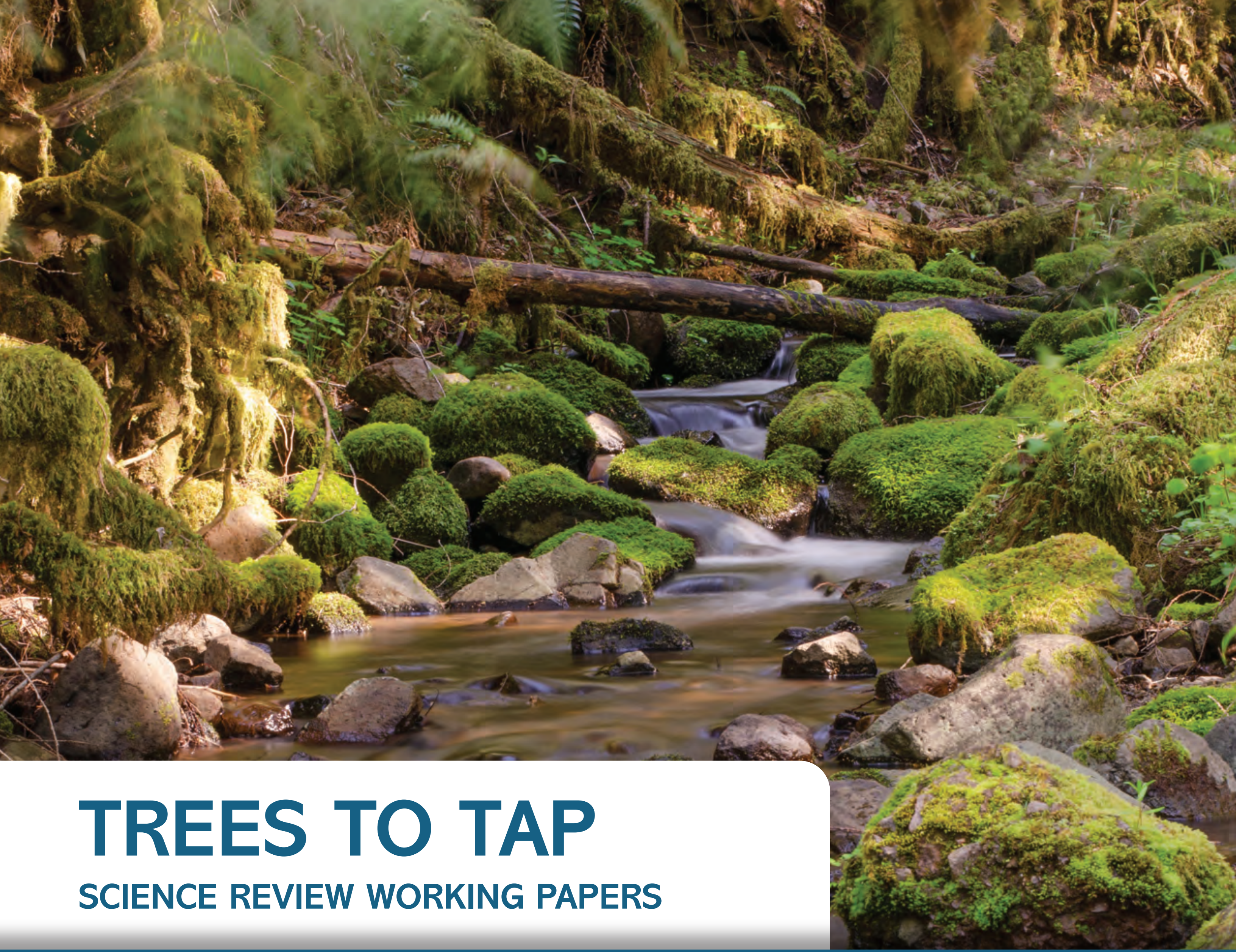


TREES TO TAP

SCIENCE REVIEW WORKING PAPERS

Institute for Natural Resources

Oregon Forest Resources Institute

Oregon State University College of Forestry

Trees To Tap: Forest Management and Community Drinking Water Supplies

Final Report to the Oregon Forest Resources Institute

June 15, 2020

Institute for Natural Resources

234 Strand Agriculture Hall

Oregon State University

Corvallis, OR 97331

Preface

This report provides the results of a literature review on the effects of active forest management (harvest, forest roads, and reforestation) on drinking water quality. In addition to the literature review, community water suppliers who rely on surface water as their primary source were surveyed to better understand their operations and priorities, and three case studies were conducted.

This Final Report is best characterized as "Working Papers" and will be formally published as a book by OSU's Extension and Experiment Station Communications after further review and editing. As such, the information provided here is subject to change and revision prior to publication. This report is provided as an interim product to support initiatives of the Oregon Forest Resources Institute (OFRI).

THIS PAGE INTENTIONALLY LEFT BLANK

CONTRIBUTORS

Jon A. Souder, Ph.D. Principal Investigator. Assistant Professor in the Forest Engineering, Resources, and Management (FERM) Department; and, Forest Watershed Specialist in the Forestry and Natural Resources Extension Program, and the concluding director of the Watersheds Research Cooperative (WRC). From 2000 until joining OSU, he was Executive Director of the Coos Watershed Association in Charleston, OR, managing their water quality assessment, restoration, and monitoring programs. He has a Ph.D. and M.S. in Wildland Resources Science from the University of California, Berkeley and a B.S. in biology (limnology) from Marlboro College, VT.

Kevin Bladon, Ph.D. Co-Principal Investigator. Assistant Professor of Forest Hydrology in FERM. He is interested in natural (wildfire, pests, pathogens) and human (conventional and salvage harvesting) disturbance effects on water quantity, water quality, and aquatic ecology. Relative to this project, he has previously been involved with transdisciplinary teams, researching the implications of forest disturbance on drinking water treatability. This research examined source water supply and protections strategies to enable both forest and community resiliency to respond to disturbances. He has a Ph.D. in Forest Hydrology from the University of Alberta.

Emily Jane Davis, Ph.D. Co-Principal Investigator. Assistant Professor in the Forest Ecosystems and Society (FES) Department, and Specialist in the Forestry and Natural Resources Extension Program. Her research and technical assistance focuses on natural resource social science, environmental governance, collaboration and partnerships, community development, wildfire response, and science delivery. She has a Ph.D. and M.A. in Human Geography from the University of British Columbia, and a B.A. in History from McGill University, Montreal, Canada.

Bogdan Strimbu, Ph.D. Co-Principal Investigator. Assistant Professor of Forest Management in FERM. His research focuses on strategic forest planning, remote sensing, and biometry. He has conducted research and published on the effects of forest management and petroleum drilling on moose and martin habitat, travel distances for debris flows, structurally complex forest stands, and remote sensing for forest inventories. He has a Ph.D. and a M.Sc. in Forest Management from the University of British Columbia, a M.Sc. in Mathematics/Statistics from Louisiana Technical University, and a B.S. in Forest Operations from Transilvania University, Romania.

Jeff Behan, M.S. Senior Policy Research Analyst, INR. Jeff has over 20 years of experience working with state and federal agencies, universities and native tribes to acquire, synthesize and deliver natural resource science knowledge. Jeff's professional interests include science-based natural resource and water policy, science synthesis and knowledge transfer, sustainability, public land management, river management and outdoor recreation policy and management.

Chapter 8 on Fire Risk was written by **Michelle A. Day**, M.S., Biological Scientist with the U.S. Forest Service, Rocky Mountain Research Station; **Chris Ringo**, M.S., Senior Faculty Research Assistant in the Department of Crop and Soil Science, OSU; and **Alan A. Ager**, Ph.D., Research Forester with the U.S. Forest Service, Rocky Mountain Research Station. In addition to Chapter 8, Ms. Day and Mr. Ringo provided fire risk maps for each of the 156 community water supply watersheds for an accompanying Atlas.

The forest cover change analysis presented in Chapter 1 was conducted by **Robert Kennedy**, Ph.D., Associate Professor in the College of Earth, Ocean, and Atmospheric Sciences, OSU; and **Peter Clary**, Faculty Research Assistant in the College of Earth, Ocean, and Atmospheric Sciences, OSU. In addition to their contributions for Chapter 1, they have created forest cover change maps and data for each of the 156 community water supply watersheds for an accompanying Atlas.

Lisa Gaines, Ph.D. is the Director of the Institute for Natural Resources at OSU and provided convening and facilitation for the Trees To Tap Steering Committee. She has dual B.A. degrees in Economics and International Relations from the University of California, Davis, an M.S. in Agricultural and Resource Economics and a Ph.D. in Environmental Sciences from Oregon State University.

STEERING COMMITTEE

We were fortunate to convene a broadly representative Steering Committee that worked well together. The role of the Steering Committee was to identify priorities for our science review, appraise the community water system survey questions, and review the draft chapters. We did not ask for their approval of the Final Report.

The Steering Committee consisted of:

Marganne Allen was the Forest Health and Monitoring Manager for the Oregon Department of Forestry prior to moving over to the Department of Agriculture towards the end of the project. She is an Oregon State University graduate, with an M.S. in Forest Management/Minor Soil Science and Masters of Forestry in Forest Hydrology.

Seth Barnes is the Director of Forest Policy for the Oregon Forest and Industries Council (OFIC), and prior was the Operations Manager for the Washington Forest Practices Program. He has a B.S. in Forest Management from Oregon State University and attended graduate school for Public Administration and Natural Resource Policy at Washington State University.

Dr. Ashley Coble is a forest watershed scientist with the National Council on Air and Stream Improvement (NCASI). Dr. Coble has a B.A. in Biological Sciences from Mount Holyoke College, an M.S. in Terrestrial Biogeochemistry from Northern Arizona University, and a Ph.D. in Aquatic Biogeochemistry from Michigan Technological University.

Mike Collier is the Deputy Director and Source Water Specialist at Oregon Association of Water Utilities (OAWU), a nonprofit, independent association of 422 water and wastewater utilities that represents water utilities' interests and provides technical assistance. He has an M.S. in Water Resources Engineering from Oregon State University.

Cathy Kellon was the Working Waters Program Director for the Geos Institute during most of this project, moving to be the Columbia Slough Watershed Council's Executive Director in January, 2019. Cathy earned her Master's degree in Geography, with a minor in Interdisciplinary Water Resources Studies, from Oregon State University.

Teresa Kubo was the Environmental Review and Sediment Manager for the Oregon Operations Office, Region 10 of U.S. Environmental Protection Agency, where she worked for last 16 years. She has an M.S. in Forest Resources Management from the University of Idaho, as well as a B.A. in Spanish and International Studies (Emphasis on Natural Resources), also from UI. She was succeeded by **Dan Brown** when she left EPA in July, 2019.

Casey Lyon is the technical services unit manager for Oregon Health Authority Drinking Water Services, currently managing the source water protection team within OHA drinking water services and coordinating drinking water implementation efforts with DEQ partner staff. He has a B.S. in Environmental Studies from the University of Oregon and is a registered Environmental Health Specialist.

Mary Scurlock is Coordinator for the Oregon Stream Protection Coalition (OSPC), representing 20 environmental and conservation organizations in Oregon and Washington. Prior to joining OSPC, Ms. Scurlock was Policy Director for 19 years at the Pacific Rivers Council. She has an A.B. degree from Duke University in Public Policy and History, and a J.D. degree, cum laude, from the Boston University School of Law.

Josh Seeds is a Nonpoint Source Analyst in the Drinking Water Protection program at Oregon DEQ where his work focuses on nonpoint source pollution risks, natural processes, land management, and drinking water provision. He has a B.S in Biochemistry, and an M.S in Environmental Science, from Washington State University.

Brian Staab has been the Regional Hydrologist for the U.S. Forest Service Pacific Northwest Region for the last 11 years, and previously the Regional Hydrologist for the Pacific Southwest Region. Mr. Staab has a B.S. in Civil Engineering from Penn State University, and an M.S. in Hydrology and Water Resources Science from Stanford University.

Mike Cloughesy served in an *ex officio* role as Director of Forestry and project manager for the Oregon Forest Resources Institute. Prior to joining OFRI in 2003, Mike was the director of outreach education at the Oregon State University College of Forestry and the assistant leader of the Forestry Extension Program and Professor of Forest Resources. He has a B.S. in Forestry from Iowa State University and an M.S. in Forest Science from Oregon State University.

Table of Contents

THIS PAGE INTENTIONALLY LEFT BLANK

Chapter 1. Introduction: Forestry and Forest Management in Oregon

Jon A. Souder

1.1 Report Purpose & Overview

There are 337 public water providers, servicing almost 3.5 million Oregonians, who rely on surface waters for some or all of their supply. These providers may own their source water watersheds, but many do not. As a result, they have little control on activities occurring in their source watersheds, many of which are forested and managed by a diversity of owners. The Oregon Forest Resource Institute (OFRI) Board has asked OFRI staff to produce a special report, *Trees to Tap*, to be a science-based summary of the effects of forest management on drinking water. The last report on this topic that OFRI commissioned was in 2000 – *Municipal Water Supplies from Forest Watersheds in Oregon: Fact Book and Catalog*, by Paul Adams and Mark Taratoot. That report summarized the findings of a survey of 30 major municipal water systems in Oregon and the literature of the day on forested watersheds and the effects of forest management. The current report is being prepared under contract by OSU's Institute for Natural Resources using faculty from the College of Forestry as subject matter experts (Box 1).

The purpose of this project is to 1) update that report by synthesizing current science about the impacts of forest management on community drinking water supplies, and 2) describe and analyze the management of forested municipal watershed systems. Our report will focus on the 156 Community Water Supplies (i.e., those with 25+ hookups and 15+ year-round residents) that rely on surface water. The project has three major components: (a) the Science Review focusing on four topics identified by the Steering Committee as priorities, sediment/turbidity, changes in water quantity, forest chemicals, and natural organic matter/disinfection by-products; (b) a survey of the 156 water utilities along with three detailed case studies to identify their needs and concerns; and (c) an Atlas of information on each of the source watersheds for the 156 CWS utilities. We will divide forest management effects analysis into three areas: harvesting, roads, and reforestation.

1.1.1 Importance of forests for clean water

Western forests are managed for many diverse purposes, including wood products, recreation, and wildlife habitat. By filtering rain and snowfall and delivering it to streams or aquifers, forests also produce the highest quality and most sustainable sources of fresh water on earth, arguably their most important ecosystem service (NRC 2008; Neary et al. 2009; Creed et al. 2001). Oregonians value water produced from Oregon forests very highly, and continue to rank water quality and quantity as primary concerns with forest management (cite OFRI or other survey?). Oregon's extensive and diverse forests generally produce very high quality water and supply the majority of the state's community water systems. Forest practices designed to minimize impacts to water quality have improved significantly in recent decades. At the same time, demand for all forest ecosystem services continues to rise, against a backdrop of a changing climate and uncertain implications for water derived from forests. Together, these trends point to the importance of maintaining and expanding public awareness of current science knowledge regarding the complex relationships between forest hydrology and forest management.

1.1.2 Approach of the report

The Oregon Forest Resources Institute (OFRI) is a state agency established by the Oregon Legislature in 1991, funded by a dedicated forest products harvest tax and governed by a 13-member board. The institute was created to enhance collaboration among forest scientists, public agencies, community organizations, conservation groups and forest landowners; to provide objective information about responsible forest management; and to encourage environmentally sound forest practices through training and other educational programs (OFRI 2018 [website]). In 2001, OFRI published a report entitled *Municipal Water Supplies from Forest Watersheds in Oregon: Fact Book and Catalog* for the purpose of helping Oregonians better understand relationships between their water supplies and forest watersheds and their management (Adams and Taratoot 2001). The report presented results of a review and summary of relevant science information, and a survey of 30 major municipal water systems in Oregon.

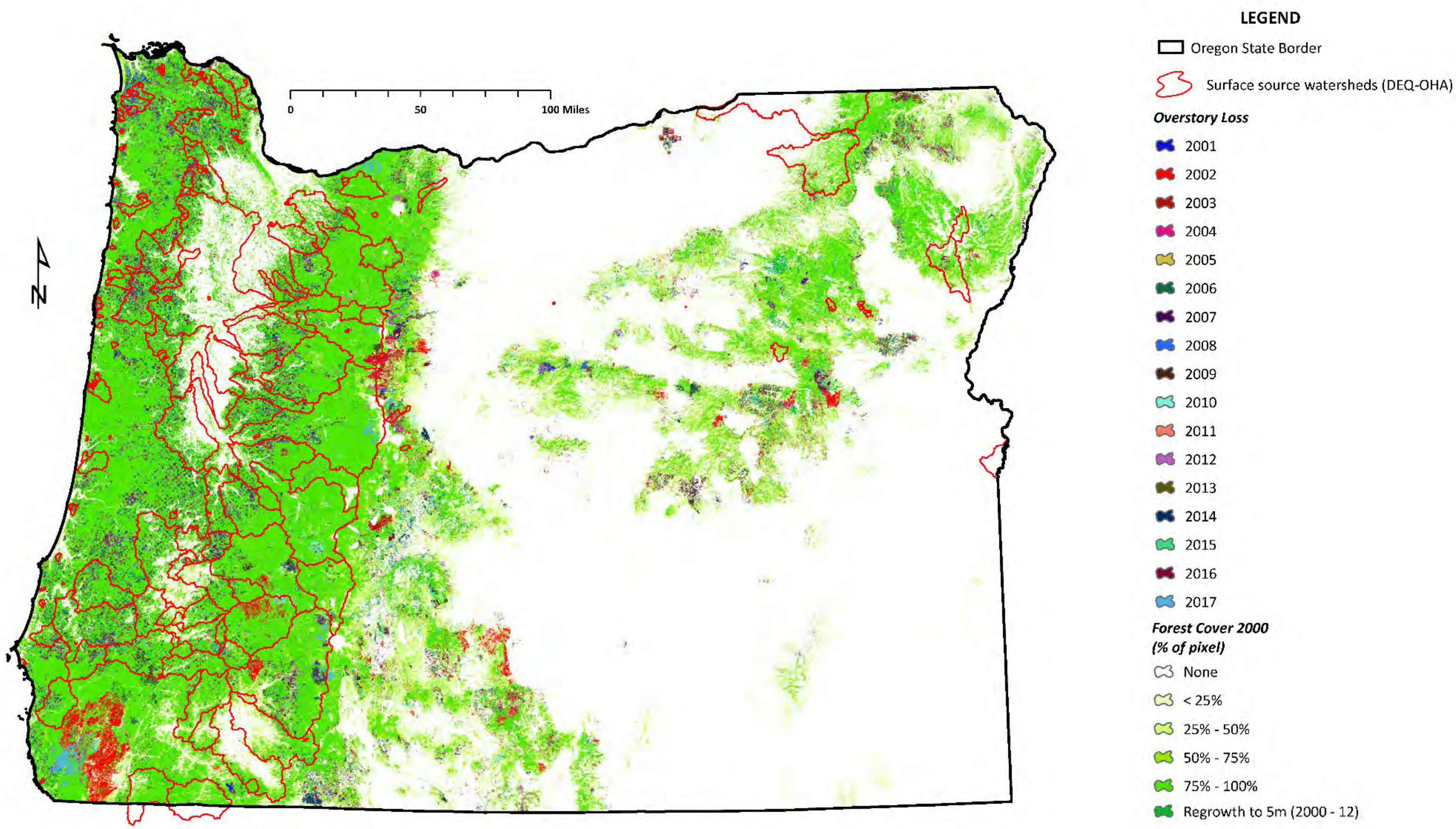
> Box 1. Project personnel and steering committee representation.
>
> Project Personnel:
>
> Jon A. Souder, PI
> Kevin Bladon, Co-PI
> Emily Jane Davis, Co-PI
> Bogdan Strimbu, Co-PI
> Jeff Behan, Senior Policy Analyst
> Lisa Gaines, INR Director, Co-PI
>
> Steering Committee Representation:
>
> Department of Forestry
> Oregon Health Authority
> Department of Environmental Quality
> U.S. Environmental Protection Agency
> U.S. Forest Service
> National Council for Air & Stream Improvement
> Oregon Forest & Industries Council
> Oregon Stream Protection Coalition
> Oregon Association of Water Utilities
> Geos Institute
> Oregon Forest Resources Institute (ex officio)

In fall of 2017, OFRI contracted with Oregon State University (OSU) School of Forestry and OSU Institute for Natural Resources (INR) to revise and expand on the 2001 report in order to reflect more recent research and refinements in Best Management Practices (BMPs). To guide the process, a ten member steering committee was formed with broad representation in spring of 2018 to assist in identifying priorities for the science review, and to review the draft project chapters (Box 1). The Steering Committee met four times from January, 2017 to June, 2018 in the lead-up to the Science Review. Members were provided with review copies of all the draft products as well as the opportunity to comment on them. These comments were incorporated into revised drafts that were then circulated back to the Steering Committee for their review. Steering Committee members were not asked to approve the final products found in this report.

1.2 The landscape of source watersheds in Oregon

Drinking water source watersheds are shown in Map 1-1 along with the amount of forest cover and overstory losses from 2001 – 2017. Source watersheds predominate in the Cascades, Coast Range, and in smaller areas of the Oregon coast. Fewer (only 12) source watersheds exist east of the Cascade ridgeline; more often communities in that part of the state rely on groundwater which is much more dependable than surface water supplies. Also evident on Map 1-1 are areas of overstory loss; for example in the southwest part of the state the area of the Biscuit Fire in 2002 (red) and the Chetco Bar Fire in 2017 (blue). It is in this landscape that we will focus our attention.

Map 1-1. Forest cover change in Oregon, 2001 – 2017. [Note: change this to just forest cover + CWS watersheds]



1.2.1 Water quality and land uses

Community Water Supply water quality at the raw water intake is clearly dependent on land cover and land uses in the contributing drainage area. The Oregon Department of Environmental Quality (DEQ) developed the "Oregon Water Quality Index" (OWQI) to describe overall conditions by stream, region, state-wide, and land use (Brown 2019b). The OWQI is based on water temperature, pH, dissolved oxygen, biological oxygen demand, total solids, nitrogen, phosphorus, and bacteria based on 160 long-term monitoring sites throughout Oregon. Scores are determined from seasonal averages (summer and fall-winter-spring) where high quality data was available for at least ten years. A site is scored from 10 to 100, with scores 90 – 100 rated *Excellent*; 85 – 89 *Good*; 80 – 84 *Fair*; 60 – 79 *Poor*; and 10 – 59 *Very Poor* (Brown 2019a).

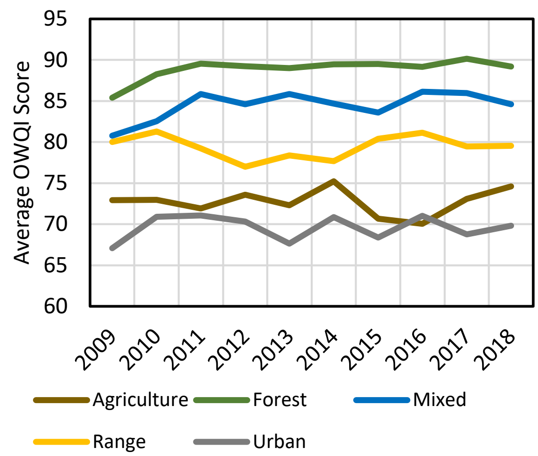


Figure 1-1. Average OWQI scores by dominant land use from 2009 – 2018. Source: https://www.oregon.gov/deq/FilterDocs/OWQIdata.xlsx

Figure 1-1 shows state-wide Oregon Watershed Restoration Inventory (OWRI) results for five different dominant land uses in a five mile upstream buffer from the sample site (Brown 2019b). The "Mixed" category is used when none of the other four land uses exceed 50% in the upstream area. Data is from over 8,500 samples collected by DEQ from water years 2009 through 2018, averaged by dominant land use for that water year. It's clear that the cleanest water is coming from samples where the dominant upstream land use is forest. Also noteworthy is that year-to-year variability in water quality is less (at least since WY2011) compared to the other land uses.

Table 1-1. Average OWRI scores by basin (region), WY2009 – WY2019.

Basin	Agriculture	Forest	Mixed	Range	Urban	*Average*
Columbia			86		86	*86*
Deschutes	47	92	92	81		*86*
Grande Ronde	91	91	87			*90*
Hood	81	85	90	84		*83*
John Day	88			87		*87*
Klamath	31	82	67	90		*62*
Malheur	28			54		*40*
Mid Coast		89				*89*
North Coast		86	78			*85*
Closed Lakes	79			83		*82*
Owyhee	52			84		*73*
Powder				73		*73*
Rogue	79	90	80			*86*
Sandy			91			*91*
South Coast		89				*89*
Umatilla	74		82			*75*
Umpqua		89	85			*87*
Lower Willamette	60	95	89		68	*70*
Middle Willamette	82	95	90		91	*85*
Upper Willamette	85	94	93			*88*
Statewide Averages	*73*	*89*	*84*	*79*	*70*	*81*

Source: https://www.oregon.gov/deq/FilterDocs/OWQIdata.xlsx

Aside from just land use, there are regional differences in average OWQI scores throughout the state. Table 1-1 provides information on these differences based on DEQ watershed basin. In any given basin (with the exception of Hood River), the water quality from dominantly forest land use matches or exceeds the scores for other uses. Removing "Mixed" from the analysis, the water coming from forest land uses is often substantially better than from agriculture and urban, and is better on average from range (the other non-intensive land use).

1.2.2 Forests in Oregon

How you define "forest" determines their extent. According to the U.S. Geological Survey's National Land Cover Database that uses 30-m resolution Landsat satellite data, forests cover about 35% of Oregon (including open water) (Oregon Explorer Land Cover 2011: https://oregonexplorer.info/tools/oe-atlas). This definition is based on having 25% or greater tree canopy cover within the 30-m pixel. Using the US. Forest Service (USFS) definition of "forest," 47% of the State of Oregon is forested, with about 38% of the state considered "commercial" quality timberland (Palmer et al. 2018). The USFS definition is based on having 10% or greater tree canopy cover, but requires a minimum one acre size and at least 120 feet of width. We'll use the USFS definition in our discussion.

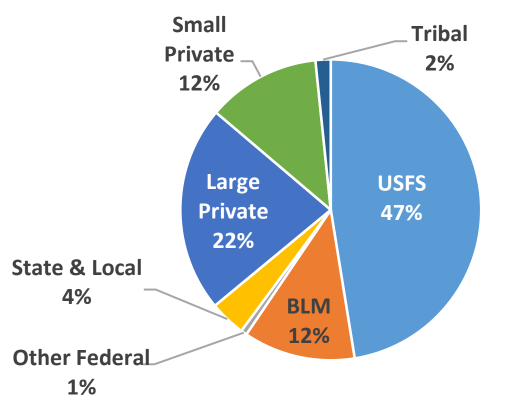


Figure 1-2. Ownership of forest lands in Oregon. Source: OFRI 2019

These forests are held and managed by a diversity of owners (Figure 1-2). The USFS manages 47%, the U.S. Department of Interior Bureau of Land Management (BLM) another 12%, and other Federal owners 1%. The State owns 3%, and counties and local government another 1%. Thus, almost two-thirds of forests in Oregon are in public ownership. Tribal forests comprise 2%, with large private owners (> 5,000 acres) owning 22% and smaller private owners 12% (OFRI 2019).

1.2.3 Land uses and ownership in Community Water Supply watersheds

Community water supply (CWS) source watersheds overlay larger land cover and land use patterns in Oregon. These source watersheds cover __% of the state, and are outlined in Figure 1-Q. Figure 1-3 shows the averages percentages by different owners and land uses in the source watersheds for the 156 community water supplies. These percentages are not area-weighted, but rather the average of the percentages for each individual CWS. The greatest proportion, approximately one-third, of source watershed areas are owned by industrial (≥ 5,000 acres) timberland owners; although in aggregate just

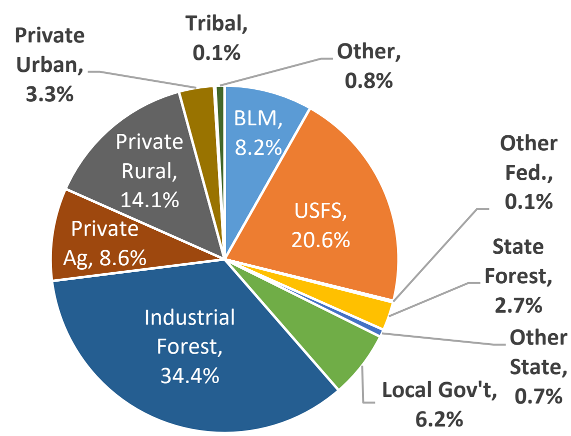


Figure 1-3. State-wide ownership and land uses of source watersheds. Source: DEQ dwpLandUseSumtable.xlsx.

under 40% of source watersheds are in public ownership (Federal, State, local). Small woodland owners and rural residential properties own 14%, with private agricultural lands almost 9%, and urban just over 3%.

The watershed ownership pattern differs, however, regionally. Table 1-2 shows this same land cover and land use pattern divided into three broad regions of the state: the Oregon Coast; Valleys, principally the Willamette, Umpqua, and Rogue; and the Dryside, which is everything east of the Cascade divide. In evaluating Table 1-2, it's important to recognize that the Dryside contains only 12 CWSs, in comparison to the 90 in the Valleys and 54 in the Coast regions.

Table 1-2. Average percent source watershed ownership by region.

Ownership Class	Coast	Valleys	Dryside	State
BLM	2.9%	13.5%	2.3%	8.2%
USFS	21.6%	19.7%	54.1%	20.6%
Other Federal	0.0%	0.3%	0.0%	0.1%
State Forest	4.0%	1.3%	0.0%	2.7%
Other State	1.1%	0.3%	0.2%	0.7%
Local Government	8.6%	3.9%	0.0%	6.2%
TOTAL PUBLIC	37.5%	38.6%	56.6%	38.1%
Industrial Forest	42.7%	26.1%	2.2%	34.4%
Private Ag Land	2.1%	15.2%	19.5%	8.6%
Private Rural Land	12.5%	15.8%	12.0%	14.1%
Private Urban Land	2.9%	3.6%	7.8%	3.3%
Tribal Land	0.1%	0.2%	1.8%	0.1%
Other Land & Water	1.6%	0.1%	0.1%	0.8%

Source: DEQ dwpLandUseSumtable.xlsx. Averages are not area-weighted, but by individual CWS percentages.

The state-wide averages from Figure 1-3 are shown in the right-most column for comparison with the three regions. Industrial timberland predominates on the Coast, is rare on the Dryside, and is about a quarter of the ownership in the Valleys region. The Valleys CWSs have greater relative public ownership compared to industrial timber, but also higher percentages of rural private owners and agriculture. Dryside source watersheds are predominantly publically owned (mostly managed by the USFS).

It's important to recognize that each CWS typically has a unique pattern of ownerships and land uses; information specific to each CWS will be provided in the Atlas accompanying this report. Some CWSs will have 100% of a single owner/manager, which can be either private or public. Others will have a diverse mix of land uses and owners. Even within a broad category, individual owners and managers will likely have different objectives and land management perspectives. Each situation will bring its own management opportunities and challenges, as well see in Chapter 3 with the CWS survey discussion, and Chapter 9 with the case studies of three CWSs.

1.2.4 Forest cover change

Oregon's forests are constantly changing at scales ranging from individual trees to stands to larger forest units. As we'll demonstrate in this report, these changes affect water quality differently depending on location, scale, and duration. Recent advances in interpretation of satellite images allow for refined analyses of forest cover change. For this project, we have partnered with the eMapr group in OSU's College of Earth, Oceans, and Atmospheric Sciences (CEOAS) to use their LandTrendr tools (Cohen et al. 2018; Masek et al. 2013) and Landsat data hosted by Google Earth Engine to identify forest cover changes statewide, as well as specifically for each of the 156 community source watersheds. Using Landsat satellite 30-m pixel data, we have constructed a time-series of forest cover change from 1986 through 2018; and applying image interpretation and ancillary data, have been able to ascribe to each pixel a cover condition and likely source of disturbance if cover has changed. Essentially, we have been able to identify abrupt changes in forest cover (one year to the next); slow changes to forest cover that occur over a number of years, and recovery from disturbed conditions until "forest" is again achieved (i.e. trees about 16 feet in height).

We are interested in three basic causal factors driving forest cover change: timber harvest, fire, and disease and insect mortality. Harvest and fire tend to be abrupt forest cover changes, while disease and insect infestations are typically gradual. Ancillary annual data on wildfires (www.mtbs.gov; Eidenshink et al. 2007) is used to separate this disturbance factor from other abrupt changes, with the residual most likely the result of timber harvest. We are currently evaluating the ability of the Oregon Department of Forestry (ODF) Forest Activity Electronic Reporting and Notification System (FERNS) harvest notification spatial data to refine this identification). Identifying insect and disease effects on forest cover is more difficult because the changes occur over a longer time period; a procedure was developed to track individual pixels over time to identify downward trends in forest condition. Similarly, the slow process of recovery of forest cover can be tracked over time using essentially the same procedure. Figure 1-4 shows the results of our forest cover change analyses on a state-wide basis over the years 1986 to 2019. Change in this context is the percent of the state disturbed (recovered) by that causal factor in any given year. Categories displayed in the figure are: 1 – Undisturbed; 40 – Fire; 100 – Unknown Slow Disturbance; 110 – Unknown Abrupt Disturbance; 111 – Unknown Abrupt Disturbance continuing a 2nd year; and 160 – Recovery.

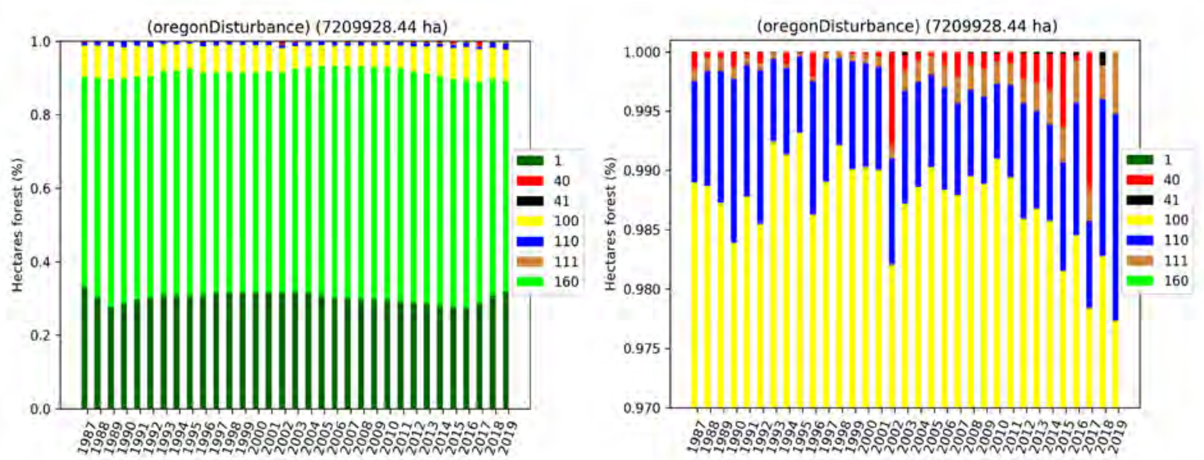


Figure 1-4. Oregon forest cover change 1987 – 2018. Note scale change on right chart to highlight disturbances.

Raw drinking water quality for any given community water system is likely to be affected by large disturbances (and even recovery) in the source watershed. The Resource Atlas accompanying this report provides information identical to Figure 1-4 for each of the 156 community water supplies (some of which have multiple source watersheds). This was an unbudgeted add-on to the project, and the results have not been thoroughly calibrated and so should be considered estimates and trends.

1.3 Overview of active forest management in Oregon

For the purposes of this report, we will divide forest management activities into three basic categories: (1) harvest; (2) forest roads; and (3) revegetation. While these activities are inter-connected, and inter-related, distinguishing between them will make our analysis clearer. Generally, the sequence of forest management operations, at least those that result in harvest, is that roads are built/reconstructed into the site, harvest activities are conducted, and then revegetation is initiated if needed. Each one of these three general categories has a host of management "activities" associated with them. And, these different activities may affect water quality at the intake for community water systems.

The Oregon Department of Forestry uses the web-based, electronic Forest Activity Electronic Reporting and Notification System (FERNS) for landowners and operators to "notify" the State Forester of forest management activities as required under the Oregon Forest Practices Act. This system, which went on-line in October, 2014, replaced the traditional paper notifications and has the benefit of providing digital, geospatial information on forest management activities, including—for forest harvest only—activities on Federal lands. A single "Notification" can include multiple units (typically harvest) and multiple management activities. For each activity, a method used to accomplish the task is also identified. Beginning and ending dates for the operation are also required. A Notification lasts for the specific calendar year (ending 12/31), however, with ODF approval, they can be extending for multiple years until all the operations "notified" are completed.

We used data from ODF Notifications for four complete calendar years (2015 – 2018) to identify the types and magnitude of forest management activities during this period. This dataset included about 59,625 unique notifications that cover 112,839 units and activities. There is no set protocol for how many units and activities are included in a single Notification. A single NOAP may contain all the units and activities that a land manager anticipates during the year; or the landowner could submit multiple NOAPs for the same unit, each covering a single activity. The number of units/activities covered in a single Notification ranges from one (1) to a high of 81; 57% of NOAPS contain only a single unit and activity, another 27% contain only two units/activities (typically two activities on the same unit); and 98% have six units/activities or less. It is important to note that not every activity notified is actually conducted; and that the dataset is not perfect: there is duplicate information in differently numbered NOAPS. Thus, our results should be interpreted as estimates, trends, and comparative magnitudes of forest management activities.

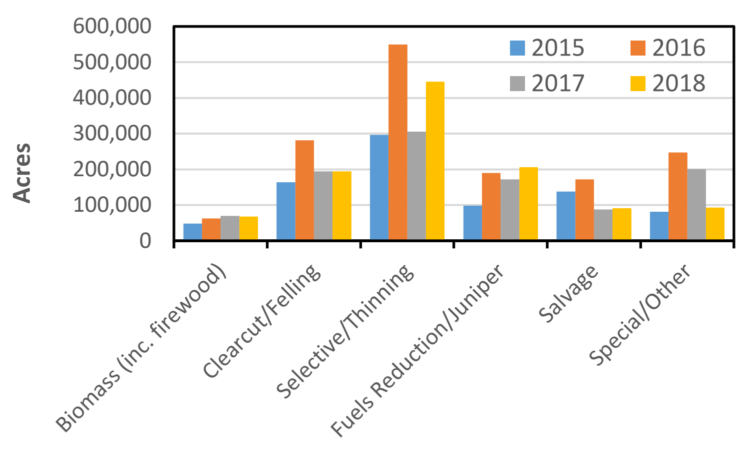


Figure 1-5. Harvest activities in Oregon, 2015 – 2018 (ODF FERNS).

1.3.1 Harvests

There are eleven different activities that we are categorizing under "harvest"; several of these are combined in Figure 1-5 in order to simplify the chart. Over the four years, harvests covered an average of 1,114,000 acres per year, with a low of 826,000 acres in 2015 to a high of 1.5 million acres in 2016. As a gross simplification, the amount of harvesting is very dependent on timber prices, which are in turn dependent on housing starts and—to a lesser extent—export markets. In terms of acreage, selective harvests and thinning are approximately twice the area compared to clearcuts (an average of almost 400,000 acres compared to 209,000), with salvage (the harvest of dead, down, or burned trees) being about half (122,000) of clearcuts. Fuels reduction and juniper treatment—typically found in the drier areas of the state—averages about 167,000 acres per year. The "Special/Other" category includes areas (map polygons) that are likely associated with harvest activities, and may occur adjacent to the harvest unit but that need to be cleared for yarding.

1.3.2 Roads

Active forest management requires access to the site of the activity, and roads must be maintained once they are built. Figure 1-6 shows the amount of road-related work notified in 2015 – 2018 in terms of length. Many forested areas are already roaded, so road reconstruction prior to harvest operations is more common (averaging about 1,500 miles/year) than new road construction (about 800 miles/year). The first step in constructing a new road (and occasionally in re-opening older roads) is to fell trees in the right-of-way and haul them to the mill; this averages about 300 miles per year state-wide. Once roads are constructed, they need to be maintained. Standard maintenance operations (grading, cleaning ditches, spot rocking, mechanical brushing) do not require a FERNS notification, but herbicide applications do, which average over 900 miles of road annually. Around 4,000 acres/year are used for rock pit development and management.

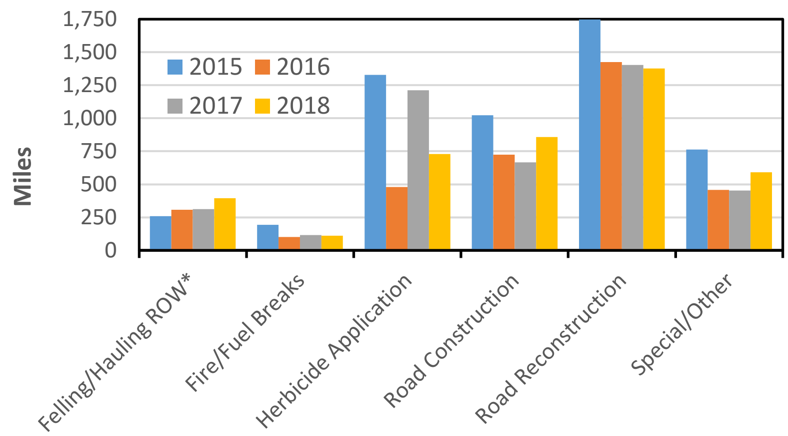


Figure 1-6. Forest road activities in Oregon, 2015 – 2018 (ODF FERNS).

Similar in physical impacts to roads, fire and fuel breaks are constructed on an average of 130 miles annually. Special Activities/Other contain a wide diversity of actions, including brushing (Power-Driven Machinery [PDM] permit), rocking, road decommissioning, and stream habitat improvements. It is also worthwhile to note that utility line and railroad line maintenance within and adjacent to forest lands also require notifications for fire and PDM; these average about 1,400 miles/year for utility lines, and 240 miles/year for rail lines.

1.3.3 Reforestation

The Oregon Forest Practices Act requires reforestation within 24 months after harvest if the remaining stock of trees is below a threshold set based on site class (i.e., the ability to grow trees on a specific area of land) (OFRI 2018). Reforestation is required after clearcutting, and may be needed in selection harvesting depending upon the remaining number of trees and their size. On highly productive land (Site Classes 1 – 3), the residual requirements are 200 trees per acre for seedlings; 120 trees per acre if they're saplings or poles 10" dbh or less; or 80 basal area per acre of trees larger than 10" dbh.

Similar to the other two categories, reforestation involves about a dozen different activities that are reported in FERNS. A typical sequence would be slash treatment and site preparation needed from the previous harvest. Then the site would be planted (an activity not requiring notification). Prior to, and after the planting, herbicide may be used to control competing vegetation to conserve growing space and moisture. Once planted, animal repellants and rodenticides may be needed to insure that the seedlings survive. Depending upon the density of seedlings planted that survive, there may be a need to pre-commercial thin the stand (usually at 10 – 15 years of age), and after thinning (both pre-commercial and commercial) fertilizer is applied to provide nutrients needed to accelerate height and crown growth. Fire may be used as part of site preparation and slash treatment, or to reduce fuel to decrease the likelihood of high intensity burns.

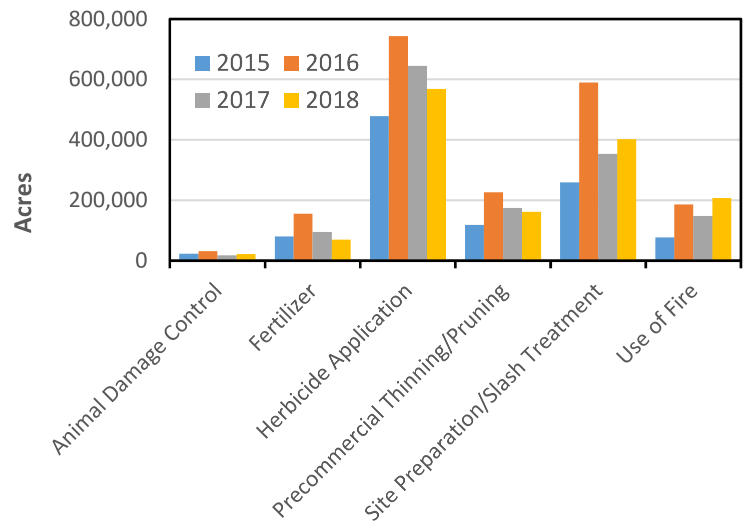


Figure 1-7. Reforestation activities in Oregon, 2015 – 2018.

Figure 1-7 simplifies reforestation activities into 6 types, eliminating insecticide and fungicide applications since they are so rare (see Chapter 7); and combines commercial thinning and pruning into a single category, and site preparation and slash treatment into another. On average, about 1.5 million acres of Oregon's private and State forests have reforestation treatments annually. By far the greatest extent is herbicide treatments, averaging about 600,000 acres annually. As we'll discuss in Chapter 7, there may be multiple applications of herbicides as part of site preparation and revegetation, which is likely why the area treated with herbicides is 50% greater than the acreage of site preparation/slash treatment (about 400,000 acres annually). Stand growth is accelerated through pre-commercial thinning (averaging 170,000 acres annually) and fertilization (about 100,000 acres annually). Prescribed fire is used on just over 150,000 acres of private and State land annually.

1.4 Context Matters

At numerous points in this report we will conclude our analysis of the potential effects of active forest management on source water quality with various qualifications or caveats that can be summarized as "it depends." This is because the diversity of source watershed sizes, land uses, geographic regions, geomorphic conditions and other factors makes generalizations difficult. Additionally, as we'll show, much of the research on active forest management effects on overall water quality has been conducted in upper watershed areas, and hasn't focused on tracing effects sufficiently downstream to raw source water intakes to reasonably infer cause and effect. Difficulties in accounting for the confounding effects of intervening land uses that occur below forestry activities in the watershed but above the drinking water intake may help explain the paucity of research on forestry and drinking water connections (Figure 1-8). We have done our best, however, to highlight possible linkages and allow the reader to make inferences based on their particular situation. There are three general categories that we emphasize as important in placing our findings in context.

1.4.1 Size of Watershed versus Scale of Forest Management Activities

First, the size of the source watershed in relation to the scale and frequency of forest management activities clearly matters. The smallest community water supply, the Bay Hills Water Association in Lincoln County, has a source watershed of only 0.04 square miles, or slightly over 26 acres. In contrast, the largest source watershed, the City of Wilsonville, is 1,641 square miles, or 40 thousand times as large! A similarly sized management activity will have vastly different potential effects on a smaller compared to a larger watershed. Similarly, cumulative effects of management activities are likely to have greater effects in smaller compared to larger source watersheds. Larger source watersheds are also more likely to have a higher diversity of land uses, which we saw in Figure 1-1 affects water quality. The forest cover change information derived from satellite imagery presented in the Atlas will be reported as the percent (%) of the source watershed affected.

1.4.2 Geography and geomorphology

Second, just as size matters, so does where the source watershed is located and the landforms within the drainage basin. Coastal watersheds tend to be rainfed, with peak flows in the winter during storms and pronounced dry periods in the late summer. Conversely, watersheds draining into the Valleys (Willamette, Umpqua, and Rogue) are more likely to have snowpacks at upper elevations that retain moisture into the spring and early winter, moderating and lengthening flows. Finally, source watersheds in Central and Eastern Oregon also have snowpacks (albeit holding less water), but rainfall patterns shift to the summer “monsoon” season. Thus, precipitation and runoff patterns vary by geography, and these variations influence how forest management activities affect source water quality.

The landforms and underlying geology of the source watershed also shape how active forest management influences source water quality in important ways. Steeper slopes and shallow soils are more likely to slide; higher gradient watersheds transport streamflow more rapidly downstream all other things being equal. Basalt geology transports groundwater more rapidly than sandstone geology, which again influences annual streamflows as well as source water quality. Some soil and rock types are much more erodible than others, which directly affects the amount of sediment mobilized by harvesting activities and forest roads. So understanding landforms and its underlying geology is important to evaluate the effects of active forest management on source water quality.

1.4.3 Land ownership

Third, who owns the drinking water sources watershed affects the types, intensity, and scale of forest management activities. From the water utility’s perspective, having control over activities in their source watershed provides the best insurance of maintaining future water quality and quantity. This can be achieved by owning their source watershed; coming to agreement with landowners in their watershed on how the lands will be managed; or increasing regulatory oversight to prevent undesirable outcomes. Public ownership provides opportunities for water utilities to participate in planning processes, and generally involves environmental analyzes that highlight management effects on water quality. In contrast, with private ownership the water utility may or may not be able to cooperatively plan forest management activities in its source watershed, and instead may have to rely solely on the regulatory process. Thus, who owns the source watershed affects the likelihood that one of these arrangements will be available.

Ownership also affects various types and intensities of forest management activities. Important factors such as the age of harvest, and harvest types, vary by owner. Revegetation, particularly the use of forest chemicals, differs among owners too. Finally, the regulatory process varies by ownership: Federal lands have to be managed by the responsible agency's regulations, as well as laws such as the National Environmental Policy Act. State lands operate under a different set of criteria, but have their own management plans that outline permissible activities. Management actions on private lands, as well as State lands, are governed by the Oregon Forest Practices Act (FPA) and its regulations. Therefore, the types and diversity of land ownership within a source watershed will influence the types and intensities of forest management activities that can potentially occur, and by extension, influence the effects of these management actions on source water quality (Figure 1-8).

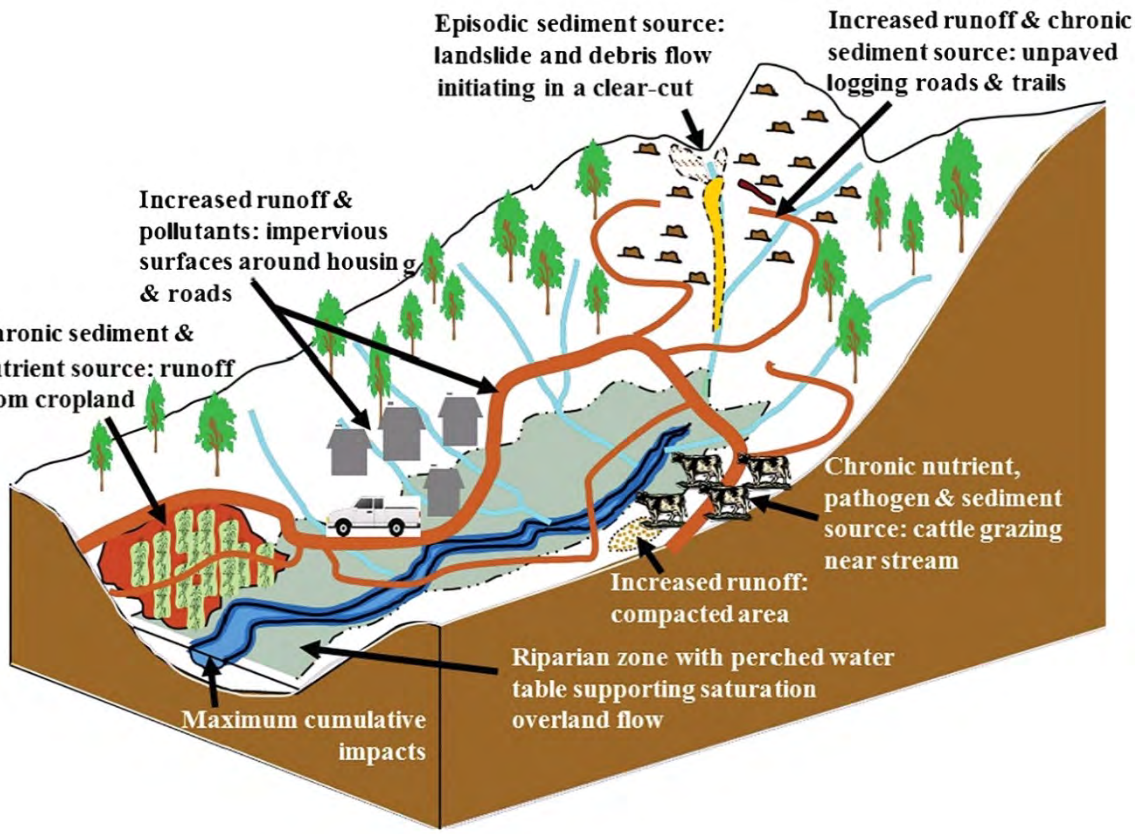


Figure 1-8. How multiple land uses within a watershed can generate cumulative effects related to streamflow regime, sedimentation, and nutrient/pollutant fate and transport. Source: Sidle, R.C. and Gomi, T., 2017. Hydrologic Processes in Forest Headwater Catchments: Implications for Policy and Management. In Mountain watersheds and ecosystem services: Balancing multiple demands of forest management in head-watersheds, R.Tognetti, G. Scarascia Mugnozza and T. Hofer (eds). EFI Technical Report 101. Publisher: European Forest Institute, Yliopistokatu 6, FI-80100 Joensuu Finland.

1.5 Organization of the report

The body of the report will include nine chapters, three major appendices, and the Atlas of Community Water Systems. Chapter titles and a brief summary of their contents follow.

Chapter 2, *Community Drinking Water Systems in Oregon*: characteristics, regulations and management, treatment processes, CWS survey results.

Chapter 3, *Active Forest Management and Community Water: Issues and Interactions*: stream sediment, water production, forest chemicals and nutrients, natural organic matter and disinfection byproducts, best management practices (BMPs), implementation of BMPs in Oregon, controversial and unresolved issues.

Chapter 4, *Water Quantity*: context, annual yields, peak flows, low flows, and timing.

Chapter 5, *Sediment and Turbidity*: effects on water treatment, harvesting effects, forest roads, increased landslides.

Chapter 6, *Forest Chemicals*: background, chemicals used in Oregon forestry, chemical descriptions, review of effects, prevalence of forest chemicals in streams, chemicals in raw water supplies and potable water treatment.

Chapter 7, *Natural Organic Matter* (NOM)*/Disinfection By-products* (DPB): overview, chemistry and issues, NOM and potable water treatment, review of forest management effects, prevalence of standards exceedance, drivers and effects by region.

Chapter 8, *Fire Risk Assessment*: modeling approach, state-wide and regional patterns, incorporation into planning.

Chapter 9, *Case Studies*: background, Ashland Water Department, Baker City Water Department, Oceanside Water District, lessons learned.

Chapter 10, *Key Findings, Recommendations, and Information Gaps*.

1.6 Literature Cited

Brown, D. 2019a. Oregon Water Quality Index: Background, Analysis and Usage. Oregon Department of Environmental Quality, Laboratory and Environmental Assessment Program, Hillsboro, OR. 7 pp. https://www.oregon.gov/deq/FilterDocs/wqmreportingmethodsF.pdf accessed 12/3/2019.

Brown, D. 2019b. Oregon Water Quality Index Data Summary, Water Years 2009 – 2018. Oregon Department of Environmental Quality, Laboratory and Environmental Assessment Program, Hillsboro, OR. 7 pp. https://www.oregon.gov/deq/FilterDocs/WQI2018DataSummary.pdf accessed 12/3/2019.

Cohen, W.B., Z. Yang, S.P. Healey, R.E. Kennedy, and N. Gorelick. 2018. A LandTrendr multispectral ensemble for forest disturbance detection. Remote Sensing of the Environment 205: 131-140.

Eidenshink, J., B. Schwind, K. Brewer, .Z-L. Zhu, B. Quayle, and S. Howard. 2007. A Project for Monitoring Trends in Burn Severity. Fire Ecology Special Issue 3(1): 3-21.

Masek, J.G., S.N. Goward, R.E. Kennedy, W.B. Cohen, G.G. Moisen, K. Schleeweis, and C. Huang. 2013. United States Forest Disturbance Trends Observed Using Landsat Time Series. Ecosystems: 1087-1104.

Oregon Forest Resources Institute (OFRI) 2019. Oregon Forest Facts, 2019-2020 Edition. Portland, OR 24 pp.

Palmer, M. O. Kuegler, G. Christensen (tech. eds.). 2018. Oregon's forest resources, 2006–2015: Ten-year Forest Inventory and Analysis report. Gen. Tech. Rep. PNW-GTR-971. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR. 54 pp.

Sidle, R.C. and Gomi, T., 2017. Hydrologic Processes in Forest Headwater Catchments: Implications for Policy and Management. In Mountain watersheds and ecosystem services: Balancing multiple demands of forest management in head-watersheds, R.Tognetti, G. Scarascia Mugnozza and T. Hofer (eds). EFI Technical Report 101. Publisher: European Forest Institute, Yliopistokatu 6, FI-80100 Joensuu Finland.

Chapter 2. Community Drinking Water Systems in Oregon

Emily Jane Davis, Jon Souder, and Jeff Behan

This chapter begins with a summary of Oregon's administrative framework for delivery of clean drinking water, and the state's community water providers and raw water treatment processes they use. Next is an overview of the diversity of landownerships in source watersheds, and commonalities and differences among smaller and larger water providers, with a particular focus on smaller systems where the source watershed is not owned by community and is potentially affected by active forest management and/or multiple uses. The chapter concludes with results and findings from a survey of Oregon water providers that was conducted specifically for this report, along with three case studies to help illustrate the diversity, challenges and successes among Oregon's community water systems.

About 35% of Oregonians rely solely on groundwater for drinking, mostly via small public water systems or private wells. About 10% rely solely on surface water. The remaining 55%—mostly large community water systems—rely on both surface water and groundwater, usually with groundwater as an emergency backup. There are 238 source watersheds that feed into 157 water treatment plants operated by 156 community water systems (i.e., those with 15 or more service connections for at least 25 people year round). These watersheds utilize surface water, and shallow wells that are influenced by surface water, to provide the raw water source for almost 3 million Oregonians. These watersheds are located throughout the state, although communities in eastern Oregon are more likely to depend on groundwater rather than surface water as their source of supply. Figure 2-1 shows these source watersheds, identifying those specifically that responded to our survey of water providers, and the three locations where in-depth case studies were conducted that will be reported in Chapter 9.

Most community water systems serve small populations. The USEPA classifies community water systems into five categories dependent upon the population they serve (Tiemann 2017). Very Small systems are those that serve less than 500 people. Small systems serve from 501 to 3,300 people. Medium systems provide water to between 3,301 and 10,000 persons. Large systems provide water to between 10,001 and 100,000 people; and Very Large systems serve communities of greater than 100,000 population. Figure 2-2 shows the proportion of population provided drinking water from surface sources based on the size of the system (a), and the proportion of CWS by their size class (b). Over 75% of the population reliant on surface water is served by large or very large community water providers, even while there is a significant number of people (14%) supplied by very small systems. In contrast, these large and very large water systems comprise only 20% of the number of community water supplies providers reliant on surface water, with only 7 very large systems supplying half (50%) the population. Almost two-thirds of the community water providers dependent on surface water serve small (35%) or very small (29%) populations that limit their infrastructure capacity.

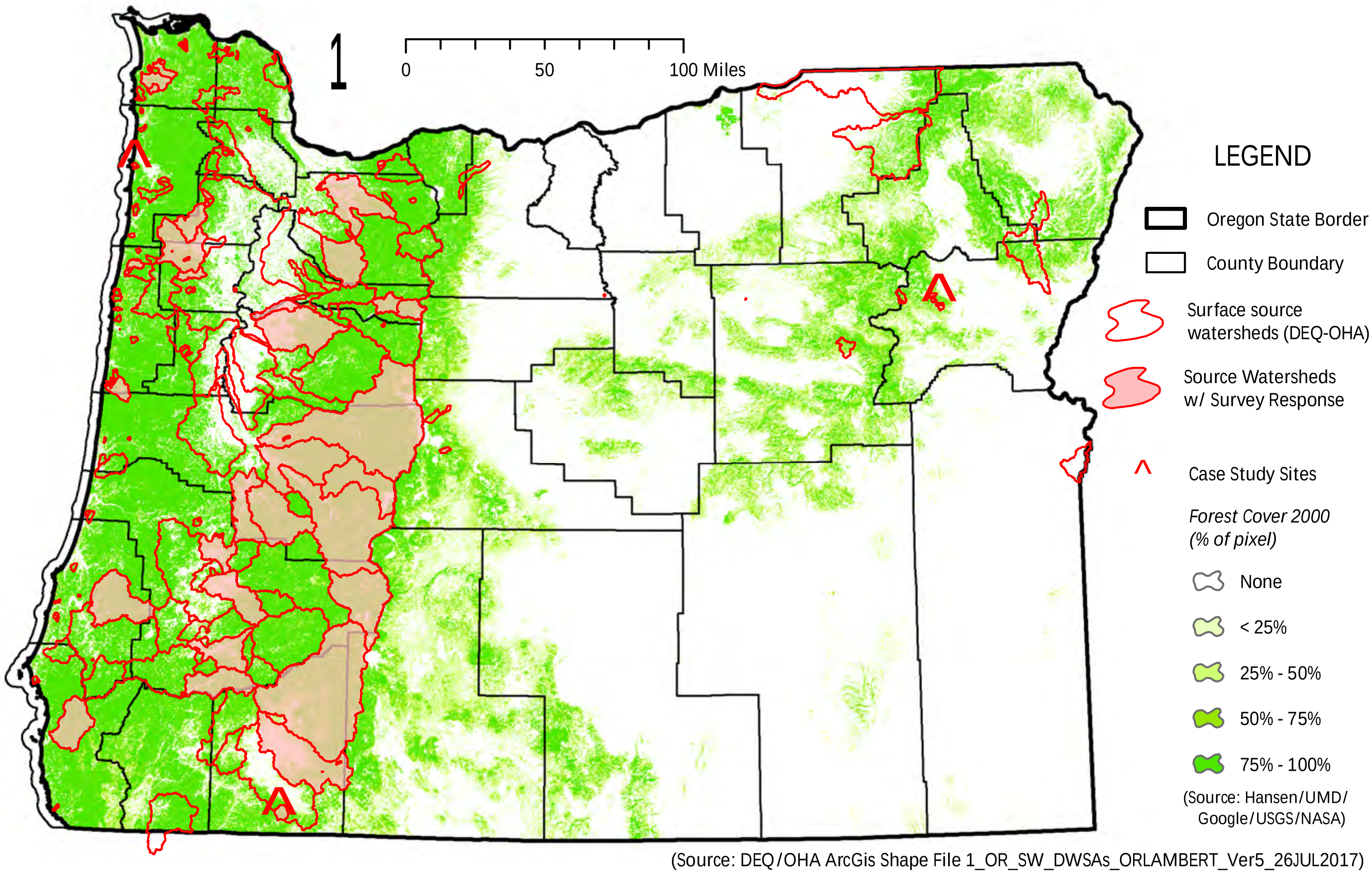
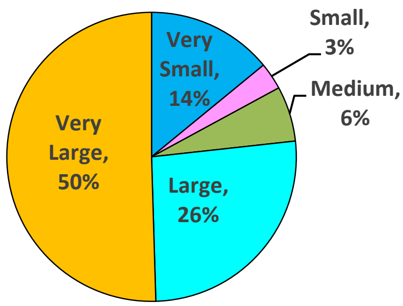
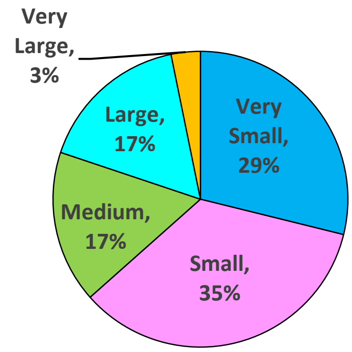


Figure **2-1**. Drinking water source watersheds showing those that responded to the water provider survey and the case study locations.

a). Population served by CWS size class.



b). Proportion of CWS systems by size class.



Source: DEQ/OHA ArcGIS Shape File 1_OR_SW_DWSAs_ORLAMBERT_Ver5_26JUL2017.

Figure 2-2. Characteristics of community water systems (CWS) reliant on surface water by size class.

2.1. Regulation and management of drinking water in Oregon

This section discusses federal statutes and regulations that pertain to drinking water, how these statutes are coordinated to address different but complimentary aspects of drinking water protection, and Oregon's administrative framework for interpreting and implementing them. The Oregon Department of Environmental Quality (DEQ) provides reports, general information and technical assistance regarding surface water systems, while the Oregon Health Authority (OHA) supplies these services for groundwater systems. (Oregon DEQ 2018b). In addition, the OHA regulates the treatment and distribution of potable water under the Federal Safe Drinking Water Act, while the DEQ has regulatory authority under the Federal Clean Water Act for point and non-point sources of pollution.

2.1.1. The Clean Water Act

The Clean Water Act (CWA) provides the basic structure for regulating discharges of pollutants into U.S. waters via national water quality criteria recommendations developed and administered by the USEPA and mostly delegated to the States and Tribes for implementation. This regulatory framework makes a key distinction between *point* sources and *nonpoint* sources of pollution. The CWA made it unlawful to discharge any pollutant from a point source into waters, unless a permit is obtained from USEPA or an authorized State or Tribe under the National Pollutant Discharge Elimination System (NPDES) permit program. Point sources are discrete conveyances such as pipes or human-made ditches. (USEPA 2018a).

The USEPA defines nonpoint source (NPS) pollution as pollution from diffuse sources resulting from land runoff, precipitation, atmospheric deposition, drainage, seepage or hydrologic modifications. (USEPA 2018b). NPS pollution is caused by rainfall or snowmelt moving over and through the ground, where it picks up and carries natural and human-made pollutants, depositing them into surface waters and ground waters. Logging operations are typically dispersed across large areas and affected by natural variables such as weather, channel morphology, or geology and soil characteristics of the watershed.

This presents challenges in clearly distinguishing harvesting impacts from natural factors. Thus, it was relatively straightforward for the USEPA to define silvicultural activities such as thinning, harvesting, site preparation, reforestation, prescribed fire, wildfire control and pest control as NPS sources. (USEPA 2018c). The USEPA also defines forest road construction, use and maintenance as NPS sources, which has been more controversial. Chapter 4 provides a more detailed discussion of this issue.

Due to its generally dispersed nature, NPS pollution is addressed through area-wide management planning processes and voluntary incentive-based, quasi-regulatory, or regulatory programs. Oregon and other western states have had regulatory programs to address NPS pollution from forest operations (in the form of forest practice acts) since the 1970s. Because NPS pollution causes about 60% of water quality impairments, Congress amended the CWA in 1987 to establish the Nonpoint Source Pollution Management Program under Section 319, which provides States and Tribes with grants to implement controls described in their approved NPS pollution management programs. (USEPA 2018c).

2.1.2. The Safe Drinking Water Act

The Safe Drinking Water Act (SDWA) was enacted in 1974, and significantly expanded in 1996, specifically to protect drinking water quality. The SDWA focuses on all U.S. surface water or groundwater sources actually or potentially used for drinking, and requires USEPA to establish and enforce standards to protect tap water. The USEPA National Primary Drinking Water Regulations (NPDWR) are legally enforceable standards, treatment techniques and water-testing schedules that apply to public water systems. The NPDWR place legal limits - "maximum contaminant levels" (MCLs) - on over 90 drinking water contaminants. The MCLs are levels that protect human health and that water systems can achieve using the best available technology. Regulated contaminants are grouped as follows:

- Microorganisms
- Disinfectants
- Disinfection Byproducts (DBPs)
- Inorganic Chemicals
- Organic Chemicals
- Radionuclides

The USEPA also established National Secondary Drinking Water Regulations (NSDWRs) that set non-mandatory water quality standards for 15 so-called “nuisance” contaminants. These "secondary maximum contaminant levels" (SMCLs) serve as guidelines to assist public water systems in managing their drinking water for aesthetic effects (e.g. taste, color, odor), cosmetic effects (e.g. skin or tooth discoloration) and technical effects (corrosion, staining, scaling or sedimentation in distribution systems or home plumbing). These contaminants can result in significant economic impacts, e.g. by reducing the efficiency of distribution systems, but are not considered to be human health risks at the SMCL. (USEPA 2017a, b).

