] | Step-pool | 633 | E | 2.9 | 23.2 | 10.6 | 476 |

[1] Located in Olympic National Park
[2] Located on DNR-managed land
[3] Located in Olympic National Forest

# Appendix F. Stream surveys from 2013 to 2020

**Table F-1.** Stream surveys completed between 2013 and 2020.

| Watershed | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 |
|---|---|---|---|---|---|---|---|---|
| - - - - - - - - - - - - - - - - - - - - - - - - - *DNR-managed watersheds* - - - - - - - - - - - - - - - - - - - - - - - - - | | | | | | | | |
| 145* | X | | | X | X | X | X | X |
| 157 | X | | | X | X | | | X |
| 158 | | X | X | X | | | X | |
| 165* | X | | | | X | | X | X |
| 196 | X | | | X | X | X | | |
| 328 | | X | | X | X | X | | X |
| 433 | X | | | X | X | X | | |
| 443 | | X | | X | | | X | |
| 488 | | X | X | X | | X | X | X |
| 542 | | X | | X | | X | | X |
| 544* | X | | | X | X | X | X | X |
| 545 | X | | | X | | X | | |
| 550 | | X | | X | X | | | X |
| 567 | | X | | X | | | X | |
| 568 | | X | | X | X | | | X |
| 582 | | X | | X | | | X | |
| 584* | | X | | X | X | X | X | X |
| 597 | | X | | X | | | X | |
| 605 | | X | X | X | | | X | |
| 619 | | | X | X | | X | | X |
| 621 | | X | | X | | | X | |
| 625 | | X | | X | | X | | X |
| 637 | | X | | X | | X | X | |
| 639 | | | X | X | | | X | X |
| 642* | | X | | X | X | X | X | X |
| 653 | | X | | X | | X | | X |
| 658 | | | X | X | | | | X |
| 687 | | | X | | X | | | X |
| 688 | | | X | X | X | | | X |
| 690 | | X | | | X | | X | |
| 694* | | X | | X | X | X | X | X |
| 716 | | | X | | X | | | X |
| 717 | X | | | X | | | X | |
| 718 | | X | X | X | X | | X | |

| Watershed | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 |
|---|---|---|---|---|---|---|---|---|
| 724 | | X | X | X | X | X | X | |
| 730 | | X | | X | | X | | X |
| 737* | | X | | | X | | X | X |
| 750 | | | X | X | | | | X |
| 760 | | X | | X | | | | X |
| 763 | | X | X | X | X | | X | |
| 767 | | X | | X | | X | X | |
| 769 | X | | | X | X | | | X |
| 773 | | X | | X | | | X | |
| 776 | | X | | | X | | X | |
| 790* | X | | | X | X | X | X | X |
| 796 | | X | | X | | | X | |
| 797 | | X | | | X | | X | |
| 804 | | X | | X | | | X | |
| 820 | | | X | X | | | | |
| 844 | | X | | X | | | X | |
| - - - - - - - - - - - - - - - - - - - - - - - - - - - *Reference watersheds* - - - - - - - - - - - - - - - - - - - - - - - - - - - | | | | | | | | |
| 566 | | | | | X | X | | |
| 744* | | | | | X | X | X | X |
| BOG | | | X | X | X | X | | X |
| HOH | | | X | | X | | | |
| NF4 | | | | | | X | X | X |
| NF5 | | | | | | X | X | X |
| NF6 | | | | | | X | X | X |
| NF7 | | | | | | X | X | X |
| NF8* | | | | | | X | X | X |
| NF9* | | | | | | X | X | X |
| QUE* | | | X | X | X | X | X | X |
| SFH | | | X | X | X | X | | X |

* Beginning in 2019, designated as an annually sampled sentinel site.

## Appendix G. Data analysis

Analysis of variance was performed with R software (R Core Team 2021), by using the *lmer* function from the *lme4* package. Model residuals were graphed using the *plot_model* function in the *sjPlot* package. Mean comparisons were made using the *emmeans* function from the *emmeans* package. The following is an example of a script used for the general ANOVA model, though it was modified for specific analyses:

```
library(lme4)
library(sjPlot)
library(emmeans)
model.out <- lmer(depvar ~ (1 | watershed) + channeltype + year +
                           year:channeltype + bfw, REML=T, data=d1)
plot_model(model.out, type='diag')    # Residual plots
summary(model.out)    # Estimates of fixed effects
anova(model.out)      # Produce F values and p-values for fixed effects
emmeans(model.out, pairwise~channeltype, adjust="scheffe") # Mean comparisons
```

# Appendix H. Estimated timeline to detect change

## BACKGROUND

In this Appendix we present a series of calculations to estimate, for each indicator metric, the duration of monitoring necessary to perform a statistically reliable test of whether change has occurred. This analysis uses statistical power calculations, combined with already observed trends and variation in each indicator from 2013-2020.

In the context of this analysis, statistical power is the probability of being able to detect change in an indicator, in the case when change actually has occurred. Statistical power is measured by a number that ranges from 0 to 1. In the natural sciences, a power of 0.80 is usually considered acceptable, and power of 0.90 is considered strong. In an experiment where statistical power is low, a real change could occur, but it is unlikely to be detected by the experiment. If the statistical power of a study were only 0.45, then, even if a real change occurred, there would only by of 45% chance of detecting that change. It is therefore desirable to have high statistical power because then, when a real change occurs, the study is likely to detect it.

Although it is always desirable to have very high statistical power, it comes at a cost, which is sample size. Assuming one is using the best available measurement approach to detect change, the only way to further increase a study's power is to increase the sample size. In Status and Trends Monitoring our sample size is fixed at 50 DNR-managed watersheds. Thus, a sample sized of 50 is used in all of our power analyses.

A power analysis is a statistical calculation made up of four components: statistical power, confidence level, sample size, and effect size. If values are known or estimated for any three of these four components, then the fourth can be calculated. In the power analysis presented here, the component calculated is statistical power.

In calculating statistical power, it is important to establish an appropriate confidence level (also known as Type I error rate) because confidence level will affect the power calculation. Using long-term monitoring as an example, confidence level is the probability of correctly finding that there was no change in an indicator when no change occurred. In the natural sciences, a confidence level of 95% is commonly used, as it is throughout the analyses in this report, including this power analysis.

A final component of statistical power calculation is effect size. In this long-term monitoring project, effect size represents the change that we are able to detect at a specified point in time. For a given indicator, effect size is calculated as the amount of change that is considered meaningful, divided by the variability of that indicator over time. If an indicator varies much differently over time among the 50 streams, then the effect size will be small and change will be difficult to detect. If, among the 50 streams, the indicator changes consistently over time, the effect size will be large and change will be more easily detected.

## POWER CALCULATIONS

For each of the indicator metrics included in the long-term Status and Trends Monitoring, we performed a series of power calculations to determine how many years would need to pass before we could reach a power of 0.80 for detecting change in each of the indicator metrics. To make such calculations, we had to assume a rate of real change in the indicator metrics. We tested three scenarios for each indicator metric, each with its own hypothetical rate of change:

1. A net change of 1% per year. This rate of change is arbitrary but was selected because it was used in previous power analyses similar to this one (Larsen et al. 2004).
2. A net change of 2% per year. Again, this rate was selected based on its use in previous power analyses for monitoring long-term change in stream habitat indicators (Larsen et al. 2004).
3. The observed annual rate of change (%) in the indicator metric from 2013-2020 continues into the future.

The change assumed in these scenarios net change in one direction, positive or negative.[38] For example, in scenario 2, a change of 20%, either positive or negative, is present after 10 years. Other fluctuations in the indicator could occur during that 10-year period, but at the end of 10 years a 20% increase or decrease exists. In scenario 3, we estimated the observed annual rate of change for each indicator metric during the sampling period from 2013 to 2020, by using a mixed-model analysis in which year was treated as a linear effect.

Power analyses were performed in R using the pwr.t.test function of the pwr package, with the 'paired' and 'two.sided' options selected (R Core Team 2021). The power analysis calculated, for the three scenarios, how statistical power increased over time with each additional year of monitoring. The analysis was performed for each indicator metric by iteratively running a power calculation 20 times, with each run representing a year. During these 20 runs, the only value in the calculation that was changed was the effect size. Effect size was directly related to the number of years elapsed because the rate of change per year was multiplied by the number of years. Thus, for each indicator metric, effect size was calculated as the number of years elapsed multiplied by the percent change per year (1%, 2%, or observed) in the indicator, with that product divided by the variation in the indicator.[39]

## RESULTS

The analysis was designed to project the rate at which statistical power accumulates over time under three scenarios, thereby providing information that could assist in determining for how long to monitor the various habitat indicators. In interpreting the results, we use a target statistical power of 0.80. Once the 0.80 threshold is reached, the study will have an 80% chance of detecting a real change in a given indicator metric. The results of the analysis for 11 different habitat indicator metrics appear in Figures H-1 through H-11. (Metrics that were not intended for frequently repeated long-term sampling, such as riparian vegetation and riparian microclimate, were not included in this analysis.)