The USEPA uses the Unregulated Contaminant Monitoring Rule (UCMR) program to collect occurrence data for contaminants suspected to be in drinking water, but for which health-based standards have not been set under the SDWA. These data are collected to support USEPA decisions regarding whether to regulate particular contaminants to protect public health. Every five years USEPA reviews the list of unregulated contaminants, largely based on the Contaminant Candidate List (CCL), a list of contaminants that 1) are not regulated by the NPDWR; 2) are known or anticipated to occur at public water systems and, 3) may warrant regulation under the SDWA. (USEPA 2017c).

In 2006, based on evidence that *Cryptosporidium* and other microbial pathogens are highly resistant to traditional drinking water disinfection practices (usually chlorination), and that the disinfectants themselves can react with naturally occurring materials in water to form byproducts that may pose health risks, the USEPA enacted updated rules to balance the risks of microbial pathogens and disinfection byproducts (DBPs). Under these rules - the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) and Stage 2 Disinfection Byproduct Rule (Stage 2 DBPR) - surface water systems are required to monitor source water for *Cryptosporidium, E. coli*, and turbidity, and to identify and monitor locations in their distribution systems likely to have high levels of DBPs. If source waters do not meet standards, surface water systems must select from an array of "microbial toolbox" treatment options to meet treatment requirements. Locations identified as DBP "hotspots" are to be monitored for compliance with maximum residual disinfectant levels (MRDLs) for disinfectants, and DBP MCLs established under the Stage 1 Disinfectants and Disinfection Byproducts Rule (Stage 1 DBPR) (USEPA 2005, 2017a; NACWA 2006).

The SDWA allows individual states to set and enforce their own drinking water standards if the standards are at a minimum as stringent as USEPA's national standards. The USEPA delegates primary enforcement responsibility for public water systems to states and Indian Tribes if they meet certain requirements. Oregon implements these primary (health-related) standards for USEPA, and also encourages attainment of secondary standards (nuisance-related). (USEPA 2018d).

2.1.3. How CWA and the SDWA overlap

In the past, the CWA and SDWA had mostly separate goals and functions. The CWA focused on environmental protection and maintaining "fishable/swimmable" waters, primarily by identifying and regulating sources of pollution in waterways. In contrast, the SDWA focused on municipal water treatment standards and providing clean drinking water at the tap. Over time, rising demand for surface water, driven by population growth and associated development, has been accompanied by increases in wastewater and stormwater, and reduced in-stream flow volumes available to keep these wastes diluted. These changes can, in turn, escalate loadings of sediment, nutrients, bacteria, and other pollutants in community water sources. In response to the increasingly interrelated nature of watershed management and provision of safe drinking water, the SDWA evolved to encompass environmental as well as consumer protection, resulting in overlaps with the CWA, and greater emphasis on cooperation and holistic water management among agencies charged with implementing the two statutes. (NACWA 2006).

Coordination across the CWA and SDWA is motivated by potential synergisms among goals and outcomes of these policies. Efforts driven by the SDWA to reduce contamination of drinking water sources can also protect aquatic ecosystems and wildlife, and provide higher quality and safer water-based recreation opportunities. Conversely, using the CWA to develop Ambient Water Quality Criteria that are protective of aquatic life can also help achieve and maintain safe drinking water. (NACWA

2018). Among implementers of both statutes, preventing contamination is widely understood to be much more cost effective at providing safe drinking water than removing contaminants or finding alternative water sources after the fact.

Collaboration among CWA and SDWA implementers also facilitates more effective action to reduce disinfection byproducts (DBPs) in drinking water. These DBPs can form when a disinfectant (e.g. chlorine, chloramine, chlorine dioxide) reacts with organic matter– often decomposing plant matter - in source water (USEPA 2005). Total trihalomethanes and haloacetic acids are widely occurring DBPs which have been linked with increased cancer risk, problems with reproductive systems and other human health risks (USEPA 2006). Dissolved organic matter (DOM) from forest detritus is a major precursor to DBPs in drinking water sources (Bardwaj 2006, Karanfil and Chow 2016). Thus, forest management activities that influence the quantity and mobility of this source of DOM in source waters can influence the potential for DBPs to form during water treatment. Addressing DBP issues efficiently requires coordination across the entire drinking water production chain from source water to tap.

2.1.4. The SDWA Source Water Assessment Program

In 1996, Congress significantly expanded the SDWA to facilitate prevention of contamination through an increased focus on drinking water source protection. The 1996 revisions were instrumental in pushing the SDWA into the realm of the CWA, most notably via the SDWA's new Source Water Assessment Program. This program, along with the UCMR Program and the LT2ESWTR (discussed above), extended the SDWA's largely post-hoc emphasis on regulating water treatment to include environmental protection focused on source waters. (NACWA 2006).

The 1996 SDWA revisions required states to develop USEPA-approved programs to carry out Source Water Assessments (SWAs) for all public water systems in the state. The SWAs focused on delineation of drinking water sources, identification of the origins of USEPA-regulated contaminants (and any additional contaminants selected by the state) in those source waters, and providing water utilities, community governments, and other stakeholders with information needed to protect drinking water sources. The 1996 amendments outline six steps for conducting SWAs for public water systems (CWAs).

<u>Step 1.</u> **Delineate the source water protection area** (SWPA). Delineation shows the area to be protected based on the area from which the CWA draws its drinking water supplies.

<u>Step 2</u> **Inventory known and potential sources of contamination**. The contaminant source inventory lists all documented and potential contaminant sources or activities of concern that may be potential threats to drinking water supplies.

<u>Step 3</u> **Determine the susceptibility of the CWA to contaminant** sources or activities within the SWPA. Determining susceptibility of the CWA to inventoried threats relates the nature and severity of the threat to the likelihood of source water contamination.

<u>Step 4</u> **Notify the public** about threats identified in the contaminant source inventory and what they mean to the CWA. Effective programs ensure that the public has information necessary to act to prevent contamination.

Step 5 **Implement management measures** to prevent, reduce, or eliminate risks to your drinking water supply. The assessment information can support formulation and implementation of measures to protect the source water. These measures can be tailored to address each threat or array of risks specific to each CWA.

Step 6 **Develop contingency planning strategies** that address water supply contamination or service interruption emergencies. Water supply replacement strategies are an indispensable part of any drinking water protection program in the event of short- or long-term water drinking water supply disruption.

The 1996 revisions also authorized voluntary source water protection partnerships between state and local governments focused on reducing contaminants in drinking water, opportunities for financial and technical assistance, and developing long-term source water protection strategies, usually documented in Source Water Protection Plans. (NACWA 2006; Tiemann 2017).

2.1.5. Programs for local source water assessment and source water protection planning

In 2015, Congress enacted the Grassroots Rural and Small Community Water Systems Assistance Act, reauthorizing and revising the small water system technical assistance program included in the 1996 SDWA expansion. Under this act, the Source Water Protection Program (SWPP) is coordinated jointly by USDA Farm Service Agency (FSA) and the National Rural Water Association (NRWA), a non-profit water and wastewater utility membership organization. The SWPP is designed to help prevent pollution of drinking water sources for rural residents. Participation in the program is voluntary. Rural source water technicians work with specialists from the USDA Natural Resources Conservation Service (NRCS) and state and county staff to identify areas where pollution prevention is most needed. These technicians then work with state rural water associations to form local teams comprised of citizens and representatives from federal, state, local, and private organizations. They collaborate on Rural Source Water Protection plans to promote clean source water through voluntary actions that local landowners can implement to prevent contamination. The goal is to work at the grassroots level to educate and inform rural residents about practical steps to prevent water pollution and improve water quality.

The Oregon Association of Water Utilities (OAWU) is a nonprofit, independent association of about 700 mostly smaller and rural public and private community water utilities in the state. The OAWU represents their members' interests in the Oregon legislature and coordinates with the National Rural Water Association (NRWA) which represents rural water systems at the national level. The OAWU also plays an important role in addressing drinking water issues at the local water system level, through onsite technical assistance in areas such as SDWA and CWA regulations, water treatment technology, distribution system operation and maintenance, and wastewater treatment and collection. The OAWU Source Water Specialist deals specifically with drinking water protection, working directly with local water systems to prepare drinking water protection plans that address all state and federal requirements including specifically addressing potential contaminants through education of local management authorities and best management practices to help reduce the likelihood of contamination.

The American Water Works Association (AWWA) mission is to support water utilities in evaluating and improving their water quality, operations, maintenance, and infrastructure. The AWWA has developed detailed guidance for local municipalities to use in developing their SWAs and protection plans - Utility Management Standard G300, Source Water Protection (AWWA 2014). This American National Standards Institute (ANSI)-approved standard and its accompanying operational guide (Gullick 2017) outline six primary components of successful source water protection (SWP) programs and requirements for meeting the standard:

- A SWP program vision and stakeholder involvement
- Source water characterization
- SWP goals
- SWP action plan
- Implementation of the action plan
- Periodic evaluation and revision of the entire SWP program

2.1.6. How Oregon agencies coordinate to provide safe drinking water

In Oregon, the SDWA is directly implemented by Oregon Drinking Water Services (DWS), within the Environmental Health Section of the Public Health Division, Oregon Health Authority (OHA) under ORS 338.277 and 448.273. Under SDWA, DWS is primarily involved with administering and enforcing drinking water quality standards for public water systems, but also with source water protection, primarily for groundwater systems. The Oregon Department of Environmental Quality (DEQ) implements CWA authorities to address pollutants that affect the quality of drinking water source waters, primarily surface waters. In practice, the DEQ Drinking Water Protection Program coordinates with OHA's DWS through an interagency agreement to carry out provisions of the two acts and jointly provide clean drinking water. Although OHA is the primary implementer of the SDWA, DEQ took the lead on the SWAs mandated by the 1996 SDWA revisions, conducting all surface water assessments and assisting on the groundwater assessments.

The DEQ also administers the Oregon Coastal Nonpoint Pollution Control Program (CNPCP). Coastal states are required to develop such programs to be eligible for federal funding to mitigate nonpoint source pollution under the federal Coastal Zone Management Act Reauthorization Amendments of 1990 (CZARA). Coastal states are also required to implement a set of management measures based on guidance published by the USUSEPA. These programs are designed to restore and protect coastal waters from nonpoint source pollution and to mitigate impacts to beneficial uses of these waters, including use for municipal drinking water. Oregon's CNPCP was developed in cooperation with the Oregon Department of Land Conservation and Development (DLCD) Oregon Coastal Management Program (OCMP). The CZARA, and how it intersects with drinking water protection in Oregon, are discussed in more detail in Chapter 4.

The DLCD also coordinates with DEQ to offer guidance to communities who may wish to enhance protection of their source watersheds through improved land use regulations such as comprehensive plan and zoning ordinance updates. (Oregon DEQ 2017).

3.1.7. Source Water Assessments in Oregon

As stipulated by SDWA and Oregon Regulations (OAR 333-061-0020(125)), Source Water Assessments (SWAs) were completed between 1996 and 2005 for community water systems in Oregon serving at least 15 hookups or more than 25 people year round (OAR 333-061-0020(25). Under the SDWA, smaller systems and transitory uses are also called public water systems (see OAR 333-061-0020(107) for a definition of these), but these are beyond the scope of this report. In following years, Oregon agencies significantly expanded their capabilities for analyzing natural characteristics and potential pollutant sources. With this expanded capacity, Updated Source Water Assessments (USWAs) with more detailed data, maps, and technical information were completed for roughly 50% of these systems in 2016-2017.

The assessments 1) defined groundwater and surface water source areas which supply public water systems, 2) inventoried each area to determine potential sources of contamination, and 3) determined the most susceptible areas at risk for contamination. For surface water systems, DEQ prioritized the 52 coastal community water systems under the rationale that these systems are challenged by geographic setting, climate and geology, and seasonal tourism in ways that other areas in Oregon do not necessarily experience.

As part of the USWAs, DEQ developed a statewide land use/ownership Geographic Information Systems (GIS) layer to evaluate land cover in drinking water source areas. Maps for each individual public water system are provided in that system's USWA report. Information from the SWAs for surface water systems is available to the public via a database maintained jointly by the DEQ and OHA. In 2018, after consulting with stakeholders, the DEQ also finalized a Surface Water Resource Guide to provide additional technical assistance and information to surface water community water systems. (Oregon DEQ 2018c). This document (and a companion Groundwater Resource Guide) will continue to be updated and improved as source water protection efforts in Oregon move forward. The USWAs and Resource Guides are ultimately intended to assist public drinking water providers, community governments, and others in the development of community-based Drinking Water Protection Plans to protect their upstream source waters.

Several rural water providers in Oregon have voluntarily worked with the Oregon Association of Water Utilities (OAWU) to take advantage of the USDA-FSA SWPP. Most of utilize groundwater, but some are surface water systems. The protection plans are based on interviews with water utility personnel, local managers and land owners, information from the SWA or USWA, and a visit to the source water intake and source watershed. The plans include (Collier 2018):

- A map of the planning area;
- An inventory of potential contaminant sources, and characteristics and sensitivity of the source water;
- A definition of areas and community profile that align with participating local entities and organizations;
- A definition of voluntary measures and best management practices that may be initiated;
- Identification of public education initiatives, entities and resources to facilitate plan implementation and sustainability; and

- A contingency and emergency response plan in the event of problems with the local drinking water supply.

2.2. Raw water treatment processes

2.2.1. Overview

The conversion of raw source water into finished potable water entails a series of steps called the "treatment train." These steps, along with source water protection activities such as identifying and reducing contamination in watersheds, are designed to provide an integrated "multiple barrier" approach so that if any one step fails there is redundancy to reduce the likelihood of contamination reaching the tap. There are various permutations of these steps depending upon the quality of the source water and the expectations of the utility's customers, but all processes are designed to at least meet the SDWA standards discussed in the prior section. Treatment is a combination of physical operations, such as screening, mixing, sedimentation, and filtration; operations such as precipitation and disinfection resulting from chemical additions; and biofiltration to remove nitrogen and organic matter. Because treatment processes may be simpler for groundwater sources, our focus will be on the treatment of surface water rather than groundwater.

Our discussion highlights how raw water is treated to remove impurities, kill pathogens, and provide safe water at the drinking water tap. General concerns include turbidity and particles; hardness and total dissolved solids (TDS); color, odor, and taste; dissolved minerals such as manganese, iron; bacteria, algae, protozoan cysts, and viruses; and anthropogenic sources such as pesticides, herbicides, volatile organic compounds, and pharmaceuticals; and natural organic matter (NOM) and disinfection by-products (DBP). We will focus on our treatments affecting our three high-priority concerns (turbidity/sediment; forest chemicals; and NOM/DBP) as they are most likely to be affected by active forest management.

Permutations of treatment processes among the 157 treatment plants that rely on surface water can be grouped into five major categories as shown in Table 2-1. Table 2-1 separates these different types of treatment processes by the size of the population served by the system, using the USEPA population size categories. The most common treatment process is the conventional and direct process with either rapid or pressurized sand filtration, utilized by 96 (over 60%) of the water treatment plants that rely on surface water. The next most common process is membrane filtration, used by 28 (18%) of the treatment plants. Nineteen plants (12%) used a slow sand filtration process, with one subsequently applying membrane ultrafiltration. Alternative methods, approved by OHA, are employed in eight treatment plants, most commonly a cartridge filtration system that uses polypropylene as a filter (discussed below) to catch sediment. Finally, three community water providers do not filter their drinking water (Portland, Baker City, and Reedsport), although it is disinfected prior to entering their distribution systems. Another three providers (Monmouth, Monument, and the Shangri La Water District) are currently not filtering their surface water, but have been required to install equipment [Note: Baker City may be in this same situation].

Table 2-1. Drinking water treatment plant technology by USEPA system size category.

Treatment Process	USEPA Drinking Water System Service Population Size					
	Very Small	Small	Medium	Large	Very Large	Process Total
1. *Conventional/Direct*						
Filtration, Rapid Sand	16	40	13	18	4	91
Filtration, Pressure Sand	4	1				5
2. *Slow Sand*						
Filtration, Slow Sand	3	10	2	2	1	18
Filtration, Slow Sand & Ultrafiltration				1		1
3. *Membrane*						
Filtration, Microfiltration	1	1		2		4
Filtration, Ultrafiltration	9	6	5	3	1	24
4. *Alternative Methods*						
Filtration, Cartridge	4	2				6
Filtration, Diatomaceous Earth	1					1
Natural Filtration	1					1
5. *Unfiltered*						
Unfiltered, Avoiding Filtration			2		1	3
Unfiltered, Must Install Filtration	2		1			3
Systems by Population Size Total	41	60	23	26	7	157

Source: DEQ/OHA ArcGIS Shape File "Treatments_07SEP2017."

Very small and small community water systems are more likely to use membrane filtration, slow sand filtration, and alternative methods. This is largely due to scalability of membrane and cartridge systems, and the comparative ease of operation of slow sand filtration. Most all (71%) of the slow sand filtration treatment plants are located in small systems, with over half serving populations of 501 to 3,300 persons; the one slow sand treatment plant in a very large system is Salem.

Figure 2-3 shows process diagrams of the two most common treatment processes: (a) conventional filtration; and, (b) membrane filtration. All systems that rely on surface water have an intake structure that controls the amount of raw water entering the treatment plant. Usually, this structure incorporates bars (called “trash racks”) and screens that intercept debris coming into the plant. Depending upon the system, there may be settling ponds just after the intake to reduce suspended particles prior to the water entering the plant. Screens come in two varieties: course screens, to remove large particles from 20 to 150 mm and larger, commonly used at the entrance of the plant; and micro screens, to remove small particles 0.025 to 1.5 mm commonly used to remove filamentous algae. When stream conditions exceed the capacity of the plant to treat the raw water, the intake is commonly closed until conditions improve. High levels of turbidity/sediment and/or debris that clog screens are common reasons to close the raw water intake, although spills or other incidents may also result in closure. When this occurs, the water utility is dependent upon alternative sources (such as wells) if available, and its storage capacity to maintain service. (Crittenden et al. 2005).

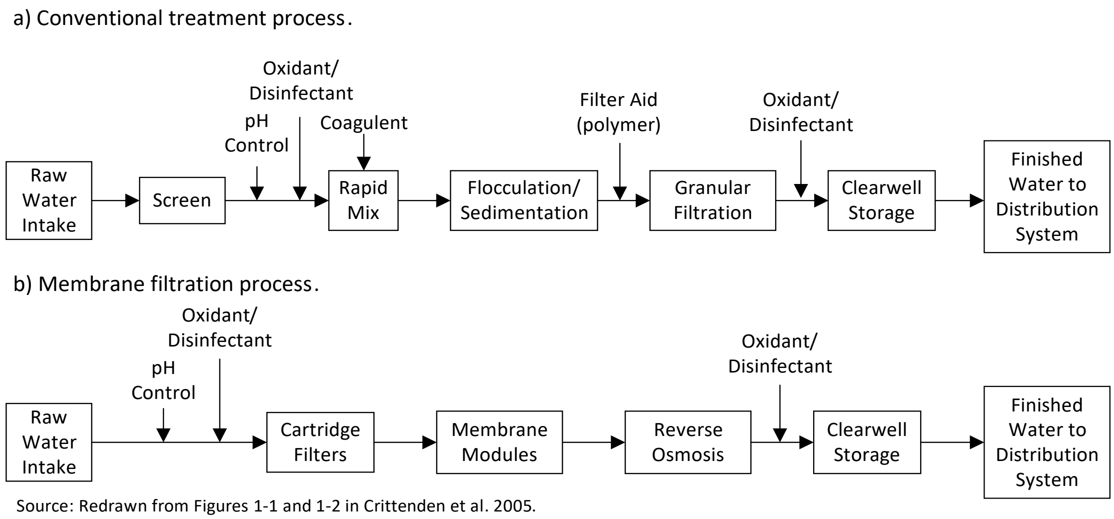


Figure 2-3. Conventional (a) and membrane (b) filtration drinking water treatment processes.

Common among treatment processes is the control of pH and addition of oxidants or other disinfectants once the raw water passes through the intake screens (Figure 3-3). In the initial treatment process, pH control is used to assist in removing undesirable particles through either precipitation or coagulation. Common oxidation additions are chlorine, ozone, chlorine dioxide, permanganate, and hydrogen peroxide. They are primarily used to control taste and odor, remove undesirable solutes (hydrogen sulfate, color, iron and manganese), and disinfect (i.e., kill pathogens such as bacteria and viruses). If ozone is used to disinfect the water, then the pH is usually lowered to avoid the formation of brominated organics—undesirable disinfection by-products (DBP)—through reaction with natural organic matter (NOM) in the water.

2.2.2. Conventional and Direct Treatment Processes

Conventional and direct treatment processes have evolved since the first ones were created in the early 1800s. These treatments typically comprised of coagulation, flocculation, sedimentation, and filtration (Figure 2-3a). The difference between “conventional” and “direct” treatment is that there is a sedimentation step between flocculation and granular filtration in the conventional process, while this step is skipped in the direct process. The filtration process removes suspended particles and dissolved substances during the treatment process.

Rapid mixing is important for coagulation and the addition of chlorine. Rapid mixing is used when chemicals need to be added to the water being treated, there needs to be uniformity in the blending, and where there may be competitive consecutive reactions among the chemicals added and there is a desire for the reaction to be irreversible (or not occur, as in the case of NOM and DBPs). Coagulants are used to condition the suspended, colloidal, and dissolved matter for subsequent removal. Most particles in natural water are negatively charged, so they naturally repel each other. This charge has to be removed before the particles coalesce sufficiently large to be removed through filtration. Coagulants provide adsorption and reaction locations for colloidal and dissolved NOM; have electrical charges that neutralize small suspended or colloidal particles so that they aggregate into larger particles; and enmesh small suspended, colloidal, and dissolved particles as they settle. Typically these are alum and iron salts (but sometimes polymers), as they produce positive charges to neutralize the negative charges on particles so they can clump and be more easily removed.

Flocculation is the process of forming larger aggregates by mixing smaller particles so that they come close enough to attach to each other. This process occurs in moving water, with higher velocities in the early stages of flocculation where aggregates are smaller (micro-flocculation), and progressively moving slower water to avoid breaking up larger aggregations (macro-flocculation). Water (and aggregates) then move to a sedimentation (settling) basin if the particles are heavier than water, or a floatation basin if they are lighter. Polymers—natural or synthetic long-chain (high molecular weight) organic molecules that can have different levels of positive (cationic) or negative (anionic) charge—may be added as a filter aid. Cationic polymers are used instead of (or in addition to) metallic salts to neutralize charged particles in both the coagulation and flocculation steps. In general, polymers act slower than the inorganic metallic salts. The remaining water is decanted for further processing while the flocculants are removed.

Granular filtration strains fine suspended particles (sand, clay, and iron and aluminum flocs) during the treatment process. Typically, granular filtration is accomplished through sand filter beds, with particles larger than the space between sand grains remain in the sand. Filter media (usually sand) ranges in grain size from (0.5 –1.2 mm diameter), with the tradeoff being that smaller sizes remove more particles, but get clogged more quickly, and the vice versa with larger grain sizes. Some treatment plants overcome these constraints by using multiple sand filter beds so that water progresses from coarse grained beds to finer grained beds. Unfiltered water input above the bed and the filtered water drawn from below with water velocities of about 5 – 15 meters per hour (m/h) from a hydraulic head of 1.8 – 3 m (Crittenden 2005). Treatment operators periodically backwash filters to clean out the filtered particles and coagulants and then dispose of the waste and backwash water.

The filtered water in conventional and direct treatment is dosed with disinfectants (typically chlorine) to kill pathogens. Clearwells temporarily store filtered water for a sufficient time to provide chlorine contract for disinfection. They are also used to buffer variations in finished water demand. The finished water may have its pH adjusted again to prevent dissolving toxic metals used in distribution pipes and household plumbing (e.g., the Flint, Michigan problem due to lead pipes). Finished water is also required to have some chlorine residual throughout the distribution system to protect against contamination. If this residual chlorine remains in contact with dissolved organic carbon (DOC) in stagnant water, then undesirable disinfection by-products (DBPs) may be created.

2.2.3. Slow Sand Treatment Process

While conventional rapid sand filtration is a physical and chemical treatment process, slow sand filtration is a physical and biological treatment process (Crittenden 2005). The treatment train for slow sand systems does not typically include the flocculation/sedimentation steps found in conventional and direct filtration. It also differs in that the sand filter media is smaller (0.3 – 0.45 mm diameter), and the hydraulic head used to push the water through the filter is less (0.9 – 1.5 m), resulting in a water velocity of between 0.05 and 0.2 m/h. Most particles are physically removed in the upper inches of the filter bed. Additional particle straining occurs in the *schmutzdecke*, a complex layer that consists of decomposing organic matter, iron, manganese and silica (Ranjan and Prem 2018) that includes a gelatinous biofilm containing algae, bacteria, fungi, protozoa, rotifer, and various invertebrates. The *schmutzdecke*, and the remainder of the filter bed, contribute to water purification through four mechanisms: (1) it creates a hostile environment for intestinal bacteria because of low temperatures; (2) the bioactivity in the layer competes for food needed by these pathogens; (3) predatory organisms feed on the pathogens; and (4) microorganisms in the slow sand filter produce compounds poisonous to intestinal bacteria (Huisman and Wood 1974).

Slow sand gravity filtration systems have a filter capacity of 0.005 – 0.018 m^3/h per square meter of granular filter area, compared to rapid sand filtration that has a throughput capacity of 5 – 15 (m^3/h per square meter of filter area (Crittenden 2005). Thus, a slow sand filtration process requires approximately 100 times the area as rapid sand filtration, but requires less expertise and has fewer operational costs. Other than land area, the primary limiting factor for slow sand filtration is that the turbidity of the raw water should be less than 50 NTU, preferably < 10 NTU, and optimally less than 5 NTU (Crittenden 2005).

2.2.4. Membrane Filtration Treatment Process

Membrane filtration is a physio-chemical process that uses a semi-permeable membrane as a mechanism to remove suspended particles. The treatment train for membrane filtration is shown in Figure 2-3b. In Oregon, both microfiltration (0.1 µm pores) and ultrafiltration (0.01 µm pores) treatment plants exist. Microfiltration will remove particles, sediment, algae, protozoa, and bacteria; ultrafiltration will additionally remove small colloids and viruses. Membrane filtration is typically used to process water containing < 1000 mg/L of total dissolved solids (TDS) (Crittenden 2005).

Cartridge filters are used in membrane filtration for pre-treatment to remove solids (Figure 2-3b). Systems are pressurized to force the water through the filters, generally moving from the outside to the inside of the cartridge. Cartridge filters consist of a filter media consisting of micro-denier (screen size) polypropylene fiber, typically with coarser outer layers and finer inner layers down to 0.2 microns (filtrasystems.com). Cartridge filters can remove turbidity (if source is < 1 NTU and without fine colloids or clays), and remove *Cryptosporidium* and *Giardia* cysts from source water, although they do not remove bacteria or viruses. Used by themselves, cartridge filters are typically used in systems of < 100,000 gpd; as shown in Table 2-1, there are 6 water treatment plants in Oregon that utilize only cartridges.

Whether or not cartridge filters are used, undergoing treatment then moves to the membrane modules. Membrane filters are usually manufactured as flat sheet stock or as hollow fibers and then formed into different types of membrane modules. Module construction typically involves potting or sealing the membrane material into an assembly, such as with hollow-fiber module. These types of modules are designed for long-term use over the course of a number of years. Spiral-wound modules are also manufactured for long-term use, although these modules are encased in a separate pressure vessel that is independent of the module itself (MNRWA 2009).

Reverse osmosis is generally only used in drinking water treatment when the raw water is high in total dissolved solids, such as seawater or brackish groundwater (TDS = 1000 – 20,000 mg/L), or the water is highly colored (TOC > 10 mg/L). As a type of membrane filtration, reverse osmosis uses a semipermeable membrane to remove dissolved ions and molecules, as well as larger particles from drinking water. However, the membrane pores are quite a bit smaller (< 1 nm). Reverse osmosis has the capability to remove hardness, natural organic matter, heavy metals, radionuclides, and pesticides (Chittenden 2005). The final treatment steps for membrane filtration—disinfection and clearwell storage—are similar to those in conventional treatment. At this time, no water treatment plants in Oregon use reverse osmosis.

A few systems in Oregon utilize powdered activated charcoal (PAC) and granulated activated charcoal (GAC) treatments to remove contaminants such as pesticides and volatile organic compounds. It should be noted that these treatments do not provide complete removal of targeted contaminants.

2.2.5. Sensitivity to various process types to sediment and turbidity.

Sediment and turbidity levels in the raw surface source water limits available treatment processes and affects operational costs (coagulants and requirement for backwashing filters). Table 2-2 compares threshold sediment and turbidity levels for the various types of treatment processes described above.

Table 2-2. Surface water thresholds for turbidity and color for various treatment processes.

Treatment Process	Turbidity Range (NTU) [1]	Color Range (CU) [1]	General Design References
Established Technologies			
Conventional Filtration	Unlimited	< 75	Kawamura 2000
Direct Filtration	< 15	< 40	Kawamura 2000
Pressure Filtration	< 5	< 10	Ten State Standards 2007
Diatomaceous Earth Filtration	< 10	< 5	AWWA 1999; Fulton 2000; WSDOH 2003
Slow Sand Filtration	< 10	< 10	Hendricks et al. 1991, WSDOH 2003
Alternative Technologies			
Bag and Cartridge Filtration	< 5	See Note 4	USEPA 2003
Membrane Filtration	< 10	See Note 4	Chittenden 2005

Source: Washington Department of Health 2009. Water System Design Manual, Table 12-3 p. 155. [Membrane filtration changed]

Other potential impacts of excess sediment include filling of reservoirs and intake ponds (requiring dredging with associated permitting), shorter filter life, and extra staff time to manage water quality and treatment processes. There are also regulatory compliance problems if MCLs cannot be met and the potential for treatment plant shutdowns, which can cause finished water supplies to run low.

2.2.6. Diversity of community water suppliers in Oregon

Community water suppliers in Oregon range in size from the Portland Water Bureau with over 500 employees and an organized, multi-stakeholder working group that collaborates on management and produces and annual report, to very small rural systems serving only a handful of households and managed by a single staff member. Some of the smallest systems in Oregon are staffed by volunteers only.

Regardless of size, every community water system requires a certain minimum level of infrastructure and treatment capability. Economies of scale often work against smaller water providers because treatment costs per unit [gallon] of finished water typically decline as system size and volume treated increase. Larger systems also usually have correspondingly larger budgets and dedicated staff. Smaller systems, in contrast, usually face higher costs per unit of finished water delivered, have smaller budgets, and operate with fewer dedicated staff. One important consequence of this disparity is that smaller systems have correspondingly less capacity to identify, publicize and mitigate threats or impacts to the quality and quantity of their drinking water sources.

2.4. Results and findings from survey of Oregon drinking water providers

This section explores some of this diversity among Oregon CWSs, and the issues they face by summarizing results from a survey of water providers conducted in 2018. A key component of the Trees to Tap project was a survey of Oregon drinking water providers that utilize surface waters as part or all of their supply. The survey was modeled after, but expanded upon, a similar survey by Adams and Taratoot (2001) and solicited input from utility managers regarding the issues they face in managing and protecting their drinking water sources. Details regarding methods used in the water provider survey are provided in Appendix __.

The following section provides detailed results and findings from the survey, including:

- respondent characteristics,
- governance of respondent water systems,
- partnerships and activities,
- data collection and notifications,
- reported issues of management concern in drinking water source watersheds,
- lessons learned, and
- a summary and discussion of survey results

2.4.1. Summary of respondent characteristics

We examined respondent location by several characteristics. These figures help contextualize the data by identifying which types of systems are better represented by survey results. We did not conduct initial sampling based on any characteristic, but as noted, conducted follow-up to target respondents with highest percentages of private industrial timberland, public land, and local government ownership based on interest of the steering committee.

The majority of survey respondents (58%) were from the Valleys region, and approximately one-third of all Valley systems responded (Table 2-3). Thirty-eight percent of respondents were from the Coast region and 39% of the systems in this region responded. Only 4% of responses were from the Dryside region and 17% of all systems in that region responded. Relative to their total proportion in the state, survey responses over-represent Coast and Dryside systems, and under-represent Valley systems. Size of populations served by respondent systems ranged from 29 (Weiss Estates Water System) to 183,523 (Eugene Water and Electric Board) with a mean population service size of 15,985. Population quintiles were fairly equal in percentage of response, excepting systems serving populations of 1,801 to 3,000 (Table 2-4). Respondents from smaller primary source watersheds (less than 10 square miles) composed the largest proportion of responses at 41%, and larger watersheds exceeding 350 square miles were the smallest proportion at 11% (Table 2-5). Responses were fairly equal across size classes of public and private industrial forest land ownerships (Table 2-6. and Table 2-7).

Table 2-3. Respondent affiliation and response rate by major region. Due to rounding, totals may not consistently be 100%.

	Survey Respondents	Proportion of responses	Systems in region	Proportion of systems	Proportion of systems responding
Coast	21	38%	54	35%	39%
Dryside	2	4%	12	8%	17%
Valleys	31	57%	90	58%	34%
Total	54		156		

Table 2-4. Respondent affiliation and response rate by population quintile size served. Due to rounding, totals may not consistently be 100%.

Population quintile	Number of respondents from quintile	% of respondents from quintile
0 to 235	12	22%
236 to 1,800	11	20%
1,801 to 3,000	4	7%
3,001 to 10,700	13	24%
10,701 to 184,000	14	26%
Total	54	

Table 2-5. Respondent affiliation and response rate by size of primary source watershed. Due to rounding, totals may not consistently be 100%.

Primary source watershed size class	Number of respondents from size class	% of respondents from size class
0 to 10 square miles	22	41%
10.1 to 100 square miles	11	20%
100.1 to 350 square miles	13	24%
350.1 to 1,150 square miles	6	11%
Left blank	2	4%
Total	54	

Table 2-6. Respondent affiliation and response rate by % of primary source watershed in public land ownership. Due to rounding, totals may not consistently be 100%.

% of primary source watershed in public land ownership	Number of respondents from class	% of respondents from class
0 to 10 %	17	31%
10.1 to 40 %	12	22%
40.1. to 80 %	12	22%
80.1 to 100 %	13	24%
Total	54	

Table 2-7. Respondent affiliation and response rate by % of primary source watershed in private industrial timberland ownership. Due to rounding, totals may not consistently be 100%.

% of primary source watershed in private industrial forestland ownership	Number of respondents from class	% of respondents from class
0 to 10 %	17	31%
10.1 to 25 %	11	20%
25.1 to 75 %	11	20%
75.1 % to 100 %	13	24%
Left blank	2	4%
Total	54	

2.4.2. Governance of respondent systems

We asked respondents a series of questions about how their drinking water system was governed, including organizational model, average annual operating budget dedicated to drinking water supply, number and type of employees, and access to the watershed.

First, the majority of respondent systems (56%) were organized as departments or units of municipal government (Figure 2-4a). A little more than a quarter were special districts, while less than 10% respectively were nonprofits or private for profits. No respondents were from tribal systems or joint (regional) entities of multiple governments. The survey population therefore largely represents public, rather than private or nonprofit entities that manage Oregon's drinking water supply. Size of budget is also important to understand, as providers with larger budgets may have correspondingly larger capacity to manage their drinking water supplies. Fifty-eight percent of Oregon CWSs operated on a budget of $500,000 per year or less. Twenty-three percent reported annual budgets of $500,001 to $2 million. Only six percent exceeded $10 million (Figure 2-4b).

a. Organization type (n=52).

b. Average annual budgets dedicated to drinking water supply over past four years (n=47).

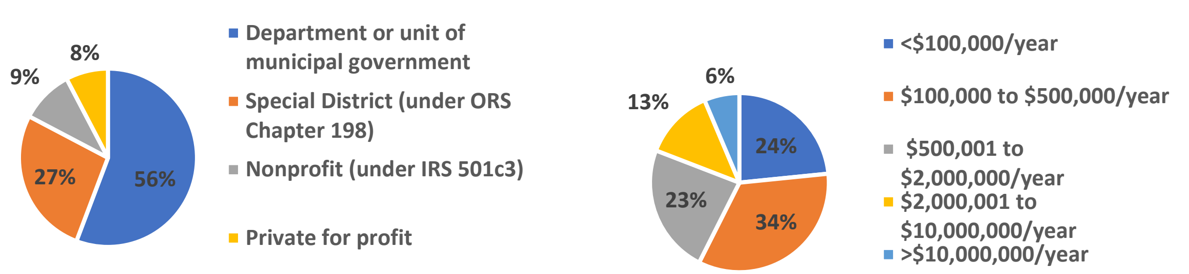


Figure 2-4. Water utility survey responses for type of organization (a) and annual budget (b).

We also asked respondents to identify the number and type of staff employed in drinking water provision (Table 2-8). Like provider budgets, staff size and type may be an indication of the capacity that a provider has to manage their drinking water supplies. Total staff sizes ranging from zero to 200 were reported. A majority (64%) had one to ten total staff. Seven respondents indicated that they had no paid employees, relying solely on volunteer homeowners or board members, and ten had only one or two staff. The mean total staff size was 13, with an average of 11 full time employees.

Table 2-8. Number of staff employed in drinking water provision Due to rounding, totals may not consistently be 100%.

Total staff employed in drinking water provision	Number of respondents	% of respondents
0	7	14%
1-10	32	64%
11-20	5	10%
21-30	2	4%
>30	5	10%
Total	50	

Respondents were also asked about the type of access available to their primary source watershed (Figure 2-5). Access to source watershed areas may pose management issues to drinking water suppliers, but recreation on forested lands is also an important activity in Oregon. Respondents were able to choose all types that applied. Nearly one half allowed open access to the public at all times, while 20% allowed no public access. A few responding "other" indicated access approaches such as voluntary permits, hours of access, and limits on types of uses (e.g., motorized).

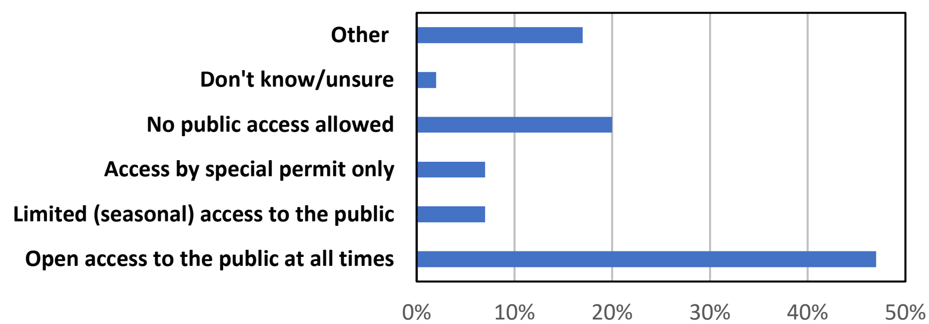


Figure 2-5. Type of access allowed on primary source watershed. n=59 for question (choose all that apply).

3.4.3. Partnerships and activities

Respondents were first asked if they worked together with other landowners in their primary source watershed, and to explain any mechanisms for doing so. A majority (51%; 27 respondents) indicated that they did, and 42% (22) did not (Figure 2-6). Additional open-ended information was provided about these approaches, showing that they range from more minimal relationships such as contacts for access through large, multi-agency and organizational partnerships to collectively manage the watershed. These included informal information sharing as needed and general "good neighbor" practices, and more formal venues such as regular meetings and established collaborations. Open-ended responses included the following:

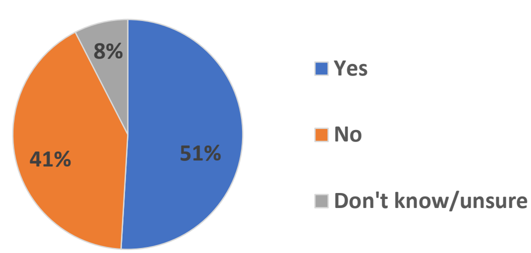


Figure 2-6. Do respondents work together with other landowners in their primary source watershed (n=53).

- Combined watershed management plan for city
- US Forest Service is sole landowner, but we work with city parks and recreation and area schools on fuels, trails, access, and education programs
- Have contacts for access through locked gates
- Attend meetings about watershed health
- Share information and current events
- Discuss every aspect of management with the private landowners
- Required to go through NEPA process with US Forest Service and consult with resource specialists
- Work with ranchers to keep runoff out
- Not much interaction
- Coordinate with the Port with vehicle access, overnight camping, and transient camp enforcement
- Cooperative arrangement for notification and monitoring of impacts of herbicide spraying with private timberland company
- Land trade with private timberland owners currently underway to transfer ownership of entire watershed
- Pesticide education
- Logging and herbicidal applications notifications
- Bi-annual meetings with private timberland owner
- Have watershed management plan and communicate frequently with other landowners and stakeholders. Also use overarching basin action plan.
- Monitor activities of neighboring land owners, communicate and coordinate with them if their activities have a potential impact to our watershed.

Respondents were then asked to name the top five partners that they interacted with most closely in the management of their primary source watershed, with #1 being the most important (Table 2-9). Table 2-9 is sorted by number of total mentions in any rank. Weighted rankings account for both number of mentions and rank of mentions; each entity was given five points for every time mentioned as #1 to one point for every time mentioned as #5. Not all respondents named all five entities; the most responses were provided for the #1 partner and descended in quantity with each subsequent ranking. The most common type of #1 partner was private timberland owners, followed by watershed councils and SWCDs. If considering top partners by combined #1 and #2 rankings, private timberland owners were first (14 respondents) followed by the US Forest Service (11 respondents). The most commonly-named partner for any ranking was also private timberland owners (21 respondents), followed by county or city government entities (19 respondents), and other (18 respondents). “Other” type entities included local fire districts, Oregon Department of Fish and Wildlife, ranchers, and consulting foresters. Although not all respondents participated in this question, these data suggest the likely importance of interaction.

Table 2-9. Top five partners with which respondents interact in management of primary source watershed, with #1 being most important.							
Entity	Ranked #1	Ranked #2	Ranked #3	Ranked #4	Ranked #5	Total Mentions	Ranked Weight
Private timberland owner	10	4	4	3	0	21	1
County or city entity	5	3	5	2	4	19	2
Other	2	3	4	4	5	18	5
Watershed council or SWCD	7	2	0	4	2	15	4
US Forest Service	5	6	1	1	0	13	3
Oregon Health Authority	4	2	2	0	0	8	6
Nonprofit organization not including watershed councils)	0	3	2	1	1	7	8
Oregon Department of Forestry	1	1	1	2	1	6	11
Oregon Water Resources Department	1	3	1	1	0	6	7
Other federal agency	1	0	2	1	2	6	10
Oregon Department of Environmental Quality	0	3	1	2	0	6	9
n =	36	30	23	21	15		

We also asked respondents if they funded or participated in forest restoration or hazardous fuels reduction activities in the primary source water watershed for the intent of drinking water protection (Figure 2-7). A majority of 56% (31 respondents) did not, while 38% (21 respondents) did. Some respondents provided open-ended information about these activities, including the following:

- Noxious weed control
- Participate in Forest Stewardship Council
- Large multi-partner project to reduce fuels
- Perform own burning of ground fuels

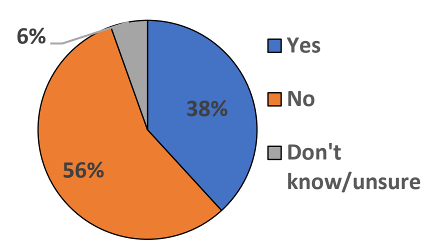


Figure 2-7. Respondent participation in forest restoration or fuels reduction activities for source water protection. n=55 for question.

- Watershed protection sign project with the US Forest Service using a Drinking Water Protection grant
- Brush clearing
- Plant vegetation to reduce temperatures and runoff
- Contract local operator to perform thinning
- Contract forester to manage watershed
- Participate in local watershed association/council or forest protection association; pay dues or provide funding to these entities
- Annual meeting with forest landowners in area about logging practices
- Riparian vegetation restoration projects with watershed council

3.4.4. Data collection and notifications

Respondents were asked a series of questions about assessments, plans, and data for their primary source watershed, and how they use these (Table 2-10). A majority had updated SWAs, but not Drinking Water Source Protection plans or collection of optional raw water quality data. Nearly a quarter respectively were unsure if they had updated SWAs or Drinking Water Protection plans. Follow-up questions allowed respondents the option of explaining how they used each of these (Appendix 2). Generally, SWAs were reportedly used to build understanding of potential risks and vulnerabilities, and as the basis for strategies to mitigate them. Drinking water source protection plans were also used to identify potential risks and strategies; but also for grant applications, program development, and outreach. Optional raw water quality data were gathered for a variety of chemicals, algae, and conditions/levels, and used for purposes including planning operational water treatment decisions, learning about potential effects of herbicide spraying, informing the public, and creating baselines.

Table 2-10. Plans and data collection

	Have updated Source Water Assessment	Have Drinking Water Source Protection plan	Collect optional raw water quality data beyond legal requirements
Yes	62%	33%	40%
No	16%	41%	58%
Don't know/unsure	22%	26%	2%

Respondents were also asked if they were aware of and how they utilized the Oregon Department of Forestry's online Forest Activity Electronic Reporting and Notification System (FERNS), which allows users to search, query, and subscribe to receive notifications of proposed forest operations on private timberland upstream of a raw water intake (Figure 2-8). Nearly half were not aware of this service, and less than 20% had a subscription. A small number of respondents had searched it for forest operations or were aware of the service, but did not use it. "Other" options filled in included USFS notices, BLM letters, private landowner notifies, observation, and word of mouth. Neither of the two Dryside respondents was aware of this service, likely because their primary source watersheds were in public land ownerships. Forty-three percent of Valley respondents were not aware of the service or did not use it. Coastal respondents were the most likely to have a subscription and/or to have queried or searched for pending forest operations.

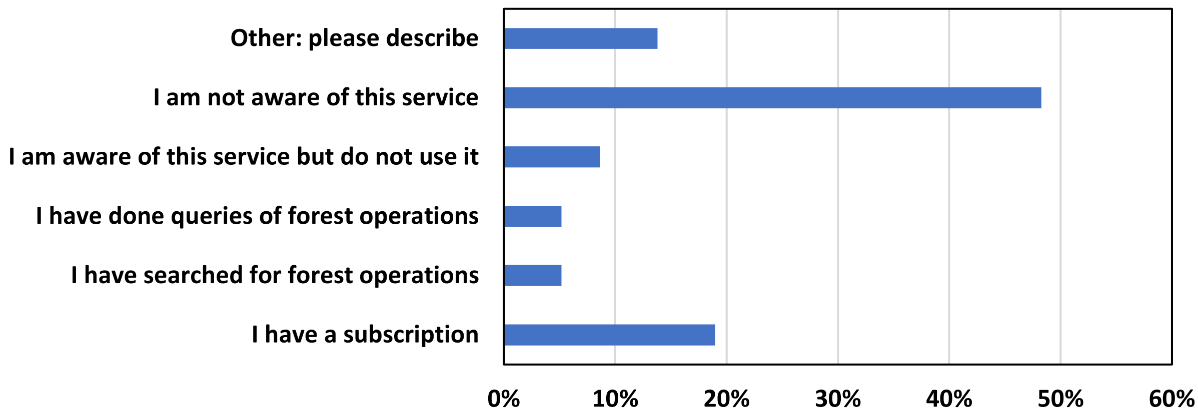


Figure 2-8. Respondents' use of FERNS notification system (n=58, multiple responses possible).

2.4.5. Reported issues of management concern in drinking water source watersheds

We asked survey respondents about issues of management concern in their source watersheds in several different ways. First, they were asked to rank ten *general issues* from 1 to 10 in order of current concern to their raw water source supply (1 being the most concerning and 10 the least concerning). The intent of this question was to observe relative levels of concern about general categories of activities that can affect source watersheds before asking about more specific management issues in following questions. This included options for "other" concerns that could be provided by the respondent, should the provided list have not included them. Responses and additional feedback were assessed and it was determined that data were most reliable for the top five concerns. Not all respondents felt that all issues were concerning. We therefore report only these top five concerns. Table 2-11 shows the results from this question. The table is sorted by weighted rankings. Weighted rankings account for both number of mentions and rank of mentions; each entity was given five points for every time mentioned as #1 to one point for every time mentioned as #5.

For their #1 general issue of concern, 37% of respondents selected forest harvest and management, followed by stormwater runoff, which was selected by 20% (Table 2-11). This was mirrored in the most commonly-selected #2 issues of concern, with nearly a quarter of respondents choosing forest harvest and management and 16% choosing stormwater runoff. Only two percent of respondents ranked cannabis cultivation or residential/commercial development in the watershed as top two concerns. "Other" responses provided for the #1 issue of concern (8%) included forest fires, algal blooms, hazardous material spills, culvert crossings, and landslides. Weighted rankings were also calculated, which account for both number of mentions and rank of mentions; each entity was given five points for every time mentioned as #1 to one point for every time mentioned as #5. Forest harvest and management, stormwater runoff, and ability of watershed to meet supply demands were the top three in both weighted ranking and number of mentions.

Table 2-11. Top 5 issues of general concern for management of source watersheds, with #1 being most important.

Concern	Ranking 1st	Ranking 2nd	Ranking 3rd	Ranking 4th	Ranking 5th	Total mentions	Weighted ranking
Forest harvest and management	37%	24%	8%	6%	12%	43	162
Stormwater runoff	20%	16%	12%	12%	14%	37	119
Ability of watershed to meet supply demands	18%	14%	16%	12%	16%	38	117
Public access and use of the watershed	4%	10%	12%	22%	8%	28	74
Climate change	2%	10%	18%	6%	18%	27	67
Agricultural land management	6%	4%	14%	14%	8%	23	62
Rural residences and septic tanks	2%	10%	6%	12%	6%	18	49
Other: please describe	8%	8%	2%	4%	0%	12	46
Residential and commercial development in the watershed	0%	2%	6%	6%	8%	11	23
Cannabis cultivation	2%	0%	2%	4%	8%	8	16

We then examined if there were differences in issues of top concern by general region (coast, valleys, or dry side) by averaging the rankings given to these issues for each region (Table 2-12). A lower value indicates that an issue was more of a concern. Forest harvest and management was the top-ranked issue for all three regions, and Dryside respondents were the most concerned with this issue, although there were only two Dryside respondents.

Table 2-12. Average rankings of issues of general management concern by regions. A lower value indicates that an issue was more of a concern.

General management issue	Coast	Dryside	Valleys	Grand Total
Forest harvest and management	2.5	1.5	3.1	2.8
Ability of watershed to meet supply demands	3.4	2.5	4.5	4.0
Stormwater runoff	3.7	7.5	4.2	4.2
Public access and use of the watershed	4.9	4.0	5.4	5.1
Agricultural land management	6.3	4.5	4.8	5.4
Climate change	5.7	5.5	5.4	5.5
Rural residences and septic tanks	6.9	9.0	5.7	6.3
Residential and commercial development in the watershed	6.7	8.0	6.4	6.6
Cannabis cultivation	7.7	7.0	7.4	7.5

Next, we asked respondents to identify their *level of concern* about a series of *more specific source water protection issues* as they may affect raw water supply by rating each issue on a five-point Likert concern scale (with 1 not a concern at all to 5 being an extreme concern; higher values indicate higher concern). These more specific issues were drawn from the literature review, prior survey/report, and input and expertise of the project steering committee. The highest average rankings, indicating issues of most concern, were for potential wildfire impacts, turbidity/suspended sediment, forest chemicals, and increased wildfire risk (Table 2-13). The issues related to wildfire (potential impacts, increased risk, and response impacts) together averaged a rating of 3.7. A few “other” responses were provided naming additional issues of an extreme concern: cyanotoxins, earthquakes, and recreation use. Issues that were the least frequently rated as an extreme concern were dissolved organic carbon, pH levels, and issues related to direct human use of the watershed (fecal contamination, unhoused people, and off-highway vehicle [OHV] use). The latter three issues together averaged a rating of 2.7, making direct human use less of a concern than almost all other issues.

Table 2-13. Ratings of level of concern for various source water management issues. Higher values indicate higher concern.

Specific issue of concern	Coast	Dryside	Valleys	All respondents
Potential wildfire impacts (e.g., erosion, cover loss)	3.8	5.0	3.9	3.9
Turbidity/suspended sediment	3.5	3.5	4.1	3.9
Increased wildfire risk	3.5	5.0	3.7	3.7
Forest chemicals (pesticides, fertilizers)	4.0	3.0	3.5	3.7
Future water quantity	3.7	3.0	3.6	3.6
Wildfire response impacts (e.g., retardants, pumping)	3.7	5.0	3.3	3.5
Other point source pollution	3.5	2.5	3.5	3.4
Flood events	3.3	3.0	3.5	3.4
Landslides and slope instability	3.6	3.0	3.0	3.2
Transportation-related fuel and hazardous material spills	2.6	3.0	3.6	3.2
Potential fecal contamination	3.1	3.0	3.1	3.1
Temperatures	3.4	1.5	2.9	3.1
Nutrient levels	3.1	3.0	3.0	3.0
Riparian buffer blow-down	3.2	3.5	2.8	3.0
Dissolved organic carbon	3.1	3.0	2.9	2.9
pH levels	3.1	2.0	2.8	2.9
Invasive species	2.9	3.0	2.8	2.9
Homeless/unhoused people using watershed	2.3	2.0	2.7	2.6
OHV impacts	1.9	3.5	2.5	2.3

Issues that were the least frequently rated as an extreme concern were dissolved organic carbon, pH levels, and issues related to direct human use of the watershed (fecal contamination, unhoused people, and off-highway vehicle [OHV] use). The latter three issues together averaged a rating of 2.7, making direct human use less of a concern than almost all other issues.

Respondent concerns showed some variability by region. For example, the highest concern for wildfire impacts, increased wildfire risk, and wildfire response impacts was from Dryside respondents. For turbidity, Valleys respondents had the highest concern, and for forest chemicals, Coastal respondents had the highest concern. Respondent comments in text and in instances where the survey was completed by phone also suggested that for some of these issues, concern varied temporally. For example, turbidity could be more of a concern in the winter season for Coastal systems; or, forest chemicals were not currently a concern, but would be in the future after a planned herbicide application.

We also attempted to identify issues of concern by asking respondents to select the *top two issues* that currently concerned them most from the same series of *more specific source water protection issues* as they may affect raw water supply (Table 2-14). Just over a quarter of respondents selected turbidity/suspended sediment as a top concern, and forest chemicals were the second most common top concern at 12%. Every other issue was selected by 10% or less of respondents as a top issue.

Table 2-14. Top two specific source water protection issues chosen.

Source water protection issue	Percent of respondents that selected as a top issue
Turbidity/suspended sediment	26%
Forest chemicals (pesticides, fertilizers)	12%
Future water quantity	10%
Potential wildfire impacts (e.g., erosion, cover loss)	7%
Transportation-related fuel and hazardous material spills	7%
Increased wildfire risk	6%
Flood events	6%
Temperature levels	4%
Landslides and slope instability	4%
Other: please describe	4%
pH levels	3%
Nutrient levels	3%
Dissolved organic carbon	2%
Wildfire response impacts (e.g., retardants, pumping)	2%
Riparian buffer blow-down	2%
Invasive species	2%
Other point source pollution	1%
Potential fecal contamination	1%
General public access effects	0%
Homeless/unhoused people using watershed	0%
OHV impacts	0%

For additional insight, we asked respondents to offer open-ended comment on why those issues concerned them and how they managed them. Full text from these responses is provided in Appendix 3; prominent themes are summarized here:

- *Impacts of prior wildfires and concerns about future fire events:* Reported challenges related to wildfires included post-fire effects such as erosion, sediment, turbidity, infrastructure destruction, and needs for filtration systems. Activities to address these risks have included participation in forest collaborative group projects on federal lands, meetings with agencies about forest conditions and risks, pursuit of alternative water sources, and support of forest thinning projects.
- *Forest harvest and management activities:* Several respondents described concerns about logging, particularly “clear cutting” practices, and use of herbicides. They have observed or anticipate future impacts such as invasive species growth, landslides, increased turbidity/sedimentation, and chemicals in drinking water supply. Few described how to address these impacts beyond increased monitoring and communicating with forest landowners about timing of activities so that mitigation actions could be taken.

- *Turbidity:* Many respondents described concerns and experiences with turbidity as a result of a variety of events such as wildfires, forest harvest activities (particularly "clear cutting"), winter storms, and landslides. Approaches to managing and responding to turbidity range from mitigation to emergency response, and include reducing wildfire risk in the watershed, diversifying water sources, having dams that reduce runoff amounts, pre-sedimentation preparation treatments before storms, new filtration systems, physical removal of silt, new storage capacity, and plant shutdowns.
- *Algae:* A few respondents described issues with algae including its growth after increases in turbidity/temperature/nutrients, resulting production of cyanotoxins, and clogging of fish screens. Responses have included new monitoring, disinfection treatments, and building buffering wetlands.
- *Water quantity:* Respondents reported concerns about future water quantity as a result of wildfire events, population growth, drought years, and forest harvest. Few options for addressing this were mentioned aside from finding additional or alternative water sources.

Next, we asked respondents to identify their *level of control* over the same series of *more specific source water protection issues* as they may affect raw water supply by rating each issue on a five-point Likert concern scale (with 1 = no control at all to 5 = total control). Looking at control, and comparing control to concern, may help indicate areas where drinking water providers are most concerned about issues that they feel they can or cannot manage effectively. Table 2-15 shows the results from this question.

Table2-15. Ratings of level of perceived control over various source water management issues

Specific issue of concern	Coast	Dryside	Valleys	All respondents
Potential fecal contamination	2.7	3.5	1.9	2.3
Turbidity/suspended sediment	2.7	1.0	1.9	2.2
Future water quantity	2.3	1.5	1.8	2.0
Increased wildfire risk	1.9	2.0	1.7	1.8
Off Highway Vehicle impacts	2.2	3.0	1.5	1.8
Invasive species	1.8	1.5	1.8	1.8
Homeless/unhoused people using watershed	2.1	2.5	1.6	1.8
Transportation-related fuel and haz. material spills	1.9	3.0	1.6	1.8
Forest chemicals (pesticides, fertilizers)	2.3	2.0	1.4	1.7
Other point source pollution	2.1	1.5	1.5	1.7
Potential wildfire impacts (e.g., erosion, cover) loss	1.9	1.5	1.5	1.7
Wildfire response impacts (e.g., retardants, pumping)	1.8	2.5	1.5	1.7
Landslides and slope instability	1.8	1.0	1.6	1.7
pH levels	2.1	1.0	1.4	1.6
Riparian buffer blow-down	1.7	1.0	1.5	1.5
Dissolved organic carbon	1.8	1.0	1.3	1.5
Flood events	1.5	1.0	1.4	1.4
Nutrient levels	1.5	1.0	1.3	1.4
Temperatures	1.3	1.0	1.4	1.3

Results in Table 2-15 largely indicate a strong sense of lack of control over issues that affect their source drinking watersheds. The top issues that respondents perceived the least control over by mean rating were flood events, nutrient levels, and temperatures. Large majorities (exceeding 70%) felt they had no control at all over multiple issues: dissolved organic carbon, temperature levels, pH levels, nutrient levels, riparian buffer blown down, and flood events. Percentages of respondents selecting "no control at all" were above 40% for every listed issue and above 50% for all but three issues. The top two issues wherein respondents perceived moderate or a lot of control were unhoused people using the watershed (24%) and turbidity/suspended sediment (20%), but the proportion of responses that saw no control over these issues was much larger.

To compare *perceived control* of with *concern about* drinking water issues, we subtracted individual ratings of concern from control. A negative difference indicates that, on average, individuals' concerns over this issue were greater than their perceived control of it. We found this to be the case for every listed issue (Table 2-16). We found the largest differences between control and concern about issues for potential wildfire impacts, forest chemicals, increased wildfire risk, and wildfire response impacts.

Table 2-16. Comparison of mean ratings of *level of perceived control* versus *level of concern* over various source water management issues.

Source water protection issue	Avg. Control	Avg. Concern	Avg. Difference
Potential wildfire impacts (e.g., erosion, cover) loss	1.7	3.9	-2.2
Forest chemicals (pesticides, fertilizers)	1.7	3.7	-1.9
Increased wildfire risk	1.8	3.7	-1.9
Wildfire response impacts (e.g., retardants, pumping)	1.7	3.5	-1.9
Flood events	1.4	3.4	-1.8
Turbidity/suspended sediment	2.2	3.9	-1.7
Water temperatures	1.3	3.1	-1.7
Future water quantity	2.0	3.6	-1.7
Nutrient levels	1.4	3.0	-1.6
Landslides and slope instability	1.7	3.2	-1.5
Other point source pollution	1.7	3.4	-1.5
Riparian buffer blow-down	1.5	3.0	-1.4
Transportation-related fuel and hazardous material spills	1.8	3.2	-1.3
Dissolved organic carbon	1.5	2.9	-1.3
pH levels	1.6	2.9	-1.2
Invasive species	1.8	2.9	-1.0
Potential fecal contamination	2.3	3.1	-0.8
Homeless/unhoused people using watershed	1.8	2.6	-0.7
OHV impacts	1.8	2.3	-0.5

3.4.6. Lessons learned

Respondents were asked a final reflective open-ended question: "What has been a key lesson learned for your utility about managing a forested watershed for drinking water supply? What would you tell someone else in your position?" Full responses are listed in Appendix 4; prominent themes across these comments included:

- *The importance of communications*: Multiple respondents discussed the need to know and communicate regularly with landowners in the watershed and other relevant entities. Some specifically suggested communications with logging foreman and crews who were on the ground in order to have real-time discussions about forest operations as they occurred. Communicating and knowing who to call prior to potential issues was advised. Relationships and communications in a more general sense were mentioned more frequently than more formal partnerships or collaborations.

- *Being proactive and prepared*: Respondents described learning from experiences where they were not prepared, indicating that these taught them to become more proactive and ready for a range of possible events and situations. Activities to foster this preparation included regular examination of the watershed, knowing who to call, practicing scenarios, stocking supplies such as filter bags, updating assessments and plans, and having all necessary documentation.
- A*ctive forest management*: Some respondents recommended hands-on, fully-engaged forest management for forest health, with proactive planning, inventory, monitoring, and activities such as invasive species control and stand improvements.

3.4.7. Summary of key survey results

We attempted to survey all 156 identified drinking water providers in Oregon about the management of their primary source watershed. We obtained a response rate of 35% (54 respondents). Targeted follow-up was conducted with systems with the largest percentages of private industrial timberland, publicly-owned land, or local government ownership of the utility. We found the following key results:

- The majority of systems surveyed were primarily organized as departments or units of municipal government. One-third (34%) had budgets of $100-$500K. Another 24% had budgets of $100k or less.
- Sixty-four percent had one to ten total staff. Seven respondents relied solely on volunteer homeowners or board members, and 19% had only one or two staff. The mean total staff size was 13, with an average of 11 full time employees.
- Nearly one-half of respondents have open access to their primary source watershed for the public at all times, while 20% allowed no public access.
- Fifty-one percent of respondents reported partnering with other entities and landowners to manage their source watersheds, using a range of approaches from informal information sharing as needed and general “good neighbor” practices, to more formal venues such as regular meetings and established, multi-partner collaborations. The most important partners that respondents interacted with most frequently were private industrial forestland owners, watershed councils or SWCDs, and the US Forest Service as a public landowner.
- A majority of respondents (56%) did not participate in or fund forest restoration or hazardous fuels reduction in their source watershed. Those that did described a range of activities including performing their own brush clearing or prescribed burning, or partnering with entities such as watershed councils, forest collaborative groups, or the US Forest Service to support these activities.
- Sixty-two percent of respondents had an Updated Source Water Assessment, 33% had a Drinking Water Source Protection plan, and 40% conducted optional raw water quality monitoring beyond what is required by law. These were variously used to prioritize

management activities, write grants, identify potential hazards, and support adaptive decision-making.

- Nearly half of respondents were not aware of the Oregon Department of Forestry Forest Activity Electronic Reporting and Notification System (FERNS) online system, and less than 20% had a subscription.
- Respondents were asked about management issues of concern that may affect their raw water supply in several ways:
  - ✓ Ranking *general issues:* Forest harvest and management and stormwater runoff were ranked most important over other general issues such as residential development and cannabis cultivation.
  - ✓ Rating level of concern about *specific issues*: Respondents rated potential wildfire impacts (e.g., erosion), turbidity/suspended sediment, forest chemicals (e.g., herbicides), and increased wildfire risk as most concerning.
  - ✓ Identifying their *level of control* over the same *specific issues*: Respondents largely indicated a strong sense of lack of control over most issues that affect source drinking watersheds. The top issues that respondents perceived the least control over by mean rating were flood events, nutrient levels, and temperatures.
  - ✓ Comparing perceived control of specific issues with concern about them: Concern over every listed issue was greater than control of it. The largest differences between control and concern were found for potential wildfire impacts, flood events, forest chemicals, and increased wildfire risk.
  - ✓ Identifying the top two *specific issues* of concern: When asked to pick their top two most important issues, approximately a quarter of respondents selected turbidity/suspended sediment and 12% selected forest chemicals.
  - ✓ Respondents' lessons learned generally emphasized the importance of communication with forest landowners, being proactive and prepared rather than reactive in the face of events and challenges, and actively managing for forest health.

3.4.8. Discussion of survey results

Perhaps the most important finding of our survey of drinking water providers was that many perceived a lack of control over issues that affected their drinking water source watersheds and thus the quality and quantity of their raw drinking water. A large majority (over 70%) perceived no control at all over multiple issues. The survey results also present some other crosscutting themes about the intersection of forest management and drinking watersheds, with implications and future questions for management and future research.

First is public water system capacity, and if it is matched to the management needs that providers may have. Smaller systems (in terms of population served or total connections) may run on a single paid staff person or on a volunteer-only basis, yet they may have to address multiple and substantial forest management and other activities that affect their source watersheds. Our respondents often did not provide complete responses about their budgets, perhaps due to an unwillingness to share this information, and our use of larger brackets precludes more fine-grained understanding of smaller budgets. Further examination could help clarify the degree to which the budgets and staff of smaller public water systems align with issues these systems face in their source watersheds.

A second theme is how assessments, plans, and monitoring may allow more structured understanding of source water management issues, prioritization of actions, and informed partnerships with landowners and other entities. Respondents were more likely to have an Updated Source Water Assessment and/or to do optional raw water quality monitoring than to have a Drinking Source Water Protection Plan. From limited open-ended responses, it seems that assessments may help providers prioritize the risks to address, which may be especially useful in capacity-constrained systems. Monitoring may offer data that allows for anticipation of and more adaptive response to potential impacts of forest management activities, and communication and partnership with a forest landowner could facilitate this. More research would be needed to understand how widespread this type of monitoring and partnership is across all systems in Oregon.

Third, there is growing academic and practitioner interest in partnerships and collaborative approaches for managing forested source watersheds. Some of these are well-publicized, large, and formal multi-stakeholder efforts in areas serving larger populations (e.g., Denver). Less is known about the functioning and approaches of partnerships that may be informal, in smaller systems, and/or involve private industrial owners. Our results show that private forestland owners were the top most important partner to respondents (likely because they own many of the drinking water source areas for providers we surveyed) followed by watershed councils/SWCDs. How providers communicate with private landowners warrants more attention. For example, given that the majority of respondents reported not using the ODF FERNS, are they instead learning about planned forest management activities more informally and directly with landowners? Or does this suggest a lack of communication and awareness of activities that could be improved? How are local entities such as these councils and districts facilitating partnerships with private industrial landowners and accessing funding for source water protection activities?

Fourth, respondents indicated the most concern about potential wildfire impacts, turbidity/suspended sediment, forest chemicals, and increased wildfire risk, and also reported a perceived lack of control over these issues. These concerns fluctuate seasonally, with expectedly more emphasis on turbidity during winter storm events and wildfire risks and impacts during summer seasons. The capacity of providers to anticipate, plan for, and respond to these concerns may depend on the ownership(s) of their source watersheds, and the relationships and communication that they have with landowners and other relevant entities.

3.5 Literature Cited

Adams, P.W. and M. Taratoot. 2001. Municipal water supplies from forest watersheds: fact book and catalog. Department of Forest Engineering, Oregon State University, Corvallis, OR. 100pp.

American Water Works Association (AWWA). 1999. *Water Quality & Treatment* 5th Edition, Chapter 16 “Water Treatment Plant Residuals Management,” McGraw-Hill. New York, NY.

American Water Works Association (AWWA). 2014. AWWA G300-14 Source Water Protection. Second Edition. https://www.awwa.org/store/productdetail.aspx?productId=39814230

Bhardwaj, V. 2006. What is the disinfection by-products rule? On Tap, Winter, 2006. National Environmental Services Center, University of West Virginia.

Collier, M. 2018. Personal communication (e-mail), 9/20/2018, from Mike Collier, Deputy Director/Source Water Specialist, Oregon Association of Water Utilities to Jeff Behan, Senior Policy Analyst, Institute for Natural Resources, Oregon State University.

Crittenden, J.C., R.R. Trussell, D.W. Hand, K.J. Howe, and G. Tchobanoglous. 2005. Water Treatment: Principles and Design. 2nd Ed. (New York: Wiley & Sons). 1968 pp.

Fulton, G.P. 2000. D*iatomaceous Earth Filtration for Safe Drinking Water*, ASCE Press, Reston, VA. 224 pp.

Gullick, R.W. 2017. Operational Guide to AWWA Standard G300-Source Water Protection. Second Edition. American Water Works Association/Colorado.

Hendricks, D., J.M. Barrett, J. Bryck, M.R. Collins, B.A. Janonis, and G.S. Logsdon. 1991. Manual of Design for Slow Sand Filtration, Chapter 4: “Pilot Plant Studies,” AWWA Research Foundation, Denver, CO.

Huisman, L. and W.E. Wood. 1974. Slow Sand Filtration. World Health Organization, Geneva, Switzerland. 120 pp.

Karanfil, T. and A. Chow. 2016. Fuel reduction techniques as effective forested watershed management practices against wildfire: Drinking water quality aspects. USEPA National Priorities: System-Based Strategies to Improve the Nation’s Ability to Plan and Respond to Water Scarcity and Drought Due to Climate Change (Powerpoint presentation). https://www.epa.gov/sites/production/files/2016-05/documents/karanfil-presentation-slides-drought-kickoff-meeting.pdf

Kawamura, S. 2000b. *Integrated Design and Operation of Water Treatment Facilities*, 2nd Edition, Chapter 2: “Preliminary Studies,” John Wiley & Sons, New York, NY.

Minnesota Rural Water Association (MNRWA). 2009. *Minnesota Water Works Operations Manual*, 4th Ed., Chapter 20 Membrane Filtration.

National Association of Clean Water Agencies (NACWA). 2006. Emerging Issues: Intersections of the Clean Water Act & Safe Drinking Water Act (White Paper). http://archive.nacwa.org/getfilecf3c.pdf?fn=2006-06cwa_sdwa_wp.pdf

Oregon Department of Environmental Quality (DEQ). 2017. Drinking water protection: Using an ordinance or overlay. (Fact sheet). Accessed 1-6-2019. https://www.oregon.gov/deq/FilterDocs/DWPOrdinanceOverlay.pdf

Oregon Department of Environmental Quality (DEQ). 2018b. Fact Sheet: Introduction to Oregon’s Drinking Water Protection Program. https://www.oregon.gov/deq/FilterDocs/dwpIntro.pdf

Oregon Department of Environmental Quality (DEQ). 2018c. Oregon public water systems surface water resource guide for drinking water source protection. February 2018, version 1.0. https://www.oregon.gov/deq/FilterDocs/SurfaceWaterResourceGuide.pdf

Ranjan, P. and M. Prem. 2018 Schmutzdecke – A Filtration Layer of Slow Sand Filter. Int. J. Curr. Microbiol. App. Sci. 7(7): 637-645. https://doi.org/10.20546/ijcmas.2018.707.077

Ten States Standards. 2007. Recommended Standards for Water Works. 2007 Edition. Policies for the Review and Approval of Plans and Specifications for Public Water Supplies. A Report of the

Water Supply Committee of the Great Lakes – Upper Mississippi River Board of State and Provincial Public Health and Environmental Managers. Health Research, Inc., Health Education Services Division, Albany, NY. 136 pp.

Tiemann, M. 2017. Safe Drinking Water Act (SDWA): A Summary of the Act and Its Major Requirements. Congressional Research Service Report 7-5700. RL31243. www.crs.gov.

USEPA. 2003 (a). Long Term 2 Enhanced Surface Water Treatment Rule: Toolbox Guidance Manual (Draft), Chapter 8: "Bag and Cartridge Filters," EPA 815-D-03-009.

United States Environmental Protection Agency (USEPA). 2005. Fact Sheet – Long Term 2 Enhanced Surface Water Treatment Rule. EPA Office of Water 4607M. EPA 815-F-05-009. Accessed 1-4-2019. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000E999.txt

United States Environmental Protection Agency (USEPA). 2006. 40 CFR Parts 9, 141, and 142 National Primary Drinking Water Regulations: Stage 2 Disinfectants and Disinfection Byproducts Rule; Final Rule. Federal Register Vol. 71, No. 2 / Wednesday, January 4, 2006 / Rules and Regulations.

United States Environmental Protection Agency (USEPA). 2017a. Drinking Water Requirements for States and Public Water Systems. Accessed 1-2-2019. https://19january2017snapshot.epa.gov/dwreginfo/drinking-water-regulatory-information_.html

United States Environmental Protection Agency (USEPA). 2017b. Secondary Drinking Water Standards: Guidance for Nuisance Chemicals. Accessed 1-3-2019. https://www.epa.gov/dwstandardsregulations/secondary-drinking-water-standards-guidance-nuisance-chemicals

United States Environmental Protection Agency (USEPA). 2017c. Monitoring Unregulated Drinking Water Contaminants. Accessed 1-2-2019. https://19january2017snapshot.epa.gov/dwucmr/learn-about-unregulated-contaminant-monitoring-rule_.html,

United States Environmental Protection Agency (USEPA). 2018a. Summary of the Clean Water Act. Accessed 12-12-2018. https://www.epa.gov/laws-regulations/summary-clean-water-act

United States Environmental Protection Agency (USEPA). 2018b. Basic Information about Nonpoint Source (NPS) Pollution: Overview. Accessed 12-18-2018. https://www.epa.gov/nps/basic-information-about-nonpoint-source-nps-pollution

United States Environmental Protection Agency (USEPA). 2018c. Watershed Academy Web, Forestry Best Management Practices in Watersheds: The Clean Water Act and Best Management Practices. Accessed online 12-18-2018: https://cfpub.epa.gov/watertrain/moduleFrame.cfm?parent_object_id=1522

United States Environmental Protection Agency (USEPA). 2018d. Summary of the Safe Drinking Water Act. Accessed 12-12-2018. https://www.epa.gov/laws-regulations/summary-safe-drinking-water-act

WSDOH. 2003. *Slow Sand Filtration and Diatomaceous Earth Filtration for Small Water Systems*, DOH 331-204, Washington State Department of Health, Olympia, WA. 118 pp.

THIS PAGE INTENTIONALLY LEFT BLANK

Chapter 3. Active Forest Management and Community Water: Issues and Interactions

Jeff Behan

Oregon has some of the most productive forestland in the world. Timber harvesting and associated activities played a key role in Euro-American settlement and development in Oregon, and remain significant as a sector in the state's economy. For years, Oregon has led the nation in softwood and plywood production. Oregon's forest industry supports more than 60,000 jobs (OFRI 2017).

Forested watersheds – both managed and unmanaged - also produce higher quality water than any other type of surface water source area and supply drinking water to a majority of Oregon's community water systems. Forest management practices, including methods for road construction and use, harvesting and site preparation, and chemical applications have markedly improved over the past few decades. But forestry can still adversely impact downstream water quality in a number of ways, primarily as a result of the construction, use and maintenance of forest roads, but also silvicultural activities, mainly from when trees are harvested through the first decade or two into the new rotation. Forest roads and active forest management have also been shown to impact the volume and timing of water delivered from watersheds.

This chapter is primarily an overview of interactions between forestry and water, best management practices to reduce water quality impacts, and some remaining issues and concerns.

Sections 3.1 and 3.2, summarizing mechanisms and functions regarding, 1) the stability and movement of soil and sediment in forest environments, 2) water collection, retention, and production in forested watersheds, and 3) how forest management can affect these functions, are drawn primarily from forest hydrology text books (Chang 2012, Amatya et al. 2016 and chapters therein) supplemented by additional references drawn primarily from the literature search described in Chapter 2.

Section 3.3 introduces the topics of the use of synthetic forest chemicals (herbicides, pesticides, fertilizers), and forest management actions that can modify the production of naturally occurring compounds (e.g. natural organic matter, nitrates) that may affect water quality. Section 3.4 discusses natural organic matter (NOM) and disinfection byproducts (DBP).

Section 3.5 provides a brief history and overview of Best Management Practices (BMPs) for mitigating the effects of forest management on water resources.

The intention with this chapter is to summarize generally established and accepted science knowledge and concepts regarding these topics to complement more detailed discussion focused more specifically on relationships to drinking water in Chapters 4, 5, 6, and 7.

3.1. Forest Management and Stream Sediment

Undisturbed forests have high infiltration rates and little overland flow, with precipitation usually passing through the soil before reaching streams, minimizing erosion and sedimentation and producing high quality water (Stednick and Troendle 2016; Williams 2016). Forestry activities such as road building and timber harvesting involve a certain amount of soil disturbance and the potential for disturbed soils to be mobilized by water or wind. For this reason, processes by which forest operations may increase the erosion, transport and deposition of forest soil into waterways have long been subject to intense focus from stakeholders, researchers, policymakers and practitioners. Wind erosion has been recognized as an issue in drier forests (e.g., Whicker et al. 2006). But water erosion is of primary concern, especially in wetter forests, and will be the focus here.

Stream sediments are soil and mineral particles, usually inorganic but sometimes partly organic, detached from the land by processes that include raindrop impact, surface runoff, streamflow, wind, and gravity; often in association with human activity. Sediment inputs that result from human activities frequently impair the physical, chemical, and biological properties of streams and degrade beneficial uses. Sediment is a leading cause of stream impairment nationwide and in Oregon (USEPA 2017). Sediment can affect water turbidity, chemical composition, taste, and odor and interfere with drinking water treatment processes. Sediment concentration and yield are widely accepted indicators for the effectiveness of watershed management practices.

3.1.1. The Water Erosion Process

Water erosion is a three-step process consisting of soil detachment, transport and deposition. Raindrops are effective at detaching soil particles, and generally increase in size and terminal velocity with rainfall intensity. Once detached by striking raindrops or overland flow, soil particles are transported by runoff, with distance varying according to soil properties, topography, runoff energy and surface conditions. Sediment is deposited when the soil carrying capacity is less than the weight of the soil particles. The ability of soil to resist detachment from raindrop impact and surface flow generally increases with increasing organic matter content and infiltration rate. Depending on conditions and disturbance history of the area, a soil particle can move into a nearby waterway during one rainfall event or, conversely, this could take decades or even centuries (Chang 2012).

Water erosion in watersheds occurs at a range of scales. Smaller-scale processes include interrill and rill erosion on side slopes and ephemeral gully erosion along shallow drainage ways. These in turn feed downslope into more deeply incised gullies which form when converging eroded drainage ways reach a certain size. Gully erosion generally occurs in well-defined drainage ways and involves soil particle detachment by flowing water and slumping of unstable banks, and transport by flowing water. Sediment loads are often greater downstream due to the additive effect of interrill and rill erosion from adjacent areas and detachment of soil particles upstream in the drainage way. Sediment transport capacity increases downstream along with flow volume.

Gullies are an advanced stage of water erosion and are permanent unless they are actively filled. Without active conservation and mitigation measures, gullies will continue to expand and grow via down-cutting and head-cutting. Down-cutting deepens and widens gully bottoms. Head-cutting extends channels upslope into headwater areas and expands the gully tributary system. Deep gullies may extend to the watershed divide. Poor road layout and construction often accelerate gully development.

In contrast to gully erosion, which occurs in the upper ends of headwater tributaries with water flowing primarily during or immediately after storms, channel erosion occurs in the lower end of headwater tributaries where water flows on a continuous basis. Channel erosion consists of soil erosion on stream banks and sediment transport in the stream channel. Stream bank erosion is frequently caused or exacerbated by removal of vegetation.

Slope failures or mass movements generally are larger scale processes involving downhill movement of significant volumes of soil, rocks, and organic matter under the direct influence of gravity, including landslides and debris flows. Slope failures occur most often in areas with steep slopes and weak geological structures. They are often triggered by some combination of factors or events, including intense and prolonged rainfall, snow buildup and melt, converging overland flows and seepage, earthquakes, and forest harvesting. Factors leading to a decrease in slope stability can include increase in water content, reduced internal soil cohesion, and higher groundwater table as a result of increased precipitation or deforestation (Chang 2012).

Pore water pressure (or just pore pressure) is the pressure exerted by water held in pore spaces in soil. When a soil is fully saturated, pore-water pressure said to be positive. Pore pressure rise resulting from rainfall or snowmelt is the most common triggering mechanism for landslides. Positive pore water pressure develops just above a restrictive layer (e.g., bedrock) in rapid response to rainfall infiltration, causing soil shear strength to decrease to the point at which the slope fails. In addition to storm intensity and duration, the extent of pore water buildup is also influenced by soil moisture conditions prior to the storm. Wetter antecedent conditions (e.g. in midwinter) promote more rapid pore pressure response during storms compared to drier conditions (e.g. at the onset of fall rains) (Sidle and Bogard 2016).

3.1.2. The Role of Forest Vegetation in Controlling Water Erosion

Maintaining forest vegetation is an effective, economical and long lasting approach to mitigating soil erosion and stream sediment loading in forest environments. Plant sizes are taller, canopy density is greater, litter floor is thicker, and root systems are deeper in forests than in any other type of vegetation cover. Thus, forests resist erosion and sediment movement much more than other vegetation types (Chang 2012). This resistance is a key reason forests are capable of producing such high quality water.

Water erosion of soil is initiated when soil particles or soil masses are detached from the soil matrix or underlying surface by some combination of precipitation, runoff energy and gravity. Forest vegetation attenuates soil detachment and movement, at scales ranging from individual soil particles to mass wasting, via several different mechanisms:

- Interception of rainfall by tree canopies above the ground, which reduce the velocity and energy of raindrops, and also the amount of precipitation that reaches the ground.
- The ability of litter, woody debris and ground-level vegetation to reduce raindrop and overland flow energy by shielding the soil and inhibiting runoff movement.
- Root systems and organic matter that increase the cohesive and frictional components of soil shear strength, which contributes to soil stability.
- Transpiration and evaporation (evapotranspiration, or ET) of water by trees and other vegetation, which reduces soil moisture content.

- Buttressing or soil arching action between tree trunks, which counteracts downslope shear forces.

Most rain falling into a forest canopy is intercepted by tree foliage. In smaller storms, nearly all rainfall may evaporate off of the foliage and never reach the ground, especially in dense coastal old-growth forests, but the percentage that evaporates decreases as storm intensity and duration increase (Moore and Wondzell 2005). The degree to which a forest canopy reduces raindrop energy by intercepting drops depends on canopy density, canopy height and tree species. Canopy heights (from ground level to the lowest tree branches) of less than 20m significantly decrease raindrop speed and impact energy and conifer forest canopies intercept more rainfall than deciduous forests. Forest vegetation of any height also helps attenuate wind and increases in raindrop impact energy caused by wind-driven drops striking the soil at an angle.(Williams 2016; Chang 2012). Canopies that are close to or in ground contact act as shields and essentially eliminate raindrop energy. Litter in the form of leaves, needles, cones and small branches that drop from forest canopies of any height increases ground surface roughness and slows runoff velocity, thereby reducing soil erosion.

Large roots from woody vegetation extend down through the soil surface horizon and anchor the soil mantle to the substrate. In conjunction with these larger taproots and lateral roots, fine roots, fungal mycelia, and decomposed organic matter help form anchored aggregates of surface soils centered around individual trees. The strong binding effects of this dense and interwoven soil–root system stabilize the forest soil mantle.

Any forest management action that reduces canopy coverage and disturbs the forest floor and soil has the potential to generate additional erosion and sediment production. The increased sediment yield resulting from a forest activity depends on the degree of forest and soil disturbance, location and proportion of the watershed affected, watershed characteristics (e.g. slope, soil type, ecological factors), weather patterns, and climate.

3.1.3. Forest Harvesting, Erosion and Sediment Production

In actively managed forests, logged hillslopes are the largest land surface area subject to potential disturbance. Under modern forest practices, the size of harvested compartments (clearcuts) is restricted to smaller sizes than in the past. These General Harvest Areas (GHAs) usually have patches of compacted soils interspersed with areas more similar to undisturbed forest floor. Runoff typically builds slowly in GHAs, even under heavy rainfall, usually starting on the more disturbed patches of the hillslope. But channelized flow tends not to develop in GHAs due to the high spatial variability in soil infiltration capacity, and presence of remaining vegetation and loose material on the soil surface. This patchy nature of runoff generation usually limits the ability of runoff in GHAs to mobilize large amounts of sediment. After harvesting, disturbed soils can recover some of their infiltration capacity over time (Croke and Hairsine 2006).

There are exceptions to these general findings regarding GHAs, especially when forests are harvested in steeper terrain. Removal of trees has consistently been shown to reduce the stability of steep slopes and increase the risk of landslides and mass movement (Goetz et al. 2015; Imaizumi and Sidle 2012; May 2002; Jakob 2000; Montgomery et al. 2000). More specifically, many studies have shown that from about 2 to 15-20 years after harvesting on steep slopes, the rate of landsliding is about 2 to 10 times higher than prior to harvest (Sidle and Bogard 2016) and that this increase is strongly linked to the loss of root reinforcement and cohesion in forest soils after the trees are removed and as the roots decompose (Sakals and Sidle 2004; Roering et al. 2003; Guthrie 2002; Schmidt et al. 2001). Intact forests also contribute to slope stability by attenuating rainfall and soil moisture (Preti 2013) although Sidle and Bogard (2016) argue that in temperate forests, root reinforcement is usually more important for slope stability than transpiration or canopy interception. Increased landslide risk associated with forest harvesting can be reduced by partial cutting of the stand and retention of understory vegetation (e.g. Dhakal and Sidle 2003; Sakals and Sidle 2004; Turner et al. 2010).

Findings linking forestry activities on steep slopes with increased occurrence of landslides are usually based on landslide inventories comparing logged and unlogged areas. Such inventories are often compiled primarily through air photo interpretation, a method which can be subject to "detection bias" - the difficulty of detecting smaller slides under the canopies of intact forests (Robison et al. 1999). Thus, rigorous studies often attempt to correct for this potential bias, by augmenting air photos with subsampling, ground truthing or some type of correction factor.(E.g., Turner et al. 2010; Miller and Burnett 2007). The use of Light Detection and Ranging (LIDAR) remote sensing techniques also shows promise for reducing detection bias in landslide delineation and inventory (Guzzetti et al. 2012; Jaboyedoff et al. 2012).

Relationships between active forest management and sediment production are discussed in more detail in Chapter 5.

3.1.4. Site Reparation and Sediment Production

Under the Oregon Forest Practices Act (FPA), industrial timberlands in the state must be replanted to trees within 24 months after clearcut harvests. Prior to replanting, sites are prepared to reduce vegetation that competes with tree seedlings, reduce habitat for animals that damage seedlings, to reduce wildfire risk and to create spots for planting (Fitzgerald 2008). Site preparation can involve the use of herbicides, mechanized equipment, fire or some combination of these methods. In general, any site preparation activities that contribute to an increase in bare mineral soil, soil compaction or soil mixing have the potential to increase sediment production. See Chapter 5 for a more detailed discussion regarding these interactions.

Industrial timberlands in western Oregon are typically treated with herbicides prior to replanting. Neary et al. (2000) maintain that in general, herbicide use ranks behind both fire and mechanized equipment in severity of impact on sediment production. But herbicide use in western Oregon forestry continues to spark controversy, especially over the potential for it to drift into drinking water sources or populated areas when applied via aerial spraying (e.g. Burns 2019; Perkowski 2018; Swanson 2017). The FPA stipulates that herbicides must be prepared for use at least 100 feet from streams that bear fish or are drinking water sources. Aerial applicators must closely monitor weather patterns and only spray when risk of drift will be minimized. They must also spray at least 60 feet from waterways and bodies of standing water larger than a quarter-acre. Any detectable concentrations of herbicides in waterways are usually short-term. Herbicide use in forestry is discussed in more detail in Chapter XX.

3.1.5. Forest Roads, Erosion and Sediment Production

Sediment input into streams from forest roads has long been of concern, and forest roads continue to be recognized as the major source of erosion in watersheds (Croke and Hairsine 2006). As Neary et al.(2009) put it, "…the study of nonpoint source pollution from forestry activities has largely been a study of runoff and erosion from bare soil areas created for roads, landings, skid trails, fire breaks, and also bare soils created by site preparation fires. In all forested areas of the United States (except for flat coastal plain areas), roads, landings, and skid trails have been repeatedly implicated as the primary source of sediment from silvicultural operations" (p. 2275).

A watershed-level network of forest roads often contains an interconnected mosaic of older and newer roads designed to different standards, sometimes for different purposes, and crossing terrain of differing sensitivities to erosion and mass wasting. The particular pattern and hydrologic connectivity of this mosaic of road segments has implications for how it will interact with the forest watershed, streams, and other downstream water uses (Endicott 2008). Impacts of roads range from chronic and long-term contributions of fine sediment into streams to catastrophic mass failures of road cuts and fills during large storms (Beschta 1978; Wemple et al. 2001; Sidle and Ochiai 2006). Megahan and King (2004) concluded that roads affect landslide creation more than any other forest management activity. Problems with drainage and transport of water—especially during heavy rainfall and floods—are primary reasons roads fail.

Roads can also alter channel morphology directly or modify channel flow and extend the drainage network into previously unchanneled parts of the hillslope. The actual magnitude and longevity of chronic effects of forest roads on suspended sediment in streams depends on many site-specific factors, including traffic, geology, road grade, road connectivity to the stream, and sediment availability for transport (Grant and Wolff 1991; Benda and Dunne 1997; Hassan et al. 2005) and also road age, construction practices, maintenance practices, climate, and storm history. Volume, weight and the timing of traffic (i.e. during dry or wet weather) also affect the amount of sediment produced.

In recent decades, management practices in road location, design, construction, maintenance and use have improved markedly (Gucinski et al. 2001). Most changes have focused on reducing hydrologic connectivity between roads and waterways. But few studies have quantified improvements in lowering mass erosion rates, and forest roads and their effects on sediment production and water quality remain controversial issues, as discussed in more detail in section 3.7.1 below. Also sometimes problematic and controversial are so-called "legacy" roads - forest roads that were planned and built without the benefit of current knowledge regarding their effects or guidance from current road-building standards. Legacy forest roads are discussed in more detail below in section 3.7.2.

3.1.6. Forestry and Sediment Production: Information Gaps

Anderson and Lockaby (2011) review existing science knowledge regarding active forest management and stream sediment, and note some information gaps. One such area is the need for longer-term studies that can better account for climatic variability and address the effectiveness of current and improved forest practices over time. They note that funding for long-term and paired watershed studies has declined, although the Alsea Watershed Study has been reinitiated on a more limited basis (e.g.

Hatten et al. 2018) and the Hinkle Creek Watershed Study was initiated in 2011. They observe that major storms are often a significant driver of sediment movement, and that whether or not one or more such storms occur during the duration of study can significantly affect results of studies that span only a few years. Knowledge is also limited regarding mechanisms of sediment production and cumulative effects resulting from forest management in larger watersheds, due in part to the variability of forestry activities (e.g. roads, harvesting, site preparation) and temporal range of their impacts on stream sediment, with some actions having an immediate effect and others taking years to manifest. Research is also needed that further clarifies how much of the sediment mobilized as a result of silviculture or forest roads is then actually delivered to streams.

Anderson and Lockaby (2011) also note that while forestry Best Management Practices (BMPs) overall have clearly resulted in significant reductions in impacts to water quality, studies that sort out the effectiveness of individual practices are still quite limited. This point is echoed by Edwards et al. (2016). In addition, even when a particular BMP is known to be effective, the exact mechanism for its effectiveness may still be unclear. For example, vegetated buffers along streams have been clearly shown to reduce sediment, but it remains unclear whether this is primarily due to reducing or intercepting overland flow, reducing bank and channel scouring, or a combination. Similarly, there are significant knowledge gaps regarding the effectiveness of different buffer widths, and the effects of thinning or partial harvest within buffer zones.

3.2. Forest Management and Water Production

Relationships between forest cover and type, forest management, and the quantity and timing of water produced by forested watersheds have been studied for at least 100 years (e.g. Bates and Henry 1928; Griffin 1918). Understanding of these relationships has been greatly enhanced by long-term, paired watershed studies (Stednick 1996, Stednick 2008, Stednick and Troendle 2016). But significant knowledge gaps remain, for example effects in larger watersheds, and how harvesting may affect mechanisms that influence the ability of watersheds to store water (McDonell et al. 2018).

3.2.1. Precipitation, Infiltration and Watershed Storage

Water in stream channels comes from at least one of the following:

- Precipitation intercepted by stream channels
- Overland flow (surface runoff)
- Interflow (subsurface runoff)
- Baseflow (groundwater runoff)

Precipitation in forests is reduced by canopy and litter interception and to a much lesser degree by wetting of the soil surface. *Effective rainfall* – the amount that reaches the mineral soil - ranges from 70% to 80% of gross rainfall in forested areas. Water enters soil by *infiltration*, a combination of capillary attraction, gravitation, and pressure from water ponding at the surface. The rate of infiltration is initially high, and then declines as soil spaces fill with water. The process of water draining to deeper layers is called *percolation*. *Macropores* are voids in the soil through which precipitation percolates, mostly tubular channels created by root mortality or activity by insects, worms or burrowing animals, but also structural cracks or fissures. Macropores are the reason intact forest soils display much higher vertical hydraulic conductivity than those obtained from sieved samples of the same soil (Williams 2016).

Surface conditions such as vegetation type, land uses, roughness, crusting, cracking, slope, water repellency resulting from fire, and chemicals have a significant impact on surface ponding, overland flow velocity, and the ability of water to infiltrate soil. Below ground conditions that affect soil water-holding capacity and water movement include soil texture, structure, organic matter content, depth, compaction, water content, groundwater table, and root systems. Forested watersheds are generally characterized by deep, loose soils, thick, loosely compacted duff layers on the forest floor, complex root systems, large canopies, and high capacities for water infiltration. For a given soil type, water infiltration in a forest can be many times greater than over bare mineral soil. As a result of these factors, infiltration-excess overland flow is rare or non-existent on undisturbed temperate forests (Neary et al. 2009; Williams 2016) including those in the Pacific Northwest.

Watershed storage is water retained within a watershed after collection from precipitation and before discharge out of the watershed as streamflow. Watershed storage consists primarily of soil moisture, but also canopy and litter interception, snowpack, ponds and wetlands, shallow aquifers, stream bank storage, channel storage, storage in fractured bedrock, and in vegetation during transpiration. Stored water can remain in a watershed for years or even decades (McConnell et al. 2018). Water storage is recognized as the key function of forested watersheds (Black 1997).

3.2.2. Runoff and Streamflow

Runoff is precipitation (rain or melted snow) running across the land surface or through the soil to nearby stream channels, and occurs when rainfall or snowmelt is greater than soil infiltration rate, or exceeds soil-infiltration or percolation capacity. Thus, the soil surface does not need to be saturated for overland flow to occur. Infiltrated water can become surface runoff again as it flows laterally and downslope or to stream channels as subsurface runoff. Channel rainfall, surface runoff, and subsurface runoff combined are *direct runoff*, a direct response of streamflow to storm precipitation over a relatively short time frame. Water flowing in streams during periods of no rainfall (base flow) comes from groundwater. The sum of direct runoff and base flow is total streamflow.

A *hydrograph* graphically illustrates streamflow discharge or stage over a particular time period, such as a single storm event, or a water year. A hydrograph for a storm event typically shows an upward sloping, then level, and then downward sloping line as discharge increases and then declines back to base flow. The *rising limb*, which shows increasing watershed discharge, begins sometime after precipitation starts, and varies with watershed characteristics and storm duration, intensity, and distribution. Due to watershed storage, lag time is longer in forests than other watershed cover types. Large watersheds may take days to respond to precipitation. The hydrograph *crest* - the highest concentration of storm runoff, also termed *peak flow* - spans from where the rising limb levels off to where the line begins to decline. The end of the crest indicates the end of direct runoff to the stream. The *recession limb,* showing the draining-off process, represents the contribution of water from watershed storage and is independent of storm characteristics (Chang 2012).

Streamflow discharge varies greatly with watershed. All else being equal, smaller watersheds tend to be more sensitive to precipitation events, with quicker responses, and sharper rises and declines in their hydrographs than larger watersheds. In general, higher elevation watersheds are cooler and have less ET, more precipitation, steeper slopes, and shallower soils, which combined result in comparatively more runoff. In general, soil infiltration tends to be lower and overland runoff greater and faster in watersheds with steep slopes, and watersheds with shallower slopes would be expected to store more water than those with steep slopes (Chang 2012). However, studies in the California and Oregon Coast Ranges (Montgomery and Dietrich 2002; Sayama et al. 2011) showed that steeper watersheds in those sites can store as much or more water than, and release it at similar rates to, watersheds with shallower slopes. These findings are attributed primarily to water storage in fractured, permeable bedrock just below the soil layer.

Oregon experiences a Mediterranean climate, resulting in distinct seasonal delivery of precipitation that can be categorized as wet and dry seasons. About 80% of annual precipitation in western Oregon falls between October and March, especially from December to March when there is ample streamflow and virtually no agricultural demand for irrigation water. At higher elevations, much of this precipitation falls as snow, which accumulates through winter then melts during spring. The timing of snowmelt thus plays a major role in shaping annual hydrographs in Oregon. Hydrographs for most Oregon streams peak in winter and spring, but demand for most water uses peaks during the late summer dry season when flows are lowest (Mucken and Bateman 2017). For water providers trying to meeting late-summer demand this misalignment poses persistent challenges, which are expected to intensify as a warming climate reduces the proportion of annual precipitation falling as snow and stored as snowpack, and increases winter rainfall which runs off without being stored (Clifton et al. 2018; Mote et al. 2018; Siler et al. 2018).

Streamflow fluctuation is important to water supply and floodplain management and can be used as an indicator for the effectiveness of watershed management conditions.

3.2.3. Forest Harvesting and Water Yield

Many studies in wetter forests have found that forest harvesting increases watershed-level water yield. Paired watershed studies indicate that a minimum of 450-500mm of annual precipitation is usually necessary for increases to be apparent. In drier forests, harvest often simply increases soil evaporation or water use by other vegetation (Stednick and Troendle 2016).

Usually, increases are greatest the year after cutting then decrease as vegetation regrows, eventually returning to pre-harvest levels. Water yield increases are attributed to increases in soil moisture due to reductions in ET and canopy interception of rain and snow after trees are removed, and vary with harvest intensity, species, amount and timing of precipitation, and soil topographic conditions (e.g. Reid and Lewis 2007). Deep and fine-textured soils can hold more water than shallow and coarse-textured soils, and thus have more potential for water yield increase. In soils less than about 1m deep, water yield increases are minimal after forest harvest. Harvesting on upper slopes increases water yield less than harvesting on lower slopes or close to stream channels (Chang 2012). Harvesting 20% of the watershed is commonly cited as the minimum necessary to detect an increase in water yield; for 12 studies in the Pacific Northwest (PNW) this figure averaged 25% (Stednick 1996).

Results of studies on which these generalizations are based vary widely, with some watersheds showing large increases in water yield after harvest, and others showing little to none at all. Further complicating the picture are studies indicating that watersheds covered with young, vigorously growing plantations of Douglas-fir significantly reduce summer low flows compared to adjacent unharvested watersheds where cover remains in old-growth forest (Moore et al. 2004; Perry et al. 2017). Few studies have addressed this issue, perhaps because any such effects appear to take around two decades or more after harvesting and replanting to become apparent (Gronsdahl et al. 2019), and availability of relevant long-term data is limited. But the potential for decreased summer low flows associated with timber plantations has sparked interest and is likely to be a focus of additional research given the critical nature of water supplies during this time of year and the potential for climate change to exacerbate such challenges.

In addition to changes in annual water yield, forest harvesting can also affect the timing of water production from a watershed. Comparing two small, snow-dominated watersheds on the Okanagan Plateau of BC, Canada, Winkler et al. (2017) found only a 5% increase in overall yield after clearcutting of 47% of the logged watershed, but dramatic changes in the timing and magnitude of April-June streamflow, which they said could increase the risk of channel destabilization during the snowmelt season, and water shortages early in the irrigation season.

Difficulties in consistently predicting the effects of forest harvest and regeneration on water yield have prompted suggestions that this approach is overly simplistic, and calls for an expanded research agenda that also encompasses relationships between forest harvesting and processes that affect watershed storage in order to maintain this key ecosystem service (McDonnell et al. 2018; see also McNamara et al. 2011, Sayama et al. 2011). Further, Chang (2012) notes that most studies on harvesting and water yield take place in the upper parts of watersheds, so effects on water quantity changes for downstream water users also warrants further research.

3.2.4. Forest Harvesting and Peak Flows

The effects of forest cover on peak flow frequency, magnitude and timing have been debated since at least the early 20th century, when these issues served as a key rationale for creation of the US National Forest system. Peak flows and flooding can affect drinking water treatment by raising turbidity levels, and introducing other pollutants mobilized by flood waters.

While not definitive, there is considerable evidence to support the notion that forestry activities can increase peak flows (e.g. Winkler et al. 2017; Zhang and Wei 2014; Schnorbus and Alila 2013; Kuras et al. 2012; Lin and Wei 2008; Moore and Wondzell 2005; Jones 2000; Burton 1997; Jones and Grant 1996). One challenge with this type of research is the difficulty of distinguishing the effects of harvesting from those of roads. Also, reviews and summaries (e.g. Stednick and Troendle 2016; NRC 2008; Grant et al. 2008) often find that results are mixed across studies from different areas and with different methods, with some studies showing significant increases in peak flows, and others no effects or decreases.

Smaller peak flows generally increase after harvesting, but Sidle and Gomi (2017) caution that "much controversy still persists around the effects of forest removal on large floods", [e.g. Alila et al. 2009; Beschta et al. 2000; Jones and Grant 1996; Kuras et al. 2012; Thomas and Megahan 1998] and that based on physical evidence, attributing increases in major peak flows to forest harvest only is “difficult to justify” (Sidle and Gomi 2017, 101). Scientific thinking on this complicated topic continues to evolve, and research results can vary widely, depending on topography, climate, site conditions, land use history, scale, study design, analysis methods, and other variables. As the National Research Council (2008) notes, scientists are quite confident regarding general hydrological responses to forest harvesting, but precise prediction of effects in areas that have not received intensive study can be problematic.

As discussed above, most Oregon streams and rivers peak during winter and spring, and decline to their lowest levels in late summer. Overlain on this seasonal pattern are numerous hydrograph peaks from individual precipitation events. Rain-on-snow (ROS) events in particular can produce large spikes in hydrographs (Marks et al. 1998). Thus, peak flows are assessed by looking at both their frequency and magnitude. Chang (2012) provides broad conclusions from research on the effects of forestry on peak flows and flooding, including: 1) forestry activities such as road construction and site preparation that cause soil compaction are more likely to affect flood generation than is forest harvesting, and 2) forests can attenuate peak flows for storms of short duration and lower intensity, but cannot prevent floods produced from storms of high intensity and long duration over a large area.

The general consensus has been that impacts of forest harvest on peak flows are more noticeable for smaller, shorter storms. When precipitation amount, intensity, and duration increase, the relative influence of human activities on runoff volume declines, as soils become saturated and flows overwhelm any incremental increase attributable to forest harvesting. Human-caused increases in flow volume and peak are less evident downstream due to cumulative effects from other tributaries, decreasing percentages of treated areas as watershed area increases downstream, and attenuating effects of channel storage (Chang 2012, Buttle 2011).

Grant et al. (2008) reviewed the effects of harvesting on peak flows in western Oregon and Washington. They found wide variability in research results, and that assessing the effects of modern forest practices was particularly problematic. They note that peak flows are also affected by overall basin condition; age and pattern of forest stands; the location, age, and extent of road networks; and the extent of riparian buffers. The review was complicated by challenges in distinguishing the effects of harvesting from effects of forest roads. General conclusions from the review include: 1) the largest peak flow increases reported were for small storms with recurrence interval much less than 1 year, 2) increase in peak flow generally decreases with time after harvest; 3) the largest increases occur in clearcut areas; and 4) watersheds in rain-dominated elevations are less sensitive to peak flow changes than those in the transient snow zone. (Lack of sufficient data precluded assessment of harvesting in the snow zone).

The review found studies in larger basins to be limited and complicated by other land uses and factors that affect peak flows but in general, reviewers concluded that the magnitude of any peak flow increase in response to forest management diminishes with increasing basin area. Grant et al. (2008) list, in order of potential likelihood of increasing peak flows, 1) high road density, 2) high road connectivity, 3) fast watershed drainage efficiency, 4) large harvest patch size, and 5) lack of riparian buffers.

The magnitude of peak flow increases generally increase with percentage of the basin harvested (Buttle 2011). Stednick and Troendle (2016) indicate that peak flow increases seem to occur less frequently under contemporary forest practices, which they surmise is due to generally smaller harvest patch sizes and proportion of the watershed harvested, reduced road lengths, and presence of streamside vegetation buffers.

Research on peak flows in transient snow and snow zones has increased over the past decade and is relevant for Oregon, where many forests receive a large percentage of their annual precipitation as snow. Studies indicate that forest harvest may increase peak flows in these zones to a greater degree than in rain-dominated zones, especially during ROS events (e.g. Marks et al. 1998; Jones and Perkins 2010). Mechanisms for these increases may include greater snow accumulation and higher wind and advective rain energy available to melt snow in open areas than under forest canopies.

More recently, debate has emerged among researchers regarding experimental methods and relevant parameters for assessing the effects of forest harvest on flooding in snow-dominated systems. Green and Alila (2012) argue for a “paradigm shift” from generally accepted methods of comparing floods by equal meteorology or storm input (“chronological pairing”; CP) to a flood frequency distribution framework (“frequency pairing”; FP). They maintain that CP approaches in paired watersheds have yielded inaccurate results that underestimate forestry effects on large flood frequency, and that FP approaches are more appropriate. Green and Alila (2012) and related work (Kuraś et al. 2012, Schnorbus and Alila 2012) in a low elevation, snow dominated system in BC, Canada found that forest harvesting has substantially increased the frequency of the largest floods. These findings are attributed to increased net radiation associated with conversion from longwave-dominated (infrared) snowmelt beneath the canopy to shortwave-dominated (visible and ultraviolet light) snowmelt in harvested areas, amplified or mitigated by basin characteristics such as aspect distribution, elevation range, slope gradient, amount of alpine area, canopy closure, and drainage density

Alila and his colleagues acknowledge that their results run counter to prevailing wisdom in hydrological science – i.e. that the effect of forest harvesting must always decrease with an increase in flood event size - which is still being taught in textbooks today, including Chang (2012) cited above. Their work spurred debate regarding the use of CP and FP approaches (Alila and Green 2014a; Alila and Green 2014b; Bathhurst 2014; Birkinshaw 2014). Despite various critiques regarding the most appropriate research questions and methods, commenters generally suggested that both approaches provide meaningful information.

The effect of roads on peak streamflow is generally assumed to be strongly related to watershed size road density, and their degree of hydrologic connectivity. Roads on steep hillsides not only contribute overland runoff from compacted areas, but also intercept subsurface flow along cutslopes, especially where cutslopes intersect with bedrock (Sidle and Gomi 2017). Forest roads can contribute significantly to increases in peak flows, sometimes at levels equal to increases attributed to forest harvest (La Marche and Lettenmaier 2001). In large watersheds, roads usually constitute a smaller proportion of the land area and have relatively insignificant effects on peak flow (Gucinski et al. 2001) but this would depend on road density in the watershed.

At a study site in the HJ Andrews Experimental Forest, Burt et al. (2015) found that large contrasts between El Nino and La Nina climate patterns were a stronger driver of variability in streamflow response than differences in forest cover. Safeeq et al. (2015) concluded that over time, snowpack changes related to climate warming are likely to result in large increases in peak flow magnitudes in areas such as the Cascades and Blue Mountains. These and similar findings suggest that any effects that forestry activities have on peak flows may intertwine with climate in increasingly complex ways. At the same time if, as expected, the frequency and magnitude of floods in Oregon increase under climate change, public and agency interest in mitigating anthropogenic factors that contribute to peak flows may intensify.

3.2.5. Forest Management and Low Flows

Active forest management also has the potential to affect late summer low flows in streams and rivers. This is particularly relevant in Oregon, where there is essentially no precipitation for about three summer months each year. Thus, late summer is a particularly challenging time for water providers.

Definitions of low flows vary, from point in time flow rates, to number of days below a certain threshold, to recurrence intervals such as 7-, 10- or 30-day average low flow. Defining low flow as a percentage of change could be misleading, because a small change in low flow volume could be expressed as a large percentage (Stednick and Troendle 2016).

Baseline low flows in unmodified landscapes are controlled by natural factors such as geology, soils, and topography (Tague and Grant 2004) and then may be modified by changes in land use and climate. Until recently, most studies on the topic of forest management and low flows have focused on effects from just after harvest through re-establishment of the new stand. There is general consensus that low flows usually increase in the first years after forest harvesting (Buttle 2011). Most studies show that removal of forest vegetation increases low flows as a result of reduced evapotranspiration which increases soil moisture content (e.g. Stednick 2008; Surfleet and Skaugset 2013). Flows generally decline toward preharvest levels within a few years as transpiration rises in the regenerating stand. But a number of studies have also reported no significant change in low flows after harvest (e.g. Lin and Wei 2008). In the snow zone, low flows typically occur from late summer through the winter until spring snowmelt. Low flows are a normal part of the yearly water cycle. Low flows are maintained in the dry season through the release of water from groundwater storage and (or) surface water discharge from lakes, wetlands, and flow from channel banks (Pike and Sherer 2003).

Stednick and Troendle (2016) maintain that because current forest practices exclude many riparian areas from harvest, flow increases may not be as common today, and any such increases appear to return to pre-harvest conditions within a few years.

There is evidence that forest practices may decrease low flows under some conditions. In a study in coastal Oregon, Harr et al. (1982) found reduced low flows after harvesting and hypothesized that reduced fog interception and canopy drip could explain these results. Jones (2000) found similar results and suggested the same causal mechanism in 2 out of 10 basins examined. Hicks et al. (1991) identified decreases in low flows that could be attributed to changes in riparian vegetation from conifer to deciduous species; with the latter transpiring relatively more water per unit of leaf area.

More recently, some studies have focused on how regenerating forests affect summer low flows after the new stand is fully re-established. Perry and Jones (2017) showed that summer low flows were lower in young, vigorously growing stands compared to older adjacent stands in the western Oregon Cascades.

In three small watersheds in southern interior BC, Canada, Gronsdahl et al. (2019) found that summer flows were reduced starting about 20 years after the onset of forest harvesting which, they surmised was a result of regenerating forests transpiring more water than the mature forests they replaced. Moore et al. (2004) showed that younger, vigorous stands use more water than adjacent older stands, which they attributed primarily to tree age and, to a lesser degree, differences in sapwood basal area and finally species composition.

Segura et al. (2020) compared responses of daily streamflow in 1) harvested mature/old forest in 1966, 2) 43 to 53 and 48 to 58 yr-old industrial plantation forests in 2006–2009, and 3) these same plantation forests in 2010 and 2014, after harvesting using contemporary forest practices, including retention of a riparian buffer. The work was part of the long-term Alsea Watershed Study in the Oregon Coast Range (Stednick 2008). Segura et al. (2020) found that daily streamflow from a 40- to 53-yr-old Douglas-fir plantation was 25% lower on average, and 50% lower during summer, relative to the mature/old forest, and that these deficits lasted at least six months of each year. Contemporary forest practices (retaining riparian buffer strips in clearcuts) had a minimal effect on streamflow deficits. Two years after logging in 2014, summer streamflow deficits were similar to those prior to harvest (under 40- to 53-yr-old plantations).

Consistent with Perry and Jones (2017) and Gronsdahl (2019), Segura et al. (2020) attributed persistent streamflow deficits after logging to high evapotranspiration from rapidly regenerating vegetation, including planted commercial timber species. The authors note that their findings for summer streamflow deficits in young stands in the Oregon Coast Range were similar in magnitude to those detected in Douglas-fir plantations in the western Cascades (Perry and Jones 2017; Jones and Post 2004) indicating that Douglas-fir plantations of similar age have similar evapotranspiration rates relative to mature and old-growth forest reference stands in all of these locations. Overall, Segura et al. (2020) found that 40- to 50-yr rotations of Douglas-fir plantations can produce persistent, large summer low flow deficits, and that clearcutting with retention of riparian buffers increased daily streamflow slightly but flows did not return to conditions when the old/mature forests were intact. The authors suggest that additional work is needed to investigate how intensively managed forests and expected warmer, drier conditions in the future may influence summer low flows.

In addition to their salience for water providers and users, summer low flows in streams are also associated with reduced turnover and mixing in the water column, and with increased potential for harmful algal blooms in receiving lakes and reservoirs. This issue is discussed in more detail below.

3.3. Forest Chemicals, Nutrients and Water Quality

A variety of chemicals are used in forestry. Fertilizers are often applied in timber plantations to enhance tree growth. Pesticides are used to control unwanted organisms, including fungi, rodents, insects and plants. Herbicides are widely used after harvest to discourage colonization of clearcuts by deciduous species until newly planted conifer trees are established. Herbicides may also be applied near forest roads to control weeds or vegetation encroachment. Fungicides and insecticides may be used locally to control for fungi or insects that attack trees.

Some of these chemicals may pose a human health hazard if drinking water sources are contaminated during or after chemical applications. During application, chemicals may drift into waterways or other non-target areas. After application, chemicals or chemical residues may enter surface water or groundwaters through runoff and leaching (USDA-FS 2012). Plant nutrients, minerals, organic chemicals, fertilizers, and pesticides can attach to soil particles and be carried into streams with sediment (Chang 2012). Chemicals applied to roads can also enter streams by various pathways. The effects of these chemicals on water quality depend on how much chemical is applied, the distance of the road from a stream, and characteristics of weather and runoff events that move chemicals and sediments (Gucinski et al. 2000). Forest harvesting machinery requires petroleum fuel and lubricants, which can leak or spill and wind up in waterways.

While the use of chemicals in forestry is usually far lower than in other forms of agriculture, the risks of contamination of water bodies by silvicultural chemicals are well-recognized. In general, research indicates that these risks are usually low, provided that the chemical is carefully applied according to manufacturer directions (by properly licensed professional applicators in some cases) and that modern best management practices are followed. However, concerns about the risks that chemicals used in forestry may pose to human health are a persistent issue. Moreover, there are knowledge gaps regarding the persistence and long-term fate of chemicals after they are applied, and a lack of consensus in some quarters regarding the toxicity of certain chemicals (e.g. glyphosate) and long-term health effects in humans.

Aerial spraying of herbicides is a common practice in western Oregon industrial forests, and can be particularly contentious. (See, e.g., Bernstein et al. 2013; Glucklich 2018). Potential impacts on drinking water have led to efforts to eliminate aerial spraying through county-level ballot initiatives. This was successful in Lincoln County. The risks that such activities could degrade water quality in small, non-fish-bearing streams, and potential impacts on drinking water, were among the factors cited in NOAA-EPA's disapproval of Oregon's Coastal Non-Point Pollution Control Program (NOAA 2015).

3.3.1. Nitrogen and OtherForest Nutrients in Drinking Water

Nitrogen (N) is essential for all living things and is a key nutrient for trees and other plants. But excess N can also impair water quality and aquatic ecosystems, and is the most common water pollutant in the US. Nitrogen occurs naturally in soil in organic forms from decaying plant and animal residues, and also in inorganic forms derived from minerals. In the soil, bacteria convert N to nitrate, which is desirable because most N used by plants is absorbed as nitrate. But nitrate is also highly leachable and easily carried by water through the soil profile. In wet climates, dissolved nitrate often percolates below the plant root zone and travels into surface waters and groundwater.

Because of its importance as both a plant nutrient and pollutant, N dynamics after forest harvest (and forest soil N processes in general) have received extensive study. With some exceptions (e.g. Binkley et al. 2004; Binkley et al. 1999) research regarding N dynamics in forests tends to focus on management effects such as harvesting, site preparation, and fertilization on the productivity and sustainability of forest soils, rather than potential effects on drinking water. Temperate conifer forests usually conserve N and other nutrients. Soil N and N leaching often increase (usually temporarily) after timber harvesting as a result of reduced uptake from vegetation, or when N is released from decomposing slash or other plant material (e.g. Mupepele and Dormann 2016). Nitrogen export also often increases after wildfires (e.g. Rhoades et al. 2011; Smith et al. 2011) or prescribed fires. Nitrate is often a major portion of the total N exported from forests to surface waters. Processes (e.g. denitrification) in riparian and wetland areas and in streams can remove nitrate, but the significance of these processes in regulating nitrate flux varies widely. This variation suggests that some watersheds with increased N inputs (e.g. fertilization) will show increased nitrate-nitrogen outputs, while others have buffering capacity within soils, riparian areas, and stream channels to mitigate such a response (Stednick 2008).

To track its various sources and fates, total dissolved nitrogen (TDN) in water is often broken out into total organic nitrogen (TON) and total inorganic nitrogen (TIN). Undisturbed, mature stands may have large stores of organic N in the soil, forest floor litter layers, and old trees, and may utilize less N than vigorously growing younger stands that have lower ecosystem N stores after removal of slash from prior harvest. Forests and tree plantations in the Oregon Cascades and Coast Ranges established after a previous harvest and site preparation are often N-limited, with trees and other vegetation taking up all available N.

Vitousek and Reiners' (1975) model of N dynamics after forest harvest suggested that there is usually an initial flush of N export (because N uptake by vegetation is interrupted) that declines a few years after a new stand is initiated, and then N often becomes limiting again as the young trees grow. Leaching of N after harvest is often observed in temperate conifer forests (Mupepele and Dormann 2016; Jerabkova et al. 2011; Stednick 2008; Binkley et al. 2004; Antos et al. 2003; Feller et al. 2000; Martin and Harr 1989; Brown et al. 1973) with most studies finding that nitrate export declines to preharvest levels within 5-7 years or less, but confounding factors and exceptions are fairly common (Binkley et al. 2004). Variables that can affect the results of different studies include soil conditions (especially initial N availability) and land use history prior to harvest, site preparation methods and length of time after harvest, sampling strategy, weather and climate, topography, hydrology, and other factors.

Recent research illustrates the complexity of this topic. At their sites in southwestern Canada, Grand et al. (2014) found overall moderate increases in N, but a dramatic increase in N variability after harvest, with some sites showing extreme inorganic N values. Consistent with studies of local drainage water chemistry, Grand et al. (2014) concluded that conifer forests export significant N after harvesting, but that leaching would likely vary significantly from plot to plot. They suggest that this small to medium scale heterogeneity in N export has implications for nutrient leaching potential as well as researchers' ability to detect and predict harvest-induced changes.

In Oregon's west central Cascades, Cairns and Lajtha (2005) found that younger watersheds with stands 10 years or more in age still lost significantly more N than did watersheds with older forests. However, building on this work, Cairns et al. (2009) found that higher N concentrations in streams draining younger stands did not correlate well with N concentrations in soil solutions from those stands that were tested by lysimeter. They surmised that the differences identified in their 2005 study may have been a result of in-stream processing (nitrification) of N, in combination with processes in the dynamic riparian vegetation zone near the streams, and also perhaps the presence of minor amounts of N-fixing red alder, which has been shown to be a significant contributor to N exports in many western Oregon watersheds (Greathouse et al. 2014; Wise and Johnson 2011; Compton et al. 2003). If alder increases after harvest, this adds to the pool of N available for export, especially if alder is a component of the riparian vegetation (Pike et al. 2010; Stednick 2008).

Nitrogen export can increase seasonally with the onset of wet weather in the fall (e.g. Vanderbilt et al. 2003) or during periods of snowmelt. Swank (2000) indicates that knowledge gaps remain regarding nutrient concentration changes associated with storm runoff events, and that such information is most important where drinking water supplies are derived from forested headwaters with rapid streamflow responses to precipitation, e.g., watersheds with shallow soils, steep slopes, intense rainfall, and rapid snowmelt.

While Oregon forestlands are some of the most productive in the world, additions of N can often promote even more vigorous tree growth. Also, intensive forest management can reduce N stores in forest soil. For these reasons, N fertilizer is commonly applied on PNW commercial timberlands. Although the amount applied is a fraction of that used in conventional agriculture, some 125,000 acres of Oregon timberland are fertilized annually, about 5% of the state's total.. Nitrogen from forest fertilization can be a significant contributor to elevated N levels in some stream reaches in Oregon's western Cascades and Coast Ranges (Anderson 2002).

Phosphorus is less frequently applied to commercial timberlands, usually as a smaller component of N-based fertilizer blends. While the focus has primarily been on N, there is evidence that phosphorus may be limiting in a significant acreage of PNW Douglas fir forests, and suggestions that adding it to these stands may be beneficial from a timber management perspective (Mainwaring et al. 2014).

3.3.2. Forest nutrients and harmful algal blooms

Certain environmental conditions in freshwater bodies (usually involving excessive nutrients) can cause algae and similar microorganisms to grow explosively, causing algal blooms. Blooms that can harm human health or aquatic ecosystems are termed *harmful algal blooms* (HABs). Phosphorus and nitrogen both contribute to HABs in freshwater systems. In these systems, naturally occurring cyanobacteria (photosynthetic bacteria formerly called blue-green algae) typically cause the most frequent and severe HABs.

Some cyanobacterial HABs (termed cyanoHABs by USEPA) can produce potent toxins called cyanotoxins. These cyanotoxins can cause sickness and death in humans, pets and livestock who drink the water or otherwise come in contact with it. CyanoHABs can also create hypoxic (low oxygen) conditions in water bodies that can kill fish and other wildlife. CyanoHABs are a growing concern in the United States and worldwide as a result of their potential to broadly impact aquatic ecosystems, drinking water supplies, property and other economic values, and water-based recreational activities (USEPA 2019a).

A range of environmental factors can contribute to cyanoHABs. CyanoHABs are usually initiated by an excess of nutrients (especially phosphorus and nitrogen), compounded by warm, stagnant water, plentiful sunlight and sometimes invasive fish species. Sources of nutrient pollution include wastewater treatment plants, septic systems, fertilizers, agricultural runoff, urban and forestry runoff, and soil erosion. The exact combination of these factors that result in an individual bloom depends on conditions at that particular waterbody. Identifying the specific causes of a cyanoHAB usually requires detailed environmental analysis (Oregon DEQ 2019).

There have been cyanoHABs in a number of Oregon lakes, reservoirs and rivers, usually in late summer when inflows, water levels, and vertical mixing in the water column are lowest. Depending on local conditions the cyanoHABs vary in appearance from green, blue-green to reddish brown colored in the form of mats, foam, slicks or scum. If cyanotoxins over the USEPA national 10-day Health Advisory levels occur in tap water, people are at risk of health impacts including upset stomach, vomiting and diarrhea, and liver and kidney damage. Oregon has several documented cases of dogs dying and humans becoming ill from exposure to cyanotoxins from cyanoHABs. Conventional water treatment can usually remove cyanobacterial cells and low levels of cyanotoxins. However, providing safe drinking water can challenge providers during a severe bloom event, when drinking water sources contain high levels of these pollutants.

Conditions that cause cyanobacteria to produce cyanotoxins are complex and not fully understood. Some species that can produce toxins may not do so under all conditions. Both toxic and non-toxic varieties of most of the common toxin-producing cyanobacteria exist, and it is not possible to determine toxicity by how the bacteria look. Even when toxin-producing cyanobacteria are present, they may not always produce toxins. To further complicate matters, some species can produce multiple types and variants of cyanotoxins. Molecular testing can establish if the cyanobacteria carry the toxin-producing gene but quantitative cyanotoxin analysis is necessary to determine if they are actually producing the toxin (USEPA 2019a).

Conditions that favor longer and more severe cyanoHABs, such as warmer temperatures and increased nutrient inputs into waterways, are increasing. Reducing excess nitrogen and phosphorus in drinking water sources is important for long-term mitigation of the risks cyanoHABs pose. As of June 2019, there were no federal regulatory guidelines for cyanobacteria or their toxins in drinking water or recreational waters. However, the USEPA published drinking water health advisories (HA) with recommended 10-day limits for children and adults for the toxins microcystins and cylindrospermopsin in June 2015. In 2016, the cyanotoxins anatoxin-a, cylindrospermpsin, microcystins and saxitoxin were listed on the US

EPA Contaminant Candidate List, requiring monitoring for them between 2018 and 2020 using analytical methods developed by EPA and consensus organizations (USEPA 2019a, b).

The OHA is responsible for posting warnings and educating the public about cyanoHABs. Once a bloom is identified, the Oregon DEQ is responsible for investigating the causes, identifying pollution sources and producing a pollution reduction plan. The DEQ and the OHA are coordinating on the handling and analysis of HAB water samples (Oregon DEQ 2019). The DEQ also focuses on addressing nutrient, sediment and other HAB-related load allocations via its TMDL process and both the Oregon Agricultural Water Quality Management Area Plans and the Oregon FPA; which the Oregon Department of Agriculture (ODA) and the ODF use to meet water quality standards (Schaedel 2011).

At the statewide level, forestry-related nutrient runoff that contributes to cyanoHABs in Oregon probably ranks well below agricultural and urban runoff in significance. But contributions from forestry activities could be important or even dominant for particular blooms at the local level. Going forward, cyanoHABs are predicted to increase as climate change progresses. With concern about cyanoHABs growing and increased scrutiny from agencies charged with oversight of drinking water, science knowledge will also expand. This may trigger additional regulatory and agency action to monitor and control HAB-related nutrient runoff from all sources, including forestry.

3.4. Natural Organic Matter (NOM) and Disinfection Byproducts (DPB)

3.4.1. Natural Organic Matter

Natural organic matter (NOM) is ubiquitous in drinking water source waters. Defined as non-living organic molecules found in the environment in soil, sediments and water, NOM is a product of plant and animal tissue decay and plays a pivotal role in the carbon cycle (Nebbioso and Piccolo 2013). Living matter is mostly composed of well-defined molecules such as proteins, nucleic acids, lipids, sugars and cellulose. In contrast, due to interactions with soil and rocks that alter its plant and animal-derived precursors, NOM is mostly composed of molecules of unknown structure. Nevertheless NOM has been extensively researched because of its ecological and geochemical importance and influences on pollutant fate and transport in the environment. Natural organic matter in water includes *particulate organic matter* (POM) and *dissolved organic matter* (DOM), each defined by isolation using filtration, with POM being the fraction caught in the filter and DOM the fraction passing through with the water.

Prior to the early 1970's, treatment of NOM in raw water focused on aesthetic issues such as color. Then, research demonstrated that NOM is a precursor constituent in the formation of hazardous disinfection byproducts (DBPs). Today, NOM is the raw water constituent that most often influences the design, operation, and performance of water treatment systems. In addition to its role in the formation of DBPs, NOM can overwhelm activated carbon beds used in water treatment and reduce their ability to remove organic micropollutants. NOM also contributes significantly to the fouling of membranes in all membrane technologies used in water treatment, and can promote microbial fouling and regrowth in water distribution systems. Expanded understanding of linkages between NOM and DBPs continues to spur changes in drinking water treatment and regulation (O'Melia 2006).

3.4.2. Disinfection Byproducts (DBPs)

DBPs are an unintended outcome of using chemical disinfectants to kill harmful pathogens (e.g. cryptosporidium) in drinking source water. DBPs form when disinfectants react with NOM (usually decaying plant matter), or with bromide, and iodide, or various pollutants. People ingest DBPs primarily through drinking water, but also via inhalation and skin exposure while bathing and swimming. Documented health risks include bladder cancer, miscarriage, birth defects, liver and kidney damage and respiratory problems. Based on existing research, DBPs such as trihalomethanes (THMs) and haloacetic acids (HAAs) are regulated by the USEPA and in other countries. Research combining toxicology and chemistry has identified other "emerging" DBPs of concern. DBPs are produced by four major disinfectants used by water providers (chlorine, chloramines, ozone, and chlorine dioxide) and also by UV treatment with post-chlorination. Each disinfectant can produce its own suite of DBPs (Richardson and Postigo 2012).

A key consideration for drinking water providers is identifying sources of, and reducing the quantity of NOM that arrives at their raw water intakes. NOM from forest detritus is a major precursor to DBPs in drinking water sources (Bhardwaj 2006, Majidzadeh et al. 2019). Thus, forest management activities that influence the quantity and mobility of this source of NOM in source waters can influence the potential for DBPs to form during water treatment.

3.5. Best Management Practices (BMPs)

3.5.1. BMPs: history and overview

Recognition that forestry activities can affect soil and water quality emerged by the early 1900s. Organized research programs into the causes and mechanisms of these effects were initiated in the 1950s, as harvesting increased to accommodate the post-war housing boom, giving greater visibility to forestry activities and awareness of their impacts. Passage of the CWA in 1972, and additional provisions under the 1987 CWA reauthorization to address NPS pollution prompted further development, implementation, and refinement of specific forestry procedures intended to minimize soil and water quality impacts. These methods are termed *best management practices* (BMPs).

A number of different definitions for forestry BMPs appear in scientific and government agency literature. The most detailed definition, and that from which several others have been derived, may be this one from the US Forest Service Soil and Water Conservation Handbook, first published in 1988:

> *A practice or a combination of practices, that is determined by a State (or designated area-wide planning agency) after problem assessment, examination of alternative practices and appropriate public participation to be the most effective, practical (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals* (USDA-FS 1988).

The most common definition is probably this one from the CWA (40 CFR 130.2[Q]; Clean Water Act: Definitions), used for many years by the EPA and currently found in some archived EPA documents, and still in use by many state forestry agencies:

> *A practice or combination of practices considered by a State [or authorized Tribe] to be the most effective means (including technological, economic and institutional considerations) of preventing or reducing the amount of pollution by nonpoint sources to a level compatible with water quality goals.*

As of 2019, this is the definition used in the CWA and by EPA, and appears to have been in use since at least 2011:

> *Methods, measures or practices selected by an agency to meet its nonpoint source control needs. BMPs include but are not limited to structural and nonstructural controls and operation and maintenance procedures. BMPs can be applied before, during and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters.*

Other definitions include:

> *Practical control measures (including technological, economic, and institutional considerations) that have been demonstrated to effectively minimize water quality impacts* (Ice 2004).

> *Proactive and often voluntary practical methods or practices used during forest management to achieve goals related to water quality, silviculture, wildlife and biodiversity, aesthetics, and/or recreation* (Smallidge and Goff 1998).

For BMPs to be successful, they need to be 1) effective, and 2) consistently implemented. By most accounts, adoption and refinement of forestry BMPs over time have been effective in reducing (although not eliminating) water quality impacts resulting from timber harvesting, forest road building and use, and other forest management activities, as compared to these activities without the use of BMPs (Ice et al. 2010; Cristan et al. 2016). Reviews also suggest that implementation rates are generally high (Cristan et al. 2018). But the term "effective" is open to different interpretations, and there is still debate regarding differences in focus between implementation monitoring and effectiveness monitoring, the role of voluntary measures, and assessment of watershed-scale and cumulative impacts (e.g., McDonald and Coe 2014). For Oregon DEQ's purposes, "effective" BMPs are those that ensure that water quality standards are met and beneficial uses of water are protected and maintained.

Another set of issues involve "lag time", i.e. the time elapsed between when a particular BMP is implemented and the first measurable improvement in water quality in the target water body. If lag time is not accounted for, assessments and monitoring may underestimate BMP effectiveness (Meals et al. 2010). Conversely, lag time can also apply to the time elapsed between when forest management activities take place and detection of any resulting impacts, e.g. residence times for eroded sediment in hill slopes or stream channels, or chemicals in forest soils, before they are detected lower in the watershed.

The concepts that underlie most BMPs emerged from the experiences of working foresters combined with results from scientific studies conducted in the 1950s through the 1970s, mainly at USDA Forest Service experimental watersheds (Jackson 2014). BMPs are generally understood to be dynamic and always subject to improvement and development (USDA-FS 2012); development of effective BMPs and protection of water quality at the watershed scale has been an iterative process (McDonald and Coe 2014). Evolution of BMPs continues to this day as understanding of environmental impacts and the effectiveness of control measures advances, resulting for the most part in ongoing refinement of previously developed practices to further enhance effectiveness (Ice 2004, Cristan et al. 2016).

Cristan et al. (2016) reviewed the effectiveness of forestry BMPs, breaking out their summary by region. They compiled results from 31 studies conducted in the west coast region, mostly the Pacific Northwest and including 5 studies from western Oregon. Cristan et al. (2016) note that BMPs differ by state and by region, but typically include similar operational categories:

- Forest road construction and maintenance;
- Log landings (decks);
- Skid trails;
- Stream side management zones (SMZs);
- Stream crossings;
- Wetland protection and management;

- Timber harvesting;
- Site preparation; and
- Reforestation.

Cristan et al. (2016) submit the following conclusions regarding overall BMP effectiveness:

- BMPs can minimize erosion and sedimentation.
- Implementation rates and quality are critical to BMP effectiveness for reduction of erosion and sediment yield.
- BMP implementation can be enhanced with pre-operation planning and with the involvement of a registered professional forester.
- Increased logger training and landowner knowledge of forestry BMPs can help improve implementation.

Cristan et al. (2016) also submit specific BMP guideline conclusions:

- Forested Streamside Management Zones (SMZs) are effective in trapping sediment and reducing stream Total Suspended Solids (TSS) concentrations.
- Critically important BMP practices for forest roads include proper drainage structures, surfacing, erosion control of cut and fill slopes, traffic control, and closure.
- Sediment control structures applied to stream crossing approaches can significantly reduce runoff and sediment delivery.
- BMPs need to be applied during forest operations, not only as a closure measure.
- Effective skid trail closure practices can include installing waterbars and/or applying slash, mulch, or a combination of mulching and seeding.
- Improved stream crossings such as portable skidder bridges and temporary culverts can decrease TSS concentrations and turbidity compared to unimproved stream crossing structures.

Oregon's best management practices (BMPs) program is primarily regulatory, buttressed by some voluntary measures. The agencies responsible for BMPs policy development in Oregon are the Oregon Departments of Forestry, State Lands, Agriculture and Environmental Quality. Some examples of specific BMPs for timber harvesting, forest roads, and forest chemicals are discussed below.

3.5.2. Best management practices: timber harvesting

BMPs for timber harvesting related to water quality are, unsurprisingly, primarily focused on harvest activities in the vicinity of streams, wetlands or other water bodies. The basic approach is the designation of buffer strips along waterways where some or all forest vegetation is left in place in order to retain mobilized sediment or forest chemicals, to provide shade to maintain or lower stream water temperatures, and to serve as a source for woody debris to maintain certain stream functions. These areas are variously referred to as Streamside Management Zones (SMZ), Streamside Management Areas (SMA), Riparian Management Areas (RMA) and similar terms. Buffer widths may vary by landownership and management strategy, with the Federal government under the Northwest Forest Plan having the widest buffers, and with state forests management plans generally requiring the buffer widths exceeding those of private lands under the FPA (Boisjolie et al. 2017).