The graphs in Figures H-1 through H-11 can be interpreted as follows, using scenario 2 (2% change per year) in Figure H-1 as an example (and the target power of 0.80): "After 11 years, it will be

---

[38] The direction of change was not specified because if we had specified either positive or negative, then we could only test for change in that direction, and if change went in the other direction we would not have been able to detect it.

[39] In our analysis, this variation was quantified as the standard deviation of the difference in an indicator between the first and last times it was measured. We followed the effect size calculation guidance for the pwr package in R. See https://cran.r-project.org/web/packages/pwr/vignettes/pwr-vignette.html.

possible to detect a significant change in in-stream wood piece frequency, with 95% confidence that we are not making a false claim. Assuming the change is real, we will have an 80% chance of detecting it." This statement can be made because the line representing the 2% scenario passes the 0.80 power value between years 10 and 11. The number of years required to reach a statistical power of 0.80 is summarized for each indicator metric and each of the three scenarios in Table H-1.

**Table H-1.** The number of years that must elapse, under three scenarios, before Status and Trends Monitoring reaches a statistical power of 0.80 for detecting change in indicator metrics.

| | **Scenario: annual rate of change** | | |
|---|---|---|---|
| **Indicator metric (observed annual rate of change)[1]** | **1%** | **2%** | **Observed rate (2013-2020)** |
| In-stream wood: frequency of pieces (**-2.3%**) | >20 | 11 | 10 |
| In-stream wood: diameter (**-1.0%**) | 11 | 6 | 11 |
| Pool habitat: pool frequency (**-4.3%**) | >20 | 18 | 8 |
| Pool habitat: pool area (-0.6%) | >20 | 18 | >20 |
| Pool habitat: residual pool depth (**+1.6%**) | 13 | 7 | 8 |
| Channel substrate: median diameter ($D_{50}$) (+0.4%) | >20 | 13 | >20 |
| Channel substrate: percent fines (+4.3%) | >20 | 20 | 9 |
| Channel morphology: bankfull width:depth (-0.1%) | 14 | 7 | >20 |
| Channel morphology: bank erosion (**-4.6%**) | >20 | >20 | 11 |
| Stream shade: canopy closure (**+0.3%**) | 3 | 2 | 6 |
| Stream temperature: 7-DADmax (-0.1%)[2] | 2 | 1 | 15 |

[1] Trends that were found statistically significant in the primary analyses of this report (see the ANOVA results Tables) appear here in bold. In the ANOVA results Tables, the significant trends are those in which the Year effect is interpreted as "significant".
[2] In Tables 19b and 20, the Year effect shows significant change, but this represents year-to-year change rather than an overall unidirectional trend which was calculated to be -0.1% so far.

The results of this analysis show a wide variation in projected monitoring years to reach statistical power of 0.80 (Table H-1). Using the observed rate of change column in Table H-1, 3 of the 11 habitat indicators are projected to reach the 0.80 power threshold by the time eight years of monitoring have elapsed (i.e., 2021): pool frequency, residual pool depth, and canopy closure.[40] It is important to understand that this does not mean we will see a significant change after eight years; rather, it means that after eight years, *if* a change has actually occurred, we will have an 80% chance

[40] Eight years of monitoring since the initial measurement in 2013.

of detecting it. Thus, for those three metrics, we can feel confident[41] in our results, whatever they may be, after eight years. For 1 of the 11 indicators, canopy closure, statistical power has already surpassed 0.80, as of 2020.

Three habitat indicators are not projected to reach a power of 0.80 at the current rate of change, even if monitoring lasted for 20 years: pool area, substrate median particle diameter ($D_{50}$), and bankfull width:depth ratio. This lack of power for these four indicators is due to a small observed rate of change (well under 1% per year) relative to their variability.

Among the 11 indicators analyzed here, stream temperature is a special case because we already know that there are large changes in stream temperature occurring from year-to-year in response to weather conditions. However, the power analysis was based on detection of a long-term trend, either positive or negative, and the resulting power was relatively low due to the very small trend (-0.1% per year) observed during the first eight years.

## IMPLICATIONS

There is a key assumption that underlies this power analysis and that must be considered when interpreting any results: what is a meaningful amount of change in each habitat indicator? Scenarios 1 and 2 are included as hypothetical scenarios that are primarily useful as standards for comparing power among the 11 indicator metrics. However, for most of the monitored habitat indicators, there are no regulatory standards and often a lack of scientific data that is specific enough to set target habitat conditions for OESF. Recognizing this lack of data, the Status and Trends Monitoring program was designed to monitor *changes* in conditions over time. Even without exact thresholds on quality habitat, we are able to observe which direction the trends are going for each habitat indicator as well as their magnitude. This power analysis is designed to help make decisions on monitoring by calculating the relative length of time until we have power to detect a change.

Although power analysis is of key importance in making decisions about long-term monitoring, it is only one source of information to be weighed against a variety of other factors. Statistical power, on its own, cannot determine the importance of monitoring a given habitat indicator metric. The decision of how to—or for how long to—monitor a given stream habitat indicator metric may be affected by:

- How realistic is the projected timeframe for monitoring, as determined by the power analysis (e.g., years to reach 0.80 power)?
- How important is the habitat indicator relative to the other indicators monitored?
- Does the indicator represent a key habitat attribute for a species that is of particular concern?
- What is the cost of monitoring that indicator relative to other indicators?
- Is there a more effective or efficient way of measuring the indicator, or is there a comparable metric that would offer advantages?

A pivotal component of this power analysis was the rate of change used in the three scenarios. Clearly, the observed rate of change was far more realistic than the hypothetical 1% or 2% rates. However, there is still a caveat in that we don't know whether the rate of change observed so far will

[41] 80% confident, anyway.

continue. But if we assume that it will continue for the foreseeable future, then scenario 3 provides the appropriate guidance for projections of future statistical power in this monitoring program.

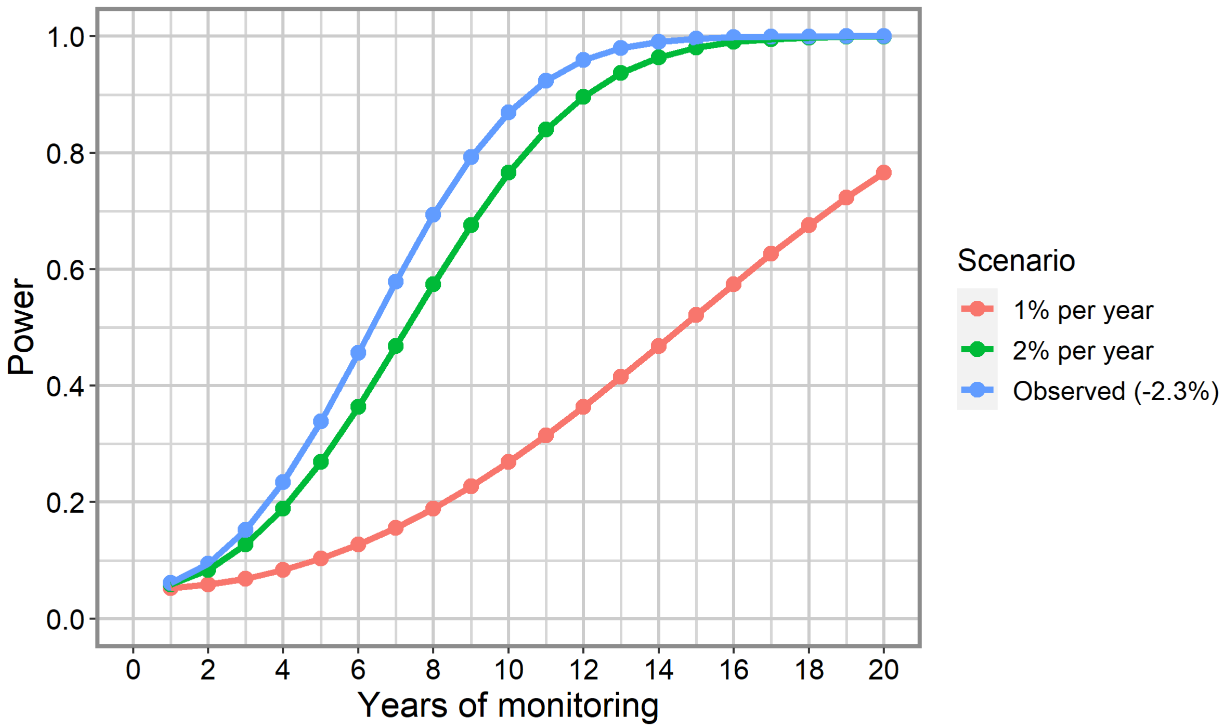


**Figure H-1.** Power to detect change in in-stream wood piece frequency, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