Typically, the width of stream buffers and the extent of forestry activities allowed within them vary according to the size of the stream, whether or not the stream contains fish species of concern, beneficial uses of the stream (including drinking water) and other factors. Smaller streams that do not support populations of salmonids and are not specifically designated as sources for drinking water often have no buffers. Harvesting can be precluded in SMZs, but in other cases allowances may be made for some limited harvesting activities, e.g. trees of a certain size class, or a certain percentage of trees. These variables have been, and continue to be researched extensively, and BMPs are updated and refined based on findings. It has been suggested that adopting a more flexible approach to buffer widths would allow site-specific tailoring to account for local conditions and management goals, but such an approach would be more complicated to administer and monitor for compliance (Richardson et al. 2012).

3.5.3 Best Management Practices: Forest Roads

Research consistently indicates that unpaved forest roads are a primary source of sediment entering streams and estuaries in forested watersheds (e.g. Reid and Dunne 1984; Amaranthus et al. 1985; Bilby 1985; Ketcheson and Megahan 1996; Luce and Black 1999; Carson and Younie 2003; Endicott 2008). Any forest road, no matter how carefully constructed, may contribute to soil erosion and potential stream sedimentation. Thus, a key tenet of road BMPs is minimizing road number and extent through careful planning (Daniels et al. 2004).

Forest road BMPs continue to be the subject of research. Over time BMPs have been developed and refined for forest road design, placement, construction practices, maintenance, temporary decommissioning, and complete decommissioning and reclamation (NCASI 2009). Three examples of significant areas of improvement are 1) actively routing runoff away from existing streams (as opposed routing it into existing channels, as was previous practice) ; 2) improving stream crossings by installation of bridges and/or culverts to keep road traffic from directly crossing stream channels, to minimize disturbance of the stream channel and maintain the integrity of stream structure and function; and, 3) upsizing culvert diameters to increase their flow capacity and reduce the likelihood that they will plug during storms, diverting water down roadways and/or causing fill failures.Other key tenets of forest road BMPs include maximizing the distance between roads and water bodies, and minimizing stream crossings, the total area of roads, and road grades (Megahan and King 2004).

Sugden (2018) provides a list of current BMPs for forest roads:

- Minimize the road density and area of road prism.
- Locate roads away from streams; i.e., outside Streamside Management Zones (SMZs) unless stream crossings are required.
- Install road drainage features at regular intervals to reduce erosion and divert overland flow from roads onto undisturbed hillslopes to promote water infiltration.
- Ensure road runoff is disconnected from streams toward filtration areas.

- Re-vegetation and ground cover establishment on disturbed areas near streams (cutslopes, fillslopes, and road ditches).
- Gravel surfacing on highly erodible soils or when wet weather use is required.
- Install supplemental filtration for suspended sediments where needed to prevent direct sediment delivery to streams. This includes slash windrows, silt fences, straw bales, etc.
- Install appropriately sized stream crossing structures that allow passage of flood flows, sediment, wood, and minimize disruptions to aquatic species movement.
- Manage/restrict seasonal road access to vehicles as needed to prevent rutting, and perform any necessary maintenance (grading) through time.
- Consider road closure or decommissioning of unneeded roads.

Edwards et al. (2016) synthesized information from almost 800 studies pertaining to BMPs for forest roads. Overall, they conclude that forest road BMPs *generally* result in some level of effectiveness, when "effectiveness" is simply defined as producing less sediment compared to not using the BMP. But they also submit some caveats, noting that despite the widespread assumption that road BMPs are well-supported by scientific research, rigorous quantitative studies of their effectiveness under different climatic, geologic and topographic conditions are limited. Sources cited as evidence of effectiveness include paired watershed studies with limited pre-treatment data and where BMPs are assessed together, making it difficult to assess which particular BMPs were most or least effective. They note that sediment measured at the mouth of a watershed does not account for hill-slope and in-channel storage of eroded sediment, and associated lag times for this sediment to reach the measurement point.

Edwards et al. (2016) also criticize statements that BMPs "minimize" sediment or pollution as misleading. They note that studies on effectiveness often find that some practices are more effective than others, or more effective in some situations than others in reducing sediment. Thus, all practices cannot be effective at "minimizing" sediment, the authors argue, so this term should be avoided because it gives a false impression about the degree of pollutant generation and transport that can be expected with BMP implementation. They note that BMPs cannot and are not intended to completely eliminate pollutants but rather to control them to levels compatible with environmental goals.

A growing area of active research and knowledge is BMPs for the decommissioning and/or removal of old forest roads. This topic is discussed in more detail in Chapter 5.

3.5.4. Best management practices: forest chemicals

Silvicultural chemical BMPs have been developed by many states for fertilizers used to improve crop tree growth and yield and pesticides used to protect trees from competing vegetation and insect pests. BMPs to protect water quality may include multiple layers of specificity based primarily on stream classification, hillside slope, soils, and presence of anadromous fish. As with sediment, the primary means of protecting streams from silvicultural chemicals is usually designation of a Streamside Management Zone (SMZ) that consists of the stream and an adjacent area of varying width where preparation and use of the chemicals is restricted.

Other BMPs for silvicultural chemicals can be categorized as follows:

- Following all product label instructions;
- Disposal of excess chemical and containers;
- How, when, and where to apply or not apply the chemical;
- Maintenance and service of application equipment;
- Prevention of direct application to surface water;
- Prevention of contamination by drift; and
- What to do in case of spills.

There is some overlap in these categories, i.e. the first is "follow label instructions", and most labels have instructions regarding disposal of containers, following recommended application rates and some of the other categories.

3.6. Implementing BMPs in Oregon

3.6.1. The Oregon Forest Practices Act: Overview and history

The Oregon Forest Practices Act (FPA) is the state's primary regulatory framework for addressing the environmental impacts of forest operations on state and private forest lands. The FPA sets standards for all commercial activities involving the establishment, management, or harvest of trees in the state. The 7-member Oregon Board of Forestry (BOF) has primary responsibility for interpreting the FPA and setting enforceable forest practice rules (FPR). Under ORS 468B.110(2), ORS 527.765, and ORS 527.770, the BOF establishes BMPs or other control measures by rule that, to the maximum extent practicable, will ensure attainment and maintenance of water quality standards.

The Oregon Environmental Quality Commission (EQC) is a five-member panel of Oregonians appointed by the governor for four-year terms to serve as Oregon DEQ's policy and rule-making board. The EQC has the authority to request rule changes to rules in the FPA, including strengthening protections for soil and waterways. If the EQC does not believe that the FPA rules will accomplish this result, it is authorized to petition the BOF for more protective rules.

When passed in 1971, the FPA was the first legislation of its kind in the USA. The FPA's first rules were implemented in 1972 and emphasized BMPs, which have since been revised repeatedly in response to emerging environmental concerns and science findings. Rules for pesticide use were strengthened in 1977 and again in 1996. In 1983, new rules focused on road and log landing parameters were added in response to heightened concern over road-related landslides in western Oregon. Rules to address landslide risks associated with harvesting in steep areas were more controversial, but were enacted two years later. The issue of linkages between forestry and landslides on steep slopes surfaced again 1996, one of the wettest years on record, when impacts from numerous slides in western Oregon increased public attention on the matter. In 1997, additional restrictions focused on public safety were placed on logging on steep slopes near roads or where people might be present (OFRI 2018a, Langridge 2011). Langridge (2011) describes scientific and policy debates associated with the 1997 rule changes and how the issue was framed primarily in terms of human safety while environmental protection was de-emphasized. As of June 2019, the FPA does not have any water quality-related landslide-prone area rules.

Over time, and continuing to this day, rules associated with riparian vegetation and buffer strips have arguably been the most contentious and evolved to the greatest degree. Riparian rules were modified in 1987 and again, more significantly, in 1994. Increasingly comprehensive and integrated science reports on topics such as the cumulative effects of forest practices (Beschta et al. 1995) and the status of salmonids and their habitat (Botkin et al. 1995), coupled with federal direction to mitigate dwindling salmon runs kept pressure on the BOF to further restrict harvesting in riparian and landslide-prone areas. But the studies also demonstrated the inherent complexity of these issues (Hairston-Strang et al. 2008).

In 2003, FPA rules were updated to require the use of higher quality rock or the suspension of log hauling during very wet weather, based on findings from an ODF monitoring study on wet season use of forest roads (Robben et al. 2003, ODF 2003).

The most recent FPA rule changes were in 2016 and 2017, and include 60' no-spray buffers for aerial herbicide use around homes and schools; a new salmon-steelhead-bull trout (SSBT) category of stream classification and wider riparian buffer strips that must be left around these streams, and additional protections for bald eagles (OFRI 2018b). The SSBT rules are the first change to FPA riparian rules since 1994.

3.6.2. FPA Administration and Compliance Monitoring

Oregon Department of Forestry (ODF) stewardship foresters administer FPA rules by working with forest landowners and operators to help them comply with FPA requirements. The Oregon Forest Resources Institute (OFRI) publishes a detailed manual to assist with planning and execution of timber harvests that comply with the FPA (Cloughesy and Woodward 2018). The ODF Forest Practices Monitoring Program reviews the effectiveness of the FPA and its rules. This program provides science information for adapting regulatory policies and management practices, education and training on FPA rules, assesses whether FPA rules and voluntary guidance sufficiently protect natural resources, and evaluates whether FPA rules are complied with and if voluntary measures are implemented. If FPA violations are identified, ODF starts with education and notices of correction before going into formal enforcement. Citations may be issued requiring cessation of the violating practice until agreement is reached on a mitigation strategy, and a legally-binding consent order signed (ODF 2019).

Since 2013, compliance monitoring has been conducted through the ODF Private Forests Monitoring Unit using contractors who audit FPA rules for road construction and maintenance, timber harvesting, some riparian management area measures, measures for small wetlands, and rules for operations near waters of the state. Audits through 2016 indicate generally high compliance rates, e.g. 97% overall compliance for 2016 (ODF 2018).

The FPA also requires forest landowners and operators to notify the ODF at least 15 days before they begin forest operations on any non-federal lands in Oregon. As defined in the FPA, forest operations include timber harvesting, road construction and reconstruction, site preparation, slash treatment, woody biomass removal, chemical application, land use changes, and certain non-commercial forest activities among other activities. In addition, permits are required for any operation using power driven machinery or fire. The Notification of Operations and Application for Permit (NO/AP) process is conducted through the ODF Private Forests, and Protection from Fire divisions. In 2014 the ODF updated the NO/AP process by implementing its Forest Activity Electronic Notification and Reporting System (FERNS), a web-based, centralized database of all forestry operations subject to ODF oversight. The FERNS application is integrated with the State's GIS system. Any interested person or party can subscribe to FERNS and then receive electronic notifications of pending forest operations in their area. Subscribers can also review and submit official comments about the forest operation work plans. Online subscriptions to FERNS are free of charge.

About 60% of Oregon's forest land is owned by the federal government, about 34% is privately owned (of which 22% is held by owners of ≥5,000 acres and 12% with <5,000 acres), 3% is owned by the state, 1% by local government, and 2% by tribes (OFRI 2017). Because the FPA and its rules apply only to non-federal forest land in Oregon, and to ensure that consistent minimum standards are met, the ODF, US Forest Service (USFS), and US Bureau of Land Management (BLM) entered into an MOU that Oregon's forest practice rules would be met or exceeded on federal land in Oregon (Hairston-Strang, Adams and Ice 2008). The Clean Water Act requires federal land managers to ensure that their practices will meet state water quality standards, laws, and rules (consistency review). In addition, state forests owned by the Department of State Lands and the BOF typically exceed FPA requirements through their management plans.

3.6.3. FPA Rules with Particular Relevance for Drinking Water

Arguably, the original FPA and most subsequent revisions to it were intended primarily to maintain or improve water quality. But certain sections are more directly to drinking water than other. Minimizing soil disturbance and erosion potential in order to protect water quality is fundamental to nearly all FPA rules for timber harvesting (Division 630). Other FPA sections that are relevant for drinking water include: Division 620 - Chemical and other petroleum product rules; Division 625 - Forest road construction and maintenance, and several divisions of the water protection rules, including Division 635 - Purpose goals, classification and riparian management areas, and Division 642 - Vegetation retention along streams, Division 645 - Riparian management areas and protection measures for significant wetlands, Division 650 - Riparian management areas and protection measures for lakes, Division 655 - Protection measures for "other wetlands," seeps and springs, and Division 660 - Stream channel changes. Provisions relating to riparian management areas, streamside buffers, and stream crossings for forest roads are often focused on maintaining conditions for coldwater fish species, but domestic water use is also explicitly referenced in the FPA stream classification system. More to the point, protection of water quality to benefit fish and maintaining safe drinking water sources for humans are not mutually exclusive goals- measures targeted toward either goal can and often do produce benefits for the other (Abell et al. 2019).

3.6.4. FPA Stream Classification System

The FPA protection goal for water quality is to ensure that, to the maximum extent practicable, non-point source discharges of pollutants resulting from forest operations do not impair the achievement and maintenance of the water quality standards (ODF 2018, p. 53).

The FPA uses a stream classification system to align the physical flow characteristics and beneficial uses of a water body to a set of appropriate protection measures. This classification system, and methods by which streams are classified, have been refined over time to reflect new science knowledge or policy imperatives. A *Type F* stream is any stream used seasonally or year-round by anadromous fish, game fish, or fish listed as threatened or endangered under the federal or state endangered species acts. Type F streams may also serve as community water sources. In July 2017, the Salmon, Steelhead and Bull Trout (*Type SSBT*) category was added along with modified stream buffer rules to better protect the cooler water quality temperatures needed by these fish. A *Type D* stream is any stream which does not contain fish (as defined above) and is located within a specified distance upstream of any domestic water intake for which an Oregon Water Resources Department (WRD) permit has been issued. All other streams are classified as *Type N*.

The distance upstream from an intake that Type D (domestic water use) classification applies varies according to whether the intake meets Oregon's definition for a community water supply: has 15 or more service connections used by year-round residents, or which regularly serves 25 or more year-round residents. If the intake meets one of these criteria, Type D classification initially applies to the length of stream that was designated Class I under the classification system in effect on April 22, 1994 (as shown on district water classification maps). If the intake is not for a community water supply (as defined above) Type D classification initially applies for the shortest of, 1) the distance from the intake upstream to the farthest upstream point of summer surface flow, 2) half the distance from the intake to the drainage boundary, or 3), 3000' upstream from the intake. Type D classification also applies to tributaries off the main channel as long as the above conditions hold.

Streams are further classified by size: *small* - average annual flow of 2 cubic feet per second (cfs) or less; *medium* – average annual flow greater than 2 but less than 10 cfs; or *large* - average annual flow of 10 cfs or greater. Criteria for establishing average annual flows are explained in Forest Practices Technical Note Number 1 (ODF 1994). Actual measurements of average annual flow may substitute for the calculated flows described in the technical note. Any stream with a drainage area less than 200 acres shall be assigned to the small stream category regardless of the flow calculated.

3.7. Forestry and Drinking Water Source Protection: Controversial or Unresolved Issues

3.7.1. Forest Roads and Sediment Input Into Streams

Among forestry-related sources of sediment inputs to streams, forest roads are recognized as a primary, if not *the* primary contributor. For this reason, runoff from forest roads continues to be a contentious issue relevant to forestry and drinking water source protection. This section summarizes a long-running legal dispute regarding forest roads in Oregon that eventually reached the Supreme Court, and where the matter stands today.

In 2006, the Northwest Environmental Defense Center (NEDC) sued the State of Oregon, the BOF, and several timber companies, claiming that forest roads, according to the CWA, are point sources of pollution and thus require an NPDES permit. This challenged decades of precedent under the EPA "Silvicultural Rule" which specifies which types of logging-related discharges EPA considers point sources and excludes forest roads (Boston 2012). The case centered on CWA language stating that *"the term* *'point source' means any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit…from which pollutants are or may be discharged"* and whether this should include logging roads that convey sediment through ditches and culverts into nearby streams.

Since the CWA exempts stormwater runoff, except when the runoff is "associated with industrial activity", the legal case also focused on whether or not transport of sawlogs constituted an "industrial activity". The EPA Industrial Stormwater Rule lists "logging" as an industry, and "transport of raw materials" as an "industrial activity", but also states that "associated with industrial activity" refers to "manufacturing, processing or raw materials storage areas at an industrial plant." Another area of dispute was whether the phrase term "at an industrial plant" referred to just "storage areas" or also "manufacturing" [or] "processing". [40 CFR § 122.26; Decker v. NEDC, 568 U.S. 597, particularly Justice Scalia's dissent in part, 616].

In 2007, the US District Court for Oregon ruled that forest roads do not require a NPDES permit. The NEDC appealed. In 2011, the US Ninth Circuit Court of Appeals overturned the district court, ruling that despite longstanding policy to the contrary, the EPA had misinterpreted clear language in the CWA when drafting its regulations and that roadside ditches on forest roads used to transport sawlogs *do* require an NPDES permit. In 2013, the US Supreme Court, in 7-1 decision, reversed the Ninth Circuit ruling. Deferring to EPA's interpretation of CWA language when the agency drafted pertinent regulations, the court cited previous case law that an agency's interpretation need not be the "best" one, only that it be "reasonable". Justice Scalia, in a detailed and strongly-worded dissent, sided with NEDC on the merits, arguing that deference was not warranted because EPA language in its Silvicultural Rule clearly conflicts with CWA definitions of "point source" in the statute and with EPA language elsewhere listing "logging" as an "industry" and detailing what is "associated with industrial activity" (Wasson 2016; Carr and Dively 2013; Boston 2012).

Complicating the Supreme Court case, the EPA amended the Industrial Stormwater Rule just before oral argument in 2012, clarifying that the NPDES requirement applied only to logging operations involving rock crushing, gravel washing, log sorting, and log storage facilities. In this amendment, EPA expressed its intention to evaluate other silvicultural discharges "under section 402(p)(6) of the Clean Water Act because the section allows for a broad range of flexible approaches that may be better suited to address the complexity of forest road ownership, management, and use." (Carr and Dively 2013).

In January 2014, Congress amended CWA Section 402(l), effectively prohibiting the requirement of NPDES permits for the discharge of runoff resulting from a range of silviculture activities, including surface drainage or road construction and maintenance. In December 2014, the Environmental Defense Center (EDC) and Natural Resources Defense Council petitioned the Ninth Circuit to compel EPA to respond, within six months, to a question remanded in a 2003 case (EDC v EPA) in which EDC contended that EPA arbitrarily failed to regulate discharges from forest roads under its 1999 Phase II stormwater rule. In the 2003 case, the court directed EPA to consider, in an appropriate proceeding, whether CWA Section 402(p)(6) requires it to regulate forest roads, then either accept or reject EDC's arguments using valid reasoning set forth in a way that permitted judicial review. Following a settlement agreement in August, 2015, the Ninth Circuit established a schedule requiring EPA to issue a final determination (Wasson 2016). On July 5, 2016, EPA issued a Notice of Decision not to regulate forest road discharges under Section 402(p)(6) of the CWA. (USEPA 2016).

In their rationale for this decision, the EPA acknowledged ongoing and significant water quality impacts attributable to forest road runoff, but argued that many states already have programs to address these impacts that are similar to options that would be available under CWA Section 402(p)(6). The USEPA stated that in general, progress continues to be made in strengthening these programs to reflect new technology and research, specifically tailored for locations in which they are implemented. Pointing to nationwide diversity in topography, climate, soil types, and intensity of timber operations and water quality impacts, the agency concluded that working with states to strengthen existing programs would be more effective than superimposing an additional federal regulatory layer over them. The EPA argued that despite the potential benefits of a more consistent and enforceable approach to mitigating forest road runoff, the complexity, cost and regulatory burden of a nationwide program could outweigh these benefits.

The EPA indicated that while it had decided not to regulate under CWA Section 402(p)(6), it would still actively work to facilitate ongoing improvements in approaches to mitigating water quality impacts from forest roads. Specifically, EPA said it plans to help strengthen existing programs by forming an ongoing dialogue with all relevant stakeholders (e.g., industry, environmental groups, academics, and government agencies at federal, state, tribal, and local levels) on program improvements, technical and policy issues, research results, state of the art technologies, success stories, and solutions to problem areas. The EPA envisions a forum where stakeholders can exchange information and expertise, primarily focused on specific problems and solutions to forest roads, such as existing/legacy roads or stream crossings as well as particularly effective forest road programs and best practices. As an example of a state-led effort to adopt newly developed methods for reducing sediment impacts, the EPA cited a stipulation in California's "Road Rules Package" (California Board of Forestry and Fire Protection 2014) that, where feasible, all forest roads must be hydrologically disconnected from streams (USEPA 2016).

In summary, as of January 2019 and in light of recent court and agency decisions, the focus of EPA efforts to regulate runoff from forest roads has shifted from consideration of a consistent, nationwide framework developed under CWA Section 402(p)(6) to one of actively working with stakeholders to strengthen state led NPS programs for forestry. But the EPA also noted that it has other tools in its toolbox with which to potentially address forest road discharges, such as Sections 303, 305 and 319 of the CWA.

3.7.2. "Legacy" Forest Roads

Nationwide, state-level monitoring shows generally high levels of compliance with forestry BMPs (Cristan et al. 2018), and regulatory frameworks for BMPs continue to be updated to reflect new knowledge and increase their effectiveness. But Oregon, like other western states, has a large number of so-called "legacy" forest roads that were constructed without the benefit of current BMPs to minimize their impacts. These substandard roads were sometimes poorly sited, e.g. along waterways, constructed with steep grades, or have poorly designed stream crossings. In other cases, problems stem primarily from a lack of maintenance.

Some unmaintained legacy road segments gradually revert back to vegetative cover, while others develop gully systems that become chronic sources of sediment. Legacy road segments that have been stabilized by revegetation can become sediment sources again if they are subsequently encompassed in new harvest units. And a significant number of legacy roads remain in use. "Problem" legacy road segments present a major challenge for managers, because they can generate many times the amount of sediment than roads constructed using modern BMPs (Ice and Shilling 2012) and resources are often scarce to repair and decommission them.

Recent interagency discussions in Oregon on the topic of forest roads and sediment resulted in adoption of the following terms for clarity:

- *Legacy road*: Built and abandoned prior to passage of the Oregon Forest Practices Act (FPA) and has not been used in the post-FPA. The FPA does not apply to these roads.
- *Old road*: Built using now obsolete techniques (e.g. built pre-FPA or pre-1984 construction standards) but in use post-FPA and therefore subject to FPA rules for water quality performance and vacating).

State and federal forest agencies are actively focused on the issue and working to inventory, then decommission or repair legacy roads. From 1997 to 2013, 2,668 miles of logging roads in Oregon public and private forests were closed or decommissioned (OFRI 2017). From 1995 through 2008, Oregon Department of Forestry (ODF) installed 63,055 cross drains on logging roads to route runoff away from streams (Mortenson 2011). But the number of such roads greatly exceeds the resources available to address them and legacy and old forest roads remain an urgent issue.

3.7.3. Non-point source pollution management in the coastal zone

The National Oceanic and Atmospheric Association (NOAA) administers the 1972 Coastal Zone Management Act (CZMA) to address the challenges of population growth and development in coastal areas by focusing on clean water and healthy ecosystems (NOAA 2018). The Coastal Zone Act Reauthorization Amendments of 1990 (CZARA) included a new Section 6217, "Protecting Coastal Waters", requiring each state with a coastal zone management program approved under section 306 of the CZMA to develop and implement a Coastal Nonpoint Pollution Control Program (Coastal Nonpoint Program) to prevent and control polluted runoff.

Section 6217 requires coastal states to implement nonpoint source pollution management measures developed by the EPA, which are organized into two tiers. If the first tier does not enable coastal waters to meet water quality standards and protect designated uses, then the state must implement a second tier of “additional” management measures targeted more specifically at restoring coastal waters to maintain water quality standards and to protect beneficial water uses designated by the state (NOAA and EPA 1993). For Oregon’s coastal waters, designated beneficial uses include “public domestic water supply” in all streams and rivers inland from the estuary or head of tidewater influence (Oregon Legislative Counsel Committee 2017; Oregon DEQ 2018b).

Section 6217 also requires each coastal state to submit their Coastal Nonpoint Program - which lays out how they intend to implement their pollution management measures - to the NOAA and EPA for approval. Failure to submit an approvable program can result in a state losing a portion of its Federal funding under section 306 of the CZMA and section 319 of the CWA.

As required, Oregon submitted its Coastal Nonpoint Program in 1995. In 1998, the NOAA and EPA conditionally approved Oregon’s program. Full approval was to be granted when the state met specific conditions, which required application of EPA management measures to address impacts stemming from a range of activities. In regards to forestry, the NOAA and EPA found that the following additional management measures were necessary to meet water quality standards and protect beneficial uses:

- Protect riparian areas for medium-sized and small fish-bearing (type "F") streams and non-fish-bearing (type "N") streams
- Address the impacts of forest roads, particularly so-called "legacy" roads
- Protect landslide-prone areas
- Ensure adequate stream buffers for the application of herbicides, particularly on non-fish bearing (type "N") streams

Oregon met nearly all conditions laid out in 1998 by modifying its program over time, but faced challenges in meeting conditions related to development, onsite sewage disposal, and forestry. In 2009, Northwest Environmental Advocates (NWEA) sued the NOAA and EPA alleging that despite Oregon's failure to submit an approvable program, the federal agencies had not disapproved the program or withheld grant funds as required and that as a consequence, Oregon had not improved its forest practices sufficiently to protect coastal water quality (NWEA 2010). In 2010, the Oregon US District Court directed the NOAA and EPA to either fully approve or disapprove Oregon's nonpoint program (NWEA 2010).

In 2015, the federal agencies found that the state had met the conditions for new development and onsite sewage disposal systems, but not for forestry. As a result, the agencies disapproved Oregon's Coastal Nonpoint Program, triggering a 30% holdback of Oregon's CZMA Section 306 funds and CWA Section 319 funds (NOAA 2015; House 2016). These funds will be withheld until the state's Coastal Nonpoint Program is approved. Programs affected are Oregon DEQ's nonpoint source reduction (NPS) program, and Oregon Coastal Management Program (OCMP) planning assistance grants for local governments in the coastal zone. As of fall 2018, the Oregon DEQ reported loss of $2,092,140 in CWA section 319 funding for its NPS program since 2015. As of spring, 2019 DLCD calculated the loss of CZMA Section 306 funds for the OCMP at $2.6 million since 2015 (Oregon DLCD 2019).

In April 2017, the Oregon Board of Forestry adopted a new set of rules to increase shade buffers on small and medium salmon, steelhead and bull trout fish-bearing streams, termed the SSBT rule. The SSBT rule covers about 1/3 of Oregon's small and medium Type F stream network according to ODF. The 80ft buffers adopted are narrower than what ODF/DEQ staff recommended (90ft) and the USEPA recommended (100ft) to ensure compliance with the Protecting Cold Water (PCW) criterion (OAR 340-041-0028). (Oregon DEQ 2018a). Some progress has been made in more clearly defining "legacy" and "old" forest roads and how these are treated under the FPA. But no action has been taken regarding additional management measures for landslide prone areas, or buffers on non-fish-bearing (Type "N") streams for protection from aerial herbicide application. Oregon has described the strategies in place (mostly voluntary rather than legally binding) to address these remaining additional management measures, and also pointed to Oregon's strong land use planning system, which has been effective in helping keep Oregon forest land in forest rather than other land uses. But to date the EPA and NOAA have not found these measures to be acceptable and have not approved Oregon's Coastal Nonpoint Program.

In summary, as result of factors including steep topography, high rainfall, and the relatively small size and close proximity of drinking water source watersheds to commercial timberlands, issues associated with forest management and protection of drinking water are likely to remain salient and a source of tension among stakeholders in Oregon's coastal zone.

3.7.4. Aerial herbicide spraying

Aerial spraying of herbicides to control understory and deciduous vegetation to promote conifer regeneration is common practice in western Oregon commercial forests, and is a perennial concern among some sectors of the public. In 2017, the BOF amended FPA rules regarding aerial spraying to require which operators leave a 60-foot unsprayed strip adjacent to inhabited dwellings or schools. But efforts to pass more restrictive county-level ordinances have continued, including Measure 21-177 that passed in Lincoln County in 2017. The Lincoln County measure is currently being challenged in court on grounds that it is pre-empted by state law (McDonald 2017), but even if the measure is overturned, the issue of aerial spraying is likely to remain active going forward.

3.7.5 Landslides

The effects of timber harvesting in steeper, landslide prone areas on landslide risk and impacts on water quality have been contentious issues in Oregon for several decades (Langridge 2011) and remain so today. Oregon's measures in the FPA to mitigate landslide risk were one facet of "additional management measures" that NOAA-EPA indicated were insufficiently addressed in the ongoing CZARA and Coastal Nonpoint Pollution program dispute. The FPA restricts harvesting in areas of landslide risk that could potentially pose a risk to lives and property, but these measures do not address water quality or aquatic habitat. The effects of landslides on sediment production and water quality are discussed in greater detail in Chapter 5.

3.8 References

Abell, R., K. Vigerstol, J. Higgins, S. Kang, N. Karres, B. Lehner, A. Sridhar, and E. Chapin. 2019. Freshwater biodiversity conservation through source water protection: Quantifying the potential and addressing the challenges. Aquatic Conservation: Marine and Freshwater Ecosystems 29(7): 1022-1038.

Alila, Y., P.K. Kuras, M. Schnorbus, and R. Hudson. 2009. Forests and floods: A new paradigm sheds light on age-old controversies. Water Resources Research 45: W08416.

Alila, Y. and K.C. Green. 2014a. Reply to comment by Birkinshaw on "A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments". Water Resources Research 50(3): 2769-2774.

Alila, Y. and K.C. Green. 2014b. Reply to comment by Bathurst on "A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments". Water Resources Research 50(3): 2759-2764.

Amaranthus, M.P., R.M. Rice, N.R. Barr, and R.R. Ziemer. 1985. Logging and forest roads related to increased debris slides in southwestern Oregon. Journal of Forestry 83(4): 229-233.

Amatya, D., T. Williams, L. Bren, and C. de Jong (eds). 2016. Forest Hydrology: Processes, Management and Assessment. CABI, Boca Raton, FL.

Anderson, C.W. 2002. Ecological effects on streams from forest fertilization—Literature review and conceptual framework for future study in the western Cascades. U.S. Department of the Interior, U.S. Geological Survey Water-Resources Investigations Report 01-4047. https://pubs.usgs.gov/wri/2001/4047/wri01-4047.pdf

Anderson, C.J. and B.G. Lockaby. 2011. Research gaps related to forest management and stream sediment in the United States. Environmental Management 47, 303-313.

Antos, J.A., C.B. Halpern, R.E. Miller, K. Cromack, and M.G. Halaj. 2003. Temporal and spatial changes in soil carbon and nitrogen after clearcutting and burning of an old-growth Douglas-fir forest. Res. Pap. PNW-RP-552. Portland, OR: USDA-Forest Service, Pacific Northwest Research Station. 19 pp.

Bates, C.G. and A.J. Henry. 1928. Second phase of streamflow experiment at Wagon Wheel Gap, Colorado. Mon. Weather Rev. Vol. 56(3): 79-85.

Bathurst, J.C. 2014. Comment on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments” by KC Green and Y. Alila. Water Resources Research 50(3): 2756-2758.

Benda, L. and T. Dunne. 1997. Stochastic forcing of sediment routing and storage in channel networks. Water Resources Research 33: 2865-2880.

Bernstein, L., L. Arkin, and R. Lindberg. 2013. Oregon’s Industrial Forests and Herbicide Use: A Case Study of Risk to People, Drinking Water and Salmon. Beyond Toxics, Eugene, OR.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14(6): 1011-1016.

Beschta, R.L., J.R. Boyle, C.C. Chambers, W.P. Gibson, and coauthors. 1995. Cumulative effects of forest practices in Oregon: literature and synthesis. Prepared for Oregon Dept. of Forestry. Oregon State Univ., Corvallis, OR.

Beschta, R.L., M.R. Pyles, A.E. Skaugset, and C.G. Surfleet, C.G. 2000. Peakflow responses to forest practices in the western cascades of Oregon, USA. Journal of Hydrology 233(1-4): 102-120.

Bhardwaj, V. 2006. Disinfection By-products. Journal of Environmental Health. Vol. 68, No. 10.

Bilby, R.E. 1985. Contributions of road surface sediment to a western Washington stream. Forest Science 31: 827-838.

Binkley, D., H. Burnham, and H.L. Allen. 1999. Water quality impacts of forest fertilization with nitrogen and phosphorus. Forest Ecology and Management 121(3): 191-213.

Binkley, D., G.G. Ice, J. Kaye, and C.A. Williams. 2004. Nitrogen and phosphorus concentrations in forest streams of the United States. J. Am. Water Resour. Assoc. 40: 1277–1291.

Birkinshaw, S.J. 2014. Comment on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments” by Kim C. Green and Younes Alila. Water Resources Research 50(3): 2765-2768.

Black PE. 1997. Watershed functions. Journal of the American Water Resources Association 33(1): 1–11.

Boisjolie, B.A., M.V. Santelmann, R.L. Flitcroft, and S.L. Duncan. 2017. Legal econtones: A comparative analysis of riparian policy protection in the Oregon Coast Range, U.S.A. Journal of Environmental Management 197: 206-220.

Boston, K., 2012. Impact of the ninth circuit court ruling (Northwest Environmental Defense Center v. Brown) regarding forest roads and the Clean Water Act. Journal of Forestry, 110(6), p.344.

Botkin, D., K. Cummins, T. Dunne, H. Reiger, and coauthors. 1995. Status and future of salmon of western Oregon and northern California. The Center for the Study of the Environment, Santa Barbara, CA. 300pp.

Brown, G.W., A.R. Gahler, and R.B. Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. Water Resources Research 9(5): 1450-1453.

Burns, J. 2019. Oregon aerial pesticide bills get hearings in Salem. Oregon Public Broadcasting. Accessed online 8/19/2019. https://www.opb.org/news/article/oregon-aerial-pesticide-bills-get-hearings-in-salem/.

Burt, T.P., N.J.K. Howden, J.J. McDonnell, J.A. Jones, and G.R. Hancock. 2015. Seeing the climate through the trees: observing climate and forestry impacts on streamflow using a 60-year record. Hydrological Processes 29: 473-480.

Burton, T.A. 1997. Effects of basin-scale timber harvest on water yield and peak streamflow. JAWRA Journal of the American Water Resources Association 33: 1187–1196. doi:10.1111/j.1752-1688.

Buttle, J.M., 2011. The effects of forest harvesting on forest hydrology and biogeochemistry. Pp. 659 – 677 *In* Levia, D.F., D. Carlyle-Moses, and T. Tanaka (eds). Forest Hydrology and Biogeochemistry: Synthesis of Past Research and Future Directions. Springer, Dordrecht, Netherlands. 733 pp.

Cairns M.A. and K. Lajtha. 2005. Effects of succession on nitrogen export in the west-central cascades, Oregon. Ecosystems 8:583–601.

Cairns, M.A., K. Lajtha, and P. Beedlow. 2009. Dissolved carbon and nitrogen losses from forests of the Oregon Cascades over a successional gradient. Plant and Soil 318(1): 185-196.

California Board of Forestry and Fire Protection. 2014. "Road Rules, 2013". Title 14, California Code of Regulations (14 CCR), Division 1.5, Chapter 4, Subchapters 1, 4, 5, 6, Articles 4, 5, 6, 8, and 12; Subchapter 7, Articles 2, 6.5, 6.8, 6.9, and 7. http://bofdata.fire.ca.gov/regulations/approved_regulations/2014_approved_regulations/roadrules2013.pdf accessed 1/30/2019.

Carr, C. and S. Diveley. 2013. Decker v. NEDC: The Supreme Court May Not Be the End of the (Unregulated) Forest Road. American Bar Association Trends 44:7.

Carson, B. and M. Younie. 2003. Managing coastal forest roads to mitigate surface erosion and sedimentation: an operational perspective. Watershed Manag Bull 7: 10-13.

Chang, M. 2012. Forest hydrology: an introduction to water and forests. Third edition. CRC Press.

Clifton, C.F., K.T. Day, C.H. Luce, G.E. Grant, M. Safeeq, J.E. Halofsky, and B.P. Staab. 2018. Effects of climate change on hydrology and water resources in the Blue Mountains, Oregon, USA. Climate Services 10: 9 – 19.

Cloughesy, M. and J. Woodward. 2018. Oregon's Forest Protection Laws: An Illustrated Manual. Revised third edition. Oregon Forest Resources Institute (OFRI). 317 SW Sixth Ave., Suite 400, Portland, OR 97204-1705 https://oregonforests.org/sites/default/files/2018-02/OFRI_IllusManual_full.pdf

Compton, J.A., M.R. Church, S.T. Larned, and W.E. Hogsett. 2003. Nitrogen export from forested watersheds in the Oregon Coast Range: The role of N2-fixing red alder. Ecosystems 6:773-785.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E. Schilling. 2016. Effectiveness of forestry best management practices in the United States: Literature review. Forest Ecology and Management 360: 133-151.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell. 2018. National status of state developed and implemented forestry best management practices for protecting water quality in the United States. Forest Ecology and Management 418: 73-84.

Croke, J.C. and P.B. Hairsine. 2006. Sediment delivery in managed forests: A review. Environmental Reviews 14(1): 59-87.

Daniels, B., D. McAvoy, M. Kuhns and R. Gropp. 2004. Managing Forests for Water Quality: Forest Roads. (Fact sheet.) Utah State University Cooperative Extension Service.

Dhakal, A. S. R.C. and Sidle. 2003. Long-term modelling of landslides for different forest management practices. Earth Surf. Process. Landforms 28: 853-868.

Edwards, P.J., F. Wood, and R.L. Quinlivan. 2016. Effectiveness of best management practices that have application to forest roads: A literature synthesis. Gen. Tech. Rep. NRS-163. Newtown Square, PA: USDA-Forest Service, Northern Research Station. 171 pp.

Endicott, D. 2008. National level assessment of water quality impairments related to forest roads and their prevention by best management practices. Report prepared for USEPA, Office of Water. Contract No. EP-C-05-066. Great Lakes Environmental Center. 250 pp.

Feller, M.C., R. Lehmann, and P. Olanski. 2000. Influence of forest harvesting intensity on nutrient leaching through soil in southwestern British Columbia. Journal of Sustainable Forestry 10(3/4): 241-248.

Fitzgerald, S.A. 2008. Successful reforestation: An overview. The Woodland Workbook. Revised October, 2008. EC 1498. Oregon State University Extension Service, Corvallis, OR. 8 pp.

Grant, G.E. and A.L. Wolff. 1991. Long-term patterns of sediment transport following timber harvest, Western Cascade Mountains, Oregon, USA. Pp. 31-40 *In*: Peters, N.E., Walling, D.E. (Eds.), Sediment and Stream Water Quality in a Changing Environment: Trends and Explanation. Proceedings of the Vienna IAHS symposium. International Association of Hydrological Sciences, Vienna, Austria. 374 pp.

Green, K.C. and Y. Alila, Y. 2012. A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments. Water Resources Research 48(10): W10503.

Glucklich, E. 2018. Rural Lane County residents fight aerial herbicide spraying. Eugene Register Guard, February 12, 2018. https://www.statesmanjournal.com/story/tech/science/environment/2018/02/12/rural-lane-county-residents-fight-aerial-herbicide-spraying/329746002/ accessed 1/24/2019.

Goetz, J. N., R.H. Guthrie, and A. Brenning. 2015. Forest harvesting is associated with increased landslide activity during an extreme rainstorm on Vancouver Island, Canada. Natural Hazards and Earth System Sciences 15(6): 1311.

Grand, S., R. Hudson, and L.M. Lavkulich. 2014. Effects of forest harvest on soil nutrients and labile ions in Podzols of southwestern Canada: Mean and dispersion effects. Catena 122: 18-26.

Grant, G.E., S.L. Lewis, F.J. Swanson, J.H. Cissel, and J.J. McDonnell. 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-science report for western Oregon and Washington. GTR-PNW 760. USDA-Forest Service, Pacific Northwest Research Station, Portland, OR, 76 pp.

Greathouse, E.A., J.E. Compton, and J. Van Sickle. 2014. Linking landscape characteristics and high stream nitrogen in the Oregon Coast range: Red alder complicates use of nutrient criteria. JAWRA Journal of the American Water Resources Association 50(6): 1383-1400.

Griffin, A. 1918. Influence of forests upon the melting of snow in the Cascade Range. Monthly Weather Review 46(7): 324-327.

Gronsdahl, S., R.D. Moore, J. Rosenfeld, R. McCleary, and R. Winkler. 2019. Effects of forestry on summertime low flows and physical fish habitat in snowmelt-dominant headwater catchments of the Pacific Northwest. Hydrological Processes 33(25): 3152-3168.

Groom, J. D., L.J. Madsen, J.E. Jones, and J.N. Giovanini. 2018. Informing changes to riparian forestry rules with a Bayesian hierarchical model. Forest Ecology and Management 419: 17-30.

Gucinski, H., M.J. Furniss, R.R. Ziemer, and M.H. Brookes. 2001. Forest roads: a synthesis of scientific information. General Technical Report PNW 509. USDA-Forest Service, Pacific Northwest Research Station, Portland, OR. 103 pp.

Guthrie, R.H. 2002. The effects of logging on frequency and distribution of landslides in three watersheds on Vancouver Island, British Columbia. Geomorphology 43(3-4): 273-292.

Guthrie, R.H., A. Hockin, L. Colquhoun, T. Nagy, S.G. Evans, and C. Ayles. 2010. An examination of controls on debris flow mobility: Evidence from coastal British Columbia. Geomorphology 114(4): 601-613.

Guzzetti F., A.C. Mondini, M. Cardinali, F. Fiorucci, M. Santangelo, and K-T. Chang. 2012. Landslide inventory maps: New tools for an old problem. Earth Sci Rev. 112:42–66.

Hairston-Strang, A.B., P.W. Adams, and G.G. Ice. 2008. The Oregon forest practices act and forest research. Pp. 95-113 *In* Stednick, J. (ed). Hydrological and Biological Responses to Forest Practices: The Alsea Watershed study. Springer, New York, NY. 316 pp.

Hanley, D.P, H.N. Chappell and E.H. Nadelhoffer. 2006. Fertilizing Douglas-fir forests: A guide for nonindustrial private forest landowners in western Washington. Publication EB 1800. Washington State University Extension. http://cru.cahe.wsu.edu/CEPublications/eb1800/eb1800.pdf accessed 7-31-19.

Harr, R.D., A. Levno, and R. Mersereau. 1982. Streamflow changes after logging 130-year-old Douglas fir in two small watersheds. Water Resources Research 18(3): 637-644.

Hassan, M.A., M. Church, T.E. Lisle, F. Brardinoni, L. Benda, and G.E. Grant. 2005. Sediment transport and channel morphology of small, forested streams. Journal of the American Water Resources Association 41: 853-876.

Hatten, J.A., C. Segura, K.D. Bladon, V.C. Hale, G.G. Ice, and J.D. Stednick. 2018. Effects of contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea Watershed Study Revisited. Forest Ecology & Management 408: 238-248.

Hicks, B.J., R.L. Beschta, and R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bulletin 27: 217-226.

Hill, B.T. (ed). 1999. Oregon watersheds: Many activities contribute to increased turbidity during large storms: Report to Congressional requesters. US General Accounting Office, GAO/RCED-98-220. 69 pp.

House, K. 2016. Oregon fined $1.2 M for failing to address coastal pollution. The Oregonian/OregonLive, March 11, 2016. Accessed online 10-22-2018: https://www.oregonlive.com/environment/index.ssf/2016/03/oregon_fined_12_m_for_failing.html

Ice, G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. Journal of Environmental Engineering 130(6): 684-689.

Ice, G. and E.B. Schilling. 2012. Assessing the effectiveness of contemporary forestry best management practices (BMPs): Focus on roads. Special Report 12-01. National Council for Air and Stream Improvement, Inc., Research Triangle Park, NC. 68p. Accessed 12-18-2018. http://www.ncasi.org/Downloads/Download.ashx?id=7589

Ice, G.G., E. Schilling, and J. Vowell. 2010. Trends for forestry best management practices implementation. Journal of Forestry 108(6): 267-273.

Imaizumi, F. and R.C. Sidle. 2012. Effect of forest harvesting on hydrogeomorphic processes in steep terrain of central Japan. Geomorphology 169: 109-122.

Jaboyedoff, M., T. Oppikofer, A. Abellán, M-H. Derron, A. Loye, R. Metzger, and A. Pedrazzini. 2012. Use of LIDAR in landslide investigations: a review. Nat Hazards 61:5-28.

Jackson, C.R. 2014. RESPONSE: Forestry Best Management Practices: A Mitigated Water Pollution Success Story. Journal of Forestry 112(1): 47–49.

Jakob, M. 2000. The impacts of logging on landslide activity at Clayoquot Sound, British Columbia. Catena 38(4): 279-300.

Jerabkova, L., C.E. Prescott, B.D. Titus, G.D. Hope, and M.B. Walters. 2011. A meta-analysis of the effects of clearcut and variable-retention harvesting on soil nitrogen fluxes in boreal and temperate forests. Canadian Journal of Forest Research 41(9): 1852-1870.

Johnson, M.G. and R.L. Beschta. 1980. Logging, infiltration capacity, and surface erodibility in western Oregon. Journal of Forestry 78(6): 334-337.

Jones, J.A., 2000. Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon. Water Resources Research 36(9): 2621-2642.

Jones, J.A. and G.E. Grant. 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research 32: 959-974.

Jones, J.A., D.A. Post. 2004. Seasonal and successional streamflow response to forest cutting and regrowth in the northwest and eastern United States. Water Resour. Res. 40(5): W05203.

Jones, J. A., and R. M. Perkins. 2010. Extreme flood sensitivity to snow and forest harvest, western Cascades, Oregon, United States. Water Resources Research, 46: W12512.

Ketcheson, G. and W. Megahan. 1996. Sediment production and downslope sediment transport from forest roads in granitic watersheds. U.S. For. Serv. Res. Pap. INT-RP-486.

Kuraś, P.K., Y. Alila, and M. Weiler. 2012. Forest harvesting effects on the magnitude and frequency of peak flows can increase with return period. Water Resources Research 48(1): W01544.

La Marche, J. L. and D.P. Lettenmaier. 2001. Effects of forest roads on flood flows in the Deschutes River, Washington. Earth Surface Processes and Landforms 26: 115–134.

Langridge, R. 2011. When do challengers succeed? Nongovernmental actors, administrative agencies, and legal change: Shifting rules for Oregon's private forests. Law & Social Inquiry 36(3): 662-693.

Lin, Y. and X. Wei. 2008. The impact of large-scale forest harvesting on hydrology in the Willow watershed of Central British Columbia. Journal of Hydrology 359(1-2): 141-149.

Litschert, S.E. and L.H. MacDonald. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. Forest Ecology and Management 259(2): 143-150.

Luce, C. H. and T. A. Black. 1999. Sediment production from forest roads in western Oregon. Water Resour. Res. 35(8): 2561–2570.

MacDonald, L. and D. Coe. 2014. RESPONSE: A suggested tiered monitoring strategy for maximizing best management practice effectiveness and protecting water quality. Journal of Forestry 112(1): 49-50.

Mainwaring, D.B., D.A. Maguire, and S.S. Perakis. 2014. Three-year growth response of young Douglas-fir to nitrogen, calcium, phosphorus, and blended fertilizers in Oregon and Washington. Forest Ecology and Management 327: 178-188.

Marks, D., J. Kimball, D. Tingey, and T. Link. 1998. The sensitivity of snowmelt processes to climate conditions and forest cover during rain-on-snow: A case study of the 1996 Pacific Northwest flood. Hydrological Processes 12(10-11): 1569-1587.

Martin C.W. and R.D. Harr. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. Canadian Journal of Forest Research 19: 35–43.

May, C. L. 2002. Debris flows through different forest age classes in the central Oregon Coast Range. JAWRA Journal of the American Water Resources Association 38: 1097–1113.

Majidzadeh, H., H. Chen, T.A. Coates, K-P Tsai, C.I. Olivares, C. Trettin, H. Uzun, T. Karanfil, and A.T. Chow. 2019. Long-term watershed management is an effective strategy to reduce organic matter export and disinfection by-product precursors in source water. Int. J. of Wildland Fire 28: 804-813.

McDonald, R. 2017. Court Challenge Filed Against Aerial Pesticide Spray Ban in Lincoln County. NPR/KLCC. Eugene, OR. Accessed 1/29/2019. https://www.klcc.org/post/court-challenge-filed-against-aerial-pesticide-spray-ban-lincoln-county.

McDonnell, J. J., J. Evaristo, K.D. Bladon, J. Buttle, I.F. Creed, S.F. Dymond, G. Grant, A. Iroume, C.R. Jackson, J.A. Jones, T. Maness, K.J. McGuire, D.F. Scott, C. Segura, R.C. Sidle, and C. Tague. 2018. Water sustainability and watershed storage. Nature Sustainability 1(8): 378-379.

McNamara, J.P., D. Tetzlaff, K. Bishop, C. Soulsby, M. Seyfried, N.E. Peters, B.T. Aulenbach, and R. Hooper. 2011. Storage as a metric of catchment comparison. Hydrological Processes 25(21): 3364-3371.

Meals, D.W., S.A. Dressing, and T.E. Davenport. 2010. Lag time in water quality response to best management practices: A review. Journal of Environmental Quality 39(1): 85-96.

Megahan, W.F. and J.G. King. 2004. Erosion, sedimentation, and cumulative effects in the northern Rocky Mountains. Pp. 201-222 *In*: Ice, G.G. and J.D. Stednick (eds). A Century of Forest and Wildland Watershed Lessons. Bethesda, MD: Society of American Foresters. 287 pp.

Michael, J.L. 2004. Best management practices for silvicultural chemicals and the science behind them. Water, Air and Soil Pollution: Focus 4(1): 95-117.

Miller, D.J. and K.M. Burnett. 2007. Effects of forest cover, topography, and sampling extent on the measured density of shallow, translational landslides. Water Resources Res. 43: W03433.

Montgomery, D.R., K.M. Schmidt, H.M. Greenberg, and W.E. Dietrich. 2000. Forest clearing and regional landsliding. Geology 28(4): 311-314.

Montgomery, D. R. and W.E. Dietrich. 2002. Runoff generation in a steep, soil-mantled landscape, Water Resources Research 38(9): WR000822.

Moore, G.W., B.J. Bond, J.A. Jones, N. Phillips, and F.C. Meinzer. 2004. Structural and compositional controls on transpiration in 40-and 450-year-old riparian forests in western Oregon, USA. Tree Physiology 24(5): 481-491.

Moore, R. and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. JAWRA Journal of the American Water Resources Association 41(4): 763-784.

Mortenson, E. 2011. Legacy of logging roads brings change to Oregon forests, and so do the courts. Oregon Live, September 27, 2011. Accessed 12-19-18. https://www.oregonlive.com/environment/index.ssf/2011/09/legacy_of_logging_roads_bring.html

Mote, P.W., S. Li, D.P. Lettenmaier, M. Xiao, and R. Engel. 2018. Dramatic declines in snowpack in the western US. NPJ Climate and Atmospheric Science 1(1): 2.

Mucken, A. and B. Bateman (Eds.). 2017. Oregon's 2017 Integrated Water Resources Strategy. Oregon Water Resources Department. Salem, OR. 190 pp. https://www.oregon.gov/owrd/WRDPublications1/2017_IWRS_Final.pdf accessed 5/6/2020.

Mupepele, A.C. and and C.F. Dormann. 2016. Influence of forest harvest on nitrate concentration in temperate streams—a meta-analysis. Forests 8(1), p.5.

NCASI (National Council for Air and Stream Improvement, Inc.). 2009. Compendium of forestry best management practices for controlling nonpoint source pollution in North America. Tech. Bull. No. 966. Research Triangle Park, NC. 230p. http://www.ncasi.org/Downloads/Download.ashx?id=10204 accessed 1/11/2018.

NOAA (National Oceanic and Atmospheric Administration Office for Coastal Management). 2015. NOAA/EPA finding that Oregon has not submitted a fully approvable coastal nonpoint program. https://coast.noaa.gov/czm/pollutioncontrol/media/ORCZARAdecision013015.pdf accessed 10/26/2018.

NOAA (National Oceanic and Atmospheric Administration Office for Coastal Management). 2018. National Ocean Service website. What is coastal management? https://oceanservice.noaa.gov/facts/czm.html accessed 10-26-2018.

NRC (National Research Council). 2008. Hydrologic Effects of a Changing Forest Landscape. National Research Council, Division on Earth and Life Studies, Water Science and Technology Board, Committee on Hydrologic Impacts of Forest Management. National Academies Press. 194 pp.

Neary, D.G., G.G. Ice, and C.R. Jackson. 2009. Linkages between forest soils and water quality and quantity. Forest Ecology and Management 258(10): 2269-2281.

Nebbioso, A. and A. Piccolo. 2013. Molecular characterization of dissolved organic matter (DOM): a critical review. Analytical and Bioanalytical Chemistry 405: 109-124.

Northwest Environmental Advocates (NWEA). 2010. NWEA v. Locke (CZARA Oregon Coastal Logging) Settlement Fact Sheet. https://www.northwestenvironmentaladvocates.org/newblog/download/nwea-v-locke-czara-oregon-coastal-logging-settlement-fact-sheet/ accessed 10/29/2018.

Northwest Environmental Advocates v. Locke et al. 2010. U.S. District Court for the District of Oregon. 2010. Final settlement agreement. Civil No. 09-0017-PK. https://www.oregon.gov/deq/FilterDocs/CZARA.pdf accessed 10/29/2018.

O'Melia, C.R. 2006. Fundamentals of particle stability. Pp. 317-362 *In*: Newcombe, G. and D. Dixonn (eds). Interface Science and Drinking Water Treatment: Theory and Applications. London, U.K.: Academic Press. 376 pp.

Oregon DEQ (Oregon Department of Environmental Quality). 2018a. Regulations; Division 41: Water Quality Standards. Accessed online 10-26-2018. https://www.oregon.gov/deq/Regulations/Pages/OARDiv41.aspx

Oregon DEQ (Oregon Department of Environmental Quality). 2018b. 2017 Oregon Nonpoint Source Pollution Program Annual Report. Submitted to USEPA Region 10, July 2018. https://www.oregon.gov/deq/FilterDocs/2017ORNonpointSourcePollutionReport.pdf accessed

Oregon DEQ (Oregon Department of Environmental Quality). 2019. Temporary rules for cyanotoxin monitoring frequently asked questions. June 29, 2018 https://www.oregon.gov/deq/FilterDocs/habfs.pdf accessed 5/6/2020.

ODF (Oregon Department of Forestry). 1994. Forest Practices Technical Note Number 1: Water Classification. https://www.oregon.gov/ODF/Documents/WorkingForests/WaterClassificationTechNote1.pdf accessed 5/6/2020.

ODF (Oregon Department of Forestry). 2003. Forest Practices Technical Note Number 9: Wet Weather Road Use. Version 1.0. Accessed 1/20/2019. https://www.oregon.gov/ODF/Documents/WorkingForests/WetWeatherRoadUseTechNote9.pdf accessed 5/6/2020.

ODF (Oregon Department of Forestry). 2009. Oregon Forest Facts. Accessed online 12-18-2018. https://www.oregon.gov/ODF/Documents/AboutODF/ForestryFactsFigures.pdf accessed 5/6/2020.

ODF (Oregon Department of Forestry). 2019. Forest practice act: Monitoring and enforcement. Accessed 1/20/2019: https://www.oregon.gov/odf/working/pages/fpa.aspx accessed 5/6/2020.

ODF (Oregon Department of Forestry). 2018. Forest Practices Implementation and Effectiveness Monitoring Update. https://www.oregon.gov/ODF/Board/Documents/BOF/20180307/BOFATTCH_20180307_06_01.pdf accessed 5/6/2020.

Oregon DLCD (Oregon Department of Land Conservation and Development). 2019. Oregon Coastal Management Program: Coastal Water Quality. Accessed online 6-4-2019: https://www.oregon.gov/lcd/OCMP/Pages/Water-Quality.aspx

OFRI (Oregon Forest Resources Institute). 2017. Oregon forest facts 2017-2018. https://www.oregonforests.org/sites/default/files/2017-05/OFRI_FactsFacts_1718_WEB_1.pdf accessed 5/6/2020.

OFRI (Oregon Forest Resources Institute). 2018a. Oregon Forest Practices Act: Adaptable and informed by sound science. (OFPA Timeline.) https://www.oregonforests.org/sites/default/files/2018-02/OFPA_Timeline_REV_2018.pdf accessed 5/6/2020.

OFRI (Oregon Forest Resources Institute). 2018b. Oregon's Forest Protection Laws are a-Changin'. https://oregonforests.org/blog/oregons-forest-protection-laws-are-changin accessed 5/6/2020.

Oregon Legislative Counsel Committee. 2017. Chapter 468B-Water Quality. https://www.oregonlegislature.gov/bills_laws/ors/ors468b.html accessed 5/6/2020.

Orem, B. 2019. Nitrates and nitrites in drinking water groundwater and surface waters. Water Research Center, BF Environmental Consultants, Inc. Dallas, PA. https://www.water-research.net/index.php/nitrate accessed 5/6/2020.

Perkowski, M. 2018. Workshop examines aerial spraying. Capital Press, Jun 26, 2018. https://www.capitalpress.com/state/oregon/workshop-examines-aerial-spraying/article_202c69e0-4f8f-503f-acec-5eb346ec67b2.html accessed 5/6/2020.

Perry, T.D. and J.A. Jones. 2017. Summer streamflow deficits from regenerating Douglas-fir forest in the Pacific Northwest, USA. Ecohydrology 10(2), p.e1790.

Pike, R. and R. Scherer. 2003. Overview of the potential effects of forest management on low flows in snowmelt-dominated hydrologic regimes. BC Journal of Ecosystems and Management 3(1): Art8. http://www.forrex.org/jem/2003/vol3/no1/art8.pdf accessed 5/6/2020.

Pike, R.G., M.C. Feller, J.D. Stednick, K.J. Rieberger, and M. Carver. 2010. Water quality and forest management. Pp. 401-440 *In*: R.G. Pike, T.E. Redding, R.D. Moore, R.D. Winker, and K.D. Bladon (eds). Compendium of Forest Hydrology and Geomorphology in British Columbia. Land Management Handbook 66. BC Ministry of Forests and Range, Forest Science Program, Victoria, BC, and FORREX Forum for Research and Extension in Natural Resources, Kamloops, BC. 2 Vols. 902 pp. https://www.for.gov.bc.ca/hfd/pubs/Docs/Lmh/Lmh66.htm accessed 5/6/2020.

Preti, F. 2013. Forest protection and protection forest: tree root degradation over hydrological shallow landslides triggering. Ecological Engineering 61: 633-645.

Reeves, G.H., B.R. Pickard, and K.N. Johnson. 2016. An initial evaluation of potential options for managing riparian reserves of the Aquatic Conservation Strategy of the Northwest Forest Plan. Gen. Tech. Rep. PNW-GTR-937. Portland, OR: USDA-Forest Service, Pacific Northwest Research Station. 97 p9.

Reeves, G.H., D.H. Olson, S.M. Wondzell, P.A. Bisson, S. Gordon, S.A. Miller, J.W. Long,. and M.J. Furniss. 2018. The aquatic conservation strategy of the northwest forest plan—A review of the relevant science after 23 years. Pp. 461-624 *In:* Spies, T.A., P.A. Stine, R. Gravenmier, J.W. Long, and M.J. Reilly (tech. cords). 2018. Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area. Gen. Tech. Rep. PNW-GTR-966. Portland, OR: USDA-Forest Service, Pacific Northwest Research Station. 1020 pp.

Reid, L. M. and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20: 1753-1761.

Reid, L.M. and J. Lewis. 2007. Rates and implications of rainfall interception in a coastal redwood forest. Pp. 107-118 *In:* Standiford, R.B., G.A. Giusti, Y. Valachovic, W.J. Zielinski, M.J. Furniss (tech eds). 2007. Proceedings of the Redwood Region Forest Science Symposium: What does the Future Hold? Gen. Tech. Rep. PSW-GTR-194. Albany, CA: USDA-Forest Service, Pacific Southwest Research Station. 553 pp.

Rhoades, C.C., D. Entwistle, and D. Butler. 2011. The influence of wildfire extent and severity on streamwater chemistry, sediment and temperature following the Hayman Fire, Colorado. International Journal of Wildland Fire 20(3): 430-442.

Richardson, J.S., R.J. Naiman, and P.A. Bisson. 2012. How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? Freshwater Science 31(1): 232-238.

Richardson, S. D. and C. Postigo. 2012. Drinking water disinfection by-products. Pp. 93-137 *In:* Barceló, D. (ed). Emerging Organic Contaminants and Human Health. The Handbook of Environmental Chemistry, Vol. 20. New York, NY: Springer. 466 pp.

Robben, J., K. Mills and L. Dent. 2003. Wet Season Road Use Monitoring Project: Final Report. Oregon Department of Forestry, Salem, OR. 34 pp. https://digital.osl.state.or.us/islandora/object/osl:19663 accessed 5/6/2020.

Robison, E.G., K.A. Mills, J. Paul, L. Dent, and A. Skaugset. 1999. Storm Impacts and Landslides of 1996. Forest Practices Technical Report 4. Oregon Department of Forestry, Salem, OR 157 pp. https://www.waterboards.ca.gov/water_issues/programs/tmdl/records/region_1/2003/ref1785.pdf accessed 5/6/2020.

Roering, J.J., K.M. Schmidt, J.D. Stock, W.E. Dietrich, and D.R. Montgomery. 2003. Shallow landsliding, root reinforcement, and the spatial distribution of trees in the Oregon Coast Range. Canadian Geotechnical Journal 40(2): pp.237-253.

Rothstein, D.E. 2005. Nitrogen management in a Fraser fir (Abies fraseri [Pursh] Poir.) Christmas tree plantation: effects of fertilization on tree performance and nitrogen leaching. Forest Science 51(2): 175-184.

Safeeq, M., G.E. Grant, S.L. Lewis, and B. Staab. 2015. Predicting landscape sensitivity to present and future floods in the Pacific Northwest, USA. Hydrological Processes 29: 5337-5353.

Sakals, M.E. and R.C. Sidle. 2004. A spatial and temporal model of root cohesion in forest soils. Canadian Journal of Forest Research 34(4): 950-958.

Sayama, T., J.J. McDonnell, A. Dhakal, and K. Sullivan. 2011. How much water can a watershed store? Hydrological Processes 25(25): 3899-3908.

Scatena, F.N. 2000. Drinking water quality. Pp. 7-25 *In:* Dissmeyer, G.E. (ed.) 2000. Drinking Water from Forests and Grasslands: A Synthesis of the Scientific Literature. General Technical Report SRS-39, USDA-Forest Service, Southern Research Station, Asheville, North Carolina. 246 pp. https://www.srs.fs.usda.gov/pubs/gtr/gtr_srs039/index.htm accessed 5/6/2020.

Schaedel, A. 2011. Oregon DEQ harmful algal bloom (HAB) strategy. Oregon Department of Environmental Quality, Portland, OR. 89 pp. https://www.oregon.gov/deq/FilterDocs/HABstrategy.pdf accessed 5/6/2020.

Schmidt, K.M., J.J. Roering, J.D. Stock, W.E. Dietrich, D.R. Montgomery, and T. Schaub. 2001. The variability of root cohesion as an influence on shallow landslide susceptibility in the Oregon Coast Range. Canadian Geotechnical Journal 38(5): 995-1024.

Schnorbus, M. and Y. Alila. 2013. Peak flow regime changes following forest harvesting in a snow-dominated basin: Effects of harvest area, elevation, and channel connectivity. Water Resources Research 49(1): 517-535.

Sedell, J., M. Sharpe, D.D. Apple, M. Copenhagen, and M. Furniss. 2000. Water and the Forest Service. Fact Sheet FS-660. USDA-Forest Service, Washington, D.C. 36 pp. https://www.fs.fed.us/biology/nsaec/assets/waterandtheforestservice.pdf accessed 5/6/2020.

Segura, C., K.D. Bladon, J.A. Hatten, J.A. Jones, V.C. Hale, and G.G. Ice. 2020. Long-term effects of forest harvesting on summer low flow deficits in the Coast Range of Oregon. Journal of Hydrology 585: 124749.

Sidle, R.C. and H. Ochiai. 2006. Landslides: Processes, prediction, and land use. Water Resources Monograph 18. Washington, D.C.: American Geophysical Union. 312 pp.

Sidle, R.C. and Bogaard, T.A. 2016. Dynamic earth system and ecological controls of rainfall-initiated landslides. Earth-Science Reviews 159: 275-291.

Sidle, R.C. and T. Gomi. 2017. Hydrologic processes in forest headwater catchments: Implications for policy and management. Pp. 94-105 *In:* R.Tognetti, G. Scarascia Mugnozza and T. Hofer (eds). Mountain Watersheds and Ecosystem Services: Balancing Multiple Demands of Forest Management in Head-watersheds. EFI Technical Report 101. European Forest Institute, Joensuu, Finland. 191 pp. https://www.efi.int/sites/default/files/files/publication-bank/2018/tr_101.pdf accessed 5/6/2020.

Siler, N., Proistosescu, C. and Po-Chedley, S. 2018. Natural variability has slowed the decline in western-U.S. snowpack since the 1980s. Geophysical Research Letters 46(1):346-355.

Simončič, T., Spies, T.A., Deal, R.L. and Bončina, A. 2015. A conceptual framework for characterizing forest areas with high societal values: experiences from the Pacific Northwest of USA and Central Europe. Environmental Management 56(1): 127-143.

Smallidge, P. and G. Goff. 1998. Forestry Best Management Practices. Cornell University. http://www2.dnr.cornell.edu/ext/info/pubs/Harvesting/BMPs.htm accessed 5/6/2020.

Smith, H., G. Sheridan, P. Lane, P. Nyman, and S. Haydon. 2011. Wildfire effects on water quality in forest catchments: A review with implications for water supply. Journal of Hydrology 396(1-2): 170-192.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology 176(1-4): 79-95.

Stednick, J.D. 2008. Hydrological and biological responses to forest practices: The Alsea Watershed Study. Berlin: Springer-Verlag. 316 pp.

Startsev, A.D. and D.H. McNabb. 2000. Effects of skidding on forest soil infiltration in west-central Alberta. Canadian Journal of Soil Science 80(4): 617-624.

Stednick, J.D. and C.A. Troendle. 2016. Hydrological effects of forest management. Pp. 192-203 *In:* Amatya, D., T. Williams, L. Bren, and C. de Jong (eds). Forest Hydrology: Processes, Management and Assessment. CABI, Boca Raton, FL. 294 pp.

Sugden, B.D., Woods, S.W., 2007. Sediment production from forest roads in western Montana. Journal of the American Water Resources Association 43: 193-206.

Surfleet, C.G. amd A.E. Skaugset. 2013. The effect of timber harvest on summer low flows, Hinkle Creek, Oregon. Western Journal of Applied Forestry 28: 13-21.

Swank, W. 2000. Forest succession. Pp. In Dissmeyer, G.E. (ed.) 2000. Drinking water from forests and grasslands: A synthesis of the scientific literature. General Technical Report SRS-39. USDA-Forest Service, Southern Research Station, Asheville, North Carolina. 246 pp. https://www.srs.fs.usda.gov/pubs/gtr/gtr_srs039/index.htm accessed 5/6/2020.

Swanson, L. 2017. Heads up! Oregon Department of Forestry provides information about upcoming aerial spraying in area. Tillamook County Pioneer, August 14, 2017. Accessed online 8/16/2019. https://www.tillamookcountypioneer.net/heads-up-oregon-department-of-forestry-provides-information-about-upcoming-aerial-spraying-in-area/ accessed 5/6/2020.

Tague, C. and Grant, G.E., 2004. A geological framework for interpreting the low-flow regimes of Cascade streams, Willamette River Basin, Oregon. Water Resources Research 40(4): W04303.

Thomas, R.B. and W.F. Megahan. 1998. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon: A second opinion. Water Resources Research 34: 3393-3404.

Thomas, J.W., J.F. Franklin, J. Gordon, J., and K.N. Johnson, K.N. 2006. The Northwest Forest Plan: origins, components, implementation experience, and suggestions for change. Conservation Biology 20(2): 277-287.

Turner, T.R., S.D. Duke, B.R. Fransen, M.L. Reiter, A.J. Kroll, J.W. Ward, J.L. Bach, T.E. Justice, and R.E. Bilby. 2010. Landslide densities associated with rainfall, stand age, and topography on forested landscapes, southwestern Washington, USA. Forest Ecology and Management 259(12): 2233-2247.

USDA-FS (United States Department of Agriculture, Forest Service). 1988. Soil and water conservation practices handbook. Forest Service Handbook 2509.22. https://www.fs.fed.us/cgi-bin/Directives/get_dirs/fsh?2509.22 accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service). 2007. Forest Service Manual, Series 2000 - National Forest Resource Management; FSM 2500 – Watershed and air management; Chapter 2540 – Water uses and development; Section 2542 – Municipal supply watersheds. https://www.fs.fed.us/im/directives/fsm/2500/2540.rtf accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service). 2010a. Forest Service Manual 7400: Public health and pollution control facilities, Chapter 7420, drinking water. https://www.fs.fed.us/cgi-bin/Directives/get_dirs/fsm?7400 accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service) 2010b. Forest Service Manual 7400: Public health and pollution control facilities, Chapter 7420, drinking water. R6/PNW Supplement No.: 7400-2010-1. Accessed online 10-30-2018: https://www.fs.fed.us/im/directives/field/r6pnw/fsm/7400/7420.doc accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service). 2012. . Volume 1: National Core BMP Technical Guide. FS-990a. https://www.fs.fed.us/biology/resources/pubs/watershed/FS_National_Core_BMPs_April2012.pdf accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service). 2014. Memorandum of understanding between state of Oregon Department of Environmental Quality and the USDA Forest Service, Pacific Northwest Region. FS-1500-15. OMB 0596-0217. https://www.oregon.gov/deq/FilterDocs/FSdeqWQmou2.pdf accessed 5/6/2020.

U USDA-FS (United States Department of Agriculture, Forest Service). 2018a. Watershed Condition Framework: 2011–2017. FS-1114.

https://www.fs.usda.gov/sites/default/files/fs_media/fs_document/5081_wcf_accomplishment_report_1.pdf accessed 5/6/2020.

USDA-FS (United States Department of Agriculture, Forest Service). 2018b. United States Department of Agriculture Wallowa-Whitman National Forest Land Management Plan. Accessed online 10-29-2018. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd584609.pdf

USDA and USDI (U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management). 1994. Record of decision for amendments for Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. 78 pp. https://www.fs.fed.us/r6/reo/nwfp/documents/reports/newroda.pdf accessed 5/6/2020.

USEPA (United States Environmental Protection Agency). 2005. National Management Measures to Control Nonpoint Source Pollution from Forestry. Office of Water, United State Environmental Protection Agency, Washington, D.D. 276 pp. https://www.epa.gov/sites/production/files/2015-10/documents/2005_05_09_nps_forestrymgmt_guidance.pdf accessed 5/6/2020.

USEPA (United States Environmental Protection Agency). 2016. Decision Not To Regulate Forest Road Discharges under the Clean Water Act; Notice of Decision. Federal Register 81(128): 43492-43510. https://www.govinfo.gov/content/pkg/FR-2016-07-05/pdf/2016-15844.pdf accessed 5/6/2020.

USEPA (United States Environmental Protection Agency). 2017. Assessment and Total Maximum Daily Load Tracking and Implementation System (ATTAINS). Information for Oregon accessed online 12-17-2018. https://ofmpub.epa.gov/waters10/attains_nation_cy.control accessed 5/6/2020.

USEPA (United States Environmental Protection Agency). 2019a. Cyanobacteria and cyanotoxins: Information for drinking water systems. EPA-810F11001. Office of Water, U.S. Environmental Protection Agency, Washington, D.C. 12 pp. https://www.epa.gov/sites/production/files/2019-07/documents/cyanobacteria_and_cyanotoxins_fact_sheet_for_pws_final_06282019.pdf.pdf accessed 5/6/2020.

USEPA (United States Environmental Protection Agency). 2019b. Cyanotoxins and the Safe Drinking Water Act: Drinking Water Protection Act, Contaminant Candidate List and the Unregulated Contaminant Monitoring Rule. https://www.epa.gov/cyanohabs/cyanotoxins-and-safe-drinking-water-act-drinking-water-protection-act-contaminant accessed 5/6/2020.

Vanderbilt K.L., K. Lajtha, and F.J. Swanson FJ. 2003. Biogeochemistry of unpolluted forested watersheds in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen fluxes. Biogeochemistry 62:87–117.

Vitousek, P.M. and W.A. Reiners. 1975. Ecosystem succession and nutrient retention: a hypothesis. BioScience 25:376–381.

Wasson, G. 2016. White Paper on CWA Regulation of Stormwater Discharges from Logging Roads. Brunini, Grantham, Grower & Hewes, PLLC. Jackson, MS. https://www.americanbar.org/content/dam/aba/administrative/environment_energy_resources/resources/wotus/forestroads/201707wasson.pdf accessed 5/6/2020.

Wemple, B.C. and J.A. Jones. 2003. Runoff production on forest roads in a steep, mountain catchment. Water Resources Research 39(8): doi:10.1029/2002WR001744.

Wemple, B. C., F.J. Swanson, and J.A. Jones. 2001. Forest roads and geomorphic process interactions, Cascade Range, Oregon. Earth Surface Processes and Landforms 26: 191–204.

Whicker, J.J., J.E. Pinder, and D.D. Breshears. 2006. Increased wind erosion from forest wildfire: implications for contaminant-related risks. Journal of Environmental Quality 35(2): 468-478.

Williams, T. 2016. Forest runoff processes. Pp. 17-31 *In:* Amatya, D., T. Williams, L. Bren, and C. de Jong (eds). 2016. Forest Hydrology: Processes, Management and Assessment. CABI, Boca Raton, FL 294 pp.

Winkler, R., D. Spittlehouse, and S. Boon. 2017. Streamflow response to clear-cut logging on British Columbia's Okanagan Plateau. Ecohydrology 10(2): e1836.

Wise, D.R. and H.M. Johnson. 2011. Surface-Water Nutrient Conditions and Sources in the United States Pacific Northwest. JAWRA Journal of the American Water Resources Association 47(5): 1110-1135.

Zhang, M. and X. Wei. 2014. Alteration of flow regimes caused by large-scale forest disturbance: a case study from a large watershed in the interior of British Columbia, Canada. Ecohydrology 7(2): 544-556.

CHAPTER 4. WATER QUANTITY

Kevin Bladon & Jeff Behan

4.1. Introduction

Relationships between forest cover and type, forest management, and the quantity and timing of water produced by forested watersheds have been studied for at least 100 years (e.g., Bates and Henry 1928; Griffin 1918). Motivations for such research have included interest in how active management affects the ways that forested catchments capture, hold, and deliver water to community water systems (Neary 2000). In order to continue to provide a safe and secure water supply to communities, water providers are concerned about, not only in maintaining access to a consistent supply of raw water over time, but also in how episodic high flows and seasonal low flows may respond to land use changes in their source watersheds.

In this chapter, current scientific understanding of how active forest management affects water delivery will be discussed in four sections: (1) annual yield of water, (2) peak flows and flooding, (3) low flows, and (4) the timing of water runoff from forested watersheds.

4.2. Overview of Literature Reviewed

Many studies contributing to current scientific understanding of the effects of active forest management on the quantity and timing of water delivered from forested watersheds were conducted when forest practices were different than they are today. This raises the question of how relevant this older research is to current practices. However, the effects of industrial forestry on sediment production and water quality have arguably received more attention, and been the focus of more significant changes in forest practices than have effects on water quantity and timing. For example, management practices for riparian areas, most notably stipulations for leaving forested buffers along waterways, were modified significantly in both the 1970s and 1990s. Rules for forest road location, construction and use have also been revised several times, primarily focused on reducing sediment impacts.

Studies have repeatedly shown that changes in water delivery resulting from forest management are driven primarily by the percentage area of the watershed that was recently harvested. In Oregon, this variable was addressed in Senate Bill 1125 in 1991, and resulting changes to the Oregon FPA in 1992, which limited clearcuts on non-federal forestland to 120 acres. Adjacent areas in the same ownership cannot be clearcut until new trees on the original harvest are at least four feet tall or four years-old and the stand is “free-to-grow”. Under these rules, an entire 6th field HUC subwatershed can still be logged within a decade. In short, from the standpoint of relevance to hydrology, forest rules which affect the amount and timing of water produced have changed relatively little, especially in comparison to practices targeted toward sediment production. This in turn suggests that older studies on linkages between forestry and water production still have at least some relevance under current practices.

For the current review, our focus is primarily (but not exclusively) on research conducted since the year 2000, in the Pacific Northwest, including studies from northern California to southwestern BC Canada. But research on relevant subtopics is often limited, so evidence is also drawn from older studies, synthesis papers, and research from outside this geographic area.

4.2.1. Forest Management and Annual Water Yields

The hydrologic response to forest management activities (e.g., road construction, harvesting, post-harvest site preparation, and silvicultural treatments) can be highly variable among watersheds due to catchment differences in forest type, soils, geology, topography, climate, hydrological regimes (e.g., rain-dominated, snow-dominated), and management approach (Stednick 2008). Specifically, forest management activities can affect hydrologic processes in several ways, including (a) decreased evapotranspiration, (b) decreased precipitation interception, and (c) increased snowpack accumulation due to decreased snow sublimation in the seasonal or transient snow zone (Jassal et al. 2009; Varhola et al. 2010; Hubbart et al. 2015; Winkler et al. 2015). These effects often lead to increased soil water content in the first few post-harvest years, especially during summer and early fall due to decreased transpiration (Harrington et al. 2013; Du et al. 2016). As a result, the deficit in soil water content necessary to exceed the soil field capacity to generate runoff is often reduced. In other words, less precipitation may be needed to produce hillslope runoff and streamflow. Thus, forest management practices often lead to increases in annual water yields and influence flow regimes for some time after harvest (Stednick 1996; Bowling et al. 2000; Brown et al. 2005).

In a chapter in a recent forest hydrology textbook, Stednick and Troendle (2016) summarized some generalized concepts regarding relationships between annual water yield and forest management gleaned from several decades of paired catchment studies. They noted that increases in water yields after harvest are often not detectable unless the catchment (1) receives annual precipitation greater than about 450–500 mm and (2) has had at least 20% of the catchment area harvested. In areas that receive less precipitation, a decrease in forest cover will usually increase soil evaporation and transpiration by residual vegetation, rather than increasing net water yield from the basin. In rain-dominated areas, increased water yields after harvest are most prominent during late fall and winter when the soil moisture deficit from the drier summer months is being recharged. In the snow zone, increases usually peak during the late spring to early summer when melting snow recharges the soil moisture. South-facing slopes in the northern hemisphere generally have less dense vegetation and receive more solar energy than north facing slopes. As a result, water yield increases are usually reduced on south facing slopes. Compared to clearcuts, forest stands subject to partial cuts usually have less response to harvesting since increased water is used by the remaining vegetation.

Changes in annual water yield following timber harvest are generally dependent on the post-harvest climate and antecedent moisture conditions. Water yield response to a given precipitation event reflects soil moisture conditions just prior to the event. Precipitation on wetter soils generally results in greater water yield than will be generated from the same event falling on drier soils and soils are typically wetter on logged watersheds. In drier forests, or during drier seasons, the difference in antecedent moisture content between forested and harvested catchments might be minimal as will be water yield responses. In wetter forests, differences in soil moisture conditions between forested and harvested catchments prior to a given precipitation event are usually greater as are differences in water yield that occur in response. Water yield and changes in yield following timber harvest generally increase with increasing precipitation, especially when differences in antecedent soil moisture exist. Where rainfall is high, or when evapotranspiration is low (winter), differences in antecedent conditions for soil moisture between forested and harvested catchments may be attenuated, as will be the difference in water yield response. (Stednick and Troendle 2016.) It should be noted that considerable variability and some exceptions to most of these generalizations can be found in the literature.

Due to the importance of water yields for downstream water supply, aquatic ecosystem health, and forest health there have been several reviews synthesizing literature regarding the effects of forest management activities on annual water yields (Stednick 1996; Brown et al. 2005; Moore and Wondzell 2005). Moore and Wondzell (2005)focused on the rain dominated regions of the Pacific Northwest and found that for each percentage of the catchment harvested by clear-cut and patch-cut harvesting, that water yields increased up to 6 mm. They also showed that selective harvesting increased water yields up to 3 mm for each percentage of the catchment harvested. Those findings were similar to an older review by Bosch and Hewlett (1982), who found an ~40 mm increase in annual water yield per 10% reduction in forest cover after reviewing 94 experimental watersheds. Moore and Wondzell (2005) also showed that increases in water yield were more muted after forest harvesting in snow-dominated catchments, ranging from ~0.25–3 mm per percentage of catchment harvested. However, most studies reviewed have concluded that annual streamflow changes are generally not detectable until at least 15–20% of a catchment is harvested (Stednick 1996; Brown et al. 2005; Moore and Wondzell 2005). The majority of past reviews have also shown that increased annual water yields can persist for ~10–20 years, with the largest increases occurring during the wet period of the year; autumn and winter in rain dominated regions. (Harr 1983; Keppeler and Ziemer 1990). Moore and Wondzell (2005) provide an important summary of research results regarding the effects of forest harvesting on annual water yields at the headwater scale (mean catchment area 0.62 km^2; range: 0.10–3.04 km^2) in the Pacific Northwest, which are relevant to many smaller water providers in Oregon. However, these findings may have less relevance for water providers with larger, basin-scale drinking water sources.

The majority of recent studies have also focused on contemporary forest harvesting effects on annual yield at the headwater catchment scale. For example, Zegre et al. (2010) assessed contemporary forest harvesting, based on the Oregon Forest Practices Act, in catchments ranging in area from 0.23–1.56 km^2. These results from the Hinkle Creek Paired Watershed Study (2004–2008) on the foothills of the west slope of the southern Oregon Cascades Mountains illustrated that ~9% of the post-harvest median model innovations (i.e., random noise component of the time series model) exceeded the 95% prediction intervals (Zegre et al. 2010). Statistically, this indicated that daily streamflow increased following harvesting, by as much as 31 mm for each model. Similar to previous studies, they also found the greatest seasonal increases occurred during winter (485 mm), followed by spring (146 mm), fall (114 mm), and summer (100 mm) (Zegre et al. 2010).

Winkler et al. (2017) investigated streamflow response to forest harvesting of 2 small (4.5 and 4.9 km^2), snow-dominated catchments on the Okanagan Plateau of British Columbia and found only a 5% increase in annual water yield after clearcutting of 47% of the logged watershed. However they identified dramatic changes in the timing and magnitude of April-June streamflow. During spring runoff (April and May) average water yield increased by ~19–29% during the first 7 years after harvesting. Winkler et al. (2017) indicated that such changes in runoff timing could increase the risk of channel destabilization during the snowmelt season, and water shortages early in the irrigation season.

Du et al. (2016) also illustrated an effect on water yield following contemporary forest harvesting of a small (28 km^2) sub-catchment in the Mica Creek Experimental Watershed in northern Idaho. However, in their study they parameterized a model (DHSVM) with 10 years (1998–2007) of data and ran a series of virtual experiments to assess various spatial and temporal patterns of forest canopy removal. Model simulations predicted increases in annual water yields of (a) 33% for gradual patch-cutting of 10% of the catchment area every 6 year, (b) 37% for the 50% forest removal scenario, and (c) 79% for the 100% clear-cut scenario (Du et al. 2016). Interestingly, model simulations also indicated the importance of the spatial location of the harvest within a catchment as annual water yields were ~4% greater if the upper half of the catchment was harvested rather than the lower half of the catchment (Du et al. 2016).

Abdelnour et al. (2011) applied the Visualizing Ecosystems for Land Management Assessments (VELMA) model to elucidate how hillslope and catchment-scale processes control stream discharge in the H.J. Andrews Experimental Forest. One interesting result of this work was that streamflow response was strongly sensitive to harvest distance from the stream channel. Specifically, they found that a 20% clearcut area near the catchment divide (average distance of 152 m to the nearest stream channel) resulted in an average annual streamflow increase of 53 mm (4%), whereas a 20% clearcut in the lowlands (average distance of 53 m to the nearest stream channel) resulted in an average annual streamflow increase of 92 mm (8%).

Unfortunately, for the current review these studies did not investigate the effects of forest harvesting at the larger basin scale, which would be relevant to larger community drinking water suppliers. Specifically, approximately 95 Oregon communities (~47.5% of state population) have a surface water supply originating in a forested watershed > 10 km^2, with a state average of ~426.4 km^2 and median area of 86.6 km^2 (Oregon Department of Environmental Quality 2018). Thus, while not directly representative of the PNW, a recent study by Zhang et al. (2017) provides insights into the potential effects of forest harvesting on annual water yields at the large watershed scale. Zhang et al. (2017) studied the effects of forest harvesting in 6 snow-dominated watersheds in British Columbia, Canada ranging from 539–3,185 km^2. They showed an increase in mean annual yields of 21–60 mm in those large basins with a cumulative equivalent clear-cut area greater than 30%. Not surprisingly, the largest changes in mean annual water yields were observed in wetter years. However, overall the results were inconsistent with no changes in water yields after substantial forest harvesting activity contrasted with significant changes in mean annual water yields with relatively small areas of forest harvesting activity (Zhang et al. 2017).

In summary, existing research is fairly consistent in showing that clearcut harvests can result in increases in annual streamflow, especially at smaller spatial scales that are most studied. These increases are typically highest just after harvest and then decline over the following decade or two as vegetation regrows. However, attempting to quantify harvesting effects on streamflow is time consuming and expensive, requiring long-term commitments from both researchers and landowners (Stednick and Troendle 2016). Study results vary considerably and are based primarily on research streams from a relatively small number of paired watershed study sites. Existing studies across the Pacific Northwest do not adequately reflect the broad range of climate, geology, topography, and vegetation, which drive highly variable hydrologic processes in the region. As such, there are still substantial information gaps, especially at the larger basin scale, most relevant to larger water providers. Given the substantial uncertainty around reliable water supplies in the PNW in coming decades it is critical to resolve some of this uncertainty through additional empirical and modeling research (Mateus et al. 2015; Vano et al. 2015).

4.2.2. Forest Management and Peak Flows

Peak flows and floods have the potential to produce extensive and costly damage to the structure and function of headwater catchments and downstream infrastructure (Downton et al. 2005; Ashley and Ashley 2008; Tullos 2018). Historically, the PNW has experienced peak flows in the upper ~90th and ~99th percentile of the contiguous U.S. (O'Connor and Costa 2004). The majority of these large flood events have occurred during winter rain-on-snow (ROS) events; however, further work is still needed to understand the relationship between ROS events and floods (McCabe et al. 2007; Jennings and Jones

2015). Regardless, recent research has projected that peak flow magnitudes may increase up to > 30–40% in some higher elevation areas of the PNW, including the Cascade Mountains, Olympic Mountains, and Blue Mountains, due to the effects of warmer temperatures on snowpack dynamics (Safeeq et al. 2015).

Given the concerns about naturally occurring high flow events, the effects of forest management activities on the occurrence and magnitude of peak flows and floods remains a contentious issue, which has led to repeated calls for the forest hydrology community to address (DeWalle 2003; Calder et al. 2007; Alila et al. 2009). The magnitude and occurrence of high flow events may be influenced by many factors, including rapidly changing forest harvesting treatment types, percent of catchment harvested, road location and construction approaches, site preparation, slope stability, vegetation species, forest re-growth rates, and the differential responses to precipitation across hydrologic zones (i.e., rain-, transient-snow, and snow-dominated) (Jones and Grant 1996; Grant et al. 2008; Kuraś et al. 2012). As a result of the complex interactions between the many influential factors and the infrequent observations of high flow events, accurate prediction and assessment of the effects of forest harvesting on peak flows remains a challenge (DeWalle 2003). Knowledge has accumulated and certain trends have been noted, but information gaps remain in the scientific community about the relationship between forest practices and peak flows.

Despite this uncertainty, regulatory agencies and land managers remain tasked with developing strategies to manage forests in ways that mitigate or avoid changes in peak flows. In the face or major revisions to regional-scale forest plans in the PNW, this provided the impetus for the most recent comprehensive review by Grant et al. (2008) entitled the "Effects of Forest Practices on Peak Flows and Consequent Channel Response: A State-of-Science Report for Western Oregon and Washington." The objective of that synthesis document was to provide guidance to forest managers and regulators for evaluating the potential risks of elevated peak flows associated with forest management. In their review Grant et al. (2008) considered factors such as different forest harvesting treatment, presence of roads, and catchment drainage efficiency.

Overall, Grant et al. (2008) found that increases in peak flows were generally smaller when a lower percentage of the catchment was harvested. The largest increases in peak flows associated with forest harvesting occurred in catchments that were clearcut (i.e., 100% harvested). With decreasing harvesting intensity, increases in peak flows were highly variable, ranging from 0–40% in the rain zone and transient snow zone, and from 0–50% in the snow zone. Unfortunately, there was insufficient research available to assess how this variability in peak flow response may be related to different forest harvesting approaches.

Additionally, Grant et al. (2008) found that forest management activities generally had less of an effect on the larger, less frequent peak flow events(. While peak flows increased ~90% in harvested catchments over reference catchments during small storm events (recurrence interval < 1 year), this effect tended to diminish as an approximate exponential function. This trend of an exponential decrease in peak flow with increasing storm magnitude was considered to be consistent from the site (< 10 km^2) to large basin scale (> 10 km^2 to < 500 km^2).

Grant et al. (2008) also found that watersheds in the transient snow zone were more sensitive to the effects of forest harvesting on peak flows compared to watersheds in rain-dominated zones of the PNW. However, the transient snow zone was the hydrologic zone most studied historically. There was not enough research or data (i.e., a lack of modeling or field studies with >50 % catchment harvested) to make interpretations about the effects of forest harvesting on peak flows in the snow zone.

Importantly for the current review, there were only a couple studies in the PNW investigating the effects of forest harvesting on peak flows at the larger basin scale (Jones and Grant 1996; Thomas and Megahan 1998), which would be most relevant to community drinking water supplies. As such, the results from the Grant et al. (2008) review may only be directly relevant to ~23 Oregon communities (~3.7% of the state population), which rely on surface water from forested watersheds with an area < 10 km^2. For comparison, approximately 95 Oregon communities (~47.5% of state population) have a surface water supply that originates in a forested watershed > 10 km^2, with a state average of ~426.4 km^2 and median area of 86.6 km^2 (Oregon Department of Environmental Quality 2018).

The lack of research at the larger basin scale creates uncertainty about how to interpret research results from the small catchment scale. Despite this, Grant et al. (2008) suggest that elevated peak flows in headwater catchments due to forest management activities are most likely to diminish with increasing basin size. The principal theories supporting the idea that peak flows diminish at the downstream basin scale, include: (a) floodplain storage, (b) transmission losses into the alluvial material of the streambed, (c) channel resistance, (d) low likelihood of sub-catchment peak flow synchrony, and (e) the proportion of basin area disturbed generally decreases with increasing basin size (Archer 1989; Shaman et al. 2004; Calder and Aylward 2006). However, the role of these different factors at attenuating peak flow magnitude at the basin scale is likely to differ depending on specific catchment characteristics, including valley width, channel morphology and complexity, stream slope, hydraulic roughness (e.g. large woody debris), amount of wetlands, and precipitation event characteristics (Woltemade and Potter 1994). As such, additional research on the scaling of peak flows from small headwater catchments to larger river basins is needed to resolve this issue.

Another consideration is that all 21 of the paired catchment studies reviewed by Grant et al. (2008) investigated effects from forest harvesting that occurred from the 1950's to the 1990's. Forest harvesting and best management practices have continued to evolve in the 21st century (Cristan et al. 2016), but there is insufficient research to determine if, or the degree to which, current forest practices may have modified the effects of harvesting on peak flows, compared to past practices. For this review we have searched the literature for research not included in previous reviews and relevant to the PNW. Unfortunately, there have been few studies investigating the effects of contemporary practices on peak flows, especially at the large basin scale.

In a valuable study Jones and Perkins (2010) analyzed more than 1000 peak flow events that occurred in the western Cascades of Oregon from 1953 to 2006. Their study sites included data from six small catchments (0.09–1 km^2) and six large basins (60–600 km^2) covering the transient snow and permanent snow zones. Their findings were mostly consistent with previous research, illustrating that forest harvesting generally had the greatest effect on the smaller, more frequent (< 1 year return interval) peak flow events. However, they did observe an increase (~10%–20%) in the magnitude of large peak flows (> 1 year return interval) during rain-on-snow events in the transient and seasonal snow zones. While this is consistent with previous research showing that the largest peak flow events were associated with rain-on-snow events, their observation of the potential synchronization of peak flows in the small catchment scale illustrates the potential for large floods at the large basin scale associated with forest harvesting (Jones and Perkins 2010). The Jones and Perkins (2010) study represents a new analysis of data not included in a previous review of peak flow effects; however, the study still relies on data from catchments harvested in the 1960's and 1970's.

Similarly, Du et al. (2016) used 10 years (1998–2007) of data from the upper sub-catchment (28 km^2) of the snow-dominated Mica Creek Experimental Watershed in northern Idaho to parameterize the Distributed Hydrology Soil-Vegetation Model (DHSVM). They used the model with this data to simulate clear-cut harvesting of the entire watershed, which predicted a ~68 % increase in peak flows (5^{th} percentile flows). They also ran scenarios with 50% vegetation removal and a gradual patch-cut of ~10% of the catchment. These two scenarios also predicted increases in peak flows of ~19% and 16%, respectively. Interestingly, the modeling exercise by Du et al. (2016) also indicated that forest harvesting away from the outlet or stream channel could potential produce larger peak flows during snowmelt. They attributed this result to a synchronicity of melt between the high and low elevations, which is consistent with historical research in snow dominated catchments (Troendle and King 1985). Specifically, the modeling scenarios suggested that harvesting of the upper portion (higher elevation) of the catchment would increase peak flows ~9 % more than scenarios where forest harvesting occurred on the lower portion (lower elevation) of the catchment (Du et al. 2016).

Green and Alila (2012) argued forcefully for a "paradigm shift" from generally accepted methods of comparing floods by equal meteorology or storm input ("chronological pairing"; CP) to a flood frequency distribution framework ("frequency pairing"; FP). They maintained that CP approaches in paired watersheds have yielded inaccurate results that underestimate forestry effects on large flood frequency. Green and Alila (2012) and related work (Kuraś et al. 2012; Schnorbus and Alila 2013) in a low elevation, snow dominated system in BC, Canada found that forest harvesting may substantially increase the frequency of the largest floods. These studies have been contentious within the forest hydrology community but may have relevance for understanding the effects of forest harvesting on peaks flows in the seasonal or permanent snow zones in the PNW and are discussed in more detail below.

Kuraś et al. (2012) used data from a harvested catchment in Penticton, BC to evaluate three modeling scenarios of increasing area harvested (20, 30, and 50% clearcut). The study catchment (241 Creek) was small (4.74 km^2), high elevation (1600–2025 m), and snow-dominated with mature lodgepole pine (*Pinus contorta* Dougl.) and small amounts of Engelmann spruce (*Picea engelmannii* Parry) and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt). The model results predicted greater effects with increasing catchment area harvested, with an increase of ~9–25% for peak flows with recurrence intervals of 10–100 years after 50% of the catchment was harvested (Kuraś et al. 2012). Interestingly, a couple of the key findings from the model simulations by Kuraś et al. (2012) were (a) an increase in peak flows of all sizes after forest harvesting and (b) a greater effect on the larger, less frequent peak flows relative to the smaller, more frequent peak flows, which was counter to the majority of current forest hydrology literature (Beschta et al. 2000; Troendle et al. 2001; Moore and Wondzell 2005; Birkinshaw et al. 2011).

Similarly, Schnorbus and Alila (2013) used data from the small (4.70 km^2), reference catchment (240 Creek) from the same study to model the effects of 11 hypothetical forest harvesting scenarios on peak flows. Again, the model suggested that annual peak flow magnitude would increase with increasing area harvested, with a threshold of ~20–30% of catchment area harvested to produce a demonstrable effect on peak flows. Additionally, the model projections from Schnorbus and Alila (2013) were also counter to the majority of past paired-catchment research, indicating a "relative increase in peak annual discharge occurs consistently across the full range of return periods". Interestingly, Schnorbus and Alila (2013) also showed increases in peak flows if forest harvesting occurred in the lower elevation bands of the catchment, which they attributed to greater channel drainage density and increased runoff efficiency at the lower elevations. This important finding of catchment physiographical control over the peak flow response to forest harvesting likely requires additional research in other regions.

More recently Yu and Alila (2019) adapted the FP approach to account for “nonstationarities” contained in peak flows that are caused by continuous harvesting and forest growth. Their nonstationary FP method for evaluating harvesting effects allowed the parameters of peak flow frequency distributions to change in time using physically based covariates. The method was demonstrated in the 37 km^2 Camp Creek (harvested) and 41 km^2 Greata Creek (reference) watersheds in the same Okanagan Valley, BC, Canada study area as that utilized by Green and Alila (2012). Yu and Alila (2019) found that both small (return periods < 10 years) and large (return periods > 10 years) peak flows are highly sensitive to harvesting in the mid-elevation south-exposed slopes of this snow-zone watershed. They purport that their nonstationary FP method is advantageous because it: (a) bypasses the need for the calibration equation traditionally used in paired watershed studies, and thus some associated sources of uncertainty; (b) enables use of longer peak flow records by explicitly accounting for physical causes of the nonstationarities, and thus more explicit inferences about effects of harvesting on the larger peak flows; and (c) allows estimation of harvesting effects on peak flows at different points during the disturbance history of a watershed, thus providing a direct evaluation of hydrologic recovery.

Alila and his colleagues (Alila et al. 2009; Green and Alila 2012; Kuraś et al. 2012; Schnorbus and Alila 2013) acknowledge that their results run counter to prevailing wisdom in hydrological science – i.e., that the effect of forest harvesting must always decrease with an increase in flood event size. These authors attribute the effects they found to increased net radiation associated with conversion from longwave-dominated (infrared) snowmelt beneath the canopy to shortwave-dominated (visible and ultraviolet light) snowmelt in harvested areas, amplified or mitigated by basin characteristics such as aspect distribution, elevation range, slope gradient, amount of alpine area, canopy closure, and drainage density. Their work spurred disagreement regarding the use of CP and FP approaches (Alila and Green 2014a; Alila and Green 2014b; Bathhurst 2014; Birkinshaw 2014) echoing similar debates over methods and statistical approaches among Jones and Grant (1996), Thomas and Megahan (1998) and Beschta et al. (2000). A persistent challenge that contributes to these disagreements is that as peak flow size increases, frequency of occurrence decreases, so the number of observations and resulting statistical power regarding the largest events are usually very limited. In these situations, trends detected and conclusions made can vary substantially depending on methodological and statistical approaches used, even with the same underlying data. While much of the current literature agrees with historical studies that forest harvesting can increase the magnitude of peak flows (Figure 4.1), the majority of research has remained focused on small catchments (< 10 km^2) (Perry et al. 2016).

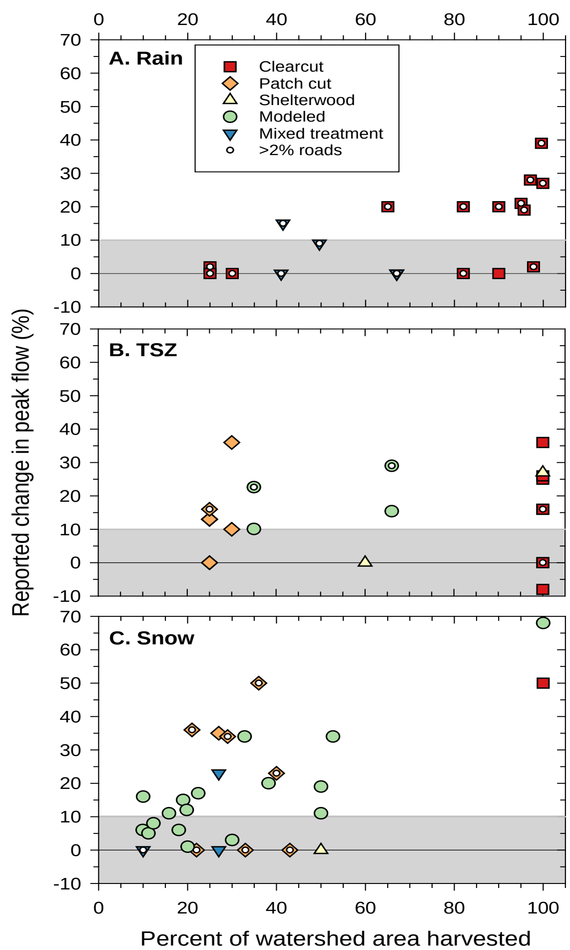


Figure 4.1. A summary of literature findings on the relationship between percent catchment harvested and percent change in peak flows in the (a) rain-dominated zone, (b) transient snow zone, and (c) snow-dominated zones of the Pacific Northwest. Symbol shapes and colors indicate the type of harvesting scenario. Figure modified from Grant et al. (2008) to include all additional studies since that publication.

Additionally, existing studies across the Pacific Northwest do not adequately reflect the broad range of climate, geology, topography, and vegetation, which drive highly variable hydrologic processes in the region. Moreover, assessing the cumulative effects of legacy impacts from historical forest management activities along with recent or proposed harvesting activities, remains a difficult challenge (Perry et al. 2016). Observations have continued to be highly variable, leading to vigorous debate focused on the analytical approach to quantitatively assessing relatively rare events with few observations (Alila et al. 2009; Lewis et al. 2010). As such, there remain gaps in our understanding of whether forest management activities influence peak flows at a scale relevant to larger downstream drinking water utilities.

Theoretical arguments have been made that peak flows in forested headwaters are unlikely to appear as integrated effects at larger basin-scales (Grant et al. 2008; Perry et al. 2016), but this is not certain as there have been observations in interior BC, Canada of peak flow effects from forest harvesting at the large watershed scale (Lin and Wei 2008; Zhang and Wei 2014). Uncertainties around predicted peak flow responses to forest harvesting are not likely to be definitively resolved without longer-term research that captures data on these relatively infrequent events in a broader range of managed forests and at larger basin scales.

4.2.3. Forest Management and Low Flows `

Low flows, which generally occur in late summer or early autumn, are increasingly of interest in the Pacific Northwest due to a greater occurrence of dry years (Mantua et al. 2010; Arismendi et al. 2013; Luce et al. 2014). Recent evidence suggests declining low flows and a lengthening in duration of the annual low flow period (Luce and Holden 2009; Leppi et al. 2012). Similar to the preceding sub-sections, most research on low flows has occurred at the small, headwater catchment scale, and primarily focused on concerns about summer stream temperature and aquatic habitat (Harr and Krygier 1972; Keppeler and Ziemer 1990; Stednick 2008). Research has not yet encompassed a broad range of geology, soils, topography, climate, or land uses, which all exert controls on low flows (Johnson 1998; Tague and Grant 2004). Much of the research comes from studies investigating older forest practices, (Rothacher 1970; Harr et al. 1979; Bowling et al. 2000). There remain significant knowledge gaps around the effects of forest management activities on low flows, especially at large basin scales.

Regardless, there is general agreement in the literature that in small catchments, forest harvesting results in increased low flows in the first ~5–20 years after harvesting, but can shift to low flow deficits in the longer-term (Moore and Wondzell 2005; Surfleet and Skaugset 2013). This is the case for both rain and snow-dominated regimes in the Pacific Northwest, where low flows have been shown to increase initially after forest harvesting as a result of decreased interception and evapotranspiration leading to increased soil moisture (Figure 4.2) (Rothacher 1965; Harr et al. 1982; Keppeler and Ziemer 1990; Bowling et al. 2000).

In a recent literature review on the potential effects of forest practices on streamflow in the Chehalis River Basin (~6,993 km^2), Perry et al. (2016) concluded that low flows in that region may increase for ~5–10 years after harvest. However they also found a broad range of low flow changes, from insignificant to more than 140% increase, with evidence of low flow deficits over time as sites revegetated (Ingwersen 1985; Fowler et al. 1987; Adams et al. 1991; Pike and Scherer 2003; Salemi et al. 2012). This latter finding was related to higher rates of transpiration from young, vigorous forests compared to older, mature forests (Moore et al. 2004; Moore et al. 2011). Perry et al. (2016) provide insights into the potential range of effects of forest harvesting on low flows, while noting, importantly, that the results

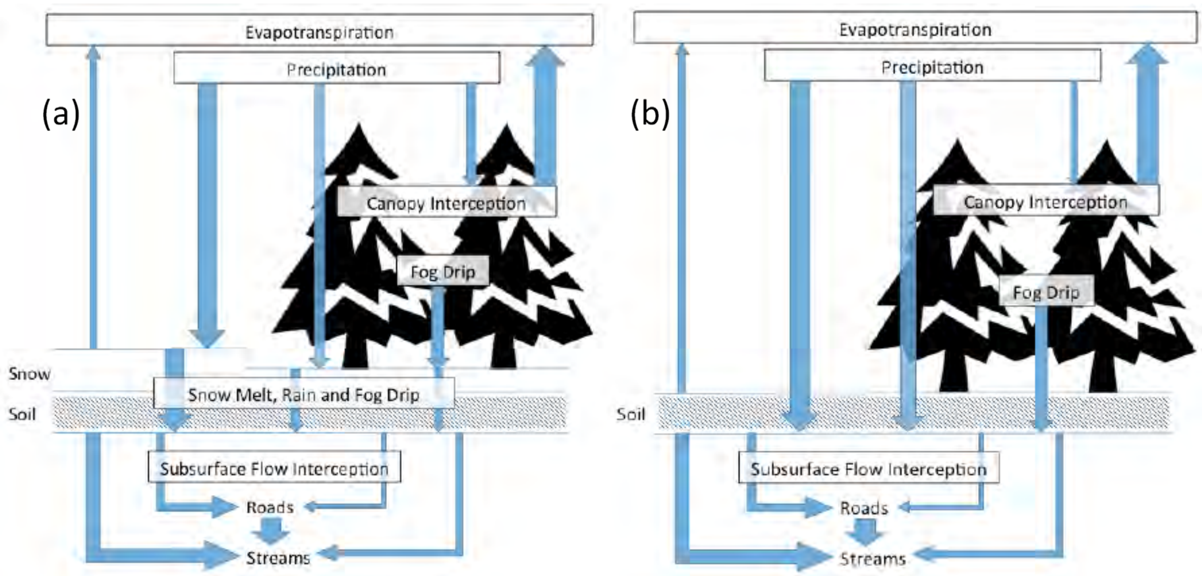


Figure 4.2. Hydrologic processes affected by forest harvesting relative to forested sites in (a) snow dominated regimes and (b) rain dominated regimes. Arrow widths denote the relative fluxes of water in each process in harvested compared to unharvested sites. From Perry et al. (2016).

from the literature reviewed were basin specific. Additionally, they note the lack of large-scale paired basin studies, principally attributed to the difficult challenge of establishing a true reference given that most large basins have experienced or will experience some forest harvesting activity (Perry et al. 2016). Some inferences can be made on the basis of smaller scale studies, but there is a paucity of direct evidence regarding the effects of forest management activities on low flows at the large basin scale.

In the western Cascades of southern Oregon, Surfleet and Skaugset (2013) also observed an increase in summer (August) low flows of ~45% (1.9 mm yr^{-1}) for three years after harvesting of ~13% of a 10.8 km^2 catchment. More specifically, summer low flows increased by 106% (4.5 mm) in the first summer and 47% (2.0 mm) during the second summer. However, the effects of forest harvesting on summer low flows weren't distinguishable 5 years after harvesting in all catchments except for the one with the greatest proportion of area harvested. Given the short duration and small spatial scale of the study, it is uncertain whether low flow deficits occurred in these catchments later as the forest revegetated or whether effects were observable at a larger, basin scale. Regardless, results from this contemporary study are consistent with historical research in the same region of southwest Oregon, which showed a ~44% increase in summer low flows (Harr et al. 1979). The results are also consistent with model simulations using data from WS10 of the H.J. Andrews, which illustrated that the largest relative increases in streamflow after harvesting occurred during the summer low flow period (Abdelnour et al. 2011).

Over the past 20 years, an increasing amount of research has focused on how regenerating forests affect summer low flows for a longer period after harvest (i.e., several decades) when the new stand is fully re-established and growing quickly. Moore et al. (2004) showed that younger, vigorous stands use more water than adjacent older stands, which they attributed primarily to tree age and, to a lesser degree, differences in sapwood basal area and finally species composition. In three small watersheds in southern interior BC, Canada, Gronsdahl et al. (2019) found that summer flows were reduced starting about 20 years after the onset of forest harvesting which, they surmised was a result of regenerating forests transpiring more water than the mature forests they replaced.

In a rigorous analysis of 60-years of daily streamflow data from eight paired watersheds in the seasonal snow zone of the Pacific Northwest, Perry and Jones (2017) showed that summer low flows were lower in young, vigorously growing stands compared to older adjacent stands. In particular, they showed that ~15 years after forest harvesting and establishment of Douglas-fir plantations, summer streamflows were in a deficit, which persisted and intensified for ~50 years (Perry and Jones 2017). The average daily streamflow during the summer (July through September) was ~50% lower in catchments with 34- to 43-year-old plantations compared to reference catchments with 150- to 500-year-old forests. This persistent decline in summer low flows was attributed to greater sapwood area, sapflow per unit sapwood area, leaf area in the upper canopy, and less stomatal control to limit transpiration in the young plantation compared to the mature forest (Perry and Jones 2017). While this study provided a much longer time series than previous observations, the potential for longer-term reductions in low flows due to vigorous re-growth following forest harvesting were noted previously in these PNW catchments (Hicks et al. 1991; Jones and Post 2004).

Segura et al. (2020) compared responses of daily streamflow in (a) harvested mature/old forest in 1966, (b) 43 to 53 and 48 to 58 yr-old industrial plantation forests in 2006–2009, and (c) these same plantation forests in 2010 and 2014, after harvesting using contemporary forest practices, including retention of a riparian buffer. The work was part of the long-term Alsea Watershed Study in the Oregon Coast Range (Stednick 2008). Segura et al. (2020) found that daily streamflow from a 40- to 53-yr-old Douglas-fir plantation was 25% lower on average, and 50% lower during summer, relative to the mature/old forest, and that these deficits lasted at least six months of each year. Contemporary forest practices (retaining riparian buffer strips in clearcuts) had a minimal effect on streamflow deficits. Two years after logging in 2014, summer streamflow deficits were similar to those prior to harvest (under 40- to 53-yr-old plantations).

Consistent with Perry and Jones (2017) and Gronsdahl et al. (2019), Segura et al. (2020) attributed persistent streamflow deficits after logging to high evapotranspiration from rapidly regenerating vegetation, including planted commercial timber species. The authors note that their findings for summer streamflow deficits in young stands in the Oregon Coast Range were similar in magnitude to those detected in Douglas-fir plantations in the western Cascades (Perry and Jones 2017; Jones and Post 2004) indicating that plantations of similar age have similar evapotranspiration rates relative to mature and old-growth forest reference stands in all of these locations. Overall, Segura et al. (2020) found that 40- to 50-yr rotations of Douglas-fir plantations can produce persistent, large summer low flow deficits, and that clearcutting with retention of riparian buffers increased daily streamflow slightly but flows did not return to conditions when the old/mature forests were intact. The authors suggest that additional work is needed to investigate how intensively managed forests and expected warmer, drier conditions in the future may influence summer low flows.

Considerable knowledge has accumulated but understanding of the magnitude, duration, physical processes, and downstream consequences associated with the short-term increases in low flows or longer-term decreases in low flows after forest harvesting remains incomplete. Additional research is necessary to examine both the upstream and downstream effects of forest management activities on low flows in a wider range of areas. Similar to the other sub-sections in this chapter, comparative studies, process studies, and modeling are all necessary to fully understand the spatial and temporal impacts. Given current projections for climate and its potential impacts on low flows (Hamlet 2011; Arismendi et al. 2013; Tohver et al. 2014), it is increasingly imperative to maintain current longer-term watershed studies and revive historical studies to capture data from a range of climates, geologies, soils, topographies, forest types, and forest ownerships. Doing so will facilitate effective management of the water supply from forests during periods of low flow, which generally coincide with the period of greatest demand by communities.

4.2.4. Timing of Water Delivery

Much of the Pacific Northwest is reliant on a community water supply originating as mountain snow. This includes many community water systems in Oregon, although mostly not in the Coast Range. The melting of the seasonal snowpack in snow-dominated catchments, combined with the onset of spring and early summer rains generates the rising limb and peak in the annual hydrograph (Kormos et al. 2016). As such, observations and projections of a declining annual snowpack along with a shift toward earlier spring snowmelt and provision of downstream water supply have generated considerable concerns (Cayan et al. 2001; Mote 2003; Stewart et al. 2005; Mote et al. 2008; Abatzoglou et al. 2014). Shifts in snowmelt timing violate the critical stationarity assumption for statistical water supply forecast models, producing concomitant challenges for downstream water supply managers (Milly et al. 2008; Barnhart et al. 2016).

Research has clearly shown the important role of forests in the PNW in influencing snow accumulation, ablation, and the timing of snowmelt (Marks et al. 1998; Storck et al. 2002; Molotch et al. 2009; Lawler and Link 2011; Gleason et al. 2013). However, predicting the effect of forest cover and the effects of forest harvesting activities on the timing of snowmelt and resulting streamflow remains particularly complex. This is because the influence of the forest on snowmelt timing is modified by a broad range of factors, including climate, topography, and specific forest characteristics (Lundquist et al. 2005; Varhola et al. 2010; Lundquist et al. 2013; Martin et al. 2013; Harpold and Molotch 2015). As such, predicting the net effect of forest management activities on forest cover and snowmelt timing requires integrating multiple forest–snow processes, which all vary in space and time (Dickerson-Lange et al. 2017). Thus, there is considerable variability and associated uncertainty in the literature regarding the effects of forest harvesting on the timing of streamflow, especially at a large, basin scales.

In a recent study, Dickerson-Lange et al. (2017) used observational data to compare snowmelt timing between forested and open areas across 14 sites in the western slopes and crest of the Cascade Range in WA and OR and central and northern ID. Overall, they found that forest modification by forest harvesting was a dominant factor influencing the timing of snow disappearance. In particular, at 12 of 14 open [or harvested] sites, melting of the snowpack was either synchronous in timing or persisted for a longer period of time (up to 13 weeks longer) relative to forested sites (Dickerson-Lange et al. 2017). This effect was most noticeable in warmer, maritime climates of the PNW and was related to greater canopy interception storage capacity, greater snow interception efficiency, and lower wind unloading of snow from the canopy due to greater snow cohesion (Kobayashi 1987; Andreadis et al. 2009; Friesen et al. 2015). However, snow disappearance occurred ~2–5 weeks earlier at two open sites compared to forested sites, which was attributed to comparatively high wind speeds (hourly average wind speeds 8 and 17 m s^{-1}). The wind effects at those sites was believed to have produced similar snow deposition in the open and the forest sites, but higher ablation rates in the open sites (Dickerson-Lange et al. 2017).

In small, snow-dominated catchments in the Okanagan Plateau of British Columbia, Winkler et al. (2017) also noted a shift in timing of snowmelt associated with forest harvesting activity. They observed an advancement in the date of peak water yield by up to one week in harvested locations with an associated increase in monthly water yields on the rising limb of the snowmelt hydrograph (April and May) along with a decrease on the falling limb (June and July) (Winkler et al. 2017). In this case, the shift in timing of snowmelt and associated streamflow was attributed to synchronization of snowmelt in the high elevation clear-cut areas (south-facing) with snowmelt from the lower elevation, unharvested forest. Similarly, Zhang et al. (2016) observed an advancement in timing of annual peak flows of approximately 9 days at the large watershed scale after forest harvesting in two snow-dominated watersheds of British Columbia. This study was not focused on timing of availability so results are limited. Additionally, this study occurred in the interior of B.C., a drier environment than much of western Oregon, but provides some indication that effects of forest harvesting on timing may be measureable at the large, basin scale.

Alternatively, in their study of the effects of forest harvesting on peak flows in the western Cascades of Oregon, Jones and Perkins (2010) found some evidence of shifts in the timing of peak flows in small catchments, but the timing of large peak flow events in large catchments remained largely unaffected (Figure 4.3). Even at the small catchment scale the effects of forest harvesting on the timing of peak flows weren't consistent. For example, peak flows occurred ~3–10 hours earlier in harvested catchments in the transient snow zone, but 6–12 hours later in the harvested catchment in the seasonal snow zone (Jones and Perkins 2010).

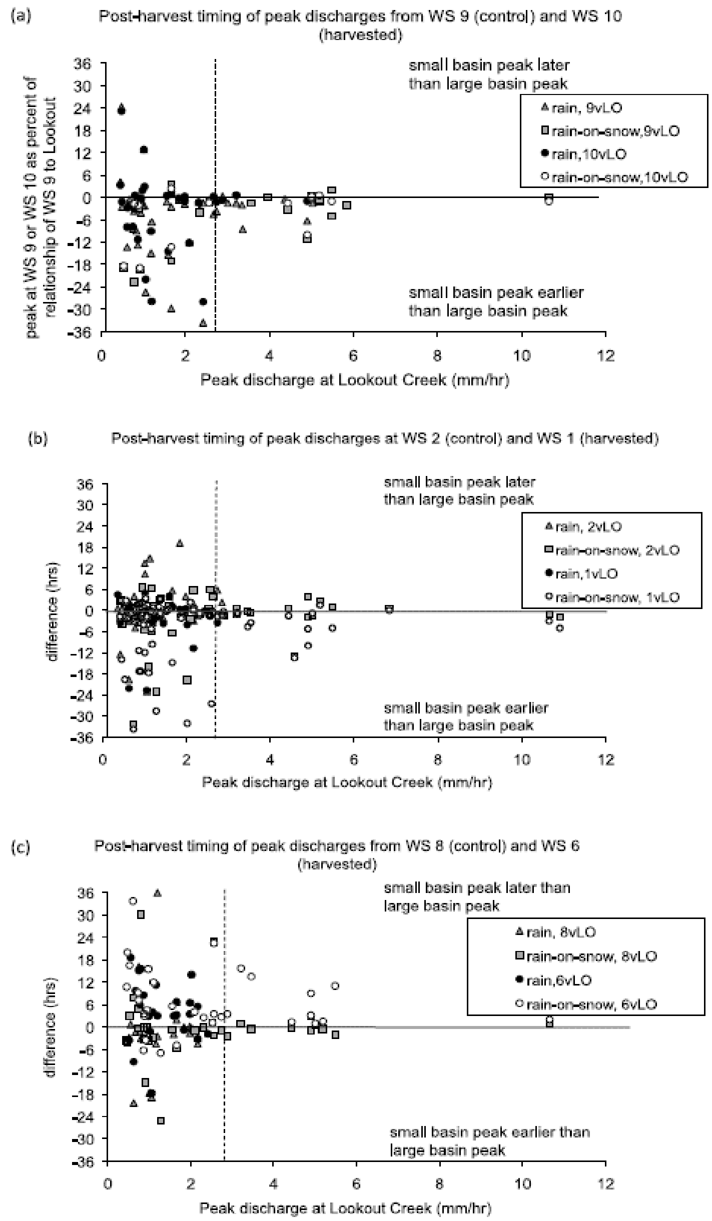


Figure 4.3. Post-harvest timing of peak flows following rain and rain-on-snow events from paired catchment studies in the (a) transient snow zone, (b) transient to seasonal snow zone, and (c) the seasonal snow zone in the H.J. Andrews Experimental Forest, OR (Jones and Perkins 2010).

Shifts in the timing of annual water yields have the potential to produce serious water supply management impacts, especially in community watersheds with limited reservoir storage capacity (Winkler et al. 2017). In communities without reservoirs, shifts to earlier timing of water supply may increasingly disconnect the timing of supply with the timing of greatest demand. Comparatively, in communities reliant on reservoirs, shifts in the timing of availability of streamflow to earlier periods of the year could potentially influence water purveyors to release water in excess of reservoir storage capacity, which would increase the risk of water shortages later in the year when demand is greatest (Winkler et al. 2017). Given the important linkage between forests and the timing of spring and summer streamflow (Whitaker et al.

2002; Lundquist et al. 2005; Lyon et al. 2008), it is critical to improve understanding and predictions of when and where forests will accelerate or delay snowmelt and streamflow timing, especially at the large basin scale (Rutter et al. 2009; Lundquist et al. 2013).

4.3. Conclusions

Relationships between forest cover and type, forest management, and the quantity and timing of water produced by forested watersheds have been studied for at least 100 years. Understanding of these relationships has been enhanced by research, especially long-term, paired watershed studies. We reviewed evidence regarding changes in (a) annual flow, (b) changes in peak flows and flooding, (c) changes in low (base) flows, and (d) changes in the timing of water delivery.

Throughout this chapter, we have noted potential sources of uncertainty in trying to extrapolate from results in the literature regarding forestry effects on these variables to effects on drinking water supplies. Key findings are derived mostly from studies in the upper parts of smaller, headwater catchments, and from a relatively limited number of geographic locations where long-term, paired watershed studies have been maintained. Even where consistent trends are noted across multiple studies, there is often considerable variability in results, with some studies finding large effects and others none at all. This suggests that effects may often be specific to the combination of conditions at a particular location. Studies we found focus on streamflow responses from headwater catchments, rather than at downstream drinking water intakes. Rigorous analysis of hydrologic responses to forest management is complex, time consuming and expensive, especially at larger scales and longer timeframes. Effects that have been quantified at smaller scales may potentially "scale up" to larger watershed scales but these larger scale effects are rarely studied and thus remain generally speculative. Lastly, conditions in many watersheds reflect the cumulative effects of actions conducted over the span of many decades of evolving forest management practices. In light of this complexity and the variability of climatic, physical and ecological factors in play, the uncertainty that remains in our understanding of the effects of active management on forest hydrology in particular locations should not surprise us.

These caveats duly noted, a substantial body of evidence has nevertheless accumulated, from an increasingly diverse array of research perspectives and methodologies. There will always be local exceptions and multiple contributing factors to any generalized conclusion, but we have some confidence that percent area of the watershed harvested is often the predominant factor affecting changes in annual flow volumes. There is general agreement that in many cases, timber harvesting temporarily increases annual water production, especially in the first few years after harvest, with these increases declining in following years, as vegetation, including planted commercial timber species, establishes and starts growing vigorously. By *volume*, these changes often peak in the fall and early winter. By *percentage,* the largest changes often occur in late summer.

Peak flows and floods have implications for community water suppliers in terms of increased sediment transport, turbidity, and mobilization of pollutants, as well as potential damage to water treatment infrastructure. The generally accepted scientific understanding regarding increases in peak flows attributable to forest management and harvesting has been that these effects are most prominent for smaller, more frequent peak flow events, and tend to decline as peak flow size and basin size increase. However, since the mid-2000s, the study designs and analysis methods used in much of the research upon which these conclusions are based have been vigorously debated. Several studies using alternative methods in snow-dominated watersheds in BC, Canada have found the opposite, i.e. that the frequency of peak flows of all sizes tend to increase after forest harvest and that these effects are most prominent for larger peak flows. Going forward, there are indications that over time, snowpack changes related to climate warming are likely to result in large increases in peak flow magnitudes in areas such as the Cascades and Blue Mountains. Predicted drivers for such a shift include greater frequency and magnitude of extreme precipitation events, and a growing proportion of winter precipitation falling as rain instead of snow. These forecasts suggest that any effects that forestry activities have on peak flows may intertwine with climate in increasingly complex ways. If, as expected, the frequency and magnitude of floods in Oregon increase under climate change, public and agency interest in mitigating anthropogenic factors that contribute to peak flows may intensify.

Seasonal low flows are of particular interest to water suppliers, because they generally coincide in late summer with the period of greatest demand for drinking and irrigation water. For at least two reasons, we may expect that relationships between active forest management and summer low flows in Oregon may be increasingly important to drinking water providers. First, while there are uncertainties regarding local and regional implications of climate change over time, there is also evidence that along with rising temperatures, dry years are increasing, low flows are declining and the annual low flow period is lengthening in duration. Secondly, recent research indicates that, in both the Oregon Coast Range and Cascades, stands of conifers established after clearcut harvests can, once they are 15–20 years old and growing quickly, significantly and persistently reduce summer low flows in comparison to the older stands they replaced. Many watersheds in these regions contain substantial amounts of timberland in this young plantation forest condition. In watersheds that serve as sources for smaller community water suppliers in Oregon and also support significant amounts of industrial forestry, climate trends and forest management may converge to further exacerbate challenges of supplying water during the critical late summer low flow period.

In summary, the weight of available evidence indicates that forest management can affect the volume and timing of water delivered from managed watersheds and by extension, community water systems that are hydrologically connected downstream. The limitations on existing knowledge described above are such that variability in local conditions can make it difficult to specify these effects for a particular water system. However, linkages between drinking water supplies and forest management (e.g., harvesting a significant percentage of the watershed) can be more readily established in smaller systems that are closer to the source watershed than in larger systems that are further away, with more intervening land uses.

Despite knowledge gaps, we understand enough to foresee that forest management activities in source watersheds will continue to be relevant considerations for water providers, and that effects may be predicted or specified with some degree of confidence in some smaller watersheds. Finally, climate change and associated shifts in snowpack levels and timing, and in the frequency and severity of extreme weather events, will further complicate an already complex set of factors that influence the amount and timing of raw water provided in actively managed drinking water source watersheds.

4.4. References

Abatzoglou, J.T., D.E. Rupp, and P.W. Mote. 2014. Seasonal climate variability and change in the Pacific Northwest of the United States. Journal of Climate 27: 2125-2142.

Abdelnour, A., M. Stieglitz, F.F. Pan, and R. McKane. 2011. Catchment hydrological responses to forest harvest amount and spatial pattern. Water Resources Research 47: 18.

Adams, P.W., A.L. Flint, and R.L. Fredriksen. 1991. Long-term patterns in soil moisture and revegetation after a clearcut of a Douglas-fir forest in Oregon. Forest Ecology and Management 41: 249-263.

Alila, Y. and K.C. Green. 2014a. Reply to comment by Birkinshaw on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments”. Water Resources Research 50(3): 2769-2774.

Alila, Y. and K.C. Green. 2014b. Reply to comment by Bathurst on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments”. Water Resources Research 50(3): 2759-2764.

Alila, Y., P.K. Kuras, M. Schnorbus, and R. Hudson. 2009. Forests and floods: A new paradigm sheds light on age-old controversies. Water Resources Research 45: W08416.

Andreadis, K.M., P. Storck, and D.P. Lettenmaier. 2009. Modeling snow accumulation and ablation processes in forested environments. Water Resources Research 45: 13.

Archer, D.R., 1989. Flood wave attenuation due to channel and floodplain storage and effects on flood frequency. Pp. 37-46 *In*: Beven, K., Carling, P. (Eds.), Floods: Hydrological, Sedimentological and Geomorphological Implications. John Wiley and Sons, Chichester, UK.

Arismendi, I., M. Safeeq, S.L. Johnson, J.B. Dunham, and R. Haggerty. 2013. Increasing synchrony of high temperature and low flow in western North American streams: double trouble for coldwater biota? Hydrobiologia 712: 61-70.

Ashley, S.T. and W.S. Ashley. 2008. Flood fatalities in the United States. J. Appl. Meteorol. Climatol. 47: 805-818.

Barnhart, T.B., N.P. Molotch, B. Livneh, A.A. Harpold, J.F. Knowles, and D. Schneider. 2016. Snowmelt rate dictates streamflow. Geophysical Research Letters 43: 8006-8016.

Bates, C.G. and A.J. Henry. 1928. Second phase of streamflow experiment at Wagon Wheel Gap, Colorado. Mon. Weather Rev., Vol. 56(3): 79-85.

Bathurst, J.C. 2014. Comment on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments” by KC Green and Y. Alila. Water Resources Research, 50(3): 2756-2758.

Beschta, R.L., M.R. Pyles, A. Skaugset, and C.G. Surfleet. 2000. Peakflow responses to forest practices in the western Cascades of Oregon, USA. Journal of Hydrology 233: 102-120.

Birkinshaw, S.J. 2014. Comment on “A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments” by Kim C. Green and Younes Alila. Water Resources Research, 50(3): 2765-2768.

Birkinshaw, S.J., J.C. Bathurst, A. Iroume, and H. Palacios. 2011. The effect of forest cover on peak flow and sediment discharge-an integrated field and modelling study in central-southern Chile. Hydrological Processes 25: 1284-1297.

Bosch, J.M. and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology 55: 3-23.

Bowling, L.C., P. Storck, and D.P. Lettenmaier. 2000. Hydrologic effects of logging in western Washington, United States. Water Resources Research 36: 3223-3240.

Brown, A.E., L. Zhang, T.A. McMahon, A.W. Western, and R.A. Vertessy. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology 310: 28-61.

Calder, I.R., and B. Aylward. 2006. Forest and floods: moving to an evidence-based approach to watershed and integrated flood management. Water International 31: 1-13.

Calder, I.R., J. Smyle, J., and B. Aylward 2007. Debate over flood-proofing effects of planting forests. Nature 450: 945-945.

Cayan, D.R., S.A. Kammerdiener, M.D. Dettinger, J.M. Caprio, and D.H. Peterson. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society 82, 399-415.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E. Schilling. 2016. Effectiveness of forestry best management practices in the United States: Literature review. Forest Ecology and Management 360: 133-151.

DeWalle, D.R., 2003. Forest hydrology revisited. Hydrological Processes 17: 1255-1256.

Dickerson-Lange, S.E., R.F. Gersonde, J.A. Hubbart, T.E. Link, A.W. Nolin, G.H. Perry, T.R. Roth, N.E. Wayand, and J.D. Lundquist. 2017. Snow disappearance timing is dominated by forest effects on snow accumulation in warm winter climates of the Pacific Northwest, United States. Hydrological Processes 31: 1846-1862.

Downton, M.W., J.Z.B. Miller, and R.A. Pielke Jr. 2005. Reanalysis of U.S. National Weather Service flood loss database. Natural Hazards Review 6: 13-22.

Du, E.H., T.E. Link, L. Wei, and J.D. Marshall. 2016. Evaluating hydrologic effects of spatial and temporal patterns of forest canopy change using numerical modelling. Hydrological Processes 30: 217-231.

Fowler, W.B., J.D. Helvey, and E.N. Felix. 1987. Hydrologic and climatic changes in three small watersheds after timber harvest. Research Paper PNW-RP-379. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, 13 pp.

Friesen, J., J. Lundquist, and J.T. Van Stan. 2015. Evolution of forest precipitation water storage measurement methods. Hydrological Processes 29: 2504-2520.

Gleason, K.E., A.W. Nolin, and T.R. Roth, T.R., 2013. Charred forests increase snowmelt: Effects of burned woody debris and incoming solar radiation on snow ablation. Geophysical Research Letters 40: 4654–4661.

Grant, G.E., S.L. Lewis, F.J. Swanson, J.H. Cissel, and J.J. McDonnell. 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-science report for western Oregon and Washington. GTR-PNW 760. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, 76 pp.

Green, K.C. and Y. Alila. 2012. A paradigm shift in understanding and quantifying the effects of forest harvesting on floods in snow environments. Water Resources Research: 48(10).

Griffin, A.A. 1918. Influence of forests upon the melting of snow in the Cascade Range. Monthly Weather Review 46(7): 324-327.

Gronsdahl, S., R.D. Moore, J. Rosenfeld, R. McCleary, and R. Winkler. 2019. Effects of forestry on summertime low flows and physical fish habitat in snowmelt-dominant headwater catchments of the Pacific Northwest. Hydrological Processes 33(25): 3152-3168.

Hamlet, A.F., 2011. Assessing the water resources adaptive capacity to climate change impacts in the Pacific Northwest Region of North America. Hydrology and Earth System Science 15: 1427-1443.

Harpold, A.A. and N.P. Molotch. 2015. Sensitivity of soil water availability to changing snowmelt timing in the western US. Geophysical Research Letters 42: 8011-8020.

Harr, R.D., 1983. Potential for augmenting water yield through forest practices in western Washington and western Oregon. Water Resources Bulletin 19: 383-392.

Harr, R.D., R.L. Fredriksen, and J. Rothacher. 1979. Changes in streamflow following timber harvest in southwestern Oregon. Research Paper PNW-RP-249. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Corvallis, OR, p. 22 pp.

Harr, R.D. and J.T. Krygier. 1972. Clearcut logging and low flows in Oregon coastal watersheds. Research Note 54. Forest Research Laboratory, College of Forestry, Oregon State University, Corvallis, OR. 3 pp.

Harr, R.D., A. Levno, and R. Mersereau. 1982. Streamflow changes after logging 130-year-old Douglas-fir in 2 small watersheds. Water Resources Research 18: 637-644.

Harrington, T.B., R.A. Slesak, and S.H. Schoenholtz. 2013. Variation in logging debris cover influences competitor abundance, resource availability, and early growth of planted Douglas-fir. Forest Ecology and Management 296: 41-52.

Hicks, B.J., R.L. Beschta, and R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bulletin 27: 217-226.

Hubbart, J.A., T.E. Link, and J.A. Gravelle. 2015. Forest canopy reduction and snowpack dynamics in a northern Idaho watershed of the Continental-Maritime Region, United States. Forest Science 61: 882-894.

Ingwersen, J.B., 1985. Fog drip, water yield, and timber harvesting in the Bull Run municipal watershed, Oregon. Journal of the American Water Resources Association 21: 469–473.

Jassal, R.S., T.A. Black, D.L. Spittlehouse, C. Brummer, and Z. Nesic.2009. Evapotranspiration and water use efficiency in different-aged Pacific Northwest Douglas-fir stands. Agricultural and Forest Meteorology 149: 1168-1178.

Jennings, K. and J.A. Jones, J.A. 2015. Precipitation-snowmelt timing and snowmelt augmentation of large peak flow events, western Cascades, Oregon. Water Resources Research 51: 7649-7661.

Johnson, R., 1998. The forest cycle and low river flows. A review of UK and international studies. Forest Ecology and Management 109: 1-7.

Jones, J.A. and G.E. Grant. 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research 32: 959-974.

Jones, J.A., and R.M. Perkins. 2010. Extreme flood sensitivity to snow and forest harvest, western Cascades, Oregon, United States. Water Resources Research 46: W12512.

Jones, J.A. and D.A. Post. 2004. Seasonal and successional streamflow response to forest cutting and regrowth in the northwest and eastern United States. Water Resources Research 40: W05203.

Keppeler, E.T. and R.R. Ziemer. 1990. Logging effects on streamflow - water yield and summer low flows at Caspar Creek in northwestern California. Water Resources Research 26: 1669-1679.

Kobayashi, D., 1987. Snow accumulation on a narrow board. Cold Reg. Sci. Tech. 13: 239-245.

Kormos, P.R., C.H. Luce, S.J. Wenger, and W.R. Berghuijs. 2016. Trends and sensitivities of low streamflow extremes to discharge timing and magnitude in Pacific Northwest mountain streams. Water Resources Research 52(7): 4990-5007.

Kuraś, P.K., Alila, Y., Weiler, M., 2012. Forest harvesting effects on the magnitude and frequency of peak flows can increase with return period. Water Resources Research 48(1): W01544.

Lawler, R.R. and T.E. Link, T.E.. Quantification of incoming all-wave radiation in discontinuous forest canopies with application to snowmelt prediction. Hydrological Processes 25: 3322-3331.