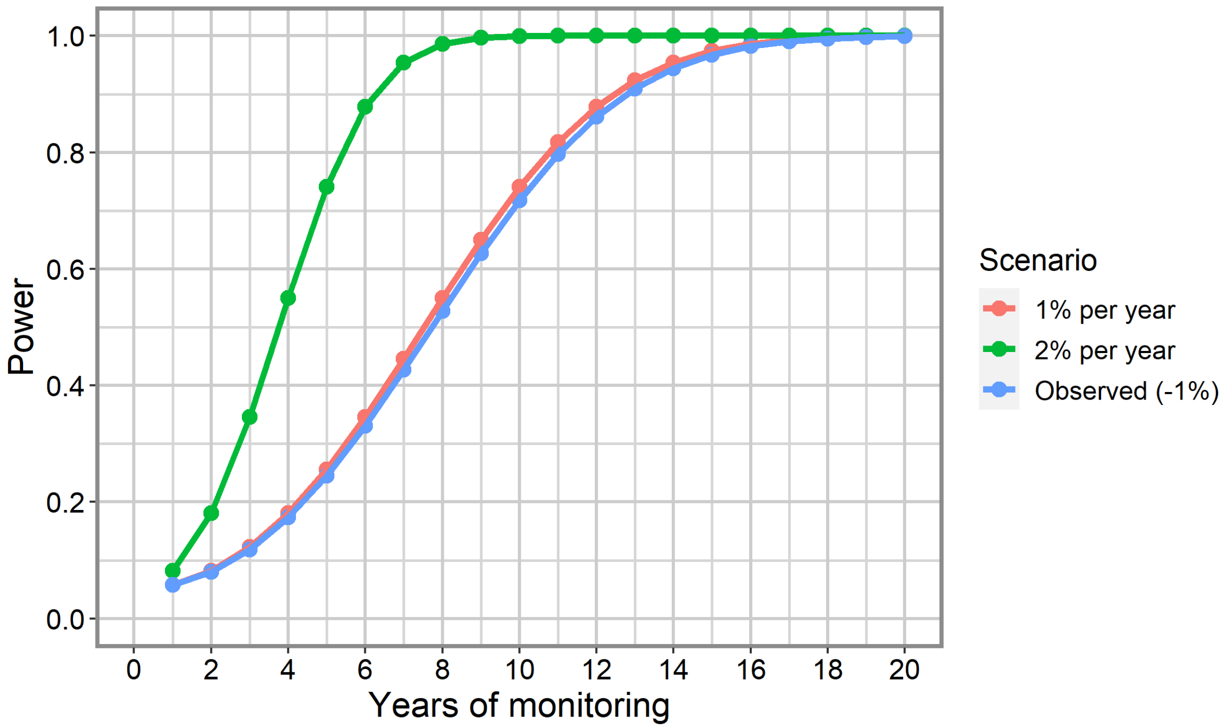


**Figure H-2.** Power to detect change in in-stream wood piece mean diameter, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