Leppi, J.C., T.H. DeLuca, S.W. Harrar, and S.W. Running. 2012. Impacts of climate change on August stream discharge in the Central-Rocky Mountains. Climatic Change 112: 997-1014.

Lewis, J., L.M. Reid, and R.B. Thomas. 2010. Comment on "Forest and floods: A new paradigm sheds light on age-old controversies" by Younes Alila et al. Water Resources Research 46: 4.

Lin, Y. and X.H. Wei. 2008. The impact of large-scale forest harvesting on hydrology in the Willow watershed of Central British Columbia. Journal of Hydrology 359: 141-149.

Luce, C., B. Staab, M. Kramer, S. Wenger, D. Isaak, and C. McConnell. 2014. Sensitivity of summer stream temperatures to climate variability in the Pacific Northwest. Water Resources Research 50: 3428-3443.

Luce, C.H. and Z.A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters 36: L16401.

Lundquist, J.D., M.D. Dettinger, and D.R. Cayan. 2005. Snow-fed streamflow timing at different basin scales: Case study of the Tuolumne River above Hetch Hetchy, Yosemite, California. Water Resources Research 41(7): W07005.

Lundquist, J.D., S.E. Dickerson-Lange, J.A. Lutz, and N.C. Cristea. 2013. Lower forest density enhances snow retention in regions with warmer winters: A global framework developed from plot-scale observations and modeling. Water Resources Research 49: 6356-6370.

Lyon, S.W., P.A. Troch, P.d. Broxton, N.P. Molotch, and P.D. Brooks. 2008. Monitoring the timing of snowmelt and the initiation of streamflow using a distributed network of temperature/light sensors. Ecohydrology 1: 215-224.

Mantua, N., I. Tohver, I., and A. Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Climatic Change 102:187-223.

Marks, D., J. Kimball, D. Tingey, and T. Link. 1998. The sensitivity of snowmelt processes to climate conditions and forest cover during rain-on-snow: a case study of the 1996 Pacific Northwest flood. Hydrological Processes 12: 1569-1587.

Martin, K.A., J.T. Van Stan, S.E. Dickerson-Lange, J.A. Lutz, J.W. Berman, R. Gersonde, and J.D. Lundquist. 2013. Development and testing of a snow interceptometer to quantify canopy water storage and interception processes in the rain/snow transition zone of the North Cascades, Washington, USA. Water Resources Research 49: 3243-3256.

Mateus, C., D.D. Tullos, and C.G. Surfleet. 2015. Hydrologic sensitivity to climate and land use changes in the Santiam River Basin, Oregon. Journal of the American Water Resources Association 51: 400-420.

McCabe, G.J., M.P. Clark, and L.E. Hay. 2007. Rain-on-snow events in the western United States. Bulletin of the American Meteorological Society 88(3): 319-328.

Milly, P.C.D., J. Betancourt, M. Falkenmark, R.M. Hirsch, Z.W. Kundzewicz, D.P. Lettenmaier, and R.J. Stouffer. 2008. Stationarity is dead: whither water management? Science 319: 573-574.

Molotch, N.P., P.D. Brooks, S.P. Burns, M. Litvak, R.K. Monson, J.R. McConnell, and K. Musselman. 2009. Ecohydrological controls on snowmelt partitioning in mixed-conifer sub-alpine forests. Ecohydrology 2:129-142.

Moore, G.W., B.J. Bond, J.A. Jones, N. Phillips, and F.C. Meinzer. 2004. Structural and compositional controls on transpiration in 40- and 450-year-old riparian forests in western Oregon, USA. Tree Physiology 24: 481-491.

Moore, G.W., J.A. Jones, and B.J. Bond. 2011. How soil moisture mediates the influence of transpiration on streamflow at hourly to interannual scales in a forested catchment. Hydrological Processes 25: 3701-3710.

Moore, R.D. and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association 41: 763-784.

Mote, P., A. Hamlet, and E. Salathe. 2008. Has spring snowpack declined in the Washington Cascades? Hydrology and Earth System Sciences 12: 193-206.

Mote, P.W., 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30(12): 1601.

Neary, D.G. 2000. Changing perspectives of watershed management from a retrospective viewpoint. Pp. 167-176. *In*: Ffolliott, P.F.; Baker, M.B. Jr. C.B. Edminster, C.B., M.C. Dillion, and K.L. Mora, K.L. (eds.) Land Stewardship in the 21st Century: the Contributions of Watershed Management. RMRS-P-13. USDA-Forest Service Rocky Mountain Research Station, Tucson, AZ. 445 pp.

O'Connor, J.E. and J.E. Costa. 2004. Spatial distribution of the largest rainfall-runoff floods from basins between 2.6 and 26,000 km^2 in the United States and Puerto Rico. Water Resources Research 40(1): W01107.

Oregon Department of Environmental Quality, 2018. Surface water drinking water source areas in Oregon. Oregon Department of Environmental Quality, Water Quality Division, Salem, OR. https://www.oregon.gov/deq/FilterDocs/swmap.pdf accessed 4/29/2020.

Perry, G., J. Lundquist, and R.D. Moore. 2016. Review of the potential effects of forest practices on stream flow in the Chehalis River Basin. Appendix A. Chehalis Basin Strategy Draft Programmatic Environmental Impact Statement (EIS). Washington Department of Ecology, Olympia, WA, 46 pp. https://fortress.wa.gov/ecy/publications/parts/1706017part27.pdf accessed 4/29/2020.

Perry, T.D. and J.A. Jones. 2017. Summer streamflow deficits from regenerating Douglas-fir forest in the Pacific Northwest, USA. Ecohydrology 10(2): eco.1790.

Pike, R.G. and R. Scherer. 2003. Overview of the potential effects of forest management on low flows in snowmelt-dominated hydrologic regimes. BC Journal of Ecosystems and Management 3: 1-17.

Rothacher, J., 1965. Snow accumulation and melt in strip cuttings on the west slopes of the Oregon Cascades. Research Note PNW-RN-23. USDA-Forest Service Pacific Northwest Forest and Range Experiment Station, Portland, OR. 7 pp.

Rothacher, J., 1970. Increases in water yield following clear-cut logging in Pacific Northwest. Water Resources Research 6: 653-658.

Rutter, N., R. Essery, J. Pomeroy, N. Altimir, K. Andreadis, I. Baker, A. Barr, P. Bartlett, A. Boone, H.P., Deng, H. Douville, E. Dutra, K. Elder, C. Ellis, X. Feng, A. Gelfan, A. Goodbody, Y. Gusev, D. Gustafsson, R. Hellstrom, Y. Hirabayashi, T. Hirota, T. Jonas, V. Koren, A. Kuragina, D. Lettenmaier, W.P. Li, C. Luce, E. Martin, O. Nasonova, J. Pumpanen, R.D. Pyles, P. Samuelsson, M. Sandells, G. Schadler, A. Shmakin, T.G. Smirnova, M. Stahli, R. Stockli, U. Strasser, H. Su, K. Suzuki, K. Takata, K.Tanaka, E. Thompson, T. Vesala, P. Viterbo, A. Wiltshire, K. Xia, Y.K. Xue, and T. Yamazaki. 2009. Evaluation of forest snow processes models (SnowMIP2). Journal of Geophysical Research-Atmospheres:114: D06111.

Safeeq, M., G.E. Grant, S.L. Lewis, and B. Staab. 2015. Predicting landscape sensitivity to present and future floods in the Pacific Northwest, USA. Hydrological Processes 29: 5337-5353.

Salemi, L.F., J.D. Groppo, R. Trevisan, J.M. de Moraes, W.D. Lima, and L.A. Martinelli, L.A., 2012. Riparian vegetation and water yield: A synthesis. Journal of Hydrology 454: 195-202.

Schnorbus, M. and Y. Alila. 2013. Peak flow regime changes following forest harvesting in a snow-dominated basin: Effects of harvest area, elevation, and channel connectivity. Water Resources Research 49: 517-535.

Segura, C., K.D. Bladon, J.A. Hatten, J.A. Jones, V.C. Hale, and G.G. Ice. 2020. Long-term effects of forest harvesting on summer low flow deficits in the Coast Range of Oregon. Journal of Hydrology 585: 124749.

Shaman, J., M. Stieglitz, M., and D. Burns. 2004. Are big basins just the sum of small catchments? Hydrological Processes 18: 3195-3206.

Stednick, J.D. 2008. Hydrological and biological responses to forest practices: The Alsea Watershed Study. Springer Science +Business Media, LLC, New York, NY. 330 pp.

Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology 176: 79-95.

Stednick, J.D. and C.A. Troendle. 2016. Hydrological effects of forest management. Pp. 192-203 *In*: Amatya, D., T. Williams, L. Bren, and C. de Jong (eds) Forest Hydrology: Processes, Management and Assessment. CABI, Boca Raton, FL. 307 pp.

Stewart, I.T., D.R. Cayan, and M.D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18: 1136-1155.

Storck, P., D.P. Lettenmaier, and S.M. Bolton. 2002. Measurement of snow interception and canopy effects on snow accumulation and melt in a mountainous maritime climate, Oregon, United States. Water Resources Research 38(11): WR001281.

Surfleet, C.G. amd A.E. Skaugset. 2013. The effect of timber harvest on summer low flows, Hinkle Creek, Oregon. Western Journal of Applied Forestry 28: 13-21.

Tague, C. and G.E. Grant. 2004. A geological framework for interpreting the low-flow regimes of Cascade streams, Willamette River Basin, Oregon. Water Resources Research 40: W04303.

Thomas, R.B. and W.F. Megahan. 1998. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon: A second opinion. Water Resources Research 34: 3393-3404.

Tohver, I.M., A.F. Hamlet, and S.Y. Lee. 2014. Impacts of 21st-century climate change on hydrologic extremes in the Pacific Northwest region of North America. Journal of the American Water Resources Association 50: 1461–1476.

Troendle, C.A. and R.M. King. 1985. The effect of timber harvest on the Fool Creek watershed, 30 years later. Water Resources Research 21: 1915-1922.

Troendle, C.A., M.S. Wilcox, G.S. Bevenger, and L.S. Porth. 2001. The Coon Creek water yield augmentation project: implementation of timber harvesting technology to increase streamflow. Forest Ecology and Management 143: 179-187.

Tullos, D., 2018. How to achieve better flood-risk governance in the United States. Proceedings of the National Academy of Sciences of the United States of America 115: 3731-3734.

Vano, J.A., B. Nijssen, and D.P. Lettenmaier. 2015. Seasonal hydrologic responses to climate change in the Pacific Northwest. Water Resources Research 51: 1959–1976.

Varhola, A., N.C. Coops, M. Weiler, and R.D. Moore. 2010. Forest canopy effects on snow accumulation and ablation: An integrative review of empirical results. Journal of Hydrology 392: 219-233.

Whitaker, A., Y. Alila, J. Beckers, and D. Toews. 2002. Evaluating peak flow sensitivity to clear-cutting in different elevation bands of a snowmelt-dominated mountainous catchment. Water Resources Research 38: 1172.

Winkler, R., D. Spittlehouse, and S. Boon. 2017. Streamflow response to clear-cut logging on British Columbia's Okanagan Plateau. Ecohydrology 10(2): eco.1836.

Winkler, R., D. Spittlehouse, S. Boon, and B. Zimonick. 2015. Forest disturbance effects on snow and water yield in interior British Columbia. Hydrology Research 46: 521–532.

Woltemade, C.J. and K.W. Potter. 1994. A watershed modeling analysis of fluvial geomorphologic influences on flood peak attenuation. Water Resources Research 30: 1933-1942.

Yu, X.J. and Y. Alila. 2019. Nonstationary frequency pairing reveals a highly sensitive peak flow regime to harvesting across a wide range of return periods. Forest Ecology and Management 444: 187-206.

Zegre, N., A.E. Skaugset, N.A. Som, J.J. McDonnell, and L.M. Ganio. 2010. In lieu of the paired catchment approach: Hydrologic model change detection at the catchment scale. Water Resources Research 46: W11544.

Zhang, M., X. Wei, and Q. Li. 2017. Do the hydrological responses to forest disturbances in large watersheds vary along climatic gradients in the interior of British Columbia, Canada? Ecohydrology 10: e1840.

Zhang, M.F. and X.H. Wei. 2014. Alteration of flow regimes caused by large- scale forest disturbance: a case study from a large watershed in the interior of British Columbia, Canada. Ecohydrology 7: 544-556.

Zhang, M., X. Wei, and Q. Li. 2016. A quantitative assessment on the response of flow regimes to cumulative forest disturbances in large snow-dominated watersheds in the interior of British Columbia, Canada. Ecohydrology 9: 843-859.

Chapter 5. Sediment and Turbidity

Kevin Bladon and Jeff Behan

Turbidity, which is a measure of water clarity as determined by the degree to which light is scattered by suspended solids in the water column, is often the most variable of all water quality constituents that are of concern to drinking water supply (Crittenden et al. 2012). Measurements of turbidity are generally used for process control and regulatory compliance, and as indicators for other water quality constituents of concern, such as bacteria *Giardia* cysts or *Cryptosporidium* oocysts (Crittenden et al. 2005). Turbidity does not necessarily indicate increased concentrations of pathogens, but the suspended solids provide refuge sites for the pathogens that make raw water more resistant to disinfection. Turbidity is also used as a surrogate for suspended sediment using established site-specific relationships.

Elevated sediment concentrations or yields, and associated increases in turbidity, in community water supply can challenge the ability of drinking water treatment operators to provide safe drinking water to communities and increase the economic costs associated with the treatment process (AWWA 1990; Borok 2014). Suspended sediment includes multiple solutes including organic matter, which can bind water contaminants and facilitate their transport. Specifically, increased suspended sediment and turbidity, and the suspended organic matter can increase the transport of nutrients, heavy metals, pesticides, and other toxic chemicals (Lick 2008; Bladon et al. 2014; Emelko et al. 2016), facilitate downstream pathogen transport (Dorner et al. 2006; Droppo et al. 2006; Wu et al. 2009), reduce the effectiveness of disinfection treatments (Lechevallier et al. 1981; Emelko et al. 2011; Leziart et al. 2019), contribute to the formation of disinfection by-products (DBPs) (Krasner et al. 2006; Singer 2006; Krasner 2009), and produce unpleasant taste and odor problems that can dramatically erode public confidence in drinking water safety (McGuire 1995; ODEQ 2010; Kehoe et al. 2015).

While the majority of treatment plants in Oregon appear to have the capacity to remove sediment and other turbidity causing constituents from source water, the effective reduction in turbidity is primarily determined by the available treatment technology in each plant (USEPA 1999; ODEQ 2010). While conventional treatment plants with advanced filtration systems can treat water with high and variable turbidity levels (>50 NTUs [Nephelometric Turbidity Units]), these types of systems are typically too expensive for most small communities in Oregon (ODEQ 2010). As such, many utilities in Oregon rely on pressurized filtration or slow sand filtration, which can be compromised at relatively low turbidity levels (e.g., < 10 NTUs). In these cases, some Oregon utilities have installed advanced filtration systems; however, these are expensive to install and maintain and can result in greater use of flocculent and coagulant with increased turbidity, resulting in increased costs to communities (ODEQ 2010; Borok 2014).

5.1. Effects related to access and harvesting

Suspended sediment has important influences on physical, chemical, and biological processes in streams (Lisle 1989; Gomi et al. 2005; Withers and Jarvie 2008). From a community water supply perspective, elevated sediment loads and associated turbidity can create challenges for the drinking-water treatment process by reducing the effectiveness of chlorination, increasing the likelihood of taste and odor issues, decreasing the operational life-span of reservoirs, and increasing treatment costs (Emelko et al. 2011; Hohner et al. 2016). Increased suspended sediment and turbidity in streams can also create many negative effects on aquatic ecosystem health (Newcombe and Macdonald 1991; Goode et al. 2012). As such, turbidity and associated sediment are considered primary pollutants, which are regulated in finished drinking water under the federal Safe Drinking Water Act (USEPA 2004; Borok 2014). In recognition of the importance of maintaining high water quality from source water catchments to help achieve the drinking water standards, turbidity water quality standards have also been developed (OAR 340-41-0036). Specifically, the turbidity water quality standard indicates that activities within a catchment can result in "no more than a 10 % cumulative increase in natural stream turbidities, as measured relative to a control point immediately upstream of the turbidity causing activity" (Borok 2014).

Given the many potential effects associated with too much sediment in water bodies, there has long been concern for increased sediment supply to streams due to forest management activities (Beschta 1978; Harr and Fredriksen 1988; Binkley and Brown 1993). In the U.S. Pacific Northwest (PNW), where forests and forest harvesting remain important for the economy, understanding the effects of current forest management practices on sediment and turbidity remains a challenge. In part, this is related to the difficulty in determining the background spatial and temporal patterns of suspended sediment and turbidity, as well as the response to disturbances (Fredriksen 1970; Harris and Williams 1971; Beschta 1978; Luce and Black 1999). In general, historical forest management practices, including road building, timber harvesting, and site preparation, led to exposure of mineral soils, decreased infiltration capacities of soils, disturbance of stream banks and channels, and increased erosion and fine sediment delivery to stream channels (Brown and Krygier 1971; Beschta 1978; Harr and Fredriksen, 1988; Binkley and Brown 1993). When conducted on steep slopes, these management practices have also been associated with significantly increased occurrence of landsliding and mass wasting, which can deliver large quantities of sediment to streams (Montgomery et al. 2000; Schmidt et al. 2001; Swanson and Dyrness 1975).

In response to the association of forest management practices with increased erosion and sediment inputs into streams, timber harvest regulations and best management practices (BMPs) were developed and implemented to reduce these sources of nonpoint source pollution (Ice 2004; Ice et al. 2004). For nonfederal timberlands in Oregon, these BMPs are codified in rules in the Oregon Forest Practices Act (FPA). Rules for perennial, fish-bearing streams generally focus on a designated riparian management area (RMA) along each side of the stream (that varies in width depending on stream size and other factors) where forest management activities are reduced or precluded. Rules for forest roads focus on locating the roads away from water bodies, and on routing runoff from the roads away from waterways. Since the 1960s, rules for fish-bearing streams and forest roads have been updated several times. However, non-fish-bearing streams do not have RMAs in most of western Oregon, while rules for forest management in steep, landslide-prone areas focus on safety for humans and their structures (Langridge 2011), and do not include provisions for protecting water quality.

Despite improved timber operations and evidence indicating that they are generally effective in reducing erosion and sediment delivery into streams (Cristan et al. 2016), there also continue to be inconsistent and even contradictory results from various studies regarding relationships between forest management, erosion and water quality (Aust and Blinn 2004; Anderson and Lockaby 2011; Cristan et al. 2016). Given that the focus of this review is on downstream community drinking water supplies, it is important to note that much of the uncertainty about the efficacy of current BMPs is partly associated with the many challenges associated with identifying the source of in-stream suspended sediment (Collins et al. 2017). Sources of suspended sediment often respond to complex interactions between numerous factors that influence sediment mobilization and delivery, resulting in high temporal and spatial variability, which can make categorical statements about the effects of forest management practices problematic (Grant and Wolff 1991; Collins and Walling 2004).

In general, downstream sediment transport is limited by the conveyance capacity of the upstream channels and floodplains (Trimble 1983). If this conveyance capacity is exceeded by sediment supply, then storage of sediment occurs (Reid and Dunne 2016). However, stored sediment can become remobilized during high flow events and increase sediment yield in the downstream direction (Bywater-Reyes et al. 2018). Additionally, while larger, heavier particles typically settle out of the water column first, smaller, fine-grained clay particles, which create the greatest challenges for downstream drinking water treatment, tend to remain suspended for longer periods of time and distances, contributing to downstream sediment and turbidity levels often many years after the upstream disturbance (Borok 2014; Emelko et al. 2016). Thus, along the course of a stream or river, suspended sediment concentrations and turbidity may increase or decrease due to many interacting factors. Despite an understanding of these fundamentals, the specific understanding of when and where forest management BMPs are likely to be successful at mitigating sediment delivery to water bodies remains limited (Edwards et al. 2016). This uncertainty is also, in part, because field based studies, which are necessary to collect representative data to further our understanding of the many interactions between forest management activities and large-scale, long-term sediment transport, have been on the decline because they are increasingly expensive and time consuming, and support for them has waned in recent decades (Burt and McDonnell 2015; Anderson and Lockaby 2011).

Below, we summarize recent literature (2000 - present) addressing the effects of forest harvesting activities on the delivery of suspended sediment and turbidity to water bodies. Contemporary forest BMPs, especially for forest roads and larger perennial and fish-bearing streams, have evolved rapidly in the 21st century (Cristan et al. 2016), and many questions remain about the effectiveness of these newer practices at mitigating effects on sediment delivery to streams. For this review, we have focused primarily on more recent research conducted in the PNW. Most of this research has been conducted at the smaller catchment scale, which is relevant to some public water systems, such as those in coastal Oregon that rely on smaller source watersheds and are closer to headwater areas. Unfortunately, as with many parameters, there have been few studies directly relating the effects of contemporary forest management practices on sediment and turbidity at the large basin scale (MacDonald and Coe 2007), which is relevant to larger drinking water treatment plants in the PNW. In reviewing the findings from previous research, it is important to keep in mind the paucity of studies at the larger basin scale, which can create uncertainty about how to interpret research results from the small catchment scale for implications to downstream drinking water treatment in systems with larger source watersheds.

5.2. Harvesting

Many studies have observed increases in runoff, soil erosion, and sediment delivery to streams due to forest management practices (Binkley and Brown 1993; Croke et al. 1999b; Megahan and King 2004; Gomi et al. 2005). The general harvest area (GHA; the area of tree harvesting, excluding primary skid trails and haul roads) generally represents the largest area of disturbance associated with forest harvesting activity, especially with the use of ground-based harvesting equipment (Miller et al. 1996; Ampoorter et al. 2012). Felling trees usually does not significantly disturb soils and expose mineral soils, but movement of logs across the ground to landings often does. It is generally known that heavy machinery, including harvesters, skidders, and forwarders, can compact soils, increase bulk density, and decrease air-filled porosity, infiltration capacity, and hydraulic conductivity across the GHA (Sidle et al. 2006; Mohr et al. 2013). However, these effects are spatially heterogeneous and difficult to study.

General harvest areas usually have patches of compacted soils interspersed with areas more similar to undisturbed forest floor. Runoff typically builds slowly in GHAs, even under heavy rainfall, usually starting on the more disturbed patches of the hillslope. But channelized flow tends not to develop in GHAs due to the high spatial variability in soil infiltration capacity, and presence of remaining vegetation and loose material on the soil surface. This patchy nature of runoff generation usually limits the ability of runoff in GHAs to mobilize large amounts of sediment (Croke and Hairsine 2006). Most available evidence suggests that forest roads, skid trails, log landings and slash burning are usually more likely to produce sediment than harvesting itself (Neary et al. 2009). Assessing nearly 200 harvest units in the Sierra Nevada and California Cascades, Litschert and MacDonald (2009) found that timber harvest alone rarely initiated large amounts of runoff and surface erosion, particularly when BMPs were utilized. Similarly, Megahan and King (2004) found that harvesting often had minor impacts on streams. Stednick and Troendle (2016) maintain that harvesting-related disturbances are usually disconnected from waterways, which reduces their potential for causing increases in sediment inputs. After harvesting, infiltration rates usually remain high enough in Pacific Northwestern forests to minimize infiltration excess overland flow and associated sediment movement.

However, there are gaps in the evidence base for this general finding, and exceptions associated with local conditions. Depending on factors that contribute to connectivity across the GHA, it may be significant source of sediment. For example, Reid et al. (2010) investigated the role of gullies in sediment production after logging, which they note is a rarely studied aspect of forest management. They found that second- cycle logging in Caspar Creek, California resulted in increased streamflow which appeared to have triggered coalescence of previously disconnected gullies that were themselves associated with first-cycle logging a century earlier, and extended these gullies significantly further upslope. They suggest that higher in-channel erosion associated with these changes compared to control sites is an important source of sediment in the logged sites, and one for which BMPs for riparian areas and roads would not be effective at reducing sediment inputs.

Few studies have explicitly quantified the proportional amount of sediment delivered directly to streams from GHAs and significant knowledge gaps remain regarding the relative importance of GHAs, skid trails and roads in contributing to overall suspended sediment concentration or turbidity (Croke and Hairsine 2006). In a study from Australia, Croke et al. (1999a, b) found that skid trails generally produce the majority of harvesting-related sediment and that GHAs tend to be sediment sinks, but noted challenges in modeling sediment production from GHAs, and in scaling up plot-level data. The relative importance of each source depends heavily on site-specific factors including geology and slope steepness, discussed in more detail below. On steep slopes, concerns over safety and higher logging costs have led to a shift away from cable yarding toward the use of tethered systems, where ground-based machinery is tethered to an anchor, usually upslope. In response to concerns about soil and water impacts associated with this new technology, Chase et al. (2019) compared soil disturbance and stream-adjacent disturbance of tethered logging and conventional cable harvest methods on steep slopes in Oregon and Washington. They found that tethered systems caused more soil disturbance than cable systems, but that impacts were still below applicable regulatory thresholds. The potential impacts of tethered logging systems on soil compaction, water routing, and associated sediment movement to streams are only beginning to be evaluated.

Anderson and Lockaby (2011) identified uncertainty of sediment sources associated with specific forest management activities as a critical research gap and suggested the use of nuclide or isotopic tracers in existing or future watershed studies to separate the various contributions to streams (Wallbrink and Croke 2002; Walling 2005). Better and more detailed information on the sources of fine sediment is critical for improving understanding of (a) the erosion and sediment delivery processes, (b) sediment-associated nutrient and contaminant fluxes, (c) the differential effects of specific sediment sources on aquatic ecosystem health, and (d) whether best management practices aimed at mitigating sediment transport to water bodies are effective (Walling 2013; Sear et al. 2016). As noted by Gomi et al. (2005), the primary external sources of sediment to streams include streambank erosion, mass movements (landslides and debris flows), roads and trails, and surface erosion on slopes of the general harvest area. The key internal sources of sediment to streams include material stored within the channel system, which may be remobilized during high flow events (Gomi et al. 2005).

Sediment stored within stream channels can originate from natural processes, from previous human land uses, or from some combination of these. Sediment eroded as a result of human land uses, or “legacy sediment” may be stored in rivers for decades or even centuries (James 2013; Wohl 2015). We found very little information regarding the residence times of sediments in Oregon streams potentially related to forest practices, perhaps due to the lack of baseline data on “natural” sediment loads, and the difficulty of distinguishing sediment contributed by other land uses such as agriculture and grazing. However, research from the Oregon Coast Range (Lancaster et al. 2010; Lancaster and Casebeer 2007) found that a significant portion of sediment from debris flows can remain in the valley bottoms of channels for many decades or centuries. And Koehler et al. (2007) found that the South Fork Noyo River watershed in coastal northern California contains large volumes of historic sediment that were delivered to channels in response to past logging operations and are presently stored beneath historic terraces and in present-day channels.

Regardless of the original sediment source or sources, increases in water yields and peak flows following forest harvesting can lead to increased suspended sediment and turbidity simply due to remobilization of stored in-channel sediment (Stednick 1996; Brown et al. 2005; Grant et al. 2008; Birkinshaw et al. 2011). Such changes in the hydrologic regime can increase in-channel sources of sediment via stream channel scouring, bedload mobilization, and remobilization of previously eroded materials that may be stored in the channel (Anderson and Lockaby 2011; Voli et al. 2013). A key point here is that while modern forest practices have clearly reduced ongoing inputs of sediment to stream channels in many cases, there may be substantial amounts of forestry-related sediments that entered Oregon streams during episodes of historic logging and which remain stored there, available for remobilization, just as Koehler et al. (2007) found in a northern California watershed. This remobilization could occur due to higher flows associated with current timber harvesting, or from infrequent large storms. Streamflow changes after harvesting are discussed in Chapter 4.

A recent study in the Oregon Coast Range used sediment source fingerprinting techniques (Walling 2005; Collins et al. 2010) to quantify the primary sources of suspended sediment in an unharvested, reference catchment and a harvested catchment (Rachels 2018). The primary sources of suspended sediments in the stream draining the harvested watershed were generally from streambank sources (90.2 ± 3.4 %), hillslopes (7.1 ± 3.1 %), and roads (3.6 ± 3.6 %). Similarly, the primary contributions of suspended sediment in the stream draining the reference watershed were streambanks (93.1 ± 1.8 %) and hillslopes (6.9 ± 1.8 %) (Rachels 2018). These findings were in agreement with previous studies from Georgia, USA (Fraser et al. 2012), North Carolina, USA (Voli et al. 2013), New Zealand (Basher et al. 2011), and Japan (Hotta et al. 2007), which all inferred from field observations and suspended sediment concentrations that streambanks could be the primary source of suspended sediment and highlight the importance of forest harvesting effects on the hydrologic regime. In harvested catchments, streambank contributions are often related to increased streambank destabilization associated with culverts, ditches, riparian vegetation disturbance, or stream crossings (Rashin et al. 2006).

Due to limitations in accurately determining sources of sediment in streams, the vast majority of research investigating forest harvesting effects on sediment and turbidity have simply focused on in-stream concentrations and yields. In one such study, Reiter et al. (2009) investigated spatial and temporal trends in turbidity using 30 years of water quality data from four locations in the Deschutes River watershed in western Washington (Figure 5.1). Importantly, for this review, the study included catchments at the small headwater scale (2.4 - 3.0 km^2) up to the larger basin scale of the Deschutes River (150 km^2). Overall, Reiter et al. (2009) provided strong evidence for a correlation between annual percent catchment harvested and the median flow adjusted turbidity during winter (p = 0.0002) and spring (p = 0.0281). Similarly, they also provided strong evidence (p = 0.0027) that median flow adjusted turbidity was correlated with the percent of annual road network constructed in the catchments. At the larger basin scale, turbidity, flow adjusted turbidity, and suspended sediment concentrations were all generally greater than observed at the headwaters scale. However, the authors did not explicitly link upstream to downstream in a manner that would facilitate assessment of the implications to community drinking water. Interestingly, though, across all sites, trend analysis provided strong evidence that for similar harvest levels the winter flow adjusted turbidity had declined in more recent years of the study relative to earlier in the record (p = 0.020). There was not a similar declining trend observed in the record for the spring flow adjusted turbidity. The authors primarily attributed the significant decreasing trend in the winter flow adjusted turbidity to improvements in road construction

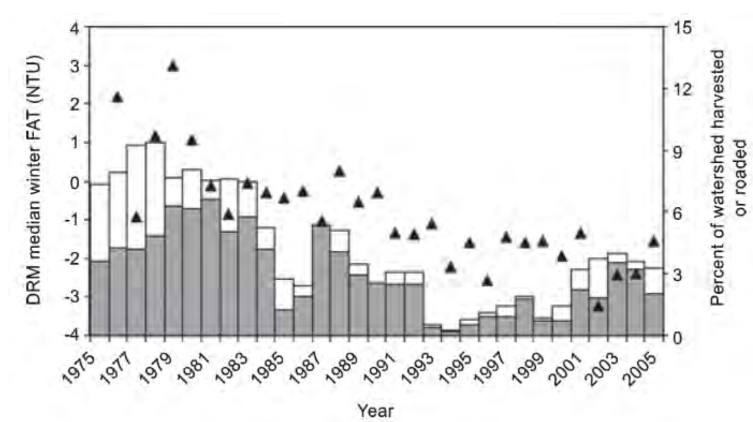


Figure 5.1. Median monthly flow adjusted turbidity (black triangle), percent of catchment harvested (grey bar), and percent of road network constructed (white bar) at the Deschutes River (WA) mainstem site. Note: percent road network constructed in 1988 was unavailable. (Reiter et al. 2009).

and maintenance practices (Reiter et al. 2009). However, it is important to note that the authors also indicated the challenges associated with isolating the specific factors contributing to the trends in turbidity due to "complex interactions of land use, landform, and natural disturbance as well as the manner in which the study was designed" (Reiter et al. 2009). Interestingly, across the entire study period, Reiter et al. (2009) also found winter turbidity values were greater in streams draining catchments dominated by more friable (easily crumbled) geology compared to streams draining catchments consisting of more resistant volcanic geologies (Figure 5.2).

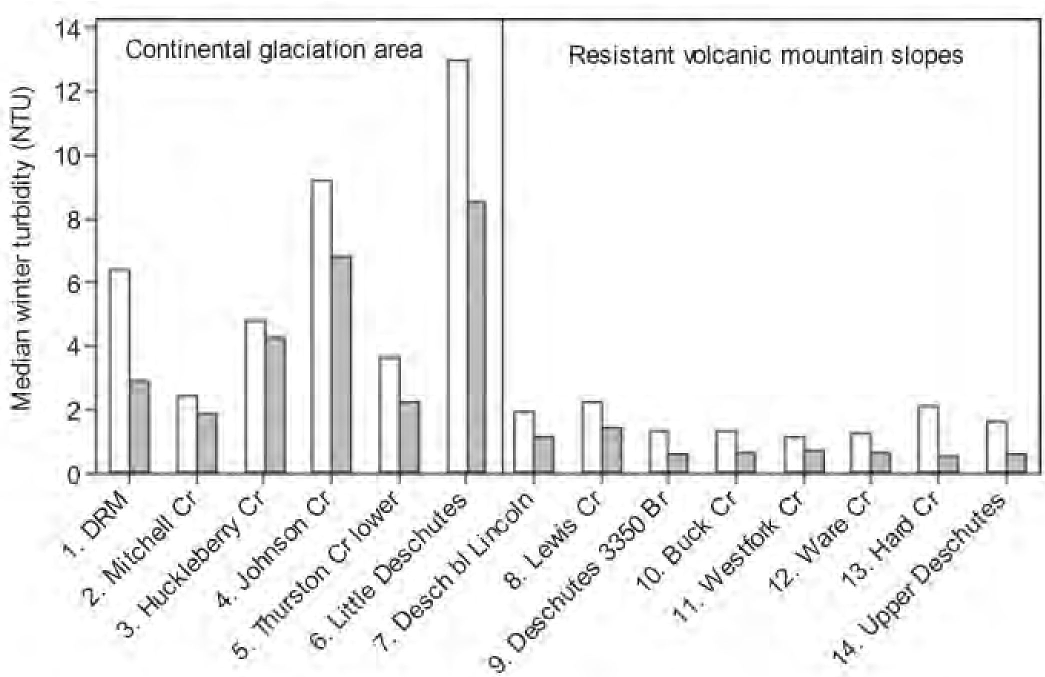


Figure 5.2. Median winter turbidity from grab samples collected during a period of high harvest activity (1982; white bars) and lower harvest activity (1997; gray bars) separated by catchments underlain by more friable lithology (continental glaciation area) and more resistant lithology (resistant volcanic mountain slopes) (Reiter et al. 2009).

Similarly, Bywater-Reyes et al. (2017) also found that the differences in the suspended sediment yield response to forest harvesting at the Trask River Watershed study in the Coast Range of western Oregon were primarily driven by catchment geology and physiography. Across six years of data from 10 sites, they found the greatest increases in suspended sediment yields after forest harvesting (up to an order of magnitude) occurred in streams draining catchments with more friable lithology (e.g., sedimentary) (Figure 3). Comparatively, catchments underlain by more resistant lithology (e.g., intrusives) had lower suspended sediment yields and were more resilient to the effects of forest management activities (Bywater-Reyes et al. 2017). They also observed increases in suspended sediment yields in three of the 10 headwater catchments (26.4 - 37.8 ha), which were harvested with contemporary forest harvesting practices in the first year after harvesting, with sediment yields increasing annually in one catchment (clearcut without a riparian buffer) for the remaining three years of the study. Consistent with the study by Reiter et al. (2009) in Washington state, Bywater-Reyes et al. (2017) also generally observed the highest sediment yields at the downstream sites, reflecting an accumulation of sediment from the upstream, headwater catchments (Figure 5.3).

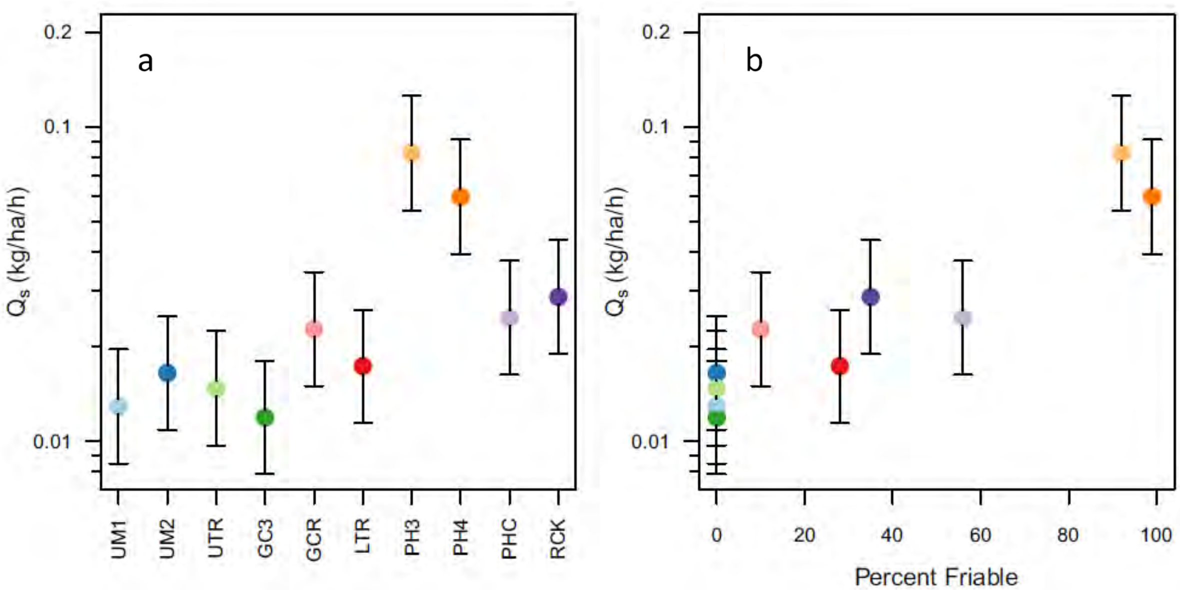


Figure 5.3. Annual suspended sediment yields in each catchment in the Trask Watershed Study as a function of (a) contributing area (catchments ordered from upstream to downstream) and (b) friability of catchment lithology (Bywater-Reyes et al. 2017).

In a follow-up study using ~60-years of data in 10 temperate mountain watersheds (8.5 - 6,242 ha) in the H.J. Andrews Experimental Watershed in the Pacific Northwest, Bywater-Reyes et al. (2018) investigated the relationship between catchment setting (i.e., lithology and physiography), forest management activities, and suspended sediment yields. Overall, annual suspended sediment yields were highly variable, fluctuating almost four orders of magnitude across the 10 catchments and through time. While the study catchments included a range of lithologies, including hydrothermally altered pyroclastic flows, welded ash-flow tuff, and ridge-capping andesite lava flows, this was a less dominant factor in driving differences in sediment yields across catchments (Bywater-Reyes et al. 2018). Rather, watersheds with greater slope variability (roughness) were more likely to have greater suspended sediment yields and tended to be less resilient to erosion and sediment delivery to streams following both natural and anthropogenic disturbances (Bywater-Reyes et al. 2018).

Richardson et al. (2018), in a unique study investigating downstream sediment transport, cross-dated ~1,500 years of sediment from cores collected from Loon Lake in the Oregon Coast Range. During a time of peak forest harvesting in the region (1939 - 1978), which coincided with a cool wet phase of the Pacific Decadal Oscillation, sedimentation rates in the lake were ~0.79 g cm^{-2} y^{-1} (0.74 - 0.92, 95 % C.I.). However, during a more recent time period (1979 - 2012), which coincided with the passing of the Oregon Forest Practices Act legislation that regulated harvesting practices in the region, sedimentation rates declined to 0.58 g cm^{-2} y^{-1} (0.48 - 0.70). The study by Richardson et al. (2018) illustrated how historical forest harvesting activities primed the landscape and lowered the threshold for sediment delivery during the high stream flow events that occurred at the end of the early study period. The study also appeared to provide evidence that forest harvest practices have improved such that sediment delivery to streams in forested headwater regions and subsequent downstream transport have substantially declined. However, it is critical to note that strong differences in climate between the historical (wet and cool) and contemporary (warm and dry) periods precluded the authors from definitively disentangling the effects of timber harvesting from climate (Richardson et al. 2018).

The paired watershed approach allows for evaluation of the role of forest management while controlling for some climate effects. In another unique study in the Oregon Coast Range, Hatten et al. (2018) returned to the same watersheds that were harvested in 1966 as part of the Alsea Watershed Study (Stednick 2008), to investigate the effects of contemporary forest harvesting. In the original Alsea Watershed Study, forest harvesting without riparian buffers, road building, and slash burning led to ~2.8-times more sediment in the streams draining the harvested catchments compared to the unharvested (reference) catchment (Brown and Krygier 1971; Beschta 1978; Hall 2008). Specifically, sediment yields increased in the post-harvest period by 253 % in Needle Branch (no buffers) and 117 % in Deer Creek (buffers) compared to the pre-harvest periods (Beschta and Jackson 2008). However, the recent harvesting practices in Needle Branch differed from the historical harvesting practices in several key ways, including: retention of vegetation as riparian stream buffers, smaller harvest units, no broadcast burning, and retention of woody materials in the stream channel. Road practices also changed. As a result of these shifts in practices, the more recent study illustrated that annual sediment yields in Needle

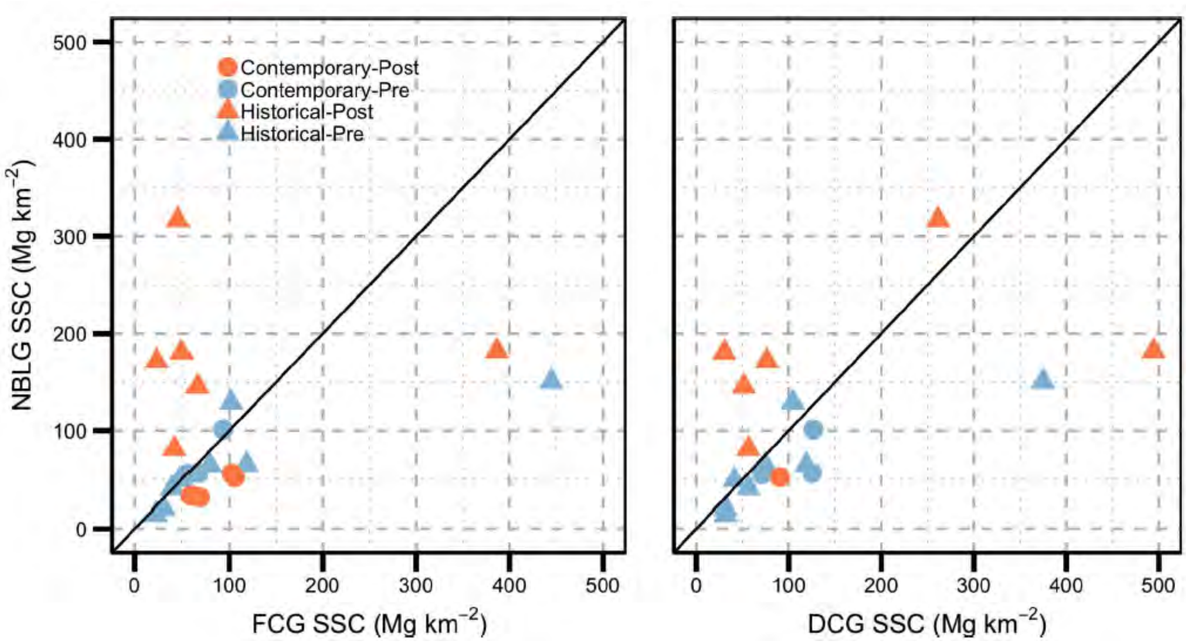


Figure 5.4. Relationships between annual suspended sediment yields in the reference catchments (Flynn Creek, FCG; Deer Creek, DCG) compared to the harvested catchment (Needle Branch, NBLG) during the historical and contemporary pre- and post-harvest periods from the Alsea Watershed Study (Hatten et al. 2018).

Branch (buffers on small-fish streams, none on non-fish streams) were lower than in Flynn Creek (reference catchment) after contemporary forest harvesting with BMPs (Figure 5.4). In fact, Flynn Creek (reference) often had the highest sediment yields, 55 - 313 Mg km^{-2} yr^{-1}, followed by Deer Creek (no contemporary harvests) at 69 - 127 Mg km^{-2} yr^{-1}, and Needle Branch (buffers on S/F, none on S/N) at 31 - 102 Mg km^{-2} yr^{-1}. The concentrations and yields of suspended sediment observed in Needle Branch after contemporary harvesting were similar to historical pre-treatment levels. As such, Hatten et al. (2018) found no evidence that contemporary harvesting techniques affected suspended sediment concentrations or yields. Overall, our understanding of the magnitude, duration, physical processes, and downstream consequences associated with both short- and long-term increases in turbidity and sediment in headwater streams after forest harvesting remains incomplete. There are many examples of improvements in forest harvesting practices, including riparian buffers, smaller harvest units, and less intensive site preparation practices (e.g., broadcast burning), which have reduced headwater-scale erosion, suspended sediment, and turbidity. However, there may be instances where current BMPs are imperfectly implemented. As Rashin et al. (2006) note, both the implementation of forestry BMPs and the erosion and sediment transport processes they are designed to address are highly variable. Moreover, current BMPs do not explicitly address the effects of tree removal on hillslope hydrologic changes, catchment water balance, or loss of root strength from decay (Klein et al. 2012; McDonnell et al. 2018). In Oregon, small non-fish streams in general and non-fish bearing streams in the upper reaches of drinking water source watersheds remain unprotected. Rashin et al. (2006) state that preventing sediment delivery to and physical disturbance of non-fish bearing streams is important in order to prevent impacts to water quality downstream. There is also evidence that some catchments are simply more susceptible to increased erosion and sedimentation following forest harvesting (e.g., Bywater-Reyes et al. 2017; 2018).

Additionally and importantly, if fine sediment is introduced into streams, it is more likely to be delivered downstream compared to coarse sediment, woody debris, or changes in water temperatures (MacDonald and Coe 2007), but this aspect of sediment mobilization and transport has been rarely quantified. One exception is a study by Jackson et al. (2001) which evaluated particle-size distributions of bed material in 15 first-or second-order Washington Coast Range streams (small streams without salmonid fish) in and nearby commercial timber harvest units prior to and immediately following harvest. Four unharvested basins served as references; 5 basins had some type of buffer and 6 basins were clearcut to the channel edge. Buffer widths were dictated by operational considerations and averaged from 15 to 21 meters; the narrowest was 2.3 meters on one side of a stream. In the clearcut streams, slash in the channel trapped fine sediment there by inhibiting fluvial transport. Fine sediment increased after harvest in 5 of the 6 unbuffered streams, an average of from 12 % to 44 %, and attributed primarily to small bank failures caused by logging operations. Only 1 of 5 buffered streams (which received drainage from a logging road and landing) showed increased fines, and unharvested reference streams showed similar or reduced fines.

Moreover, current BMPs do not explicitly address the effects of tree removal on hillslope hydrologic changes, catchment water balance, or loss of root strength from decay (Klein et al. 2012). As frequently noted in reviews and syntheses of knowledge regarding relationships between forest practices, sediment and water quality (e.g. Anderson and Lockaby 2011), there is a general paucity of research at the larger basin scale, occasionally due to confounding cumulative effects, which creates uncertainty about how to apply research results from the small catchment scale to larger areas. But catchment scale research is relevant to smaller drinking water source watersheds and community water systems that rely on them. As a result of modern BMPs, sediment production from forest operations appears to be much less frequent, but still occurs in areas with certain types of erodible soil and rock, in steeply-sloped watersheds, and in areas with substantial soil disturbance. In all of these instances, impacts are exacerbated during large storms, especially if they occur immediately after harvesting.

In an unusual study that did generate larger-scale findings, Wheatcroft et al. (2013), using sediment cores and ^{210}Pb geochronology, detected the cumulative effects of timber harvesting at the basin scale in continental shelf sediments of the Pacific Ocean off the Umpqua River, expressed as an increase in sediment accumulation and a shift in sediment grain size toward finer particles. These findings are discussed in more detail below in the section on landsliding.

5.3. Roads (fill failures, chronic sediment, hydrologic connectivity)

Despite many economic and social benefits of forest roads, they also represent a potential hazard to hydrologic, geomorphic, and ecologic processes (Jones et al. 2000; Baird et al. 2012). In particular, unpaved forest roads have long been considered one of the primary sources of suspended sediment and elevated turbidity in streams (Brown and Krygier 1971; Beschta 1978; Reid and Dunne 1984; Lane and Sheridan 2002; Gomi et al. 2005). In the western United States, it has been estimated that 18 - 75 % of forest roads are hydrologically connected to the stream network (Coe 2006). Because roads are nearly impervious surfaces they often lead to increased overland flow, which can cause chronic fine sediment contribution to nearby streams, lakes, and reservoirs (Luce 2002). Moreover, when coupled with forest harvesting or active hauling, sediment delivery to water bodies is often magnified (Bilby et al. 1989; Ziegler et al. 2001).

Impacts of roads range from chronic and long-term contributions of fine sediment into streams to catastrophic mass failures of road cuts and fills during large storms (Beschta 1978; Wemple et al. 2001; Sidle and Ochiai 2006). Many studies have shown an increase in sediment availability and delivery to streams with greater road traffic due to crushing, abrasion, and forcing of fine sediment to the surface (Ziegler et al. 2001; Sheridan et al. 2006; Sosa-Perez and MacDonald 2017a). Additionally, the lateral redistribution of runoff from roads can decrease slope stability and increase peak flows in small streams, ultimately leading to greater occurrence of mass movements or elevated in-channel erosion and sediment transport (Brown and Krygier 1971; Beschta 1978; Montgomery 1994; Croke and Mockler 2001). Indeed, there is evidence that the majority of sediment delivered to water bodies from roads is related to episodic, mass movement events (Swanson et al. 1987; Mills, 1997; Fransen et al. 2001). However, the actual magnitude and longevity of effects of forest roads on suspended sediment in streams depends on many site-specific factors, including traffic, geology, road grade, road connectivity to the stream, and sediment availability for transport (Grant and Wolff 1991; Benda and Dunne 1997; Hassan et al. 2005).

The effects of roads on forest hydrology and causes of their sediment impacts include:

- Low permeability of the road surface to intercepted rainfall and overland flow;
- The susceptibility of road cutbanks and fill-slopes to erosion from rainfall and overland flow;
- Changes in how subsurface water moves downslope; e.g. interception by cutbanks and conversion to faster surface flow;
- Concentration of overland flow, either on the surface or in adjacent ditches, channels or culverts;
- The construction and maintenance of stream crossings;
- Diversion or rerouting of water from natural surface drainage paths; and
- Undercutting and overloading of steep slopes which contributes to increased landsliding (Stednick and Troendle 2016; Chang 2012; Wemple and Jones 2003; Guthrie 2002; Gucinski et al. 2001).

Indeed, the amount of road use as well as road density, have been previously shown to be major factors in delivering fine sediment to streams (Bilby et al. 1989; Luce and Black 1999; Dubé et al. 2004). Recently, Araujo et al. (2014) developed a simulation model from time series data of hydrologic variables, suspended sediment, and road and terrain characteristics to quantify suspended sediment concentration (SSC) generated from forest roads in medium sized coastal watersheds of British Columbia and the broader PNW. Their results also illustrated that road traffic was a more important factor than road density in the delivery of fine sediment from roads to streams (Figure 5.5). As an example, their model projected a ~12 mg l^{-1} increase in SSC with moderate use of roads and an increase in road density from 15 % to 30 %. Comparatively, they projected a ~55 mg l^{-1} increase in SSC with heavy use of roads with the same increase in road density. Similarly, Miller (2014) observed a 3.3-times increase in sediment yield from forest roads in Hinkle Creek, OR if logging trucks drove on the segments during the week prior to a storm. However, there was high

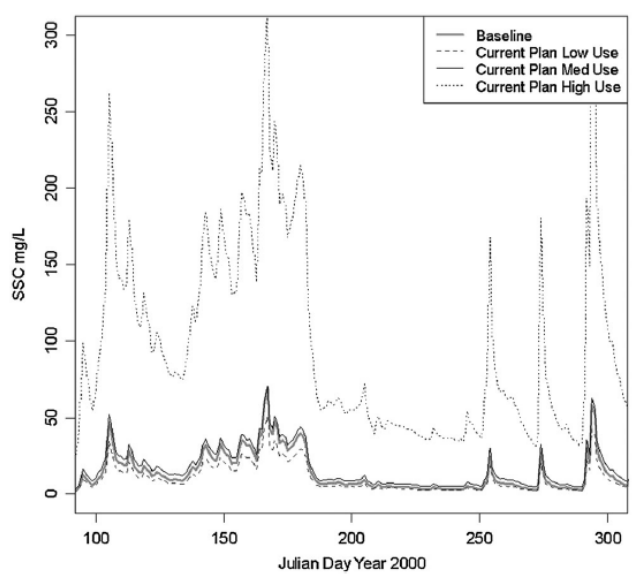


Figure 5.5. Simulation model results illustrating the potential effect of road use on mean daily suspended sediment concentration (Araujo et al. 2014).

variability (95 % CI 1.9 - 4.7-times increase) among road segments and between storm events (Miller 2014). This is consistent with several other studies in the PNW, which have shown 2- to 130-times more sediment from forest roads with heavy traffic compared to roads with little to no logging truck traffic (Reid and Dunne 1984; Bilby et al. 1989; Luce and Black 1999; Luce and Black 2001; Sugden and Woods 2007).

Similarly, the frequency of road maintenance operations can be a critical factor influencing the amount of sediment delivered from roads to ditches and streams. Maintenance of the roadbed is critical to prevent rut formation, overland flow, and road erosion (Burroughs Jr. and King 1989; Ziegler et al. 2001). However, this type of maintenance is achieved by periodic grading, which was shown on forest roads in the Oregon Coast Range to result in breaking up of the armor layer, increasing the sediment supply, and temporarily increasing sediment yields from roads to streams (Luce and Black 1999). Such increases in sediment yields are often short-lived. As the armor layer redevelops, sediment yields have been shown to decline as an exponential decay function, with reported declines in sediment yields of ~63 - 89 % in the second year and 86 - 99 % in the third year after grading (Megahan and Kidd 1972; Megahan 1974; Luce and Black 2001; Sugden and Woods 2007).

The type and quality of road surfacing material, as well as the erodibility of the underlying parent material (soil and geology), can also have large effects on erosion and sediment yields from roads. For example, Brown et al. (2014) observed 2.6 - and 3.5 - times higher median suspended sediment concentrations in road surface runoff from unsurfaced (native) roads compared with suspended sediment from roads with low gravel and high gravel surfaces, respectively. Comparatively, Luce and Black (1999) observed 9-times greater sediment yields from roads covered with aggregate on a fine textured silty clay loam base compared to roads constructed on a coarser, gravelly loam in the Coast Range of Oregon. This is consistent with most research, which has shown that erosion from roads tends to be highest in regions where soils are silt dominated, while erosion rates in regions with clay dominated soils are intermediate, and lowest in gravel dominated regions (Burroughs Jr. and King 1989; Dubé et al. 2004). In forested, mountainous regions the majority of road prisms are graded into the sub-soil—as such, in these regions the local geology is often the dominant factor affecting sediment yields from roads. Summarizing results from 15 studies and 10 parent materials in the PNW, Dubé et al. (2004) showed the highest rates of road erosion tended to occur in weathered granite, fine-grained or deeply weathered sedimentary, ash, and tuff dominated geology.

Due to the many potential effects of forest roads on sediment delivery to streams, there have been substantial efforts over the last several decades to modify forest road construction, road maintenance, and hauling practices (Gucinski et al. 2001; Wear et al. 2013; van Meerveld et al. 2014). In particular, practices have changed in many regions to reduce hydrologic connectivity of roads to streams by routing runoff from roads and into the forest as rapidly and frequently as possible (Gillies 2007; Baird et al. 2012). Further improvements in forest management practices aimed at reducing sediment delivery to water bodies include locating roads further away from streams, avoiding impacts to natural drainage patterns, minimizing total area disturbed by roads, avoiding steep slopes (>60 %), avoiding wet areas, limiting the number of stream crossings, using less erosive surfacing material, and providing more frequent road maintenance (Keller and Sherar 2003; Wear et al. 2013). Maintenance to repair damaged drainage structures or mitigate obvious sediment source points can reduce sediment production, while frequent grading or ditch cleaning may exacerbate it. Additional mitigation efforts include the use of sediment traps in ditches to dissipate energy and reduce sediment transport and the installation of ditch-relief culverts (Reiter et al. 2009). Below, we summarize the findings from current research from the Pacific Northwest investigating the efficacy of current road construction and maintenance practices at mitigating sediment transport to streams.

Reiter et al. (2009) used a water quality dataset collected over 30 years at four locations in the Deschutes River watershed (western Washington) to examine the role of forest management practices on turbidity and suspended sediment transport in streams. Increases in median monthly turbidity and the highest maximum monthly turbidity values tended to coincide with periods of active road construction (Reiter et al. 2009). In all four sub-catchments, road upgrades over the course of the study included: (a) use of less erosive surfacing material, (b) limited wet weather hauling, (c) outsloping of road surfaces, use of water bars or frequent ditch-relief culverts for the rapid diversion of water off roads surfaces, out of ditches, and onto the forest floor to facilitate infiltration, and (d) use of sediment traps and energy dissipation at relief culvert outlets (Reiter et al. 2009). These sediment control efforts applied to the road system, in part, contributed to a consistent decline in suspended sediment and turbidity over the 30 year study. Reiter et al. (2009) also attributed the reduction in sediment and turbidity to a consistent decline in road use over time.

Toman and Skaugset (2011) compared alternative designs of the pavement for unbound aggregate forest roads designed to specifically to minimize turbid runoff caused by subgrade mixing during wet-weather hauling. Alternative designs influenced sediment production but results were not consistent. The treatments produced different results across different research locations and there was no statistically significant treatment effect, suggesting that fine sediment in surface runoff did not originate from the subgrade but rather from the surface aggregate. Toman and Skaugset (2011) suggest that to minimize sediment production from forest roads, managers should be concerned with the unbound aggregate pavement rather than the subgrade. Also, they found that road segments that developed ruts produced considerably more sediment than road segments where ruts did not form, suggesting that managers should design the aggregate pavement to resist rut formation and also consider the availability of fine sediment in the aggregate.

A recently completed study by Arismendi et al. (2017) assessed both suspended sediment concentration (SSC) and turbidity in five non-fish bearing streams in the Coast Range of Oregon. Uniquely, they quantified SSC and turbidity both above and below road crossings during three successive time-periods, including before road construction/maintenance, after road construction/maintenance, and after forest harvesting and hauling. Many roads existed previously and were reconditioned, improved, or surfaced. Counter to their hypothesis, Arismendi et al. (2017) did not find strong statistical evidence that SSC or turbidity increased at the downstream sites relative to the upstream sites after road construction/maintenance, forest harvest, or hauling. In another analysis, focused on suspended sediment yields (SSY) at the sub-catchment scale from some of the same sites, Bywater-Reyes et al. (2017) also found no evidence for increases in SSY associated with roads. Moreover, Arismendi et al. (2017) also concluded that the absolute magnitude of change in SSC after road improvements, forest harvest and hauling in the treatment sites was small and likely had minimal biological relevance. Interestingly, the greatest concentrations of suspended sediment and turbidity occurred in their unharvested reference site, which they attributed to an exposed tree root-wad in the stream channel due to windthrow (Arismendi et al. 2017). As a result, they suggested that similar local disturbances in headwater streams, which often occur during discrete spatial and temporal events, could dominate the SSC and turbidity response in headwater streams (Benda and Dunne 1997; Benda et al. 2004; Arismendi et al. 2017). While this study provided evidence that current BMPs associated with forest roads may be effective at mitigating sediment transport to streams, the authors caution against broad generalizations from their findings due to the high spatial and temporal variability in SSC and turbidity they observed across a small number of study catchments (Arismendi et al. 2017).

Road upgrades and improved BMPs associated with road building have shown promise for decreasing sediment delivery to streams. However, most PNW watersheds contain an interconnected mosaic of older and newer roads designed to different standards, sometimes for different purposes, and crossing terrain of differing sensitivities to erosion and mass wasting. The particular pattern and hydrologic connectivity of this mosaic of road segments has implications for how it will interact with the forest watershed, streams, and other downstream water uses (Endicott 2008). Older so-called “legacy roads” are often the primary source of sediment due to poor water and grade control, as well as road location (Brown et al. 2014). In western Oregon forests, Luce and Black (1999) found high variability in sediment production from road segment to road segment with most segments producing little sediment, and a few key segments producing a great deal. Longer, steeper road segments, cutslopes without vegetation, cleaned ditches, and finer-grained soils were all associated with much higher sediment production.

Using the Washington Road Surface Erosion Model (WARSEM), Sugden (2018) modeled changes in sediment delivery to streams in response to systematic BMP upgrades to a 28,000 km legacy forest road network in western Montana and northern Idaho. The roads were on Plum Creek Timber Company lands where BMPs were applied over time in response to BMP legislation, Sustainable Forest Initiative (SFI) requirements and a Native Fish Habitat Conservation Plan (NFHCP) agreement with the US Fish and Wildlife Service (USFWS). Key BMPs included installing more frequent road drainage features, managing public road access, increasing road surface vegetative cover, and installing supplemental filtration near streams. The WARSEM modeling was locally validated based on comprehensive field surveys which indicated that sediment delivery in these watersheds is dependent on site-specific BMP conditions and that most such delivery occurs at a minority of crossing locations. Results from 10 repeated watersheds (inventoried and modeled before and after BMPs) estimated that sediment delivery (weighted by watershed road length) was reduced by 46 % (watershed range:

- 84 % to +57 %) over a 10 - 15-year period. Delivery rates from these watersheds were similar to an additional 22 watersheds inventoried after BMP upgrades were completed.

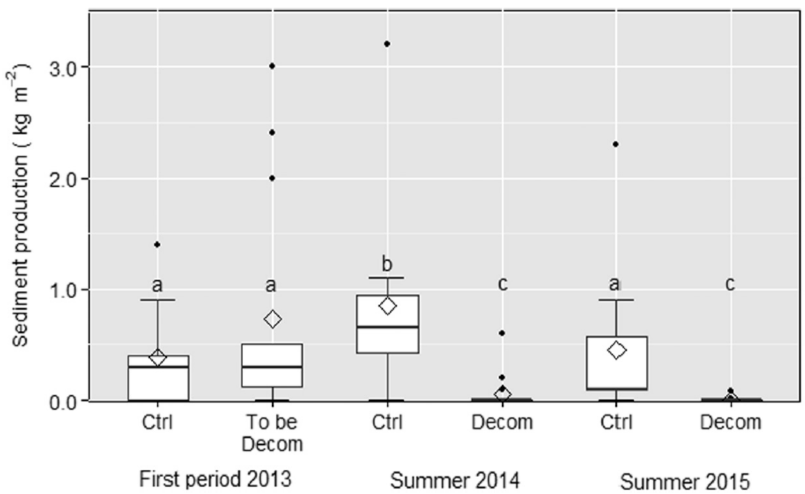


Figure 5.6. Sediment production (kg m^{-2}) during three time periods (before decommissioning, first year after decommissioning, second after decommissioning) from control (Ctrl; n = 10) and decommissioned (Decom; n = 19) road segments. Different letters above the boxplots indicate statistically significant differences (Sosa-Perez and MacDonald, 2017b).

Oregon agencies including DEQ and ODF are further distinguishing between "legacy roads"—those built and abandoned before the Oregon Forest Practices Act (and therefore not regulated by it), and "old roads"—those built before current road standards but still in use. Road deactivation, especially of legacy roads, is often suggested as a way to potentially decrease road density, erosion, and sediment delivery to streams (Switalski et al. 2004). Deactivation implies an attempt to both limit road access, but also to reestablish some of the natural hydrogeomorphic characteristics of the site (Allison et al. 2004). Thus, treatments may include gating or permanent traffic barriers, ripping of the roadbed, restoration of stream crossings, or full road recontouring (Switalski et al. 2004). A comparison of three erosion control mulches on decommissioned forest road corridors in the northern Rocky Mountains (Foltz 2012) showed that wood based alternatives are as effective at reducing sediment production as straw, and that the amount of effective ground cover provided by mulch, plants, and litter appeared to be more important than the type of mulch. A recent study in Colorado found that ripping of the roadbed was effective at trapping almost all of the eroded sediment (Figure 5.6) (Sosa-Perez and MacDonald 2017a, b). However, deactivation treatments are not always effective. In a northern California study, Madej (2001) observed no detectable erosion on 80 % of treated road reaches, but observed road fill failures on 20 % of road reaches after a 12-year recurrence interval storm event.

Again, there are many research questions on road deactivation and restoration that remain to be addressed, and knowledge regarding mechanisms for the effectiveness of specific BMPs remains limited. There is a pressing need to identify where sediment originates, understand why and how sediment delivery is controlled, and explain exactly how BMPs protect water quality. Understanding these mechanisms and differences between short- and long-term effectiveness will move the science toward the ability to develop the most effective site-specific BMP prescriptions (Edwards et al. 2016). For example, replicated research is needed across various temporal and spatial scales, topographies, soil types, and climates to more fully understand the benefits of road decommissioning (Switalski et al. 2004). Additionally, given the associated costs and uncertainty around effectiveness, additional attempts to develop decision trees and other prioritization methods to facilitate decision-making by forest resource managers about which road segments to consider for deactivation or restoration may prove valuable (Thompson et al. 2010). For example, the Geomorphic Road Assessment and Inventory Package (GRAIP) is a process and a set of tools for analyzing the impacts of road systems in forested watersheds in terms of erosion and sediment delivery to streams. The GRAIP is a collaboration between the USFS Rocky Mountain Research Station and Utah State University, and can be locally calibrated in a repeatable fashion with minimal effort. It combines a road inventory with a powerful GIS analysis tool set to predict sediment production and delivery, mass wasting risk from gullies and landslides, stream diversion potential, culvert maintenance, and fish passage at stream crossings. The road inventory protocol describes how to systematically field inventory a road system using GPS and automated data forms. Quality checked data can then be analyzed in a program implemented in ArcGIS, producing a map of surface erosion, accumulated road sediment in streams, and contributing length by segment, which relates directly to slope stability and gullying risks (Black et al. 2012).

In another example, Takken et al. (2008) present a methodology based on the principle of hydrological connectivity to evaluate the risk of road-derived runoff delivery. Their process allows estimation of runoff volume that may reach a stream through each of three different delivery pathways - stream crossings, gullied pathways and diffuse pathways - during a one in 10 year, 30 minute event. Degree of connectivity of a road depends on catchment characteristics such as topography, road placement, drain spacing and road and drainage density. Risk assessment maps outlining the distribution of different delivery pathways within a catchment are used to assess potential runoff connectivity, highlight hot-spots for runoff and sediment delivery, and evaluate different procedures for road rehabilitation or deactivation. Some such decision support tools have attempted to include estimates of the potential costs to community drinking water treatment facilities due to increased sediment inputs to the water supply (Allison et al. 2004)—these efforts could continue to be refined.

5.4. Site preparation effects on soils and erosion

The Oregon Forest Practices Act (FPA) stipulates that after heavy thinnings or clearcuts, industrial timberlands must be replanted to trees within 24 months. Prior to replanting, activities are usually conducted to reduce vegetation that competes with tree seedlings, reduce habitat for animals that damage seedlings, and to create spots for planting (Fitzgerald 2008). To reduce wildfire risk and increase plantable area, site preparation usually includes treatment to reduce the amount of slash (limbs, tops and poor quality logs) leftover from harvest operations. Site preparation can involve the use of herbicides, mechanized equipment, fire or some combination of these methods.

In the past, site preparation in western Oregon was usually done via broadcast burning. There are longstanding concerns about the impacts of this activity on forest soil protective layers and capacity for infiltration (e.g. Isaac and Hopkins 1937) and its contributions to erosion (e.g.

Bennett 1982; Beschta and Jackson 2008). Slash burning often exposes the mineral soil by consuming forest floor material and severe fires can cause soils to become hydrophobic, increasing the chances of sediment production (Neary et al. 2000). Under current practices, slash is usually piled prior to burning (Fitzgerald 2008), which significantly reduces the areal extent of exposed mineral soil, and slash fires in general are used less extensively than in the past (Swanson et al. 2000). In some cases, some or all of the slash can be distributed onsite. The FPA prohibits placing or leaving slash in or near streams.

Mechanical site preparation (e.g. with a rubber-tired skidder or crawler tractor) is used primarily to remove slash or heavy accumulations of non-tree understory “brush” vegetation. Disadvantages of mechanical methods include removal of topsoil and soil compaction (Fitzgerald 2008). Tractors and skidders can displace considerable amounts of forest floor organic debris and topsoil into slash piles, and can leave larger areas of bare soil than does harvesting itself, increasing the potential for runoff and erosion. Where soil is compacted over an extended area, mechanical treatments such as disking can improve soil porosity and infiltration rate (Neary et al. 2000). Soil compaction from heavy mechanized equipment can be reduced by conducting treatments when soils are frozen or moisture content is low (Rose and Haase 2006).

Industrial timberlands in western Oregon are typically treated with an herbicide or herbicide blend prior to replanting in order to suppress competing native and invasive species. Neary et al. (2000) maintain that herbicide treatments do not alter the integrity of the forest floor or increase the extent of bare mineral soil left after harvesting and argue that, in general, herbicide use ranks behind both fire and mechanized equipment in severity of impact. But understory plants mitigate erosion by attenuating raindrop energy and reduce soil moisture via transpiration, so the degree to which the soil remains protected following herbicide use is partly a function of slope and how much litter and duff cover remains after the vegetation is killed. For example, Slesak et al. (2015) found that vegetation control with herbicides increased erosion after post-wildfire salvage logging on steeply sloped sites in southern Oregon where there was no forest floor layer. Schmidt et al. (2001) observed reduced root cohesion following herbicide application, consistent with modeling results by Sidle (1992) indicating that suppressing understory vegetation drastically reduces slope stability, which together indicate that herbicide application can act to extend the window of landslide hazard after logging. Chapter 6 discusses forestry pesticides, including herbicides used in site preparation, in greater detail.

Research that distinguishes the effects of site preparation from those of harvesting and roads on water quality appears to be relatively limited. In general, any site preparation activities that contribute to an increase in bare mineral soil, soil compaction or soil mixing have the potential to increase sediment production. As with harvesting activities, if conducted according to current BMPs the potential for site preparation to generate significant additional sediment is probably not large in most cases, especially compared to the effects of roads. But as with all such generalizations, there can be exceptions in specific cases, especially on steeper slopes.

5.5. Increased landslides

In forested headwater catchments, mass wasting processes (e.g., translational slides, debris flows) may be the dominant processes responsible for sediment delivery from hillslopes to the stream network (Dietrich and Dunne 1978; Benda et al. 2005). Many studies have found that unpaved haul roads in steep, unstable terrain can increase the occurrence of mass movements by 25- to 350-times (Gray and Megahan 1981; Amaranthus et al. 1985; Wemple et al. 2001). Landings and skid trails have also been identified as sources of landslides (Keppeler et al. 2003). Across a broad range of conditions, removal of trees has also been shown to reduce the stability of steep slopes and increase the risk of landslides and mass movement (Goetz et al. 2015; Guthrie 2002; Imaizumi and Sidle 2012; Jakob 2000; May 2002; Montgomery et al. 2000; Schmidt et al. 2001) with the potential to significantly impact downstream resources (Benda et al. 2005). Numerous investigations have shown that for a period of from about 2 to 15-20 years after harvesting, the rate of landsliding is about 2 to 10 times higher than prior to harvest (Sidle and Bogard 2016). The time and duration of increased landslide hazard after harvesting are thought to be primarily functions of the rates of root decay and new root growth, and also species composition and distribution (Chang 2012; Roering et al. 2003; Schmidt et al. 2001). It has been estimated that forest harvesting and forest road construction can increase the densities of landslides impacting streams and the delivery of sediment to stream channels due to mass movement events by ~0.6 - 138-fold (Swanson and Dyrness 1975; Beschta 1978; Guthrie 2002; May 2002; Brardinoni et al. 2003; Hassan et al. 2005).

In the Oregon Coast Range it has been estimated that debris flows can entrain ~2 - 15 m^3 of sediment per meter of channel length (Benda 1990; May 2002; MacDonald and Coe 2007). However, prediction of the downstream transport rates of this material is challenging due to the typically high flow resistance and roughness in headwater channels (e.g., large woody debris, channel steps, large clasts) (Curran and Wohl, 2003; Benda et al. 2005; Hassan et al. 2005). In fact, large wood in streams can be effective at reducing downstream transport of sediment by decreasing stream velocity and increasing sediment storage (Davidson and Eaton 2013), with estimates in the Pacific Northwest for sediment storage of ~0.5 m^3 of sediment per meter of stream channel (May and Gresswell 2003), but this may be episodically released during mass movements and high flow events (Benda et al. 2005).

As a result of splash damming and other historic practices, many western Oregon streams remain deficient in large wood compared to conditions prior to Euro-American settlement (Montgomery et al. 2003). Landslides that originate in clearcuts contain less large wood and therefore travel farther and are more likely to enter streams than slides originating in intact forests. Landslides also terminate sooner when they enter areas with forest cover (Guthrie et al. 2010). Large wood and other factors that contribute to flow resistance play a major role in retaining coarser material that forms salmonid spawning gravels but are less effective at inhibiting the transport of very fine-grained material. Historic removal, and current and future supply of large wood in Oregon streams, and the role this key aspect of stream structure plays in sediment storage and release, are fundamental ways in which forest management continues to interact with drinking water source quality.

Increases in occurrence of mass movements following forest harvesting activities have been attributed to changes in hydrologic regimes, rather than due to specific mechanical or construction activities (Sidle and Ochiai 2006; Araujo et al. 2014). After forest harvest, soils become saturated more quickly (Johnson et al. 2007). When soils are saturated, slopes become more susceptible as soil pore pressures rise and cohesion drops, usually during intense rain, snowmelt, or rain-on-snow events. Intact forests on steep slopes contribute to slope stability via both geo-mechanical and hydrological processes. Tree root systems help to anchor forest soils to the slopes, and the tree overstory attenuates rainfall and soil saturation (Preti 2013). There is considerable evidence showing that increased landsliding after harvesting is strongly linked to the loss of root reinforcement and cohesion in forest soils after the trees are removed and as the roots decompose (Sakals and Sidle 2004; Roering et al. 2003; Guthrie 2002). In a study in the Oregon Coast Range, Schmidt et al. (2001) found that some 100-year old industrial forests had lateral root cohesion and root diameters very similar to 10-year old clearcuts, indicating that harvesting can modify root cohesion for at least a century and that the influence of root cohesion variability on landslide susceptibility cannot be accurately assessed solely on the basis of age class or the presence of one species of vegetation. Root reinforcement also decreases in areas of higher soil moisture because the tensile strength of roots decreases (Hales and Miniat 2017). The amount of reinforcement supplied by roots depends on the tensile strength and distribution of roots in the soil column. Small roots provide proportionally greater cohesive strength than larger roots.

The other primary mechanism by which forests contribute to slope stability is by attenuating rainfall and soil moisture (Preti 2013), which is important because the most common proximate cause of landslides is rainfall and snowmelt (Sidle and Bogard 2016). Mature stands of Douglas-fir and hemlock can reduce the amount of rainfall reaching the ground by 20 - 30 % or more (Link et al. 2004 and citations therein). Reduction or loss of this canopy interception after harvest increases rainfall intensity and contributes to elevated pore pressure in the soil and reduced slope stability (Baum et al. 2011; Keim and Skaugset 2003). Loss of tree evapotranspiration after harvest also increases soil saturation and reduces shear strength. Sidle and Bogard (2016) argue that in temperate forests, root reinforcement is usually a more important slope stabilizing agent than transpiration or canopy interception. Increased landslide risk associated with forest harvesting can be reduced by partial cutting of the stand and retention of understory vegetation (e.g. Dhakal and Sidle 2003; Sakals and Sidle 2004; Turner et al. 2010).

Where landscape disturbance (e.g., logging, fires) releases sediment in debris flows, some of this is stored in the steep valley network where it is removed by subsequent debris flows and fluvial entrainment. Sediment storage volumes and transit times determine both the magnitude and duration of downstream effects of the disturbances. Lancaster and Casebeer (2007) argue that as research on debris flows and fluvial sediment transport begins to influence land-use practices, there is a need to understand how much sediment is stored and the characteristics of its release. This study, and that reported in Lancaster et al. (2010) used systematic cross sections coupled with ^{14}C dating of random samples from bank, terrace riser, and in-channel materials in coastal Oregon watersheds to show that substantial volumes of sediment mobilized by mass wasting after disturbance remain stored for periods of centuries or more, and also that more recently deposited sediment is more likely to be remobilized than older sediment.

Capacity for storage of sediment delivered to streams by landslides and debris flows, and the rate at which it moves through a stream network vary with watershed size and topography, land use history, climate events and other factors. Thresholds for sediment movement and mobility vary significantly with grain size and flow volume; fine sediment is much more mobile. Introduction of new sediment and propagation of sediment through a forested watershed are largely episodic and associated with infrequent large storms (MacDonald and Coe 2007; Benda et al. 2005; May and Gresswell 2004). In between debris flow events, fine sediment may be transferred by fluvial flow in pulses during smaller precipitation events (Nistor and Church 2005). Mass wasting processes dominate in many headwaters, giving way to fluvial processes where debris flows form fans at junctions with larger streams. Sediment production was almost certainly quite high in watersheds where significant historic logging occurred, while sediment storage capacity was reduced in watersheds where splash damming resulted in removal of large wood. Owing to the temporal and spatial complexity of these processes, the amounts and locations of sediment mobilized by historic logging that remains stored in Oregon watersheds are likely highly variable across different stream systems and reaches; studies focused on these questions are very limited. However, in light of Oregon's extensive history of industrial logging and known linkages between harvesting in steep coastal watersheds and increases in mass wasting, evidence (e.g. Koehler et al. 2007) suggests that some fraction of the sediment delivered to Oregon waterways under historic practices may remain stored there today. Such "legacy sediment" is deposited when intensified land-use results in sediment deliveries greater than sediment transport capacity and may lead to valley-bottom aggradation, ultimately followed by channel incision when the sediment wave passes and sediment loads decrease. These aggradation - degradation episodes can leave substantial volumes of sediment in storage because vertical channel incision proceeds more quickly than channel widening (Wohl 2015). Modern forest practices appear to significantly reduce sediment production related to timber harvesting. The dynamics of fine-grained and coarse-grained sediment storage, residence times and mobilization differ significantly. However, even in the absence of additional sediment production, increases in peak flows associated with tree removal can remobilize sediment currently stored in streams but associated with timber harvesting decades ago. The likelihood of this may be compounded by predicted increases in peak flows associated with infrequent large storms and climate change.

While their role in slope stability is generally accepted as significant, the precise ways that root reinforcement and anchoring interact with topography, forest structure, soil depth, geology, changes in water movement and soil moisture after harvest, and the relative influence of these factors on slope stability across different sites, are complex and not fully understood (Hales and Miniat 2017; Moos et al. 2016; Schmidt et al. 2001). Despite the knowledge we have amassed regarding the effects of forest management activities on mass movements and sediment delivery to streams, quantitative evidence of the explicit linkages between upstream inputs and downstream fluxes of sediment relevant to community drinking water supply remains quite limited (MacDonald and Coe 2007). The linkage between mass movements in headwater streams related to forest harvesting activities and downstream water supply is complicated due to multiple factors, including: the random and episodic nature of mass movements that makes them difficult to study, cumulative effects from multiple disturbance agents, heterogeneous in-channel storage and release of sediment, and "increasing temporal and spatial variability in the delivery of sediment from hillslopes to headwater streams and from headwater streams to downstream reaches" (MacDonald and Coe 2007; Klein et al. 2012). Moreover, existing studies across the PNW do not adequately reflect the broad range of climate, geology, topography, and vegetation which drive highly variable hydrologic and mass movement processes across the region.

Much remains to be learned regarding the extent to which forest management activities, which influence mass movements, ultimately impact turbidity and sediment at a scale relevant to most downstream drinking water utilities. There are also large information gaps regarding historic and current sediment production from forest practices, sediment storage capacity, and rates of sediment movement through different stream networks in Oregon. However, an interesting study by Wheatcroft et al. (2013) sheds some light on these issues. They quantified sediment accumulation rates (SARs) over the past 125 years at depths of 70-200m on the continental shelf of the Pacific Ocean off the Umpqua River. Using ^{210}Pb geochronology at a dense array of sampling stations (73), Wheatcroft et al. (2013) identified a 2 - 4-fold increase in SAR and a shift toward finer sediments that occurred, on average, in 1967 ± 13 y, consistent with the history of industrial logging in the Umpqua basin, which peaked in the two decades after World War II and coincided with a wet phase of the Pacific Decadal Oscillation (1944 - 1978) when average and peak river flows were elevated. Their analysis indicated that hydroclimatic changes alone could not explain the increase in SARs; changes in sediment yield must have occurred, most likely caused by widespread logging in the Umpqua basin uplands.

Wheatcroft et al. (2013) point out that detecting a logging signal on the continental shelf is notable because, despite considerable evidence (e.g. from paired watershed studies) that logging has led to elevated sediment production from disturbed headwaters, it generally remains uncertain whether these effects scale up to encompass entire river basins 1000s km^2 in area. The authors list some reasons for this uncertainty. First, in any given year just a small fraction of the basin is disturbed by logging; evidence indicates that only about 1 % of the Umpqua basin was logged even in peak harvesting years, far less than typical in paired watershed studies. Another factor is the storage capacity of large basins, whereby sediment mobilized by harvesting activities is deposited before reaching the channel network, or stored in valley bottoms or estuaries. Lastly, intervening processes such as landslides and bank failures may confound or obliterate environmental signals as they propagate through sediment routing systems. All of these potentially contributing factors (and the fact that a significant portion of the watershed had been logged prior to construction of the reservoir) were used by Ambers (2001) to help explain the lack of a logging signal in a flood control reservoir in the Western Cascades.

Despite these potentially confounding variables, Wheatcroft et al. (2013) were able to detect the cumulative effects of timber harvesting at the basin scale in Umpqua River continental shelf sediments, expressed as an increase in sediment accumulation and a shift in sediment grain size toward finer particles. The authors also comment on the relatively short time lag between the period of maximal upland disturbance (1945 - 1955) and estimated age of the SAR increase on the continental shelf (~1967 ± 13 yrs). They attribute this finding to limits on fine-grained sediment storage capacity in the Umpqua basin and the fact that the fines they found on the shelf are more likely to be readily propagated through the system than coarser material. Noting similar patterns on the Eel River (California) margin, the authors favored the conclusion that timber harvesting results in delivery of more fine grained sediment to river channels and that this material is simply propagated through the sediment routing system. But they also allowed that timber harvesting, by increasing landslide frequency, could simply lead to an overall increase in sediment export but no change in grain size, and that the fining trend offshore could arise from the inability of post depositional reworking to winnow fines under increased deposition rates.

The study by Wheatcroft et al. (2013) indicates that large volumes of fine grained sediments mobilized as a result of forestry activities in a coastal Oregon watershed can readily move through the entire stream and river system. Their results focus on a time period when harvesting intensity was higher than today and prior to development of BMPs to mitigate sediment production. Nevertheless, their findings link sediment produced by forestry in an upper watershed to its ultimate fate on the oceanic continental shelf, implying that forestry-related fine sediments can also reach municipal water systems in this and similarly-managed coastal Oregon watersheds. Still, adapting such knowledge to forest management today will require the filling of major information gaps regarding how particular components and aspects of forest operations produce such sediment, and how it propagates through watersheds. MacDonald and Coe (2007) argue that more studies are needed to directly measure the effects of current forest operations on sediment production in headwater areas, explicitly link these sources to the channel network, evaluate sediment routing, and then document whether there is a resulting downstream physical response. This will require explicit consideration of hillslope-channel connectivity (Bracken and Croke 2007) rather than simply using watershed-scale mean or total sediment production.

5.6. Summary and conclusions

Linkages between active forest management and increased sediment loading in streams have been studied extensively and are well-established in broad terms. There is also an expanding body of evidence indicating that modern practices such as improved road building methods and stream buffers have significantly reduced sediment production from forest management activities, and the chances that this sediment will enter waterways. But these effects and findings are highly variable due to the complexity of interactions among factors such as site-specific ecology, geology and geomorphology, management prescriptions and land use histories. The specific sources of mobilized sediment within an actively managed area are also often not clear. Considerable uncertainty remains in predicting precisely how a particular set of forest management actions will affect sediment production in specific cases. Further, there is a paucity of research focused on linkages between sediment inputs related to timber harvesting and associated activities in headwater areas of watersheds and increases in suspended sediment or turbidity in water withdrawn downstream for domestic uses.

A range of potential contributing factors may help explain the lack of research focused on forestry and drinking water linkages. As watershed size and distance from forest management activities increase, it becomes progressively more challenging to isolate and quantify the effects of particular actions (Sidle and Gomi 2017). There are usually cumulative effects resulting from forest management in larger watersheds, partly due to variability in forestry activities (e.g. road building and use, harvesting, site preparation) and timing of their impacts on stream sediment, with some actions having immediate effects and others taking years to become apparent. Timber has been harvested for a century or more in many Oregon watersheds, historically without BMPs in place, with a legacy of sediment production and sediment transfer downstream in many watersheds. Over time, affects accumulate in complex patterns across forestlands managed through multiple harvests and rotations. Distinguishing effects of modern forest practices from those used earlier, and whether increased sediment and turbidity originates primarily from remobilized natural or anthropogenic sediments within streams, streambank erosion, or sources external to the waterway is difficult and complex. Climate variability, the generally episodic nature of sediment movement, and the outsize influence of stochastic events such as infrequent large storms can introduce additional uncertainty into research findings (e.g., Grant and Wolff 1991). Finally, in larger watersheds, forest management is often not the only land use or potential source of sediments.

For these reasons, it is difficult to make specific, firm conclusions regarding how, where and the extent to which sediment produced by active forest management in a headwater area affects water quality downstream at the drinking water intake. There is, however, an extensive body of evidence accumulated through forestry and sediment-focused research conducted in upper watersheds that is highly relevant to drinking water quality (Swanson et al. 2000). Reasoned inferences can be drawn from this evidence base regarding effects on drinking water sources because hillslopes, headwaters, and larger downstream waterways are all elements of fundamentally connected and integrated hydrological systems (Bracken and Croke 2007). Headwater streams comprise about 60 - 80 % of total stream length in a typical river drainage (Benda et al. 2005) and generate most of the streamflow in downstream areas, and these first and second-order streams cumulatively contribute to, and can profoundly affect water quality downstream (Nadeau and Rains 2007).

Headwater streamflow is usually routed efficiently downstream, meaning that management-induced changes in streamflow parameters will accumulate downstream (Reiter et al. 2009; Bywater-Reyes et al. 2017; Bywater-Reyes et al. 2018). Because turbidity and fine sediment can be readily transported downstream, changes in headwater inputs of these constituents may be directly linked to downstream conditions. In contrast, linkages between upstream inputs and downstream fluxes for coarse sediment and large woody debris are considerably weaker (MacDonald and Coe 2007). It is also important to note the substantial variation in distances between actively managed forests and drinking water intakes across the range of different municipal water suppliers in Oregon. Findings from studies showing that forest management activities or forest roads can increase sediment production and reduce stream water quality in headwaters can be more reliably extrapolated to indicate that drinking water may also be impacted where intakes are in relatively closer proximity to these management activities and have fewer intervening land uses.

In general, due primarily to the complex interplay of factors outlined above and difficulties in isolating and quantifying the sources and fates of mobilized sediment, we found little direct, quantitative evidence that forestry activities and forest roads impact community drinking water in Oregon. But there is considerable indirect evidence that forestry can have such affects, and likely continues to have effects in certain cases, inferred from the following:

1. Extensive findings regarding linkages between forest harvest activities, forest roads and increases in mass wasting in upper watersheds.
2. Cumulative and legacy effects of harvesting, site preparation and forest roads dating from periods when BMPs were not as robust.
3. Inevitable variability in BMP implementation and effectiveness across different site factors such as land use history, geology, topography (i.e. slope) and also different forest operators, harvesting technologies and climatic conditions.
4. The ability of fine sediment and turbidity to be carried considerable distances, especially during peak flow events.
5. The inherent connectivity of hillslopes, headwaters and larger downstream waterways.
6. The lack of provisions to protect small, non-fish bearing, ephemeral and intermittent streams during harvesting, and the lack of water quality protection provisions for operations in landslide-prone areas.

5.7. References

Allison, C., Sidle, R.C., Tait, D., 2004. Application of decision analysis to forest road deactivation in unstable terrain. Environmental Management 33, 173-185.

Amaranthus, M.P., Rice, R.M., Barr, N.R., Ziemer, R.R., 1985. Logging and forest roads related to increased debris slides in southwestern Oregon. Journal of Forestry 83, 229-233.

Ambers, R.K., 2001. Using the sediment record in a western Oregon flood-control reservoir to assess the influence of storm history and logging on sediment yield. Journal of Hydrology, 244(3-4), pp.181-200.

Ampoorter, E., de Schrijver, A., van Nevel, L., Hermy, M., Verheyen, K., 2012. Impact of mechanized harvesting on compaction of sandy and clayey forest soils: results of a meta-analysis. Annals of Forest Science 69, 533-542.

Anderson, C.J. and B.G. Lockaby. 2011. Research gaps related to forest management and stream sediment in the United States. Environmental Management 47, 303-313.

Araujo, H.A., Page, A., Cooper, A.B., Venditti, J., MacIsaac, E., Hassan, M.A., Knowler, D., 2014. Modelling changes in suspended sediment from forest road surfaces in a coastal watershed of British Columbia. Hydrological Processes 28, 4914-4927.

Arismendi, I., Groom, J.D., Reiter, M., Johnson, S.L., Dent, L., Meleason, M., Argerich, A., Skaugset, A.E., 2017. Suspended sediment and turbidity after road construction/improvement and forest harvest in streams of the Trask River Watershed Study, Oregon. Water Resources Research 53, 6763-6783.

Aust, W.M., Blinn, C.R., 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: an overview of water quality and productivity research during the past 20 years (1982-2002). Water Air and Soil Pollution 4, 5-36.

AWWA, 1990. Water Quality and Treatment. A Handbook of Community Water Supplies. American Water WorksAssociation, McGraw-Hill, Inc., New York, NY.

Baird, E.J., Floyd, W., Meerveld, I.v., Anderson, A.E., 2012. Road surface erosion, part 1: Summary of effects, processes, and assessment procedures. Streamline: Watershed Management Bulletin 15, 1-9.

Basher, L.R., Hicks, D.M., Clapp, B., Hewitt, T., 2011. Sediment yield response to large storm events and forest harvesting, Motueka River, New Zealand. N. Z. J. Mar. Freshw. Res. 45, 333-356.

Baum, R.L., Godt, J.W. and Coe, J.A. 2011. Assessing susceptibility and timing of shallow landslide and debris flow initiation in the Oregon Coast Range, USA. In 5th International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment (pp. 825-834).

Benda, L. 1990. The influence of debris flows on channels and valley floors in the Oregon Coast Range, USA. Earth Surf. Process. Landf. 15, 457-466.

Benda, L., Dunne, T., 1997. Stochastic forcing of sediment routing and storage in channel networks. Water Resources Research 33, 2865-2880.

Benda, L., Hassan, M.A., Church, M., May, C.L., 2005. Geomorphology of steepland headwaters: The transition from hillslopes to channels. Journal of the American Water Resources Association 41, 835-851.

Benda, L., Poff, N.L., Miller, D., Dunne, T., Reeves, G., Pess, G., Pollock, M., 2004. The network dynamics hypothesis: How channel networks structure riverine habitats. Bioscience 54, 413-427.

Bennett, K.A., 1982. Effects of slash burning on surface soil erosion rates in the Oregon Coast Range. MS thesis submitted in partial fulfillment of Master of Science, Soil Science. Oregon State University.

Beschta, R.L., 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14, 1011-1016.

Beschta, R.L., Jackson, W.L., 2008. Forest practices and sediment production in the Alsea Watershed Study. In: Stednick, J. (Ed.), Hydrological and biological responses to forest practices: The Alsea Watershed Study. Springer Science +Business Media, LLC, New York, NY, pp. 55-66.

Bilby, R.E., Sullivan, K., Duncan, S.H., 1989. The generation and fate of road-surface sediment in forested watersheds in southwestern Washington. Forest Science 35, 453-468.

Binkley, D., Brown, T.C., 1993. Forest practices as nonpoint sources of pollution in North America. Journal of the American Water Resources Association 29, 729-740.

Birkinshaw, S.J., Bathurst, J.C., Iroume, A., Palacios, H., 2011. The effect of forest cover on peak flow and sediment discharge-an integrated field and modelling study in central-southern Chile. Hydrological Processes 25, 1284-1297.

Black, T.A., R.Cissel, and C.H.Luce. 2012. The Geomorphic Road Analysis and Inventory Package (GRAIP) volume 1: Data collection method. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-280WWW. 110p.

Bladon, K.D., Emelko, M.B., Silins, U., Stone, M., 2014. Wildfire and the future of water supply. Environmental Science & Technology 48, 8936-8943.

Borok, A., 2014. Turbidity Technical Review: Summary of Sources, Effects, and Issues Related to Revising the Statewide Water Quality Standard for Turbidity. In. State of Oregon, Department of Environmental Quality, Portland, OR.

Bracken, L.J. and Croke, J., 2007. The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. Hydrological Processes: An International Journal, 21(13), pp.1749-1763.

Brardinoni, F., Slaymakerl, O., Hassan, M.A., 2003. Landslide inventory in a rugged forested watershed: a comparison between air-photo and field survey data. Geomorphology 54, 179-196.

Brown, A.E., Zhang, L., McMahon, T.A., Western, A.W., Vertessy, R.A., 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology 310, 28-61.

Brown, G.W., Krygier, J.T., 1971. Clear-cut logging and sediment production in Oregon Coast Range. Water Resources Research 7, 1189-1198.

Brown, K.R., McGuire, K.J., Aust, W.M., Hession, W.C., Dolloff, C.A., 2014. The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings. Hydrological Processes.

Burroughs Jr., E.R., King, J.G., 1989. Reduction of soil erosion on forest roads. In. United States Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, p. 21.

Burt, T.P., McDonnell, J.J., 2015. Whither field hydrology? The need for discovery science and outrageous hydrological hypotheses. Water Resources Research 51, 5919-5928.

Bywater-Reyes, S., Bladon, K.D., Segura, C., 2018. Relative influence of landscape variables, discharge, and forest management on sediment yields in temperate mountain catchments. Water Resources Research 54, 5126-5142.

Bywater-Reyes, S., Segura, C., Bladon, K.D., 2017. Geology and geomorphology control suspended sediment yield and modulate increases following timber harvest in Oregon headwater streams. Journal of Hydrology 548, 754-769.

Chang, M. 2012. Forest hydrology: an introduction to water and forests. Third edition. CRC Press.

Chase, C.W., Reiter, M., Homyack, J.A., Jones, J.E. and Sucre, E.B. 2019. Soil disturbance and stream-adjacent disturbance from tethered logging in Oregon and Washington. Forest Ecology and Management, 454, p.117672.

Coe, D.B.R., 2006. Sediment production and delivery from forest roads in the Sierra Nevada, California. In, Department of Forest, Rangeland, and Watershed Stewardship. Colorado State University, Fort Collins, CO, p. 110.

Collins, A.L., Foster, I.D.L., Gellis, A.C., Porto, P., Horowitz, A.J., 2017. Sediment source fingerprinting for informing catchment management: Methodological approaches, problems and uncertainty. Journal of Environmental Management 194, 1-3.

Collins, A.L., Walling, D.E., 2004. Documenting catchment suspended sediment sources: problems, approaches and prospects. Progress in Physical Geography 28, 159-196.

Collins, A.L., Walling, D.E., Webb, L., King, P., 2010. Apportioning catchment scale sediment sources using a modified composite fingerprinting technique incorporating property weightings and prior information. Geoderma 155, 249-261.

Cristan, R., Aust, W.M., Bolding, M.C., Barrett, S.M., Munsell, J.F., Schilling, E., 2016. Effectiveness of forestry best management practices in the United States: Literature review. Forest Ecology and Management 360, 133-151.

Crittenden, J.C., Trussell, R.R., Hand, D.W., Howe, K.J., Tchobanoglous, G., 2005. Water Treatment: Principles and Design. John Wiley & Sons, Inc., Hoboken, NJ.

Crittenden, J.C., Trussell, R.R., Hand, D.W., Howe, K.J., Tchobanoglous, G., 2012. MWH's Water Treatment: Principles and Design. John Wiley & Sons, Hoboken, NJ.

Croke, J., Hairsine, P., Fogarty, P., 1999a. Runoff generation and re-distribution in logged eucalyptus forests, south-eastern Australia. Journal of Hydrology 216, 56-77.

Croke, J., Hairsine, P., Fogarty, P., 1999b. Sediment transport, redistribution and storage on logged forest hillslopes in south-eastern Australia. Hydrological Processes 13, 2705-2720.

Croke, J., Mockler, S., 2001. Gully initiation and road-to-stream linkage in a forested catchment, southeastern Australia. Earth Surf. Process. Landf. 26, 205-217.

Croke, J.C., Hairsine, P.B., 2006. Sediment delivery in managed forests: a review. Environmental Reviews 14, 59-87.

Curran, J.H., Wohl, E.E., 2003. Large woody debris and flow resistance in step-pool channels, Cascade Range, Washington. Geomorphology 51, 141-157.

Davidson, S.L. and Eaton, B.C. 2013. Modeling channel morphodynamic response to variations in large wood: Implications for stream rehabilitation in degraded watersheds. Geomorphology, 202, pp.59-73.

Dhakal, A. S. and Sidle, R. C. 2003. Long-term modelling of landslides for different forest management practices. Earth Surf. Process. Landforms, 28: 853-868.

Dietrich, W.E., Dunne, T., 1978. Sediment budget for a small catchment in mountainous terrain. Z. Geomorph. N.F. 29, 191-206.

Dorner, S.M., Anderson, W.B., Slawson, R.M., Kouwen, N., Huck, P.M., 2006. Hydrologic modeling of pathogen fate and transport. Environmental Science & Technology 40, 4746-4753.

Droppo, I.G., Liss, S.N., Williams, D., Leppard, G.G., 2006. River sediment/pathogen interactions: importance for policy development on safe water practices. In, Sediment Dynamics and the Hydromorphology of Fluvial Systems. International Association of Hydrological Sciences (IAHS), Dundee, UK, pp. 314-321.

Dubé, K., Megahan, W., McCalmon, M., 2004. Washington road surface erosion model (WARSEM) manual. In. State of Washington, Department of Natural Resources, Olympia, WA, p. 189.

Edwards, P.J., Wood, F. and Quinlivan, R.L. 2016. Effectiveness of best management practices that have application to forest roads: A literature synthesis. Gen. Tech. Rep. NRS-163. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 171 p., 163, pp.1-171.

Emelko, M.B., Silins, U., Bladon, K.D., Stone, M., 2011. Implications of land disturbance on drinking water treatability in a changing climate: Demonstrating the need for "source water supply and protection" strategies. Water Research 45, 461-472.

Emelko, M.B., Stone, M., Silins, U., Allin, D., Collins, A.L., Williams, C.H.S., Martens, A.M., Bladon, K.D., 2016. Sediment-phosphorus dynamics can shift aquatic ecology and cause downstream legacy effects after wildfire in large river systems. Global Change Biology 22, 1168-1184.

Endicott, D. 2008. National level assessment of water quality impairments related to forest roads and their prevention by best management practices. Report prepared for US EPA, Office of Water. Contract No. EP-C-05-066. Great Lakes Environmental Center. 250p.

Fitzgerald, S.A. 2008. Successful reforestation: an overview. EC 1498. Oregon State University Extension Service. Accessed online 8/18/2019. https://catalog.extension.oregonstate.edu/sites/catalog/files/project/pdf/ec1498.pdf.

Foltz, R.B. 2012. A comparison of three erosion control mulches on decommissioned forest road corridors in the northern Rocky Mountains, United States. Journal of Soil and Water Conservation, 67(6), pp.536-544.

Fransen, P.J.B., Phillips, C.J., Fahey, B.D., 2001. Forest road erosion in New Zealand: Overview. Earth Surf. Process. Landf. 26, 165-174.

Fraser, N., Jackson, R., Radcliffe, D., 2012. A paired watershed investigation of silvicultural best management practices revisited: BF Grant Memorial Forest, Georgia. Forest Science 58, 652-662.

Fredriksen, R.L., 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds. In. U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR, p. 15.

Jackson, C., Sturm, C.A. and Ward, J.M. 2001. Timber harvest impacts on small headwater stream channels in the Coast Ranges of Washington. JAWRA Journal of the American Water Resources Association, 37: 1533-1549.

Gillies, C., 2007. Erosion and sediment control practices for forest roads and stream crossings: A practical operations guide. In. FPInnovations, FERIC Division, Pointe-Claire, QC, p. 87.

Goetz, J. N., Guthrie, R.H., Brenning, A. 2015. Forest harvesting Is associated with increased landslide activity during an extreme rainstorm on Vancouver Island, Canada. Natural Hazards and Earth System Sciences 15.6: 1311-1330.

Gomi, T., Moore, R.D., Hassan, M.A., 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. Journal of the American Water Resources Association 41, 877-898.

Goode, J.R., Luce, C.H., Buffington, J.M., 2012. Enhanced sediment delivery in a changing climate in semi-arid mountain basins: Implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology 139, 1-15.

Grant, G.E., Lewis, S.L., Swanson, F.J., Cissel, J.H., McDonnell, J.J., 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-science report for western Oregon and Washington. In. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, p. 76 p.

Grant, G.E., Wolff, A.L., 1991. Long-term patterns of sediment transport following timber harvest, Western Cascade Mountains, Oregon, USA. In: Peters, N.E., Walling, D.E. (Eds.), Sediment and stream water quality in a changing environment: trends and explanation, Proceedings of the Vienna IAHS symposium. International Association of Hydrological Sciences, Vienna, Austria, pp. 31-40.

Gray, D.H., Megahan, W.F., 1981. Forest vegetation removal and slope stability in the Idaho Batholith. In. U.S. Dept. of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT, p. 30.

Gucinski, H., Furniss, M.J., Ziemer, R.R., Brookes, M.H., 2001. Forest roads: a synthesis of scientific information. In. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, p. 103 p.

Guthrie, R.H., 2002. The effects of logging on frequency and distribution of landslides in three watersheds on Vancouver Island, British Columbia. Geomorphology 43, 273-292.

Guthrie, R.H., Hockin, A., Colquhoun, L., Nagy, T., Evans, S.G. and Ayles, C., 2010. An examination of controls on debris flow mobility: Evidence from coastal British Columbia. Geomorphology, 114(4), pp.601-613.

Hales, T.C. and Miniat, C.F. 2017. Soil moisture causes dynamic adjustments to root reinforcement that reduce slope stability. Earth Surface Processes and Landforms, 42(5), pp.803-813.

Hall, J.D., 2008. Salmonid populations and habitat. In: Stednick, J. (Ed.), Hydrological and Biological Responses to Forest Practices: The Alsea Watershed Study. Springer, New York, NY, pp. 67-93.

Harr, R.D., Fredriksen, R.L., 1988. Water-quality after logging small watersheds within the Bull Run Watershed, Oregon. Journal of the American Water Resources Association 24, 1103-1111.

Harris, D.D., Williams, R.C., 1971. Streamflow, sediment-transport, and water-temperature characteristics of the three small watersheds in the Alsea River basin, Oregon. In. U.S. Geological Survey, Washington, DC, p. 21 p.

Hassan, M.A., Church, M., Lisle, T.E., Brardinoni, F., Benda, L., Grant, G.E., 2005. Sediment transport and channel morphology of small, forested streams. Journal of the American Water Resources Association 41, 853-876.

Hatten, J.A., Segura, C., Bladon, K.D., Hale, V.C., Ice, G.G., Stednick, J.D., 2018. Effects of contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea Watershed Study Revisited. Forest Ecology & Management 408, 238-248.

Hohner, A.K., Cawley, K., Oropeza, J., Summers, R.S., Rosario-Ortiz, F.L., 2016. Drinking water treatment response following a Colorado wildfire. Water Research 105, 187-198.

Hotta, N., Kayama, T., Suzuki, M., 2007. Analysis of suspended sediment yields after low impact forest harvesting. Hydrological Processes 21, 3565-3575.

Ice, G., 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. J. Environ. Eng.-ASCE 130, 684-689.

Ice, G., Dent, L., Robben, J., Cafferata, P., Light, J., Sugden, B., Cundy, T., 2004. Programs assessing implementation and effectiveness of state forest practice rules and BMPs in the West. Water, Air and Soil Pollution: Focus 4, 143-169.

Imaizumi, F, Sidle, R.C. 2012. "Effect of forest harvesting on hydrogeomorphic processes in steep terrain of central Japan." Geomorphology 169-170: 109-22

Isaac, L.A. and Hopkins, H.G. 1937. The forest soil of the Douglas fir region, and changes wrought upon it by logging and slash burning. Ecology, 18(2), pp.264-279.

Jackson, C.R., Sturm, C.A. and Ward, J.M. 2001. Timber harvest impact on small headwater stream channels in the Coast Ranges of Washington. JAWRA Journal of the American Water Resources Association, 37(6), pp.1533-1549.

Jakob, M. 2000. The impacts of logging on landslide activity at Clayoquot Sound, British Columbia. Catena 38.4: 279-300.

James, L.A. 2013. Legacy sediment: Definitions and processes of episodically produced anthropogenic sediment. Anthropocene, 2, pp.16-26.

Johnson, A.C., Edwards, R.T. and Erhardt, R. 2007. Ground-water response to forest harvest: Implications for hillslope stability. JAWRA Journal of the American Water Resources Association, 43(1), pp.134-147.

Jones, J.A., Swanson, F.J., Wemple, B.C., Snyder, K.U., 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14, 76-85.

Kehoe, M.J., Chun, K.P., Baulch, H.M., 2015. Who smells? Forecasting taste and odor in a drinking water reservoir. Environmental Science & Technology 49, 10984-10992.

Keim, R.F. and Skaugset, A.E., 2003. Modelling effects of forest canopies on slope stability. Hydrological Processes, 17(7), pp.1457-1467.

Keller, G., Sherar, J., 2003. Low volume roads engineering: Best management practices field guide. In. U.S. Department of Agriculture, Forest Service and U.S. Agency for International Development (USAID), Washington, DC, p. 158.

Keppeler, E.T., Lewis, J., Lisle, T.E., 2003. Effects of forest management on streamflow, sediment yield, and erosion, Caspar Creek Experimental Watersheds. In: Renard, K.G., McElroy, S.A., Gburek, W.J., Canfield, H.E., Scott, R.L. (Eds.), First Interagency Conference on Research in the Watersheds. U.S. Department of Agriculture, Agricultural Research Service, Benson, AZ, pp. 77-82.

Klein, R.D., Lewis, J., Buffleben, M.S., 2012. Logging and turbidity in the coastal watersheds of northern California. Geomorphology 139, 136-144.

Koehler, R. D., Kelson, K. I., Matthews, G., Kang, K. H., & Barron, A. D. 2007. Mapping Prehistoric, Historic, and Channel Sediment Distribution, South Fork Noyo River: A Tool For Understanding Sources, Storage, and Transport. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y., Zielinski, W.J.; Furniss, M.J., Tech. eds. 2007. Proceedings of the redwood region forest science symposium: What does the future hold? Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station, Forest Service, US Department of Agriculture; p. 371-382 (Vol. 194).

Krasner, S.W., 2009. The formation and control of emerging disinfection by-products of health concern. Philos. Trans. R. Soc. A-Math. Phys. Eng. Sci. 367, 4077-4095.

Krasner, S.W., Weinberg, H.S., Richardson, S.D., Pastor, S.J., Chinn, R., Sclimenti, M.J., Onstad, G.D., Thruston, A.D., 2006. Occurrence of a new generation of disinfection byproducts. Environmental Science & Technology 40, 7175-7185.

Lancaster, S.T. and Casebeer, N.E. 2007. Sediment storage and evacuation in headwater valleys at the transition between debris-flow and fluvial processes. Geology, 35(11), pp.1027-1030.

Lancaster, S.T., Underwood, E.F. and Frueh, W.T., 2010. Sediment reservoirs at mountain stream confluences: Dynamics and effects of tributaries dominated by debris-flow and fluvial processes. Geological Society of America (GSA) Bulletin, 122(11-12), pp.1775-1786.

Lane, P.N.J., Sheridan, G.J., 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrological Processes 16, 2599-2612.

Langridge, R. 2011. When do challengers succeed? Nongovernmental actors, administrative agencies, and legal change: Shifting rules for Oregon's private forests. Law & Social Inquiry, 36(3), pp.662-693.

Lechevallier, M.W., Evans, T.M., Seidler, R.J., 1981. Effect of turbidity on chlorination efficiency and bacterial persistence in drinking-water. Appl. Environ. Microbiol. 42, 159-167.

Leziart, T., de la Rochere, P.M.D., Cheswick, R., Jarvis, P., Nocker, A., 2019. Effect of turbidity on water disinfection by chlorination with the emphasis on humic acids and chalk. Environ. Technol. 40, 1734-1743.

Lick, W.J., 2008. Sediment and Contaminant Transport in Surface Waters. CRC Press, Boca Raton, FL.

Link, T.E., Unsworth, M. and Marks, D. 2004. The dynamics of rainfall interception by a seasonal temperate rainforest. Agricultural and Forest Meteorology, 124(3-4), pp.171-191.

Lisle, T.E., 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resources Research 25, 1303-1319.

Litschert, S.E. and MacDonald, L.H. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. Forest Ecology and Management, 259(2), pp.143-150.

Luce, C., Black, T., 2001. Spatial and temporal patterns in erosion from forest roads. In: Wimosta, S., Burges, S.J. (Eds.), Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. Water science and application. American Geophysical Union, Washington, DC, pp. 165-178.

Luce, C.H., 2002. Hydrological processes and pathways affected by forest roads: what do we still need to learn? Hydrological Processes 16, 2901-2904.

Luce, C.H., Black, T.A., 1999. Sediment production from forest roads in western Oregon. Water Resources Research 35, 2561-2570.

MacDonald, L.H., Coe, D., 2007. Influence of headwater streams on downstream reaches in forested areas. Forest Science 53, 148-168.

Madej, M.A., 2001. Erosion and sediment delivery following removal of forest roads. Earth Surf. Process. Landf. 26, 175-190.

May, C.L., 2002. Debris flows through different forest age classes in the central Oregon Coast Range. Journal of the American Water Resources Association 38, 1097-1113.

May, C.L., Gresswell, R.E., 2003. Processes and rates of sediment and wood accumulation in headwater streams of the Oregon Coast Range, USA. Earth Surf. Process. Landf. 28, 409-424.

May and Gresswell. 2004. Spatial and temporal patterns of debris-flow deposition in the Oregon Coast Range, USA. Geomorphology 57: 135-149.

McDonnell, J. J.; Evaristo, J.; Bladon, K. D.; Buttle, J.; Creed, I. F.; Dymond, S. F.; Grant, G.; Iroume, A.; Jackson, C. R.; Jones, J. A.; Maness, T.; McGuire, K. J.; Scott, D. F.; Segura, C.; Sidle, R. C. and Tague, C. 2018. Water sustainability and watershed storage. Nature Sustainability. 1(8): 378-379.

McGuire, M.J., 1995. Off-flavor as the consumers measure of drinking-water safety. Water Science and Technology 31, 1-8.

Megahan, W.F., 1974. Erosion over time: A model. In. United States Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, p. 14.

Megahan, W.F., Kidd, W.J., 1972. Effect of logging roads on sediment production rates in the Idaho Batholith. In. U.S. Department of Agriculture, Forest Service, Intermountain Forest & Range Experiment Station, Ogden, UT, p. 14.

Megahan, W.F., King, J.G., 2004. Erosion, sedimentation, and cumulative effects in the Northern Rocky Mountains. In: Ice, G.G., Stednick, J.D. (Eds.), A century of forest and wildland watershed lessons. Society of American Foresters, Bethesda, MD, pp. 201-222.

Miller, R.E., Scott, W., Hazard, J.H., 1996. Soil compaction and conifer growth after tractor yarding at three coastal Washingon locations. Canadian Journal of Forest Research 26, 225-236.

Miller, R.H., 2014. Influence of log truck traffic and road hydrology on sediment yield in Western Oregon. In, Water Resources Engineering. Oregon State University, Corvallis, OR, p. 116.

Mills, K., 1997. Forest roads, drainage, and sediment delivery in the Kilchis River Watershed. In. Oregon Department of Forestry, Tillamook, OR, p. 38.

Mohr, C.H., Coppus, R., Iroume, A., Huber, A., Bronstert, A., 2013. Runoff generation and soil erosion processes after clear cutting. J. Geophys. Res.-Earth Surf. 118, 814-831.

Montgomery, D.R., 1994. Road surface drainage, channel initiation, and slope instability. Water Resources Research 30, 1925-1932.

Montgomery, D.R., Massong, T.M. and Hawley, S.C., 2003. Influence of debris flows and log jams on the location of pools and alluvial channel reaches, Oregon Coast Range. Geological Society of America Bulletin, 115(1), pp.78-88.

Montgomery, D.R., Schmidt, K.M., Greenberg, H.M. and Dietrich, W.E. 2000. Forest clearing and regional landsliding. Geology, 28(4), pp.311-314.

Moos, C., Bebi, P., Graf, F., Mattli, J., Rickli, C. and Schwarz, M., 2016. How does forest structure affect root reinforcement and susceptibility to shallow landslides? Earth Surface Processes and Landforms, 41(7), pp.951-960.

Nadeau, T.L. and Rains, M.C., 2007. Hydrological connectivity between headwater streams and downstream waters: how science can inform policy. JAWRA Journal of the American Water Resources Association, 43(1), pp.118-133.

Neary, D.G., L.R. DeBano and P.F. Ffolliott. 2000. Fire impacts on forest soils: a comparison to mechanical and chemical site preparation. In Fire and forest ecology: Innovative silviculture and vegetation management. Tall Timbers Ecology Conference Proceedings (No. 21, pp. 85-94).

Neary, D.G., Ice, G.G. and Jackson, C.R. 2009. Linkages between forest soils and water quality and quantity. Forest Ecology and Management, 258(10), pp.2269-2281.

Newcombe, C.P., Macdonald, D.D., 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11, 72-82.

Nistor, C.J. and Church, M. 2005. Suspended sediment transport regime in a debris-flow gully on Vancouver Island, British Columbia. Hydrological Processes, 19(4), pp.861-885.

Oregon Department of Environmental Quality (ODEQ) 2010. Turbidity analysis for Oregon public water systems: Water quality in Coast Range drinking water source areas. In. Water Quality Division, Oregon Department of Environmental Quality, Portland, OR.

Preti, F. 2013. Forest protection and protection forest: tree root degradation over hydrological shallow landslides triggering. Ecological Engineering, 61, pp.633-645.

Rachels, A.A., 2018. Quantifying impacts of timber harvesting operations on suspended sediment inputs to Oregon Coast Range headwater streams. In, Water Resources Science. Oregon State University, Corvallis, OR, p. 83.

Rashin, E.B., Clishe, C.J., Loch, A.T., Bell, J.M., 2006. Effectiveness of timber harvest practices for controlling sediment related water quality impacts. Journal of the American Water Resources Association 10, 1307-1327.

Reid, L.M., Dunne, T., 1984. Sediment production from forest road surfaces. Water Resources Research 20, 1753-1761.

Reid, L.M., Dewey, N.J., Lisle, T.E., Hilton,S. 2010. The incidence and role of gullies after logging in a coastal redwood forest. Geomorphology 117.1-2: 155-69.

Reid, L.M., Dunne, T., 2016. Sediment budgets as an organizing framework in fluvial geomorphology. In: Kondolf, G.M., Piégay, H. (Eds.), Tools in Fluvial Geomorphology. John Wiley & Sons, Ltd., Hoboken, NJ pp. 357-379.

Reiter, M., Heffner, J.T., Beech, S., Turner, T., Bilby, R.E., 2009. Temporal and spatial turbidity patterns over 30 years in a managed forest of western Washington. Journal of the American Water Resources Association 45, 793-808.

Richardson, K.N.D., Hatten, J.A., Wheatcroft, R.A., 2018. 1500 years of lake sedimentation due to fire, earthquakes, floods and land clearance in the Oregon Coast Range: geomorphic sensitivity to floods during timber harvest period. Earth Surf. Process. Landf. 43, 1496-1517.

Roering, J.J., Schmidt, K.M., Stock, J.D., Dietrich, W.E. and Montgomery, D.R. 2003. Shallow landsliding, root reinforcement, and the spatial distribution of trees in the Oregon Coast Range. Canadian Geotechnical Journal, 40(2), pp.237-253.

Rose, R. and Haase, D.L. 2006. Guide to reforestation in Oregon. College of Forestry, Oregon State University and Oregon Forest Resources Institute. 48p

Sakals, M.E. and Sidle, R.C. 2004. A spatial and temporal model of root cohesion in forest soils. Canadian Journal of Forest Research, 34(4), pp.950-958.

Schmidt, K.M., Roering, J.J., Stock, J.D., Dietrich, W.E., Montgomery, D.R. and Schaub, T. 2001. The variability of root cohesion as an influence on shallow landslide susceptibility in the Oregon Coast Range. Canadian Geotechnical Journal, 38(5), pp.995-1024.

Sear, D.A., Jones, J.I., Collins, A.L., Hulin, A., Burke, N., Bateman, S., Pattison, I., Naden, P.S., 2016. Does fine sediment source as well as quantity affect salmonid embryo mortality and development? Science of the Total Environment 541, 957-968.

Sheridan, G.J., Noske, P.J., Whipp, R.K., Wijesinghe, N., 2006. The effect of truck traffic and road water content on sediment delivery from unpaved forest roads. Hydrological Processes 20, 1683-1699.

Sidle, R.C. 1992. A theoretical model of the effects of timber harvesting on slope stability. Water Resources Research, 28: 1897-1910.

Sidle, R.C. and Bogaard, T.A. 2016. Dynamic earth system and ecological controls of rainfall-initiated landslides. Earth-Science Reviews, 159, pp.275-291.

Sidle, R.C. and Gomi, T. 2017. Hydrologic processes in forest headwater catchments: Implications for policy and management. In Mountain Watersheds and Ecosystem Services: Balancing multiple demands of forest management in head-watersheds, R.Tognetti, G. Scarascia Mugnozza and T. Hofer (eds). EFI Technical Report 101. Publisher: European Forest Institute, Yliopistokatu 6, FI-80100 Joensuu Finland. p.94.

Sidle, R.C., Ochiai, H., 2006. Landslides: Processes, prediction, and land use. American Geophysical Union, Washington, DC.

Sidle, R.C., Ziegler, A.D., Negishi, J.N., Nik, A.R., Siew, R., Turkelboom, F., 2006. Erosion processes in steep terrain - Truths, myths, and uncertainties related to forest management in Southeast Asia. Forest Ecology and Management 224, 199-225.

Singer, P.C., 2006. DBPs in drinking water: Additional scientific and policy considerations for public health protection. Journal American Water Works Association 98, 73-+.

Slesak, R.A., Schoenholtz, S.H. and Evans, D. 2015. Hillslope erosion two and three years after wildfire, skyline salvage logging, and site preparation in southern Oregon, USA. Forest Ecology and Management, 342, pp.1-7.

Sosa-Perez, G., MacDonald, L.H., 2017a. Effects of closed roads, traffic, and road decommissioning on infiltration and sediment production: A comparative study using rainfall simulations. Catena 159, 93-105.

Sosa-Perez, G., MacDonald, L.H., 2017b. Reductions in road sediment production and road-stream connectivity from two decommissioning treatments. Forest Ecology and Management 398, 116-129.

Stednick, J., 2008. Hydrological and biological responses to forest practices: The Alsea Watershed Study. Springer Science +Business Media, LLC, New York, NY.

Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology 176, 79-95.

Stednick J. D., Troendle C. A. 2016. Chapter 12. Hydrologic effects of forest management practices on water resources. In: Forest Hydrology: Processes, Management, and Assessment. CABI Publishing, 192-203.

Sugden, B.D., 2018. Estimated sediment reduction with forestry Best Management Practices implementation on a legacy forest road network in the northern Rocky Mountains. Forest Science, 64(2), pp.214-224.

Sugden, B.D., Woods, S.W., 2007. Sediment production from forest roads in western Montana. Journal of the American Water Resources Association 43, 193-206.

Swanson, F.J., Benda, L.E., Duncan, S.H., Grant, G.E., Megahan, W.F., Reid, L.M., Ziemer, R.R., 1987. Mass failures and other processes of sediment production in Pacific northwest forest landscapes. In: Salo, E.O., Cundy, T.W. (Eds.), Streamside Management: Forestry and Fishery Interactions, Proceedings of a Symposium, Seattle, WA, pp. 9-38.

Swanson, F.J., Dyrness., C.T., 1975. Impact of clear-cutting and road construction on soil erosion by landslides in the western Cascade Range, Oregon. Geology 3, 393-396.

Swanson, F.J., Scatena, F.N., Dissmeyer, G.E., Fenn, M.E., Verry, E.S. and Lynch, J.A. 2000. Chapter 3: Watershed processes—Fluxes of water, dissolved constituents, and sediment. In Dissmeyer, G.E. (ed.) 2000. Drinking water from Forests and Grasslands: A Synthesis of the Scientific Literature. USDA Forest Service General Technical Report SRS-39, Asheville, North Carolina.

Switalski, T.A., Bissonette, J.A., Deluca, T.H., Luce, C.H., Madej, M.A. 2004. Benefits and impacts of road removal. Frontiers in Ecology and the Environment 2, 21-28.

Takken, I., Croke, J. and Lane, P. 2008. A methodology to assess the delivery of road runoff in forestry environments. Hydrological Processes: An International Journal, 22(2), pp.254-264.

Thompson, M., Sessions, J., Boston, K., Skaugset, A., Tomberlin, D., 2010. Forest road erosion control using multiobjective optimization. Journal of the American Water Resources Association 46, 712-723.

Toman, E.M., Skaugset, A. 2011. Reducing sediment production from forest roads during wet-weather hauling. Transportation Research Record 1.2203: 13-19.

Trimble, S.W., 1983. A sediment budget for Coon Creek basin in the Driftless Area, Wisconsin, 1853-1977. Am. J. Sci. 283, 454-474.

Turner, T.R., Duke, S.D., Fransen, B.R., Reiter, M.L., Kroll, A.J., Ward, J.W., Bach, J.L., Justice, T.E. and Bilby, R.E. 2010. Landslide densities associated with rainfall, stand age, and topography on forested landscapes, southwestern Washington, USA. Forest ecology and management, 259(12), pp.2233-2247.

United States Environmental Protection Agency (USEPA) 2004. Understanding the Safe Drinking Water Act. In. Office of Water, U.S. Environmental Protection Agency, Washington, DC, p. 4.

United States. Environmental Protection Agency (USEPA). 1999. Guidance manual for compliance with the Interim Enhanced Surface Water Treatment Rule: Turbidity provisions. In. Office of Water, U.S. Environmental Protection Agency, Washington, DC.

van Meerveld, H.J., Baird, E.J., Floyd, W.C., 2014. Controls on sediment production from an unpaved resource road in a Pacific maritime watershed. Water Resources Research 50, 4803-4820.

Voli, M.T., Wegmann, K.W., Bohnenstiehl, D.R., Leithold, E., Osburn, C.L., Polyakov, V., 2013. Fingerprinting the sources of suspended sediment delivery to a large municipal drinking water reservoir: Falls Lake, Neuse River, North Carolina, USA. Journal of Soils and Sediments 13, 1692-1707.

Wallbrink, P.J., Croke, J., 2002. A combined rainfall simulator and tracer approach to assess the role of Best Management Practices in minimising sediment redistribution and loss in forests after harvesting. Forest Ecology and Management 170, 217-232.

Walling, D.E., 2005. Tracing suspended sediment sources in catchments and river systems. Science of the Total Environment 344, 159-184.

Walling, D.E., 2013. The evolution of sediment source fingerprinting investigations in fluvial systems. Journal of Soils and Sediments 13, 1658-1675.

Wear, L.R., Aust, W.M., Bolding, M.C., Strahm, B.D., Dolloff, C.A., 2013. Effectiveness of best management practices for sediment reduction at operational forest stream crossings. Forest Ecology and Management 289, 551-561.

Wemple, B.C. and Jones, J.A. 2003. Runoff production on forest roads in a steep, mountain catchment. Water Resources Research, 39(8).

Wemple, B.C., Swanson, F.J., Jones, J.A., 2001. Forest roads and geomorphic process interactions, Cascade Range, Oregon. Earth Surf. Process. Landf. 26, 191-204.

Wheatcroft, R.A., Goñi, M.A., Richardson, K.N. and Borgeld, J.C. 2013. Natural and human impacts on centennial sediment accumulation patterns on the Umpqua River margin, Oregon. Marine Geology, 339, pp.44-56.

Withers, P.J.A., Jarvie, H.P., 2008. Delivery and cycling of phosphorus in rivers: a review. Science of the Total Environment 400, 379-395.

Wohl, E. 2015. Legacy effects on sediments in river corridors. Earth-Science Reviews, 147, pp.30-53.

Wu, J.Y., Rees, P., Storrer, S., Alderisio, K., Dorner, S., 2009. Fate and transport modeling of potential pathogens: The contribution from sediments. Journal of the American Water Resources Association 45, 35-44.

Ziegler, A.D., Sutherland, R.A., Giambelluca, T.W., 2001. Interstorm surface preparation and sediment detachment by vehicle traffic on unpaved mountain roads. Earth Surf. Process. Landf. 26, 235-250.

THIS PAGE INTENTIONALLY LEFT BLANK

Chapter 6. Forest Chemicals

Jon Souder & Bogdan Strimbu

Few issues in contemporary forestry are as contentious as chemical use in management activities. Concerns over chemicals range from their effect on aquatic life, on domestic water supplies on adjacent properties, and on downstream community water supplies. Often, criticism of chemical use is conflated with opposition to clear cutting and even-aged forest management. From the perspective of many forest managers, chemicals provide an effective and safe tool to increase growth and yield, allowing forest lands to remain productive in difficult financial environments. Furthermore, chemicals play a significant role in maintaining forest health (e.g., root rot stump treatments, pheromone baits, and herbicide treatments of host plants e.g. Sudden Oak Death) or the control of invasive species (e.g., gypsy moth treatments). Nevertheless, the competing perspectives associated with usage of chemical in forest management come into play particularly when potable water sources may be affected. Given the range of land uses that can occur on the watersheds supplying drinking water, understanding how chemicals are used in active forest management may assist in resolving concerns about their use.

This chapter will begin with a section contextualizing the use of chemicals in active forest management. We will start this by describing the typical cycle of chemical applications in even-aged management in the Pacific Northwest. Then we will review four years of Oregon Department of Forestry Notifications of Operations (NOAPs) that involve chemical activities. Section two will describe the characteristics of chemicals typically used in forest management. Using the chemicals identified in the NOAPs, in concert with a structured literature search, we will assess peer-reviewed scientific studies related to the effects of chemical use in active forest management in section three, with a focus on water quality in streams adjacent to applications and transport downstream that can potentially result in effects at the raw water intakes of community water supplies. After the science review, section four will examine data from four case studies where water sampling was conducted with the intent to evaluate effects of chemical applications in forest management. One of these studies, in the McKenzie River drainage, was specifically focused on drinking water source protection, while the other three were concerned about effects on aquatic life, particularly ESA-listed fish species. Section five will discuss studies that have identified pesticides at raw water intakes, as well as how these chemicals are treated in the water plant if their levels exceed USEPA drinking water standards. The chapter will conclude with a summary and findings.

6.1. Background

Human activities, such as forestry or agriculture, alter the chemical and physical properties of water in many ways, one of which is the usage of compounds targeting different aspect of forest management, such as, pesticides and fertilizers. Forest pesticides, which include a large set of chemicals such as herbicides, fungicides, insecticides, and rodenticides, are used to aid the re-establishment and management of forest tree species (Dent and Robben 2000). Insecticides are primarily used to control episodic infestations, such as bark beetles and defoliating insects. Fungicides are similarly used in isolated cases to control plant diseases. Many forest landowners use herbicides to control unwanted vegetation competing with tree seedlings. The broad view on herbicides is that they are the most cost-effective means of achieving reforestation objectives. Rodenticides are used during the initial stages of reforestation to control small mammals (mice, mountain beaver) that girdle seedlings; another category of reforestation chemicals are deer and elk repellents to reduce browse damage by large mammals such as deer, elk, and bear. Fertilization is present in some intensively managed plantation forests (Binkley et al. 1999); nitrogen or nitrogen plus phosphorus being the most popular.

6.1.1 Typical Sequence of Forest Chemical Use

The *Pacific Northwest Weed Handbook*, Section M, is the standard reference for vegetation control on forestlands (Kelpsas and Landgren 2019), and includes various types of herbicide treatments and chemical mixtures. Figure 6-1 shows the typical sequence of chemical application used even-aged forest management on private lands, and for some state forestlands (note: forest management activities are significantly different on Federal lands).[1] For intensive forest management as practiced on the Oregon Coast Range, treatment may begin even prior to harvest by suppressing hardwoods, particularly bigleaf maple, in the understory that are likely to be released when the overstory is cut. This is usually done by "hack and squirt", which is a method that introduces the herbicide into the plants by using spaced cuts made at a convenient height. If troublesome grass species are present (such as false brome), then they may be treated pre-harvest by ground-based back pack spray. Site preparation herbicide treatments are usually conducted after harvest, in the summer or early fall prior to seedling planting, usually through aerial application, although sometimes ground-based equipment is used depending on terrain and local regulation. The intent of the site preparation applications is to reduce herbaceous plants (grass and forb) that compete with seedlings for moisture, and to eliminate brush and non-desired tree species that compete with the desired trees for growing space.

Once seedlings are planted, small mammals can girdle stems, "boomers" (mountain beaver) bite off, and deer and elk can either browse seedling tops or pull seedlings out of the ground. Animal repellents and rodenticides are used to reduce these losses until the seedlings are "free to grow," typically by age five to seven. After planting, especially if site preparation treatments were less than effective, a "spring release" herbicide treatment the first year targets grasses and forbs that compete with the seedlings for moisture. A second release spray two to five years after planting may be used if brush competition is still

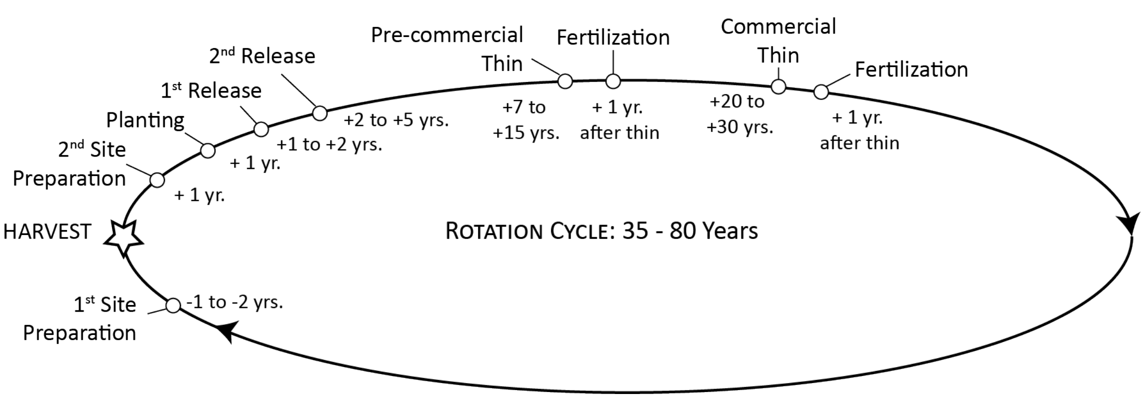


Figure 6-1. Typical even-aged harvest and re-vegetation cycle.

[1] Based on personnel communications with Dr. Carlos Gonzalez-Benecke, Director of the Vegetation Management Research Cooperative in the Oregon State University College of Forestry (2/22/2019), and Mr. Mike Cloughesy, Director of Forestry, Oregon Forest Resources Institute (2/27/2019). It should be noted that in most situations site preparation and release consists of herbicide applications.

high. On the coast, there may be an additional glyphosate spot spray treatments to control unwanted hardwoods. Most forest managers would only conduct one thinning operation, choosing between pre-commercial thins seven to 15 years after planting, or a commercial thin from 20 to 30 years after planting, depending upon site quality and markets. Fertilization is typically just used after thinning to accelerate canopy closure in the residual trees. Once these treatments are done, it is unlikely that further chemical applications will be made in the next 20 to 60 years until just before harvest when the cycle begins again.

6.1.2 Chemicals Used in Oregon Forestry

This section is based on an analysis of ODF Notifications of Operations (NOAPs) covering four calendar years from 2015 to 2018. The ODF provided us with those notifications that involved the application of chemicals covering this four-year period.[2] This data provides (with some limitations) a good overview of how chemicals are used by forest managers on private and State land in Oregon. Federal and most Tribal land managers are not required to submit notifications to the State on their chemical operations; they follow their management plans and ESA biological opinions. With respect to this analysis, the primary limits are that: the notified application may not actually occur; post-application verification of actual type and acreage of chemical application only occurs when an inspection is done by an ODF Stewardship Forester or a complaint is received. **As such, the data provided here should be used as an indication of the types and extent of chemical uses, rather than exact amounts.**

During 2015 to 2018 there were 11,728 chemical application notifications covering 29,511 activities (usually an individual harvest unit or road) submitted through the FERNS e-Notification system. While most NOAPs (60%) cover a single activity—and 91% include three or fewer activities—in extreme cases there can be hundreds of activities included in a single NOAP (the largest was 486). When chemical application is included in the activity, multiple chemicals are usually listed (Figure 6-2). For the 11,728 notifications involving 29,511 activities, there were 222 distinct chemicals in almost 160,000 mentions. These chemicals can include one or more herbicide (or mixtures) as well as "adjuvants," additions to the herbicide formulation to improve its efficacy and/or application. About a third of the activities list up to three, two-thirds list six or fewer, and 75% list seven or fewer chemicals.

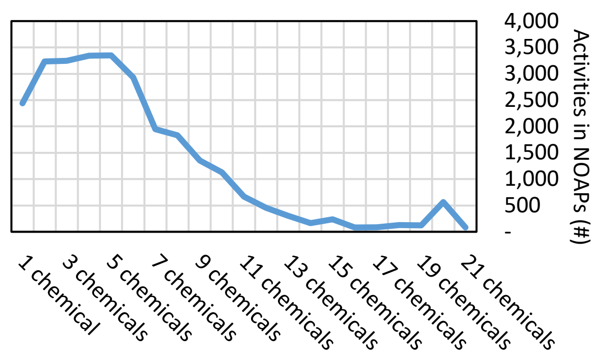


Figure 6-2. Number of chemicals "notified" in NOAP activities, 2015 – 2018.