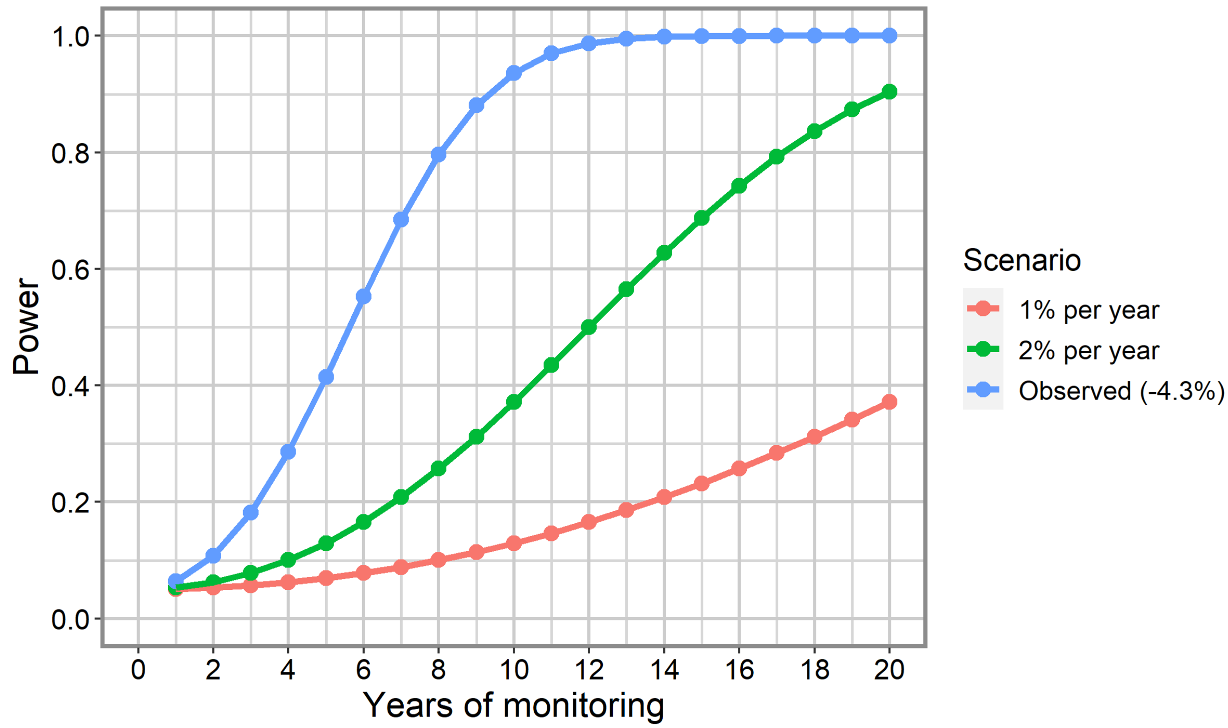


**Figure H-3.** Power to detect change in pool frequency, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

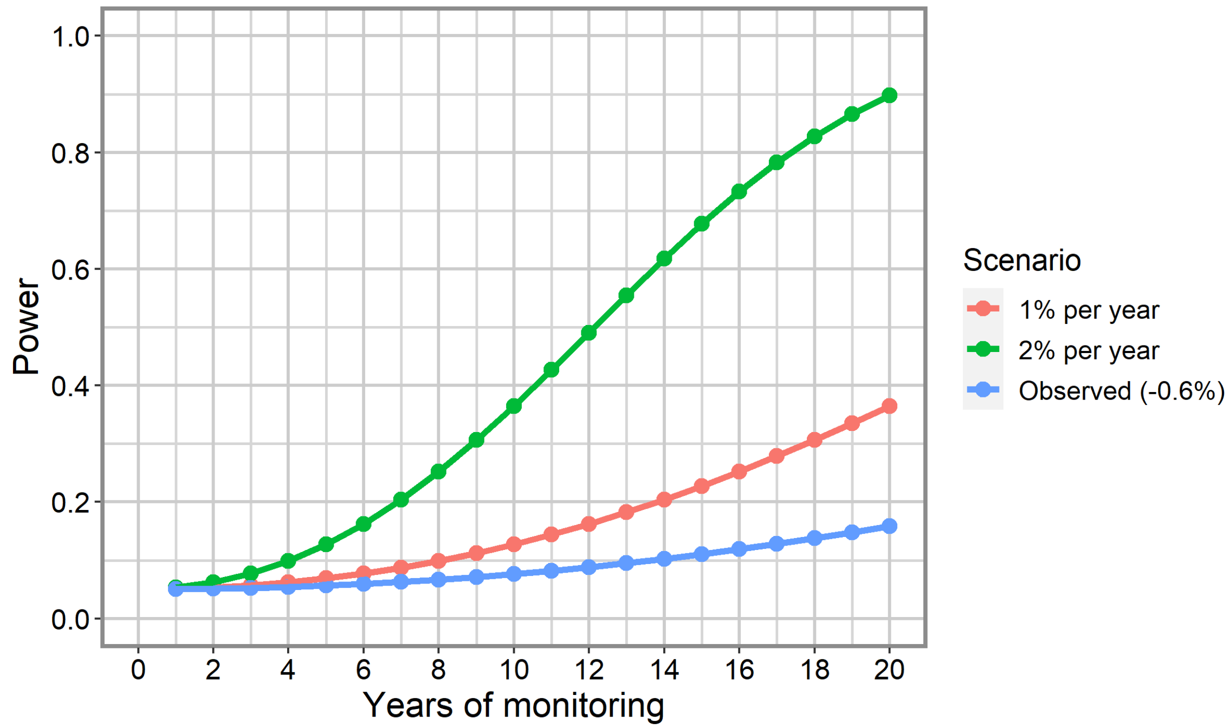


**Figure H-4.** Power to detect change in pool area, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

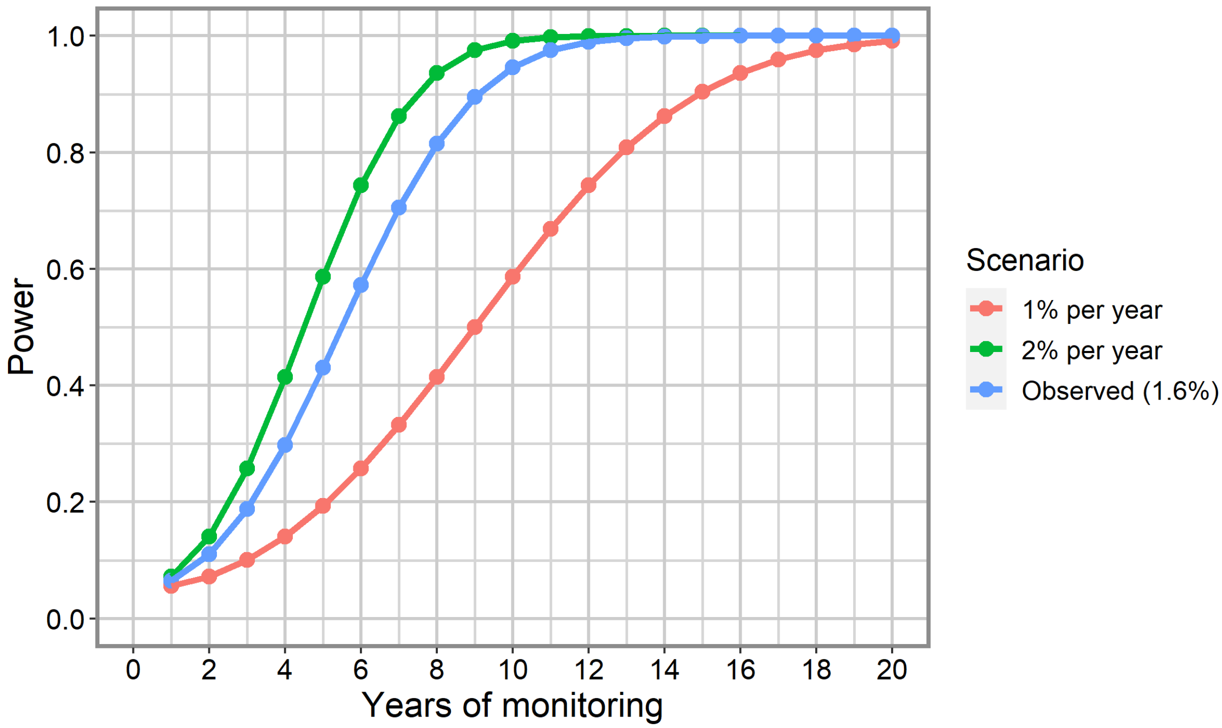


**Figure H-5.** Power to detect change in residual pool depth, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

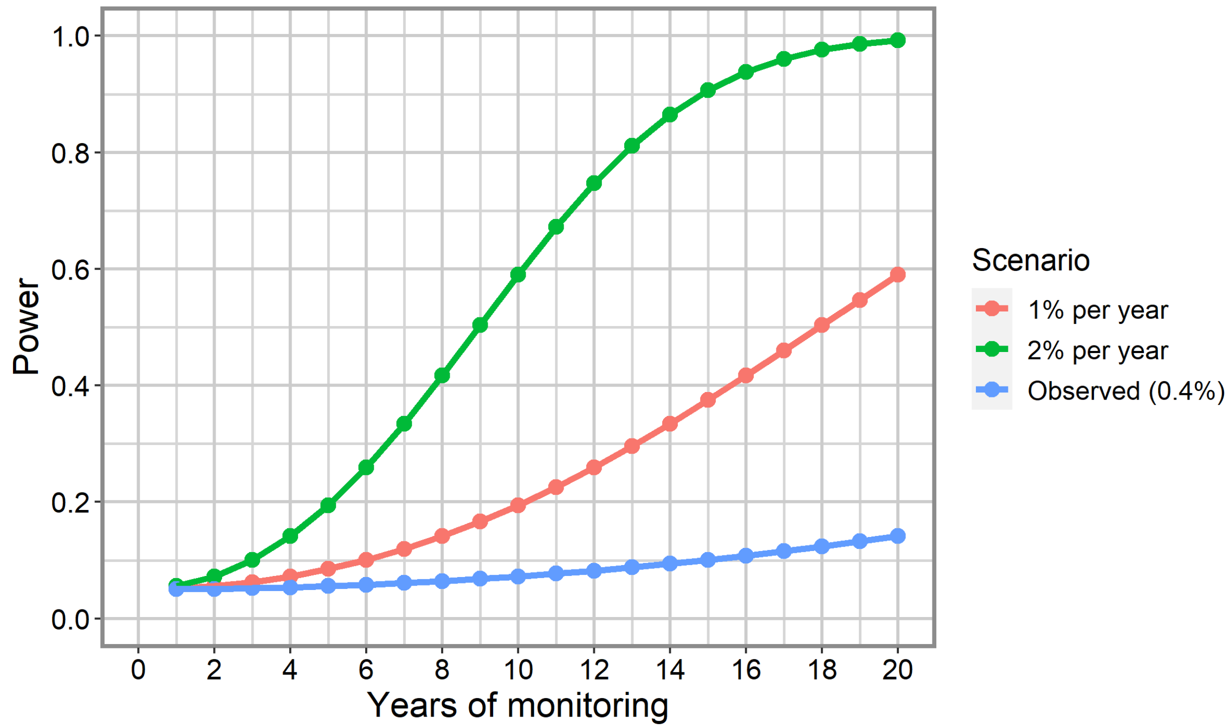


**Figure H-6.** Power to detect change in median particle size ($D_{50}$), over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

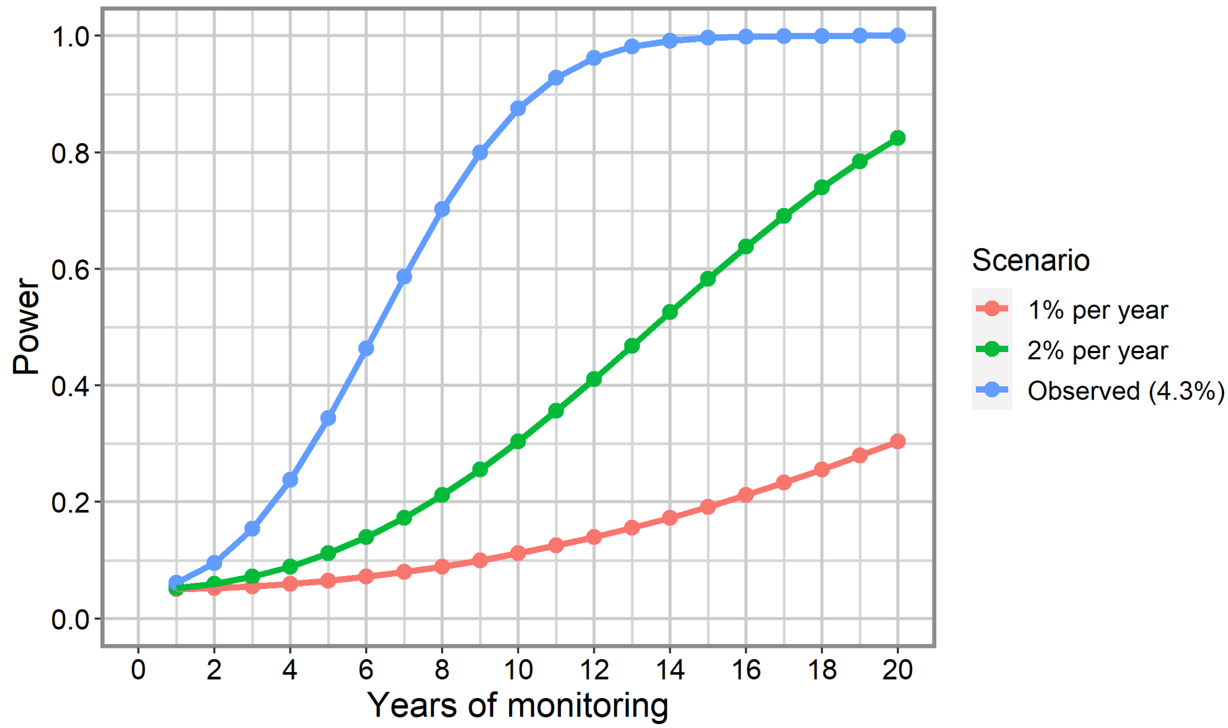


**Figure H-7.** Power to detect change in percent fines, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

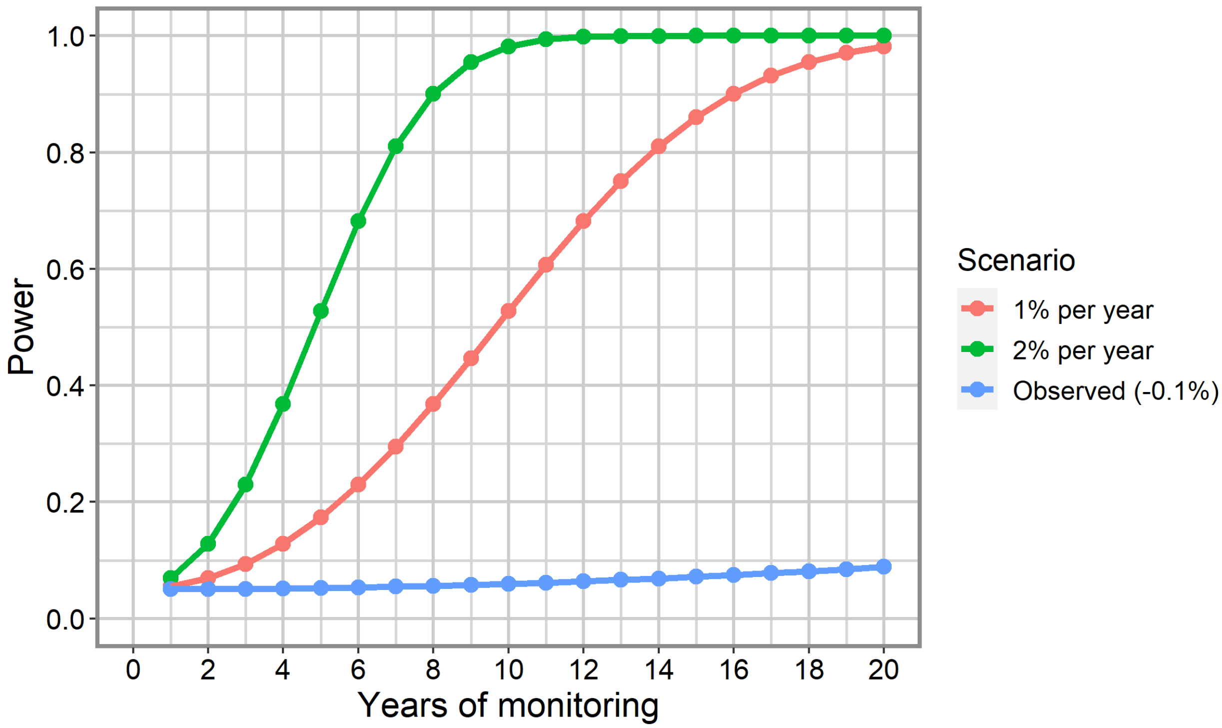


**Figure H-8.** Power to detect change in bankfull width:depth ratio, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

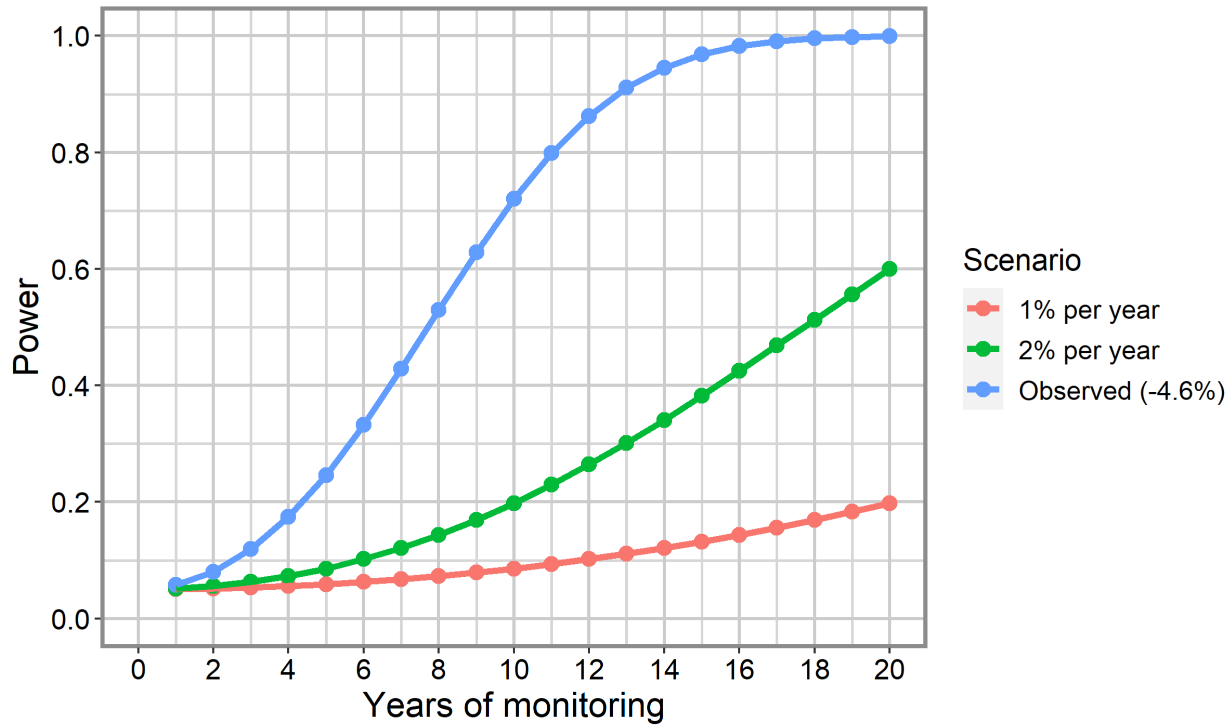


**Figure H-9.** Power to detect change in streambank erosion, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

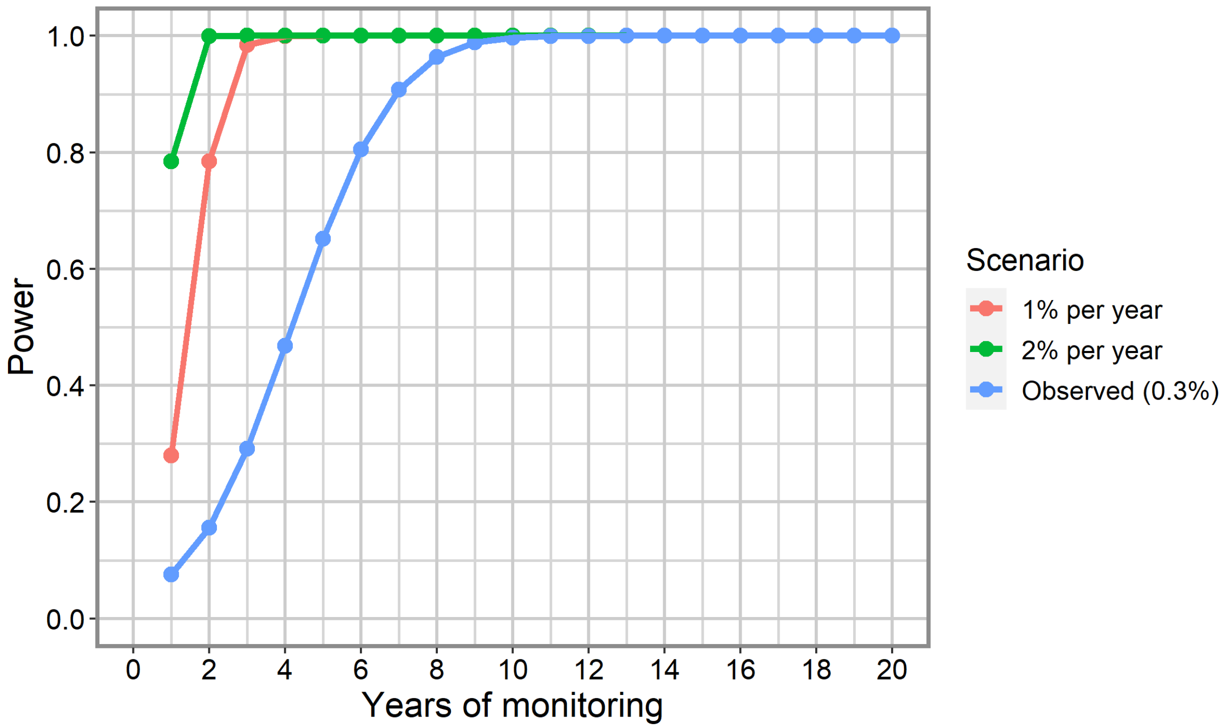


**Figure H-10.** Power to detect change in canopy closure, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

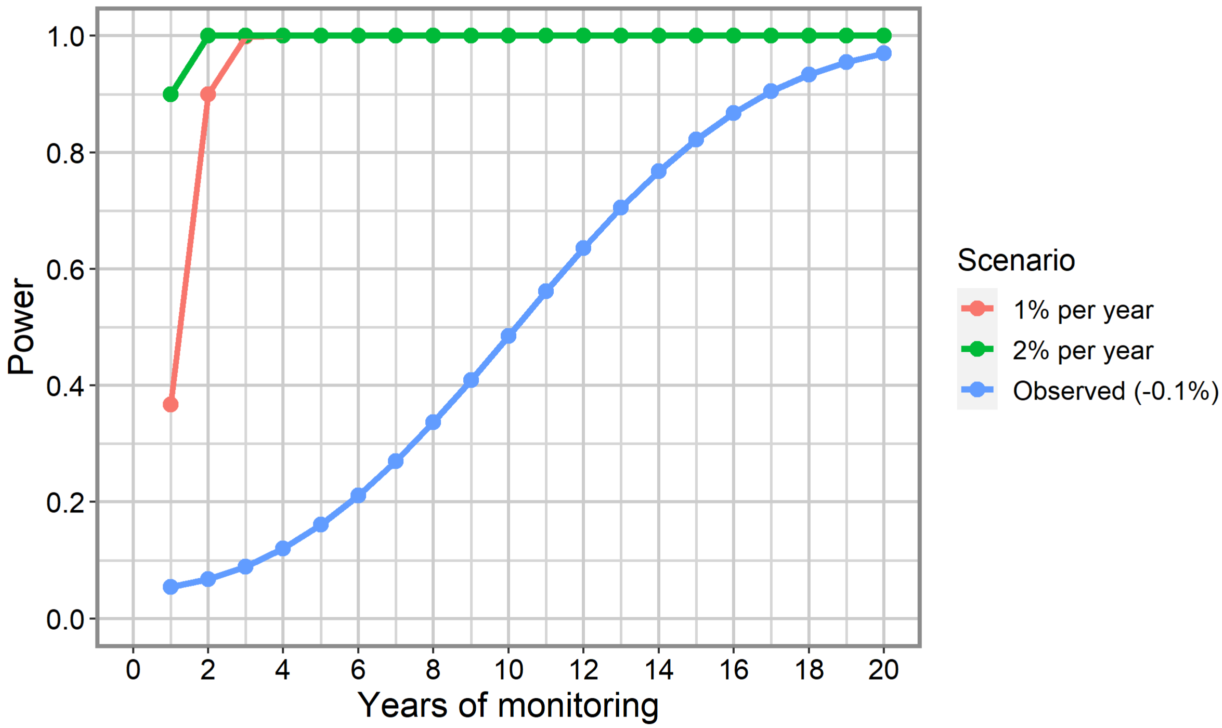


**Figure H-11.** Power to detect change in maximum 7-day average maximum stream temperature, over time, for Status and Trends Monitoring of 50 DNR-managed watersheds; three scenarios are presented: 1% per year unidirectional change, 2% per year unidirectional change, and continuation of the annual rate of change observed from 2013-2020. The confidence level was 95%. Years of monitoring refers to years elapsed since the initial measurement; thus, one year is equivalent to measurements in 2013 and 2014.

# Appendix I. Maps of overall habitat condition score

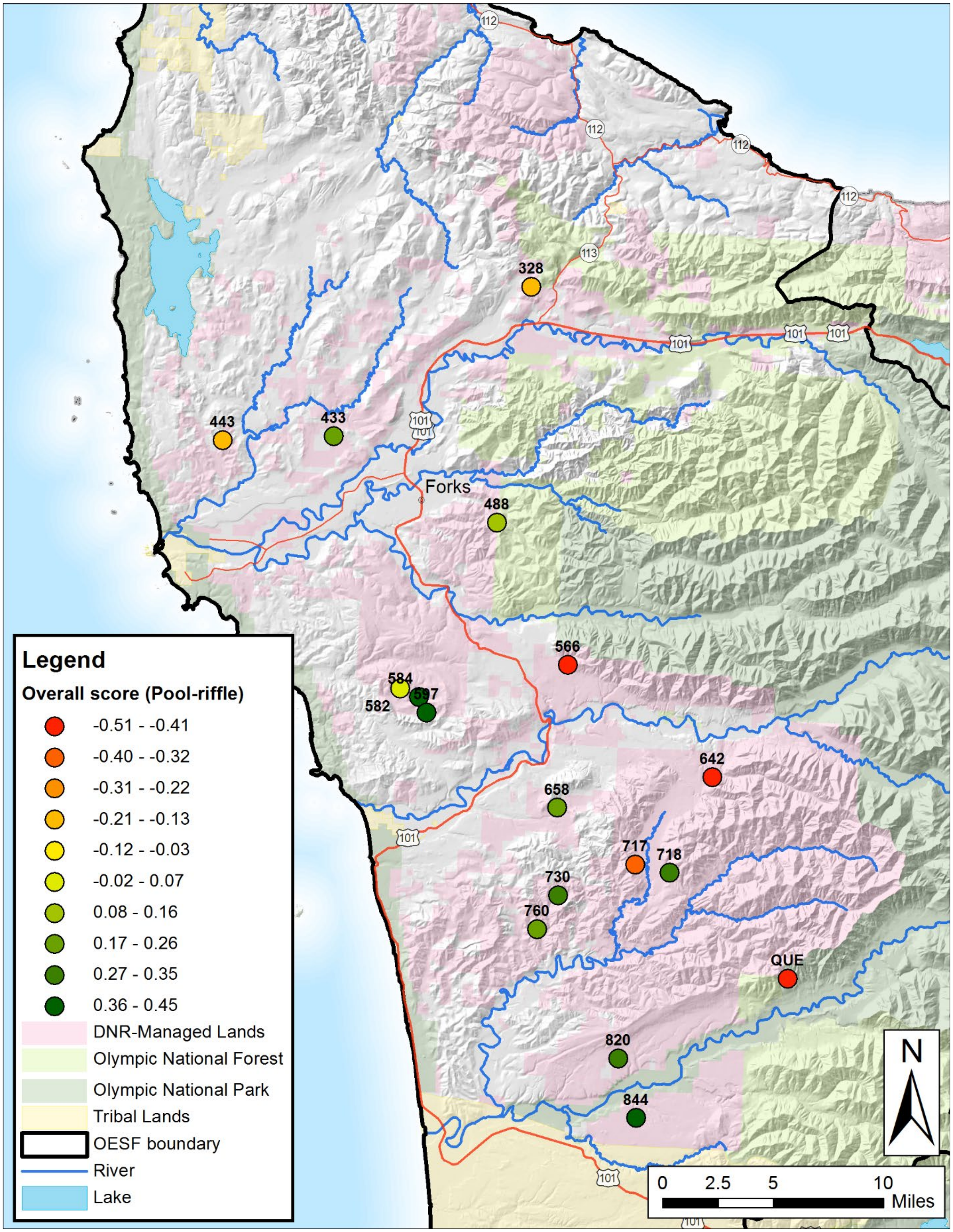


**Figure I-1.** Overall score from the habitat condition model for 17 watersheds of the pool-riffle channel type.

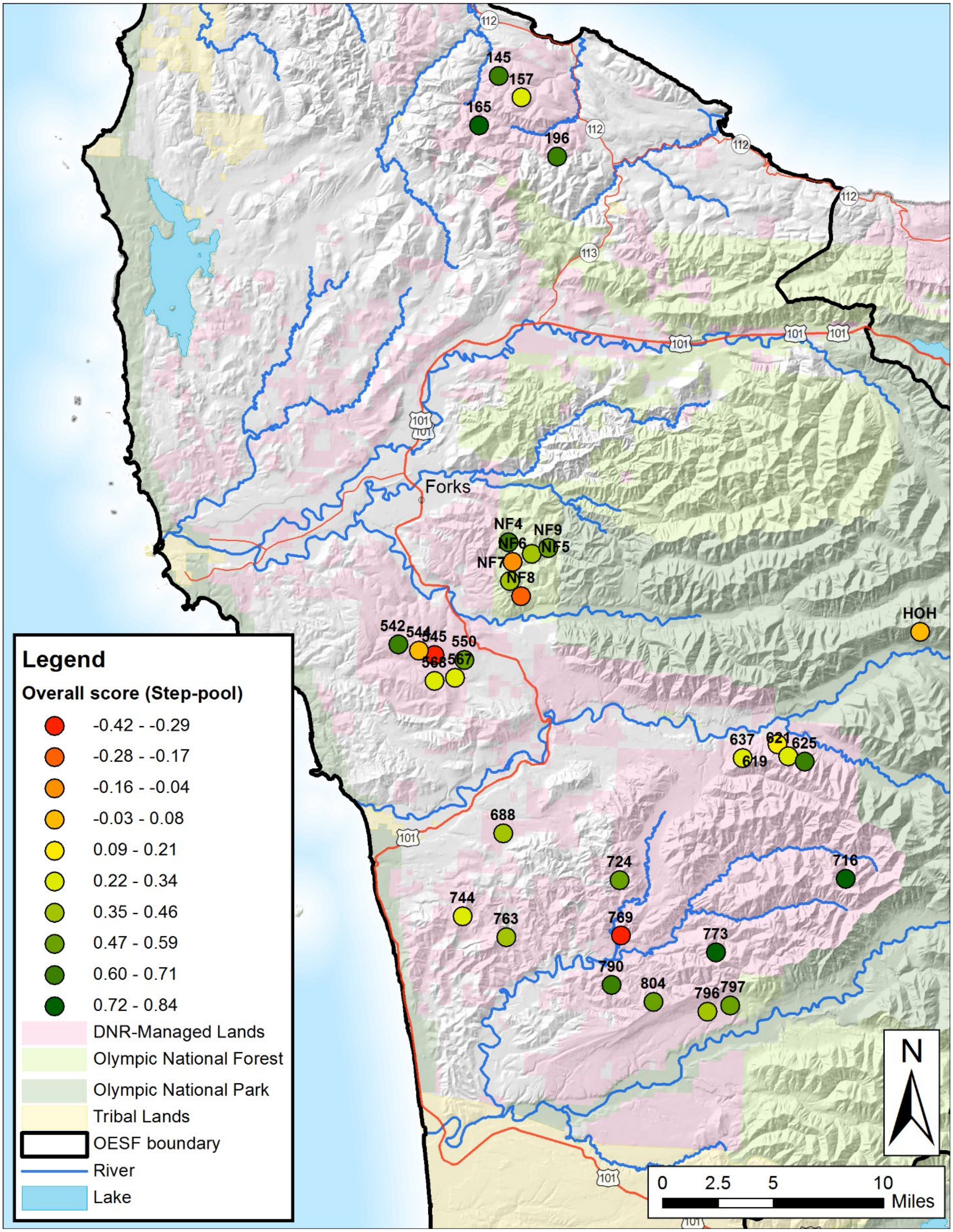


**Figure I-2.** Overall score from the habitat condition model for 32 watersheds of the step-pool channel type.

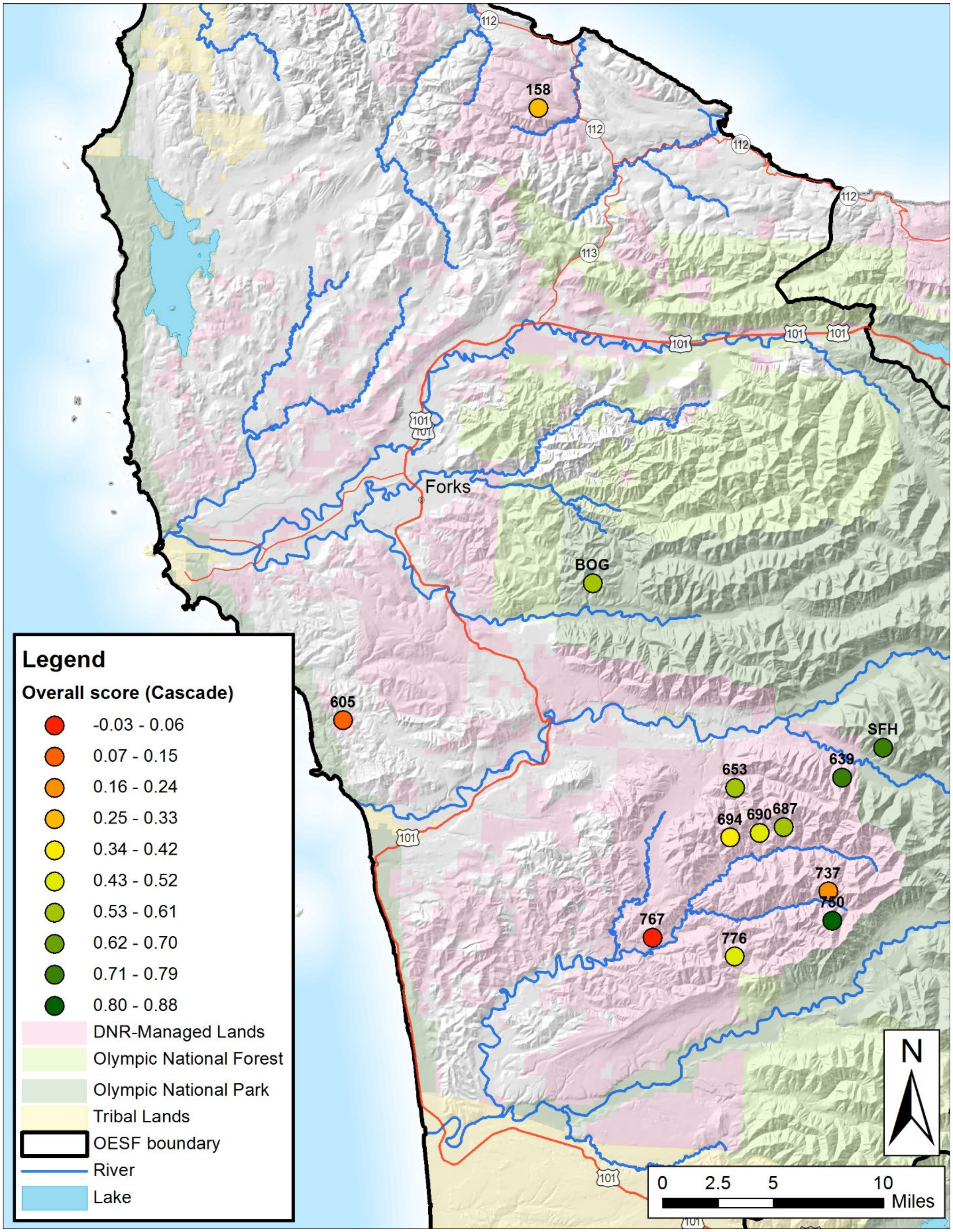


**Figure I-3.** Overall score from the habitat condition model for 13 watersheds of the cascade channel type.

## Appendix J. Projected climate changes in the OESF

To discuss what the potential effects of a changing climate will be on the riparian and aquatic habitats of streams in the OESF, we must first know what the projected climate changes are for the OESF. For the purpose of this assessment, we identified two types of potential climate change that could have direct or indirect effects on stream habitat: (1) air temperature, and (2) precipitation and snowpack. Here, we only discuss effects on stream habitat; climate has other effects on anadromous salmonid species beyond the scope of this discussion, as adult fish at sea are also affected by climate (Crozier et al. 2008; Shelton et al. 2021).

Finding reliable climate projections for the OESF is a significant challenge, for several reasons. First, projections must be made at an appropriate scale. In western Washington, there are dramatic changes in climate over relatively short distances owing to the topography of the Olympic and Cascade mountain ranges. Therefore, climate projections for the OESF must consider its mild, maritime climate conditions, which are different from the mountainous interior of the Olympic Peninsula, the rain shadow of the northeastern peninsula, the Puget Trough, and other parts of western Washington. For example, since weather conditions have been officially documented in Forks beginning in 1908, average annual air temperature there has increased at half the rate of the statewide average, +1.0 °F (+0.6 °C) compared to +2.0 °F (+1.1 °C).[42]

Another problem is the scarcity of local historical climate data. To accurately project future climate, it is necessary to know what the past climate has been. There is only one weather station in the OESF area (USC00452914; Forks 1 E) that has been in continuous operation throughout most of the 20th century. Thus, climate analyses made for the western Olympic Peninsula rely heavily on data from only one location. Additionally, the nearest data on snowpack in the Olympic Range is collected near 5,000 feet elevation in the central part of the range. Snowpack there is much different from that of the rain-on-snow zone in the higher-elevation portions of the OESF; thus, lacking local data on snowpack, we can't effectively relate our habitat results to snowpack.

In this report, we use the best available published climate projections made during the past decade: sources cited in the OESF Forest Land Plan EIS (DNR 2016b) and in the Third and Fourth National Climate Assessments (Dalton et al. 2013; Wuebbles et al. 2017).

### AIR TEMPERATURE

In the Pacific Northwest, air temperatures trend upward and downward at a multi-decadal timescale that is primarily driven by El Niño-Southern Oscillation and the Pacific-North American teleconnection patterns (Mote et al. 2013; Abatzoglou et al. 2014). However, at a longer timescale—from 1901 to 2012—there has been a warming trend of 1.3 °F (0.7 °C), averaged across the Pacific Northwest (Washington, Oregon, and Idaho) (Abatzoglou et al. 2014).

During the time that air temperature data has been recorded in Forks (1908-2020), average air temperature has increased by 1.0 °F (0.6 °C). Among the four seasons, average temperature has changed the most in Forks during summer (+1.5 °F; +0.8 °C); the least change has occurred during

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[42] All air temperature trend data for Forks, WA in this section of the report were collected from the NOAA Forks 1 E weather station, part of the United States Historical Climatology Network. Data were acquired using the University of Washington online tool at https://climate.washington.edu/climate-data/trendanalysisapp/

fall (0.0 °F; 0.0 °C). Overall, daily minimum temperatures in Forks have increased much more than daily maximum temperatures since 1908 (Table J-1).

**Table J-1.** Climate summary for data collected at the Forks 1 E station (USC00452914) from 1908 to 2020.[1]

| **Climate variable** | **Season (months)** | **Average for 1908-2020[1]** | **Trend from 1908-2020[2]** |
|---|---|---|---|
| - - - - - - - - - - - - - - - - - - - - - - - - - - *English units* - - - - - - - - - - - - - - - - - - - - - - - - - - | | | |
| Air temperature, average daily maximum | Summer (June-Aug.) | 70.4 °F | +0.8 °F |
| | Winter (Dec.-Feb.) | 46.2 °F | +0.7 °F |
| Air temperature, average daily minimum | Summer (June-Aug.) | 48.5 °F | +2.3 °F |
| | Winter (Dec.-Feb.) | 34.0 °F | +1.6 °F |
| Precipitation, cumulative | Fall (Sep.-Nov.) | 33.5 in. | +3.2 in. |
| | Winter (Dec.-Feb.) | 50.0 in. | -7.4 in. |
| | Spring (Mar.-May) | 27.0 in. | +4.8 in. |
| | Summer (June-Aug.) | 8.3 in. | +0.9 in. |
| - - - - - - - - - - - - - - - - - - - - - - - - - - *Metric units* - - - - - - - - - - - - - - - - - - - - - - - - - - | | | |
| Air temperature, average daily maximum | Summer (June-Aug.) | 21.3 °C | +0.4 °C |
| | Winter (Dec.-Feb.) | 7.9 °C | +0.4 °C |
| Air temperature, average daily minimum | Summer (June-Aug.) | 9.2 °C | +1.3 °C |
| | Winter (Dec.-Feb.) | 1.1 °C | +0.9 °C |
| Precipitation, cumulative | Fall (Sep.-Nov.) | 85.1 cm | +8.1 cm |
| | Winter (Dec.-Feb.) | 127.0 cm | -18.8 cm |
| | Spring (Mar.-May) | 68.6 cm | +12.2 cm |
| | Summer (June-Aug.) | 21.1 cm | +2.3 cm |

[1] Average values are calculated from data downloaded from National Oceanic and Atmospheric Administration (https://www.ncdc.noaa.gov/cdo-web/).

[2] Trend data for temperature are from https://climate.washington.edu/climate-data/trendanalysisapp/. Because precipitation data from that website have not been checked for errors (and some errors were found), trend data for precipitation are instead taken from National Oceanic and Atmospheric Administration (https://www.ncdc.noaa.gov/cdo-web/) and subsequently checked for errors before calculating summary values.

Note: Summary does not include years in which the percentage of days with missing data (for the seasons listed) surpassed specified thresholds. For statistics based on maximum or minimum values, the missing data threshold was 20% of days. For cumulative precipitation, the missing data threshold was 5% of days.

For the Pacific Northwest, annual air temperature is projected to increase by 3.7-4.7 °F (2.0-2.6 °C) by mid-21st century (2036-2065) and by 5.0-8.5 °F (2.8-4.7 °C) by late 21st century (2071-2100) under the RCP8.5 scenario (Vose et al. 2017). Similarly, extreme high temperature events, such as was observed in summer 2021, are projected to increase in the region (Vose et al. 2017; Overland 2021). Projections of future climate are normally made and published at a regional scale (e.g., the Pacific Northwest) because models become less certain at smaller scales. Based on the historical climate record for Forks, it is likely that increases in average annual air temperature there will be less than the regional average. One reason for this may be the marine low-cloud layer, which, near the coast, has not decreased in recent decades as it has inland (Dye et al. 2020).

## PRECIPITATION

Projections of future precipitation in the Pacific Northwest are much less certain than those of temperature. For example, the projected change for mean annual precipitation during the 21st century is less than the historical inter-annual variation (Mote et al. 2013).[43] However, climate models generally agree in projecting a decrease in summer precipitation by the year 2100 (model average is a 7.5% decrease under the RCP8.5 scenario) (Mote et al. 2013). The average change in precipitation projected for other seasons is much less certain, though the majority of models project increases (Mote and Salathé 2010; Mote et al. 2013; Easterling et al. 2017).

Extreme precipitation events are one of the greatest climate-related concerns at a national scale. Extreme precipitation events also are relevant to stream habitat because they are associated with peak flows. For the eastern U.S., the historical record (1901-2016) shows substantial increases in frequency of extreme precipitation events (Easterling et al. 2017). In contrast, the frequency of extreme precipitation events in the Pacific Northwest region has increased only slightly (Easterling et al. 2017). Projections for the future frequency of extreme precipitation events in the Pacific Northwest vary widely among models, but the model average suggests an increase during the 21st century (Dalton et al. 2013; Janssen et al. 2014; Easterling et al. 2017). Shifts in the annual timing and form of precipitation (i.e., rain vs. snow) could also impact hydrology in the region (Elsner et al. 2010). The frequency of twenty-year flood events for major rivers on the western Olympic Peninsula is projected to increase 11-15% by the 2040s (Halofsky et al. 2011).

Between 1908 and 2020, average precipitation in Forks increased for each season except winter (Table J-1). Overall, the trend in total annual precipitation shows an increase of 1.5 inches (3.8 cm) during that time period. The frequency of extreme precipitation events observed in Forks is in general agreement with the larger trend for the Pacific Northwest, showing no clear patterns in frequency of the largest events (i.e., 5- and 10-year events) between 1930 and 2019 (Figure J-1).[44]

Projected increases in peak streamflows due to changes in snowpack for the coastal Olympic Peninsula are not expected to occur until after the 2040s and by the 2080s are still projected to be relatively small (0 to 10%) due to relatively low snowfall at lower elevations (Safeeq et al. 2015).

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[43] Historically, the standard deviation of mean annual precipitation is 14%; projections are for change of between -5% and +14% (Mote et al. 2013).

[44] The 5- and 10-year events referred to here are events with that specified return interval; for example, a 10-year event is one that, based on the historical record, occurs on average once every 10 years. This is the same terminology used to describe flood events.

However, timing of streamflow is likely to be affected by a warming climate, particularly in higher-elevation watersheds where more of the annual precipitation occurs as snowfall (Halofsky et al. 2011).

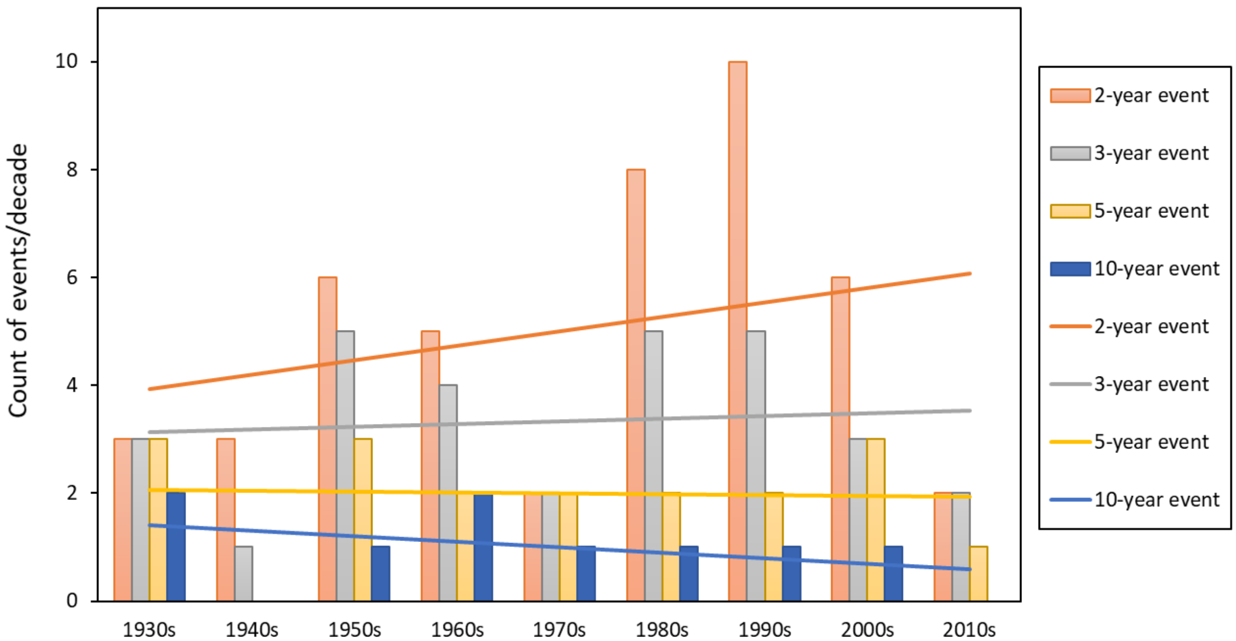


**Figure J-1.** Frequency, by decade, of largest 2-day precipitation totals ("events") over the 90-year period for which data are available for every year from the Forks 1 E weather station (USC00452914). Events are defined by return interval (e.g., there were nine 10-year events over the 90-year period).

Notes: This summary was created by using the same procedure as that of Mass et al. (2011). For the Pacific coast, extreme precipitation events are best captured by using two-day totals (Mass et al. 2011). For each of the events summarized, the precipitation events for the five days before and after were not considered in the analysis; this was done to prevent double- or triple-counting events from a single rainstorm lasting more than two days.