There are eight categories of chemical activities used in the FERNS system (Table 7-1). Forest chemicals are typically used in planting harvested areas and maintaining roads, but may also be used to treat infestations of insects, fungi, or rodents. Of the notifications submitted during 2015 – 2018, almost 92% were re-vegetation related (animal repellent, fertilizer, herbicide [unit], and rodenticide), road related activities covered another 5.6%, with only three notifications involving forest health (fungicide and

[2] File: FERNS_Chemical_NOAPSs_201902151004 created on 2/15/2019 by Nick R. Wadge, ODF.

insecticides) (Table 6-1). Of the re-vegetation notifications, 87% were for herbicide applications to previously harvested units, covering almost four million acres. However, as we'll discuss below, there may be multiple chemical applications for the same unit within a single notification.

Table 6-1. Purposes of forest chemical application NOAPs, submitted in Oregon during the four year period 2015-2018.

Purpose	Acres	% Acres	Activities	% Activity
Animal Repellent Application	23,925	0.5%	180	0.6%
Fertilizer Application	483,611	10.9%	853	2.9%
Fungicide Application	218	0.0%	1	0.003%
Herbicide Application*	-		1,100	3.7%
Herbicide Application (Road)	-		550	1.9%
Herbicide Application (Unit)	3,843,672	86.5%	26,041	88.2%
Insecticide Application	161	0.0%	2	0.01%
Rodenticide Application	92,632	2.1%	784	2.7%
Grand Total	**4,444,219**		**29,511**	

* County-wide roadside or spot treatment for noxious weeds.

There are 91 different application methods listed in the NOAPs submitted during 2015-2018, many of which are duplicative. In general, they can be divided into aerial- versus ground-based application. All animal repellents are delivered by ground-based spot applications, while the vast majority (≈98%) of fertilizer application is done aerially, typically using a bucket suspended from a helicopter. The fungicides and insecticide treatments were notified as a ground-based spot applications, with only one rodenticide application (<0.1% of the total area) aerially. In the Unit herbicide applications, about 28% were notified as aerial applications, with the remainder ground-based. Ground-based herbicide applications range from stem injection, hack and squirt, backpack sprayers, to ATV and truck-mounted pressurized sprayers (the last two applications are for roads maintenance). For the County-wide and Road herbicide applications, only two out of 1,650 NOAPs notified for aerial application (and these may have been mistakes in the Notification). The vast remainder (82%) were manual spot applications, with another 16% pressurized broadcast.

The 29,511 activities identified in ODF notifications submitted from 2015 to 2018 mentioned the potential application of 222 different chemical formulations. As noted previously, a single activity often listed multiple chemicals that potentially could be applied (Figure 6-2), resulting in a total of 159,014 mentions in the NOAP dataset. **It's important to recognize that not every chemical listed in a notification was applied; actual forestry use statistics are not routinely reported to any governmental agency**, **but must be retained for 3 years and made available upon request by ODF or ODA**.

Our discussion will focus on two different classes of forest chemicals since they constitute the vast majority of those applied: herbicides (71%) and adjuvants (29%). Within the herbicides, there are 27 different active ingredients, with an additional 10 mixtures of two to three active ingredients (see Appendix Table 6-A for the complete list). In terms of their frequency in the notifications, the top ten herbicide active ingredients are shown in Table 6-2, and the number of different formulations (products) for each of the chemicals is shown in Appendix Table 6-A. The active ingredients in herbicides are sometimes mixed (about 11% of total mentions) to obtain synergistic effects, or broaden the range of target weeds. Formulations may differ in the percent of the active ingredient(s), how it bonds with the target weeds (generally amine salt or ester), whether it contains additives that affect its efficacy or volatility, and variations in the composition of its inert compounds. (Martin et al. 2011). Tank mixes are a legal, accepted practice by EPA and ODA. Finally, the same manufacturer may market multiple formulations of the same active ingredient under different names targeted to different uses.

Table 6-2. Top 10 active ingredients in ODF Notifications, 2015-2018.

Active Ingredient	% of Total
Glyphosate	16.64%
Sulfometuron Methyl	15.44%
Triclopyr	13.91%
Imazapyr	12.72%
Metsulfuron Methyl	12.10%
Clopyralid	9.32%
Hexazinone	8.81%
2,4-D	8.54%
Atrazine	5.80%
Aminopyralid	2.85%

* Total percent exceeds 100% due to double counting from mixtures.

The other major category of chemicals applied in Oregon forestry are adjuvants, or additions to the active ingredient (Jordan 2001; Curran and Lingenfelter 2009). Adjuvants represent 29% of the chemical applications mentioned in the NOAPs submitted from 2015 through 2018. There are nine basic types of adjuvants identified in the 45,955 mentions in the NOAPs; the nine types contain 82 different products or formulations (see Table 6-3). *Surfactants* are added to spray mixes to reduce surface tension for better contact with the plant surface, and are over half (58%) of the adjuvants mentioned in the NOAPs. The second most common adjuvants (18%) are *carriers*, used to transport the active ingredient to the target weed. *Deposition aid agents* (17%) increase the proportion of the spray that reaches the target weeds, and work similarly to *drift inhibitors* (0.19%). *Anti-foaming agents* (or defoamers) are added to suppress surface foam and air entrapment (Curren and Ligenfelter 2009); while *buffers* are added to alkaline (hard) water to avoid having the active ingredient bind with chemicals in the water rather than the target plant. *Deodorizers* are used to control odors in the spray formulation, while *emulsifiers* aid in the effective mixing of the spray batch.

Table 6-3. Adjuvants types found in ODF Notifications, CY 2015 – CY2018.

Adjuvant Type	# Products	Frequency	% of Total
Surfactant	46	26,494	57.65%
Carriers	6	8,221	17.89%
Deposition Aid Agents	3	7,662	16.67%
Spray Indicator	5	1,994	4.34%
Anti-foaming Agents	7	1,096	2.38%
Buffers	7	293	0.64%
Deodorizers	3	105	0.23%
Drift Inhibitors	4	86	0.19%
Emulsifier	1	4	0.01%
Totals	82	45,955	

6.2. Forest Chemical Descriptions

The fate of pesticides once applied can be in one of four forms: (1) they can attach to solid matter such as soil or carbon particles; (2) they can dissolve into water; (3) they can vaporize; or (4) they can be taken up by biota such as plants and animals (Ongley 1996). The primary determinant for uptake is their behavior with water: *hydrophilic* pesticides form ionic bonds with water, while *hydrophobic* pesticides repel water molecules because they have no charge (i.e., nonpolar). Hydrophobic pesticides are more likely to attach to soil particles and can be transported as suspended sediments in water; while hydrophilic pesticides dissolved in water can move easily through soil and surface water (ExToxNet 1993). The amount of the pesticide that attaches to soil particles is dependent upon the size of the particle and the amount of organic carbon contained in the particle (Karickhoff 1981). Pesticides degrade through sunlight, water, other chemicals, and microorganisms. How quickly a pesticide degrades, either in soil or water, is based on its partition coefficient and half-life (Hansen et al. 2015). Table 6-4 shows the modes of action; soil sorption coefficients (K_{oc}); solubility in water; vapor pressure; degradation half-life in water (in the presence of light); and degradation in soil (aerobic conditions) for pesticides commonly used in Oregon forestry. The notes to Table 6-4 provide generally accepted thresholds for these attributes by chemical (Lewis et al. 2016). These rates reflect whether the chemical is likely to persist in soil and/or water; vaporize after application; and attach to sediment particles that could be transported downstream.

6.2.1 Insecticides

According to Dent and Robben (2000) and Sundaram and Szeto (1987), three chemically-based pesticides are commonly applied in Oregon forests: carbaryl, diflubenzuron, and chlorothalonil (chlorothalonil, a fungicide, is currently not registered for forestry use in Oregon). A natural, soil-born bacterium, *Bacillus thuringiensi* (Bt), is also used in the Pacific Northwest to control insects. As Table 7-1 shows, there were only two NOAPs for chemical activities involving insecticides (covering at most 146 acres) during the four years, from 2015 – 2018.

Carbaryl (chemical formula $C_{12}H_{11}NO_2$), commonly known under the brand name Sevin, is a 1-naphthyl methylcarbamate from the carbamate family. Carbaryl is solid, white in color, and is primarily used as an insecticide. It is toxic to insects but rapidly eliminated by vertebrates. The main species controlled by carbaryl are aphids, fire ants, fleas, ticks, and spiders. However, carbaryl kills not only the target species but also some beneficial species, such as honeybees or crustaceans (USDHHS 2007). Though toxic to humans, carbaryl is currently approved for use in US. The US Environmental Protection Agency (EPA) has initially classified carbaryl as potential carcinogen (USEPA 2004a) and in 2016 as “not classifiable as to human carcinogenicity” (USEPA 2016), which could increase the risk for diabetes and metabolic disorders as well as impacting circadian rhythms (Popovska-Gorevski et al. 2017).

Table 6-4. Summary of pesticides encountered in Oregon forest management.

Chemical	Mode of Action	Absorption Coefficient (K_{oc})[1]	Solubility Water (mg/L)[2]	Vapor Pressure (mPa)[3]	Water Half-life (days)[4]	Soil Half-life (days)[5]
2,4-D	Growth regulator	24	24,300	0.009	38	4.4
Aminopyralid	Growth regulator	8.3	2,480	2.59×10^{-09}	0.6	35
Atrazine	Photosynthesis inhibitor	174	35	0.039	2.6	75
Carbaryl	Cholinesterase inhibitor	211	9.1	0.0416	10	16
Clopyralid	Growth regulator	5.0	7,850	1.36×10^{-09}	271	23.2
Diflubenzuron	Insecticide: inhibits chitin synthesis	4,620	0.08	0.00012	80	3.0
Glufosinate-ammonium	Nitrogen metabolism	600	500,000	0.0031	Stable	7.4
Glyphosate	Amino acid synthesis inhibitor	16,331	10,500	0.0131	69	15
Hexazinone	Photosynthesis inhibitor	54	33,000	0.03	56	105
Imazapyr	Amino acid synthesis inhibitor	125	9,740	0.013	2.1	11
Metsulfuron methyl	Amino acid synthesis inhibitor	12	2,790	1.40×10^{-08}	Stable	10
Sulfometuron methyl	Amino acid synthesis inhibitor	85	244	7.3×10^{-11}	N/A	24
Triclopyr	Growth regulator	48	8,100	0.1	0.1	39

Sources: National Pesticide Information Center (NPIC). http://npic.orst.edu/HPT/table.html. Accessed 4/22/2019.
Pacific Northwest Weed Management Handbook. https://pnwhandbooks.org/weed/agrichemicals-properties. Accessed 4/22/2019.
Lewis et al. 2016. Pesticide Properties DataBase. https://sitem.herts.ac.uk/aeru/ppdb/en/Reports. Accessed 4/22/2019.

Notes: Definitions and thresholds for herbicide characteristics (Lewis et al. 2016).

1. Freundlick (log K_{oc}) used if available in PPDB; otherwise linear K_{oc}. Values are in mL/g. < 15 = very mobile; 15 - 75 = mobile; 75 - 500 = moderately mobile; 500 - 4000 = slightly mobile; > 4000 = non-mobile.
2. Solubility in water at 20°C. ≤ 50 = low; 50 – 500 = moderate; > 500 = highly soluble.
3. Vapor pressure at 25°C in mPa. < 5.0 = low volatility; 5.0 – 10.0 = moderately volatile; > 10 = highly volatile.
4. Aqueous photolysis (DT50) at pH7. Based on degradation in water exposed to light. < 1 = fast; 1 - 14 = moderately fast; 14 - 30 = slow; > 30 = stable.
5. Soil degradation (aerobic), DT50 (Typical) if available in PPDB. < 30 = non-persistent 30 - 100 = moderately persistent; 100 - 365 = persistent; > 365 = very persistent.

6.2.2 Herbicides

Herbicides are pesticides that target plant pests. Among all of the pesticides applied to the forest, herbicides are the most prevalent in the Pacific Northwest (Temple and Johnson 2011). Herbicides are produced in a variety of states, including liquids, granules, and powders. In addition to the active substances, pesticides include “inert” ingredients. These inert ingredients are not required to be identified but some are known to have toxic properties (Bernstein et al. 2013). To improve the performance of herbicides, other substances called adjuvants are added, as discussed earlier in section 6.1.2. We provide a brief description of each of the major herbicides used in Oregon forest management; included are widely recognized trade name(s) to aid understanding.

2,4-D (sold under various names, such as Crossbow, Weedone, or Brushmaster) kills plants by stimulating uncontrolled growth, and is generally used on broadleaved weeds and woody plants. Some of the formulations are strictly confined to vegetation control in road maintenance. The effects of 2,4-D depend on the form (acid, ester, amine) and product (liquid, dust, granules) (http://npic.orst.edu/factsheets/24Dgen.html). Ester products perform better in early spring and on woody species (Kelpsas and Landgren 2019). It is highly soluble in water, volatile and has a low potential to leach to groundwater based on its chemical properties (Table 6-4). It is non-persistent in soil but may persist in aquatic systems under certain conditions. It is moderately toxic to mammals but should not bioaccumulate (Lewis et al. 2016). 2,4-D was originally patented in 1942 in Great Britain, and in the U.S. in 1945; its original formulation patent expired in 1962, however, other derivations remain patented.

Aminopyralid (Milestone) is a pyridine carboxylic acid herbicide for the long-term control of noxious and invasive broad-leaved weeds. It’s mode of action is systemic, post-emergent absorption by leaves and roots with some residual action. It is non-volatile, very soluble in water and has a high potential for leaching to groundwater. It is moderately persistent in soil, but has low volatility, high solubility in water (and thus high leachability) but degrades rapidly in surface waters (Lewis et al. 2016; NPIC 2019). Aminopyralid was originally registered in 2005, and is considered a low-risk pesticide (USEPA 2005).

Atrazine (Drexel Atrazine 5L) is the second most used herbicide in the US, only after glyphosate (Atwood and Paisley-Jones 2017). It is best used on germinating grasses and broadleaf weeds when they are small (Kelpsas and Landgren 2019). Atrazine is absorbed by plants through roots and foliage, accumulates in the new twigs and leaves where it inhibits photosynthesis. However, in tolerant plants it can be metabolized. Atrazine is soil active, requiring rainfall for activation. It is a restricted-use chemical and can only be purchased and applied by licensed operators.

Atrazine is considered a potential risk to public health through drinking water, and was found to be the most common pesticide detected nationally in drinking water in 2001 (Gilliom et al. 2006). An on-going National program begun in 2003 monitors approximately 150 community water systems (CWS) on a weekly basis during seasons when applications are likely, and biweekly during the remainder of the year. The trigger for monitoring is 2.6 parts per billion (ppb) for finished water or 12.5 ppb for raw water over a 90-day rolling average. Continued exceedance can result in a ban on use of atrazine in the source watershed. About 100 CWSs have been determined to no longer require monitoring; however, another 30 have been added. No CWSs in Oregon are listed in either to 2003-2007 initial monitoring, or the latest 2017 list (https://www.epa.gov/ingredients-used-pesticide-products/atrazine-background-and-updates#drinking-water).

Clopyralid (Stinger, Transline) is a synthetic auxin, killing plants by stimulating uncontrolled growth. Clopyralid is used to control selected broadleaf weeds (including thistles) and elderberry (Kelpsas and Landgren 2019). It is highly soluble in water, volatile in air, and has a high risk of it leaching to groundwater (Table 6-4). It can be persistent in both soil and water systems depending upon conditions. It has a low mammalian toxicity and is not expected to bioaccumulate (Lewis et al. 2016).

Glyphosate (Roundup, and a wide variety of product names) is the most sold herbicide in the US (Atwood and Paisley-Jones 2017). It is poorly absorbed by the digestive tract and is almost entirely eliminated unchanged through mammal excrements (Extension Toxicology Network 2019a). Minute amounts of glyphosate can be found in tissues ten days after treatment. Numerous field and laboratory experiments on animals suggest that glyphosate has no impact on reproduction, which led to the assumption that the compound was unlikely to have any reproductive effects in humans (Extension Toxicology Network 2019a). Glyphosate went off U.S. patent in 2000, which has led to the development of several hundred products (http://www.glyphosate.eu/history-glyphosate; http://npic.orst.edu/factsheets/glyphogen.html).

A byproduct of glyphosate, aminomethylphosphonic acid (AMPA) is formed by the breakdown of glyphosate by microorganisms in soil and water, with one molecule of glyphosate creating one molecule of AMPA. Grandcoin et al. (2017) published a recent review of AMPA sources, behavior, and fate in natural waters. In addition to glyphosate as a source, AMPA is also formed from the breakdown of phosphonates, typically found in detergents and other industrial uses, and enter streams through waste water treatment plants. Strongly adsorbed to soil particles, AMPA can persist in the soil and move into streams through erosion and sedimentation. While little is known about the toxicity of AMPA, it appears to be readily removed by most potable water treatment processes (Grandcoin et al. 2017).

Hexazinone (Velpar and others) is a broad-spectrum herbicide used to control grasses, broad-leaved weeds, and woody plants by inhibiting photosynthesis (Kelpsas and Landgren 2019). It is active on contact and in the soil, absorbed through plant roots and foliage (Lewis et al. 2016). It can be long lived in soil, rated as having high leachibility, low volatility, and very soluble and moderately persistent in water (Table 6-4) (Lewis et al. 2016; NPIC 2019).

Imazapyr (Arsenal, Chopper, Habitat) is an herbicide used to control a broad range of annual and perennial weeds, as well as some woody species. Imazapyr went off U.S. patent in 2002, leading to the development of hundreds of new formulations. Imazapyr acts as a meristem inhibitor through inhibition of amino acid branched chain biosynthesis (NCBI 2020a). Imazapyr degrades in clear waters and is persistent and mobile in soil (Table 6-4). According to the EPA, there is little risk of toxicity to fish and aquatic invertebrates at maximum application rates. Imazapyr is categorized by the EPA as practically non-toxic to avian species, small mammals, and honey bees. The EPA assessment on carcinogenicity states that imazapyr is of “no concern for human carcinogenicity.” Research coordinated by the US Forest Service suggests that imazapyr does not degrade quickly in soils (Durkin 2011), which is supported by the findings of Jarvis et al. (2006) who found an initial half-life of approximately 123 days and a terminal half-life of approximately 2,972 days.

Metsulfuron Methyl (Escort XP) is an acetolactate synthase (ALS) inhibitor that obstructs a key enzyme required for amino acid synthesis (UC-IPM 2019). It is readily absorbed by both roots and foliage and translocated to leaves and stems. It is used to control ferns, and is especially effective on all *Rubus* (blackberry, salmonberry, etc.), as well as other herbaceous species (Kelpsas and Landgren 2019). Metsulfuron-methyl is moderately soluble in water, and unlikely to volatilize (Table 6-4) (NPIC 2019).

Sulfometuron Methyl (Oust) is a broad spectrum urea-based herbicide used in forestry to control woody tree species by inhibiting the synthesis of branched-chain amino acids, such as leucine and isoleucine. Microorganisms from soil and hydrolysis occurring inside water break down sulfometuron (NPIC 2020). Depending on water acidity, sulfometuron has a half-life between 10 days and 8 weeks (NCBI 2020b; NPIC 2020). The compound is non-toxic to birds and slightly toxic to fish. The EPA detected no carcinogenic effects on humans from sulfometuron (NCBI 2020b).

Triclopyr (various product names) is a selective herbicide that controls woody and broadleaf plants. Triclopyr converts rapidly to a salt in natural soil and in aquatic environments. In water, breakdown by the action of sunlight is the main source of triclopyr degradation. The half-life in soil is from 30 to 90 days, while in water is less than one day (Table 6-4). Triclopyr is relatively toxic to birds, such as mallards, but not to bees, fish or aquatic invertebrates (NCBI 2020c)). According to the EPA, triclopyr is "not classifiable as to human carcinogenicity" (USEPA 1998).

Mixtures of Herbicides. Herbicides are commonly mixed with the intent to improve the control (Damalas 2004). This occurs in two ways: commercial products that contain multiple active ingredients; and "tank mixtures" where the applicator determines the chemicals and their concentrations. Active ingredients are "tank mixed" to combine desirable properties, usually to widen the range of target species killed (Damalas 2004).

Damalas (2004) reviewed common interactions among herbicide tank mixtures. He determined that there were three times more cases where the mixes were antagonistic (i.e. reduced activity) as compared to those where the interactions were synergistic (i.e., increase activity). In general, herbicides from the same chemical group were more likely to by synergistic, while combinations from different groups may interact with each other and become deactivated, leading to antagonistic outcomes (Damalas 2004). Kelpsas and Landgren (2019) provide specific tank mix suggestions by target weed species found in forests of the Pacific northwest.

6.2.3 Adjuvants, including surfactants.

Adjuvants are defined as "a material added to a tank mix to aid or modify the action of an agrichemical, or the physical characteristics of the mixture" (ASTM 2016). The American Society of Testing and Materials (ASTM International, www.astm.org) provides standard terminology and definitions related to adjuvants. Adjuvants can also be certified for their performance and applicator safety by the Council of Producers & Distributors of Agrotechnology (CPDA 2019). Adjuvants are commonly included in spray mixtures (Hartzler 2020). Adjuvants help herbicides pass into leaf cells through the leaf surface. They were developed to improve herbicide penetration of leaves, as well as aid in the spreading, wetting, and adhesion of herbicides to leaves. Furthermore, some adjuvants serve to reduce herbicide drift, eliminate foaming problems in spray tank mixtures, or reduce alkaline hydrolysis (CPDA 2019). Penner (2000) categorizes adjuvants into three classes: (1) activators that increase herbicide activity, absorption, and spread; and decrease photo-transformation of the herbicide; (2) spray modifiers that alter the physical characteristics of the spray; and, (3) utility modifiers that widen the conditions under which the herbicide is useful. Activator adjuvants (e.g., surfactants, spreader-stickers, wetting agents or penetrants) are commonly used to improve the performance of post-emergent herbicides by increasing herbicide retention or penetration on or into leaf surfaces, rainfastness, or to decrease photo-degradation of herbicides.

A wide array of adjuvants are available to enhance herbicide efficacy, including surfactants, oil concentrates, ammonium-N fertilizers, spreader-stickers, wetting agents, and penetrants (Curran et al. 1999; Hartzler 2020). Surfactants, particularly nonionic ones, are suitable as dispersing agents aimed at improving plant coverage and foliar penetration with low toxicity to the crop plants themselves. Oil concentrates usually improve penetrability of the herbicide into the leaves. Some fertilizers, such as liquid N fertilizer products, can act as adjuvants and improve the performance of some herbicides, particularly if the mix water is hard. Solutions of liquid N fertilizer are commonly encountered in combination with nonionic surfactants and oil concentrates. Adjuvants can be combined to provide multiple functions, such as ammonium sulfate, which is used to improve herbicide performance in drought conditions or in tank mixtures.

Some of the most popular adjuvants encountered in forestry applications in Oregon are listed in Table 6-5 (Bernstein *et al.*, 2013). The purpose of each adjuvant depends on the application method and intended use of the active ingredient of the herbicide according to the manufacturer's label.

Table 6-5. Manufacturer stated purpose for adjuvants in combination with herbicides.

Adjuvants	**Manufacturer stated purpose**
Methylated seed oil	"Enhances the consistency or performance of certain post-emergence herbicides" and "improves leaf coverage and absorption."
Foambuster	Helps "defoamer for use in aqueous solutions".
Dyne-Amic	Serves as "nonionic surfactants".
Grounded	Is "designed to enhance the deposition and absorption of both ground and aerial spray applications".
Sta-put	"Improves deposition in the target swath and can retard, but not totally prevent drift."
Syl-Tac	"Provides spreading, wetting, and penetration on the leaf surface."

6.2.4 Fertilizers

The practice of using fertilizers in forest management is widespread in the southern region of the US, with more than 1.2 million acres treated annually with nitrogen or phosphorus (Fox et al. 2007), as well as in the Pacific Northwest (Binkley et al. 1999). About 125,000 acre of forestlands are fertilized annually in Oregon (Table 7.1). In addition to interest in the effects of fertilization on tree growth for commercial production, increasing emphasis has been placed on effects on carbon sequestration and carbon and water fluxes. Nutrient dynamics are covered in Chapter 3, Section 3.3.1. In this chapter we will focus on the addition of fertilizers and their potential effects.

Nitrogen/Urea. Nitrogen and urea are the most commonly used fertilizers. In the Pacific Northwest, it is believed that tree growth is constrained by available nitrogen; therefore, young plantations are sometimes fertilized with nitrogen, which is often delivered as pellets of urea, $(NH_2)_2CO$ and have a 46% content of N (Anderson 2002). The most common rate of application is approximately 200 lb N/ac (or 224 kg N/ha), an amount that balances tree growth with N-losses (Anderson 2002; Flint et al. 2008; Cornejo-OIviedo et al. 2017; Putney 2019). The EPA states that there is "inadequate information to assess the carcinogenic potential" of urea (Persad et al. 2011). Urea can produce skin irritation, but is more likely to do so when petroleum is part of the formulation (Persad et al. 2011).

6.2.5 Rodenticides

The usage of rodenticides in the Pacific Northwest is not as widespread as fertilization, with about 25,000 ac treated annually in Oregon (Table 7.1), since most applications are site-specific rather than broadcast. There are three general types of animals damage seedlings and small trees: voles (*Microtus* spp.), pocket gophers (*Thomomys* spp.), and mountain beavers (*Aplodontia rufa*) (Arjo and Bryson 2007). Most rodenticides are applied underground in the target species' burrows. There are three different types of rodenticides registered for use in Oregon forest management (https://ferns.odf.oregon.gov/E-Notification).

Zinc phosphide (various product names) is commonly used for rodent and lagomorph (rabbit) control. For rodenticides, zinc phosphide is commonly applied as granules that are ingested. Zinc phosphide produces phosphine gas in the presence of moisture, which then disrupts mitochondrial respiration and blocks protein and enzyme synthesis (NPIC 2019). Zinc phosphide has low solubility in water, low volatility, and is considered non-persistent in soil (Lewis et al. 2016).

Chlorophacinone (Rozol) is an anti-coagulant used for gopher and mice control. It acts by stopping the enzyme that produces vitamin K, needed for blood clotting (NPIC 2019). It requires multiple days of eating before it becomes effective (NPIC 2019). Chlorophacinone has low solubility in water, low volatility, low leachability, and is moderately persistent in soils (Lewis et al. 2016).

Strychnine (RCO Omega Gopher Grain Bait) is also used as a rodenticide, and has documented efficacy against pocket gophers (Evans et al. 1990). It works by causing cells in the spinal cord to fire rapidly, causing muscle spasms that can result in asphyxia and death (NPIC 2019). Strychnine is a restricted-use chemical, can only be formulated as less than 0.5% active ingredient, and must be applied only below ground (NPIC 2019).

6.2.6 Animal Repellents

There are four types of animal repellents: those that work on fear, those that create a conditioned response due to prior consumption of the repellent, those that cause instantaneous pain on contact, and those that taste bad (Trent et al. 2001). The fear-based repellents usually contain sulfurous compounds such as urine from predators, meat proteins (and blood meal), garlic, or putrescent egg solids. Conditioned response repellents are designed to make the animal ill so that they will avoid the treated plant in the future. Thiram (tetramethylthiuram disulfide) is the chemical most used for this; [note it's registered by USEPA as both an animal repellent and fungicide, but doesn't show up on the

FERNS list of chemicals, USEPA doesn't consider it a potential threat to drinking water quality, but its restricted-use to only commercial operators]. Contact repellents mostly rely on capsaicin (chili) or ammonia to immediately irritate the throat or nostrils of the animal. Taste repellents are usually bitter: bitrex (denatonium benzoate) has the Guinness Book of Records for the most bitter substance and is widely used to prevent children from ingesting products such as antifreeze, detergents, cleaners, and scented markers. Denatonium benzoate is considered to have toxicological concern, but little risk, due to it usage pattern (Lewis et al. 2016).

6.2.7 Formulation used in forestry applications

The pesticides commonly encountered in forestry applications are combinations chemicals—called formulations—that effectively control the pest. A pesticide formulation is a mixture of active and inactive ingredients: the former prevents, kills, or repels a pest to act on a plant; and, the latter enhances the effectiveness of the active ingredient or ensure an easier and safer manipulation or application. The presence of many formulations is driven by three factors: variations in solubility of the active ingredient, ability to control the pest, and easiness to handle and transport. The formulations are delivered in two states: fluid or solid. Liquid formulations are solution when chemicals are generally mixed with water. However, there are formulations when crop oil, diesel fuel, or kerosene are present (Fishel, 2013). The liquid formulation can be separated in several categories, based on the combination of chemical components: emulsifiable concentrates, ready-to-use solutions, ultra-low volume, invert emulsions, aerosols, and liquid baits. The solid formulations can be grouped in ready to use and concentrates, which requires further mixture with a fluid (usually water). The solid formulations are encounters under the following forms, which depends on the size of the particle: dusts, granules, pellets. There are authors that include the soluble or wettable powders and water dissolvable granules as solid formulations, but in essence they are a combination of liquid and solids. According to Perry and Randall (2000), the main formulations encountered in forestry applications are solutions (i.e., substances soluble in water or other solvents, such as fuel oil), emulsions (i.e., two unlike liquids mixed together), wettable powders (i.e., finely divided solid particles that can be dispersed in a liquid), and granules (i.e., crystals of the effective chemical bound together with an inert carrier).

6.3. Science Review of the Effects of Forest Chemicals on Source Water Quality

The present study is based on 116 articles and reports, of which 96 were published following peer review. Because the impact of forest activities on the chemical composition of water is a major topic of interest, several major review papers were written in the last two decades (Binkley et al. 1999; Anderson 2002; Michael 2004; Tatum et al. 2017). Among these, two focused on fertilizers, Binkley et al. (1999) and Anderson (2002), and two on herbicides, Michael (2004) and Tatum et al. (2017).

6.3.1 Forest Chemicals and Changes to the Composition of Water

Many issues of concern associated with the application of herbicides and fertilizers to manage vegetation involve the unintended collateral effects on other plants, animals, water, and air (Lautenschlager and Sullivan 2004; Tatum 2004; Louch et al. 2017). Because the objective of this report is to assess the impact of chemicals on drinking water quality, we will limit the discourse only to herbicides and fertilizers that may affect raw drinking water quality. Since there were only two NOAPs over the four year period (covering a maximum of 161 acres), insecticides will not be covered. It is important to note, however, that their application could have adverse effects on water quality.

6.3.2.1. Fate and Movement of Forest Chemicals. The movement of herbicides through the soil profile depends on a variety of degradative and dilution processes. Biological and chemical processes play a large role in impeding herbicide movement through soil profiles by destroying the herbicide molecule (Michael 2004). The residence time of chemicals in a given environment is measured with the half-life, which is the time needed for dissipation of half of the amount applied (Michael and Neary 1993). Half-life is measured in days, and for most herbicides commonly used in silvicultural applications, is less than 90 days (Wauchope et al. 1992). Some are as low as 10 or 20 days, for example 2,4-D or sulfometuron (NCBI 2020b, 2020d).

A vast array of mechanisms impact the fate and toxicity of herbicides, which can be grouped in biotic and abiotic processes (Fenner *et al.*, 2013). The main abiotic mechanisms (i.e., without involving organisms) occurring in the forested environment are hydrolysis and photolysis (Büyüksönmez *et al.*, 1999). *Hydrolysis*, which is the major transformation process for organophosphate and carbamate pesticides, cleaves chemical bonds by the addition of water. *Photolysis* is a transformation of a molecule when excited by ultraviolet light, which can transform carbaryl, for example, into 1-naphthol and methyl isocyanate (MIC), which is highly toxic. However, Büyüksönmez *et al.* (1999) argued that photolysis does not play an important role in actual degradation, except in limited cases, because usually “only a small portion of the substrate is exposed to light”. Biotic mechanisms that transform the herbicides are processes occurring in the presence of microorganisms. The biological transformations breakdown the pesticides when the chemical compound is bioavailable and is compatible with enzymes produced by the microorganisms.

In general, herbicide movement through soil is slow and most forest herbicides have not been detected deep into the soil (Vasilakoglou et al. 2001; Beulke et al. 2004; Weber et al. 2007). Glyphosate, almost immobile in soil, has not been found below 15 cm (USEPA 1993). Triclopyr has been found to depth up to 30 cm (Lee et al. 1985, Stephenson et al. 1990), whereas hexazinone has been detected as deep as 75 cm (Roy et al. 1989; Feng and Navratil 1990; Allender 1991; Michael et al. 1999). Imazapyr is rarely found below 50 cm, but has been detected under 30 cm in several soil types (Rahman et al. 1993). Similarly, sulfometuron and metsulfuron move up to 50 cm, but are not commonly found below 70 cm (Walker and Welch 1989; Lym and Swenson 1991).

An important role in the biological degradation of herbicides is advective dispersion, which slows the movement of herbicide through the soil profile. In advective dispersion, the solute front is partially slowed by interaction with the soil. This advective slowing acts on two directions. First, it dilutes the front; second, it retains the herbicide in the root zone for a longer period of time during which degradation can occur. Soils with more organic matter and clay have superior advective dispersion of nonpolar pesticides. Advective dispersion and the lack of significant movement through the soil profile impede groundwater contamination by forestry herbicides (Michael 2003). Weber et al. (2007) found that atrazine mobility depends on the type of soil and water solubility of the chemicals. They also found that the amount of herbicide present in soil after 4 months is a function of the amount of organic matter and is inversely related with soil pH and soil leaching potential. However, their findings are not robust, as a repeated analysis was used in interpretation of the data, which is sensitive to violation of assumptions. Because the authors did not provide any evidence that the assumptions were met, the interpretation is not necessarily expandable to other areas or chemicals; however, they do provide a perspective on the physicochemical properties of soils on herbicide mobility.

Michael (2004) argued that “the maximum concentrations of herbicide observed in streams is related to the method of application” particularly if applied to ephemeral or intermittent streams. Broadcast applications are generally associated with the highest concentration observed during the day of application because control on where the herbicide will land is more limited. If application occurs when the ephemeral or intermittent channels contain water, then the herbicides may reach perennial streams. Several studies suggest that on the application day, aerial broadcast applications may result in concentrations of herbicide in streams that are twice as large as concentrations resulting from overland flow during a first storm in the absence of buffer areas (Michael et al. 2006; McBroom et al. 2013; Scarbrough et al. 2015; Louch et al. 2017). Baseflow between storms contains herbicide with concentrations near or below analytical detection limits (Michael and Neary 1993, Michael et al. 1999; Michael 2003). Storms after herbicide application may contaminate the stream until, at most, the 5th storm, when herbicides are typically no longer detected in streams (Louch et al. 2017). Nevertheless, irrespective of the recording time of the application timing, the maximum concentrations observed in streams last from a few minutes to a few hours (Louch et al. 2017). The largest concentrations occur during storm runoff and seldom last longer than 30 minutes, but even these highest concentrations rarely exceed drinking water quality standards (Michael 2004).

Downslope movement of herbicides occurs principally in the form of overland flow or macropore flow (Bastardie et al. 2002; Buttle and McDonald 2002). Overland flow, when occurring immediately after herbicide application, can contain high concentrations of herbicide that could reach streams (Michael 2004). Overland flow depends on the antecedent soil moisture conditions, precipitation rate, infiltration rates, and drainage capacity. However, overland flows almost always occur when the instantaneous precipitation rate exceeds the infiltration rate, which in Pacific Northwest rarely corresponds to spraying periods. Current modeling approaches of water movement in the Pacific Northwest mention Hortonian overland flow, but there is no specific term representing it numerically, which suggests that overland flow is a rare occurrence (Wu et al. 2012). For low antecedent soil moisture and high infiltration rates, almost no downslope movement occurs. Alternatively, for high antecedent soil moisture and saturated soil, the infiltration rates may be exceeded, which results in overland flow. Overland flow will almost always appear on poorly drained soils compared with the well-drained soils, given similar slope, precipitation intensity and duration. Fast movement of the overland flow leads to higher contamination levels of streams than when herbicides reach streams through baseflow by leaching through the soil (Michael et al. 1999; Michael 2003).Beside overland flow, the macropore flow can also contribute to downslope movement of herbicides (Shipitalo et al. 2000).

6.3.2.2. Herbicide Active Ingredients used in Forestry. In this study we will review the fate and toxicity of most commonly used herbicides (Clark et al. 2009; Dinger and Rose 2010; and Bernstein et al. 2013), namely 2,4-D, atrazine, glyphosate, hexazinone, imazapyr, sulfometuron, and triclopyr.

2,4-D is a phenoxyacetic acid compounds that controls broadleaf weeds. The Oregon Department of Forestry monitored several herbicide applications (Dent and Robben 2000) and found that aerial broadcasts of 2,4-D – as the formulation Low Vol 6 (Loveland Products), at a rate of 38.4 and 56.8 oz

/ac, and a concentration less than 90% – resulted in insignificant surface water values compared with the water criteria for human health (i.e., 0.14 ppb measured vs. 300 ppb threshold).

Aminopyralid is a pyridine carboxylic acid herbicide aiming at management of rangeland, pastures, and natural areas (wildlife management areas, natural recreation areas, campgrounds, trailheads, and trails). Aminopyralid controls broadly the systemic post-emergence of a number of noxious and invasive species (USEPA 2005). A benefit of applying aminopyralid is its residual weed control, which limits re-infestations and reduces the subsequent re-treatment (USEPA 2005). The EPA found that aminopyralid is practically non-toxic to non-target animals and is less likely to impact terrestrial and aquatic plants (USEPA 2005).

Atrazine is triazine chemical, used for controlling broadleaf and grassy weeds**.** When a 10 m buffer are used, stream management zones have been effective at reducing the amount of atrazine reaching the stream by at least 25% for slopes less than 22% (Matos et al. 2008; Pinho et al. 2008). Atrazine can contaminate surface water and groundwater by runoff from row crops (NCBI 2020d). Atrazine was found to be slightly to moderately toxic to humans through oral, dermal, and inhalation exposure; only slightly toxic to birds and fish, and practically non-toxic to bees (NCBI 2020d).

Clopyralid, which contains hexachlorobenzene and pentachlorobenzene as active ingredients, is an herbicide targeting primarily broadleaf weeds (Durkin and Follansbee 2004). Typical application of clopyralid is through backpacks, even though aerial broadcastings may also be used. The toxicity of clopyralid is relatively well-studied for mammals, which suggests that for humans the cancer risk is estimated to be low (Durkin and Follansbee 2004). Clopyralid is expected to have high mobility in soil and is not expected to be adsorb by the suspended solids and sediment from stream water. Clopyralid exhibits low toxicity to fish, and is relatively non toxic to birds, bees and spiders (Durkin and Follansbee, 2004).

Glyphosate is an aminophosphonic analogue of the natural amino acid glycine that acts by inhibiting the enzymes used to metabolize amino acids, thus regulating plant growth. It is the most sold herbicide in the US (Atwood and Paisley-Jones 2017). Glyphosate is poorly absorbed by the digestive tract and is almost entirely eliminated unchanged through mammal excrements (NCBI 2020e). Minute amounts of glyphosate can be found in tissues ten days after treatment. Numerous field and laboratory experiments on animals suggest that glyphosate has no impact on reproduction, which led to the assumption that the compound was unlikely to have any reproductive effects in humans (Extension Toxicology Network 2019a). Glyphosate, measured in stream as pulses defined by the storm events, does not seem to be short lived, as Louch et al. (2017) in the Alsea Watershed Study found that glyphosate is present in water after almost one month and after six rain events. In contrast, Caldwell and Courtner (2020) didn't find glyphosate in stream water after application; both the Louch et al. (2017) and Caldwell and Courtner (2020) studies will be reviewed in detail below.

Hexazinone is a triazine herbicide used against a series of weeds and some woody plants (Tu et al. 2001). Hexazinone is a systemic herbicide that inhibits photosynthesis of the targeted plants. (NCBI 2020f) mentions that hexazinone is "unlikely to be carcinogenic to humans under normal circumstances." Furthermore, Hexazinone is considered slightly to nontoxic for birds and bees but slightly toxic to fish and other freshwater organisms.

Imazapyr, which is member of the imidazolinone class of herbicide, is used extensively in both the southern US and Pacific Northwest. As with other herbicides, there are many formulations of imazapyr, the most popular one being Arsenal, which is produced by BASF. In several experiments reported by the USEPA in their Registration Review (USEPA 2014), imazapyr in its isopropylamine salt form, more so than its acid form, is likely to damage aquatic macrophytes (i.e., aquatic plants growing in or near water), as well as some species of algae. Both forms are considered to be toxic to terrestrial plants (USEPA 2014). The USEPA considers imazapyr as "practically non-toxic to mammals, birds, honeybees, and fish", it can be inferred that it poses little risk to humans or other species of animals (USEPA 2014). Tatum (2004) notes that similar to glyphosate, imazapyr poses insignificant risk to invertebrates when exposed to environmentally relevant concentrations. The exposures to imazapyr based on recommended dosages within the best management practices (BMP) framework raise only minimal concern for animals, except for reptiles and amphibians (Trumbo and Waligora 2009), for which there is a lack of data (Durkin 2011; Tatum et al. 2017).

Metsulfuron methyl is a sulfonylurea compound used as a herbicide for broadleaf weeds and some grasses (NCBI 2020g). Metsulfuron methyl acts by inhibiting cellular division of the shoots and roots. Metsulfuron-methyl has low toxicity for birds, aquatic organisms, and honey bees. The U.S. Environmental Protection Agency classifies metsulfuron-methyl as toxicity class III, being unlikely "to be carcinogenic to humans" considering that tests on rats did not exhibited increase in the number of tumors (NCBI 2020g).

Sulfometuron methyl, a benzoate ester that is the methyl ester of the benzoic acid**,** is active at very low concentrations, and is broadcast on forest sites at rates of as low as 26 g/ ha (Paranjape et al. 2015). Sulfometuron is relatively low soluble in water and increasing pH (10 mg L^{-1}at 25°C and pH 5.5 according to the manufacturer (NCBI 2020b). In pine plantations, Michael (2003) found that the type of application has a significantly different impact on water composition. Sprayed sulfometuron leads to only 12.5% of the water samples with quantifiable residues of sulfometuron, whereas the pelleted application, even at lower rates, delivered higher concentrations in more than 70% of the samples. Considering that the sulfometuron applications were combined with a BMP of a 15 m untreated stream management zone, Michael (2003) concluded that "adverse impacts on watersheds in the southern USA are unlikely." Based on the assumptions in the Michael (2003) study, this finding cannot necessarily be extrapolated to the Pacific Northwest.

Triclopyr, a monocarboxylic acid, is a selective systemic herbicide used for control of woody and broadleaf (NCBI 2020c). Triclopyr is slightly to practically nontoxic to birds, fish, and bees (USEPA 1998). The US EPA classified triclopyr as a "Group D chemical (not classifiable as to human carcinogenicity)", based on studies executed on rats (USEPA 1998).

6.3.2.3. Changes to Water Quality Due to Herbicides. Besides the review papers mentioned above, three studies have focused on the impact of herbicides used in silvicultural practices on water quality. Relevant to the Pacific Northwest, are Thistle et al. (2009), Louch et al. (2017), and Caldwell and Courter (2020). We will go into greater detail reviewing the last two papers since their results are directly applicable to the most commonly used chemicals and application techniques (aerial spraying) in the Pacific northwest.

Thistle et al. (2009) evaluated the efficacy of riparian buffers to reduce spray drift into live streams. Conducted in the Coast Range west of Corvallis, Oregon, with stream buffers representing FPA required widths for medium (70') and small (50') fish-bearing streams, the study used fine droplets of water containing sulfoflavine fluorescent dye as a proxy for aerially applied herbicides. Using fine droplets allowed for more precise evaluation of drift since they become entrained in airflows traveling towards the riparian buffer. Thistle et al. (2009) results showed that the riparian buffers captured or deflected approximately 90% of fine spray drift, with weak evidence that the wider buffers captured marginally more fine spray droplets. Intermediate density buffers (not clearly defined in the paper) appear to capture a greater fraction of fine droplets compared to open buffers (where there is little interception by vegetation) and dense buffers (where air flows are diverted above and over the riparian area. These findings are consistent with a review stream management zones and herbicide applications in the U.S. and New Zealand (Tatum et al. 2017).

Louch et al. (2017) carried out an extensive study the impact of aerially-applied herbicides in Oregon's Coast Range using the Needle Branch watershed that was part of the Alsea Revisited paired watershed study (http://www.watershedsresearch.org/watershed-studies). An earlier, and much more extensive, reporting of the sample results is found in NCASI (2013). Glyphosate, imazapyr, sulfometuron methyl, and metsulfuron methyl, and the glyphosate breakdown chemical, aminomethylphosphonic acid (AMPA) were evaluated in a single herbicide application to the 91 acre unit on August 22, 2010. Three stream gaging stations had been established as part of the larger Alsea Revisited paired watershed study, and sampling for herbicides in stream water was conducted at these three sites. The upper portion of the unit (above the High gaging station) is classified as a Small Nonfish stream under the FPA, as such is does not require a riparian buffer (although there is a statutory 10' setback from spraying open water areas). The High sampling site was at the boundary between the SN and SF stream segments, above this site no riparian buffer is required; the Mid site was at the bottom of the harvest unit containing the SF stream segment where there was a 50' buffer required; and the Low sampling site was approximately 1 km downstream from the spraying.

Samples were automatically collected hourly from just before the application and continuing 24 hours at the three sites. Automatic sampling was manually triggered when storms were predicted, with collection intervals ranging from hourly to every six hours. Subsequently, grab samples were taken approximately weekly during base flows between storms. Two methods were used to analyze the glyphosate and AMPA samples: high performance liquid chromatography coupled with fluorescence (LC/F) for all the samples; and, liquid chromatography-tandem mass spectrometry (LC/MS-MS) for a smaller subset of the samples. The subset of samples analyzed by LC/MS-MS are more precise, however, less sensitive than LC/F, and only about 7% of samples collected were analyzed by LC/MS-MS, and then only for glyphosate and AMPA (NCASI 2013).

Table 6-5 shows the highest herbicide concentrations found in Needle Branch Creek during and after the spray application. Unfortunately, a background “interferent” of unknown composition that appeared to vary from sample-to-sample for glyphosate and AMPA, and affected the imazapyr analyzes as well; leading the authors to contend that the LC/F results are “high biased” by unknown amounts (NCASI 2013). Imazapyr, SMM and MSM concentrations were below the method detection limits (MDL) so they were not analyzed with LC/MS-MS. “Thus, the absolute bias in the LC/F result for any given sample is unknown” (Louch et al. 2017, 400)

They found that glyphosate was present in water above the regulatory thresholds in SN stream locations close to the application sites where no riparian buffer was required. Unfortunately, due to equipment failure, no glyphosate samples were available during and after herbicide application at the SF sample site. The other four herbicides had concentrations so low that they did not expect impacts on any other organisms, other than aquatic plants. Furthermore, the concentrations were so minute that even pulses of any exposure could be mitigated (Table 6-5). To summarize, Louch et al. (2017) concluded that glyphosate had no impact on site-specific aquatic organisms (in water) and little risk to the Needle Branch aquatic community (in suspended sediment); that AMPA also most likely hadt “no effect”; iImazapyr most likely result was “no effect”; sulfometuron methyl (SMM) was wWell below the levels shown to have adverse effects on fish, amphibians, or invertebrates; and metsulfuron methyl (MSM) was well below the levels shown to have adverse effects on fish, amphibians, or invertebrates.

As in many field-based evaluations, there were problems in Louch et al (2017) that likely affected their results:

1. Auto-sampler for glyphosate and AMPA failed at MID during application that precludes evaluating the effects of a riparian buffer on in-stream concentrations. There were pulses of glyphosate at HIGH where there was no buffer (other than about 3m boom turned off on stream side). Because of sampler failures, it’s not possible to determine if there were pulses at MID immediately after application, with or without the contribution from the unbuffered upper reach, and one significant N tributary that enters Needle Branch.

2. Auto-sampler for Imazapyr, SMM, and MSM failed at HIGH during application. Precludes understanding impacts of potential drift on these three chemicals when no stream buffer is present. Non-detect at MID during application may not have captured what had happened above due to time-of-travel from the unbuffered upper reach. No analysis was conducted of samples during the application period at LOW since the MID samples were considered Non-Detect.

3. Disturbing difference in results based on two different techniques run by different organizations. The LC/F (liquid chromatography/fluorescence) results run by NCASI were approximately twice the concentrations of the LC/MS-MS (liquid chromatography/tandem mass-spectrometer). Justification for this approach is in two NCASI internal reports.

Table 6-5. Highest herbicide concentrations (µg/L) in Needle Branch samples (Louch 2013).

Sample Site	Chemical:	Glyphosate	AMPA	MSS	SMM	Imazapyr
	Formulation:	Accord XRT II	n/a	Sulfomet Extra	Sulfomet Extra	Chopper Gen II
High	Application	0.062	0.007	Equip. Fail	Equip. Fail	Equip. Fail
	Baseline (after)	0.030	0.007	<1.000	<0.500	0.200
	1st Storm	0.084	0.010	<1.000	<0.500	0.400
	2nd Storm	0.062	0.009	No Analysis	No Analysis	No Analysis
	3rd Storm	0.021	0.004	No Analysis	No Analysis	No Analysis
	4th Storm	0.041	No Analysis	No Analysis	No Analysis	No Analysis
	5th Storm	No Analysis	No Analysis	No Analysis	No Analysis	No Analysis
Middle	Application	Equip. Fail	0.007	<1.000	<0.500	0.200
	Baseline (after)	0.019	0.007	<1.000	<0.500	0.200
	1st Storm	0.149	0.008	<1.000	<0.500	0.400
	2nd Storm	0.036	0.005	No Analysis	No Analysis	No Analysis
	3rd Storm	No Analysis	No Analysis	No Analysis	No Analysis	No Analysis
	4th Storm	0.045	0.004	No Analysis	No Analysis	No Analysis
	5th Storm	0.029	<0.004	No Analysis	No Analysis	No Analysis
Low	Application	<0.018	<0.004	No Analysis	No Analysis	No Analysis
	Baseline (after)	0.034	0.008	<1.000	<0.500	<0.200
	1st Storm	0.058	0.012	No Analysis	No Analysis	Equip. Fail
	2nd Storm	0.040	0.007	No Analysis	No Analysis	No Analysis
	3rd Storm	Equip. Fail	No Analysis	No Analysis	No Analysis	No Analysis
	4th Storm	0.042	0.006	No Analysis	No Analysis	No Analysis
	5th Storm	No Analysis	No Analysis	No Analysis	No Analysis	No Analysis
Blank Control	MDL	0.018	0.004	1.000	0.500	0.200
	LCL	0.015	0.015	0.625	0.625	0.625

Source: NCASI 2013, Appendices D, E, F, and H. MDL = Method detection level; LCL = Lower calibration level. AMPA = amino-methylphosphonic acid (a breakdown product of glyphosate; MSS = sulfometuron methyl; and MSM = metsulfuron methyl.

4. The authors discount a LC/F glyphosate pulse at LOW during the first storm due to a “background interferent to be present in samples” (pg. 400). They compare this to a LC/M-M sample collected two hours earlier that was Non-Detect. This previous LC/M-M result was used to justify stating that glyphosate did not move the 1 km between MID and LOW. However, the Supplementary material explanation says that the concentrations of this “interferent” were variable, and unpredictable, over the course of the study, and the LC/M-M split samples were only conducted on a portion of the samples (about 7%).

5. In the Supplemental file, it appears that the Louch (2017) study used spray buffers on the order of 15m – 18m. Based on Bladon et al (2016), there was a ≈ 15m buffer left on this section of Needle Branch. Needle Branch is a Small N stream in the upper reaches (above LOW), then becomes a Small F stream at the site of the LOW stream gauge.

Similar to Louch et al. (2017), Caldwell and Courter (2020) evaluated four chemicals applied in 2016 and 2017 for silvicultural applications on the northwest Oregon Coast. Active ingredients applied were glyphosate, clopyralid, sulfometuron methyl (SMM), and metsulfuron methyl (MSM). These herbicides were evaluated in three harvest units, and one control, on Stimson Lumber Company property in the Tillamook region of Oregon's north coast (including two harvest units within the City of Tillamook's source watershed). Caldwell and Courter (2019) do not specify the FPA classification for the streams in these units. The ODF FERNS notifications and written plans for the harvests and chemical applications were reviewed for the three harvest sites. We also obtained the spray specifications and GPS-based flight line maps for the herbicide applications from Wilbur-Ellis Company (Napavine, WA). A spreadsheet of the sample analyses supplementary to the journal article was also used.

The 63 acre Powerline unit (NOAPs 2015-511-12269C and 2016-511-05927) has a Small Fish (SF) stream in the lower end of the unit, with six other drainages classified as Small Nonfish (SN). The SF stream has a 3.4 acre buffer that also encompasses the lower portions of three SN streams. Above the buffer, it appears that there was a 75' to 150' no spray zone along the SN stream at the bottom of the unit, but that other SN streams in the unit were sprayed because they did not contain water at the time of application on July 28, 2016. Glyphosate 5.4™ (Alligare) and SFM Extra™ (Alligare) were applied at an elevation of approximately 6m above the vegetation at a target rate of 4.7L/ha. and 280 g/ha, respectively. The adjuvants Crosshair® (Wilbur-Ellis) for drift and deposition control, and Syl-Tac® (Willbur-Ellis) as a surfactant were included in the application at 290 mL/ha and 440 mL/ha, respectively (Caldwell and Courter 2019). Water samples were collected at the Powerline unit at a site just below the treated unit (Upstream), and at a second site 3km downstream (Downstream).

The 42 acre Crowbar (Crow) Unit (NOAP 2017-511-07450) has a Medium Fish (MF) stream, Killam Creek, that also is classified as Domestic water use, as well as a unnamed Small Fish (SF) stream, both at the bottom of the unit; there are five additional SN streams draining the interior of the unit. The written plan indicates that no spraying will be conducted within 60' of any fish or domestic use stream. Review of the flight lines on the spray map shows a spray buffer of about 100' from any fish or domestic use stream; all NS within the interior of the unit were sprayed. On July 13, 2017 Oust® XP (Bayer) and Transline® (Dow Agro) were applied by helicopter at between 9 m and 15m above the canopy at a rate of 200 mL/ha and 4.7 L/ha, respectively. Oust® XP's active ingredient is SMM, while Transline's® is clopyralid. Crosshair® was also added to the tank mix at a rate of 290 mL/ha. Water samples were collected at Crowbar Creek at the bottom of the unit just above the tributary's confluence with Killam Creek (Treatment), and at a site 1.6 km below the treatment area on Killam Creek (Downstream).

The 82 acre 120 Wasp unit (NOAP 2016-511-13178C) contains SF streams on its east and west sides (along with a SF tributary into the interior from the west), and nine SN or SU streams throughout the interior of the harvest unit. The F streams contain a 50' riparian buffers along a total of 5,280' of length. The chemical application NOAP written plan for 120 Wasp (NOAP 2017-511-06271) specifies a no-spray buffer of 60' from F streams if the wind is less than 5 mph blowing away from the buffer, and 100' if it is less than 2 mph and blowing towards the buffer. The unit was sprayed on July 17, 2017 by helicopter from an elevation of 9m to 15m with a mixture of Oust Extra (Bayer) and Glyphosate 5.4 at a rate of 290 mL/ha and 4.7 L/ha, respectively. Oust Extra's active ingredients are SMM and MSM, while Glyphosate 5.4's is glyphosate. Water samples were taken approximately 300m from the bottom of the unit (Upstream), and at a second site 1.9 km downstream (Downstream).

The basic sampling design for the study was that water samples would be collected prior to the spray applications (grab samples), and during and subsequent to the application (automated samplers), post application monthly grab samples, and automated sampling during the first two (2016) and three (2017) storms that were predicted to have greater than 0.5 inches of rain during 24 hours. The autosamplers for the 2016 Powerline treatment collected five samples at varying intervals after spray application (0, 6, 12, 24, and 32 hours); however, due to equipment malfunction, only one sample was collected after application. Grab samples were collected monthly after the spray application during baseflow conditions. Storm event samples for the 2016 treatment were taken at 0, 6, 12, 24, 48 and 72 hours after initiation. The autosampling interval changed in 2017 for the Crowbar and 120 Wasp treatments to collect samples hourly for a 12 hour period after the unit was sprayed. During the three storm events, samples were taken at 2 hour intervals (24 samples) for 1st storm; 5 hour intervals (22 samples) for the 2nd storm (2b); and reverted to 2 hour intervals (24 samples) for Storm 3 (Caldwell and Courter 2020, Supplementary Data File).

Caldwell and Courter (2020) referenced contact with the Louch et al. (2017) group in designing their study and analytical methodology. Sample retrieval and storage followed standard practices. They used the LC/M-M sample analysis method for MSM and SMM; GC/MS/MS for clopyralid and triclopyr, and HPLC for glyphosate and AMPA. Practical quantification levels (PQL), i.e., the lowest equipment calibration levels, were 0.1 µg/L for clopyralid, 5 µg/L for glyphosate and AMPA, and 0.01 µg/L for SMM and MSM. Note that the PQL is different from the minimum detection level (MDL): for glyposhate, the MDL was 1 µg/L.Samples were analyzed by Anatek Labs, Inc. of Moscow, ID. Table 6-6 summarizes the results from this study showing the highest concentrations of the active ingredients at the three sites during application, baseline samples, and the first three storms.

Caldwell and Courter (2020) report that, “Additionally, glyphosate and AMPA were not detected in any surface water samples from pre-application through post-storm grab samples in both study years” (pg 6). More accurately, no glyphosate was detected above the 5 µg/L practical quantification level in the Powerline unit, and none was found at or above the MDL of 1 µg/L in the 120 Wasp unit in 2017 (L. Courter, personal communication, 4/2/2020). “If glyphosate were mobilized, however, this likely occurred during the first or second storm event when TSS concentratins ranged approximately 350 to 500 mg/mL, indicating substantial surface soil runoff” (Caldwell and Courter 2020, pg. 14).

Suflometuron methyl (SMM) was applied at all three sites, and was detected in water samples above the PQL at all three sites. For the Powerline unit, SMM began to be detected at the proximal sampling site 32 hours after application, but then the auto-sampler stopped. SMM continued to be present at low concentrations throughout the remainder of the study period at the Upstream sampling site. However, SMM was found only in two grab samples at the distal sample site (Downstream), 69 and 76 days after application. At the Upstream site on the 120Wasp unit, SMM began to be detected 5 hours after the application began, and peaked at 7 hours (Table 6-6). Compared to the application concentrations, SMM was found at four times higher concentrations during the first storm (and at the last sample collected during the storm), twice as high during the second storm, and only a third as high during the third storm. The pattern at the 120Wasp Downstream site showed two hits during the application period, the first 3

Table 6-6. Highest herbicide concentrations (µg/L) after silvicultural applications on the northern Oregon coast (Caldwell & Courter 2019).

Site & Chemical	Formulation	Upstream Sample Site Appli-cation	Base-flow	1st Storm	2nd Storm	3rd Storm	Downstream Sample Site Appli-cation	Base-flow	1st Storm	2nd Storm	3rd Storm
Powerline*											
Glyphosate	Glyphosate 5.4	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	n/a
SMM	SFM Extra	0.03	0.07	0.08	0.03	n/a	<0.01	0.03	<0.01	<0.01	n/a
MSM	SFM Extra	0.01	0.01	0.07	0.01	n/a	<0.01	0.00	<0.01	<0.01	n/a
Crowbar											
SMM	Oust XP	0.32	<0.01	0.10	0.01	<0.01	0.07	0.00	0.17	0.00	n/a
Clopyralid	Transline	1.41	<0.1	0.80	<0.1	<0.1	0.35	0.00	0.98	0.00	<0.01
120 Wasp*											
Glyphosate	Glyphosate 5.4	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
SMM	Oust Extra	0.04	0.01	0.15	0.08	0.01	0.01	<0.01	0.03	0.03	0.01
MSM	Oust Extra	0.02	0.00	0.01	0.01	0.00	0.01	<0.01	0.01	0.01	<0.01

* All results from Caldwell & Courter (2020) data from Supplementary Data, ieam4196-sup-0001-ieam-2019-029-suppdata_anon.xlsx, "Detection Data" sheet. Storm sampling reported for the highest recorded value at either upstream or downstream; same with baseline. Practical quantification levels are 0.1 µg/L for clopyralid and triclopyr, 5 µg/L for glyphosate and AMPA, and 0.01 µg/L for SMM and MSM. Minimum detection level for glyphosate at 120Wasp as 1 µg/L.

hours after the start of spraying and the second at 7 hours; with trace amounts during the first storm, but concentrations at levels approximately half those of the Upstream site during the second storm, returning to trace levels during the third storm. At Crowbar, SMM concentrations at the site proximal to the application were approximately 8 to 10 times those at Powerline or 120Wasp, with the highest concentration beginning immediately after (or even during) the application. The first flush of SMM at the Downstream site was about 5 hours after application began, again with the first positive reading being the highest.

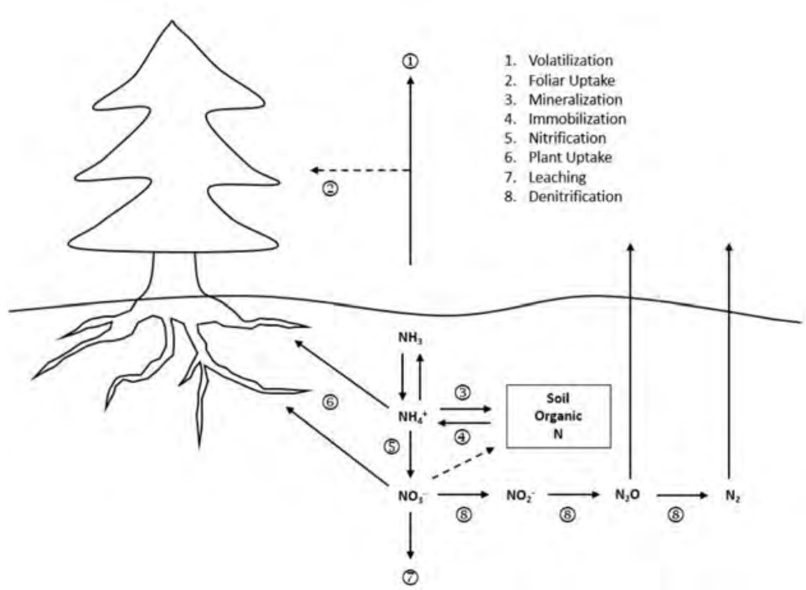
Metasulfuron methyl (MSM) was detected at both the Powerline and 120 Wasp units after application. After spraying at Powerline, trace concentrations were detected at the Upstream site about 12 hours after application during the last collection at the auto-sampler. MSM was again detected in trace amounts during all baseflow grab samples, then peaked approximately 24 hours into the first storm. Concentrations during the second storm were about one-tenth for first; no analysis was conducted during the 3rd storm. No MSM was ever detected at the Downstream sampling site at the Powerline unit. At the 120 Wasp site, MSM began being detected at the Upstream site 5 hours after spraying began, and peaked at 7 hours. No MSM was detected during Baseflow sampling at either the Upstream or Downstream sampling sites. During the first storm, MSM c at the proximal site began to be detected 9 hours after initiation, and peaked at 11 hours at the Upstream site; at the Downstream site, MSM was only detected at the last auto-sample collection 50 hours after the storm began. In contrast, during the second storm at the 120Wasp unit, MSM was detected at the Upstream site immediately (i.e. potential remobilized from sediments) and throughout for the 115 hours, and at the Downstream site a similar pattern was seen, although the concentrations were one-third to one-half those at the Upstream site, and the duration was only 75 hours. By the third storm, MSM had apparently washed through the system as no detections above the PQL were seen at either the Upstream or Downstream sample sites. Caldwell and Courter (2020) concluded that both SMM and MSM, as a result of their high sorption factor, primarily moved through the Powerline and 120 Wasp sites with the first storm event. The highest concentrations found during application at Crowbar are likely to result from the drainage pattern and steep stream gradient.

Crowbar was the only unit where clopyralid was applied. Similar to SMM, clopyralid began to be detected at the Upstream site almost immediately after application began at 10:10AM and by 11:30AM the concentration was 1.41 µg/L, receding during the next 7 hours. At the Downstream site, clopyralid detections began about 3.5 hours after spraying started, peaking at 4.5 hours, and continuing for the remaining 21 hours of sample collection. No clopyralid was found at any time in the baseflow samples at either Upstream or Downstream Crowbar sample sites. During the first storm at the Crowbar unit, clopyralid detections began at the Upstream site 4 hours after initiation, peaked at 16 hours (with a secondary peak at 32 hours), and continued through the remainder of the 44 hour sample collection. Clopyralid detections at the Downstream Crowbar sample site began four hours after the first storm began, peaked at 16 hours, and continued intermittently through the remainder of the 44 hour sample collection. No clopyralid was detected above the PQL during the second storm at either the Crowbar Upstream or Downstream sample locations. And, no clopyralid was detected at the Upstream sample location during the third storm; however, at the Downstream site, trace amounts (0.002 µg/L) were detected at the beginning of the storm and lasting for 14 hours until going below PQL. Caldwell and Courter (2020) theorize that due to its low sorption potential clopyralid mobilized during the first storm event, and thus was not detected during subsequent storm events.

Caldwell and Courter (2020) conclude that while herbicides (possibly with the exception of glyphosate) were found at all sites during application and early season storms, “maximum herbicide concentrations in our study were four orders of magnitude below [human health] benchmarks” (pg. 12). While not noted in the study, it’s quite likely based on early detections during application at the Crowbar site, along with its high stream network density, that overspraying occurred in live NS streams. Also apparent from the data is that those herbicides (again possibly with the exception of glyphosate) that are highly sorption onto sediment particles are remobilized during storm events (i.e., the Downstream sampling sites quickly detecting concentrations at the beginning of storms.

6.3.2 Fertilizers

Figure 6-3 shows nitrogen cycling in a typical PNW forest environment (Nason & Myhold 1992). In Figure 6-3, N is elemental nitrogen, having three unpaired electrons that result in large electrostatic attractions; N_2 is dinitrogen, a gas that forms 78% of the Earth’s atmosphere; N_2O^- is nitrite, commonly converted from ammonium by bacteria



Source: Nason & Myhold 1992.

Figure 6-3. Nitrogen cycling in a PNW forest environment.

through nitrification; NO_2 is nitrogen dioxide; NO_3^- is nitrate, the base for nitric acid and commonly forms water soluble salts; NH_3 is ammonia, an uncharged molecule, and a gas at room temperature; and NH_4^+ is ammonium, a positively charged molecule that is most frequently found as crystallized salt compounds.

An artificial source of nitrogen to water comes from fertilization with urea that reaches the streams either through run-off or thru direct application on the streams (Flint et al. 2008). Besides the peak levels that occur soon after urea applications, a prolonged higher nitrogen levels are present for months after the broadcast, which suggests the existence of other pathways for nitrogen, such as lateral movement or leaching into the ground water (Flint et al. 2008). To study the movement of nitrogen and nitrogen derived products thru the soil towards the streams is commonly studied with lysimeters (Perakis and Sinkhorn 2011; Devine et al. 2012). Even imperfect, as they provide a punctual representation of a continuous environment (Kitanidis 1997), lysimeters supply a process based picture of nitrogen movement through the soil matrix. Flint et al. (2008) suggest that approximately 2% of total applied nitrogen leached beyond the rooting zone as nitrate nitrogen (NO_3–N) and ammonium nitrogen (NH_4–N). They found that more than half of the administered nitrogen was accounted for, with 26% in the overstory and 27% in the soil. The distribution of nitrogen among various ecosystem components was measured 6 months after urea broadcasting, which suggests a long term impact on stream water. Nevertheless, the results are not necessarily convincing, as the significance was merely below the commonly stated level of 0.05 (i.e., p-value = 0.03), and the assumptions needed for analysis were not verified, particularly homogeneity of the varainace (i.e., heteroskedasticity), which could change the significance (Neter et al. 1996).

The current studies revealed that forest fertilization increases nutrient concentrations in stream water. Binkley et al. (1999) mentioned three main sources for increase in nutrient concentration:

- application of fertilizer directly into streams;
- the use of ammonium nitrate forms of fertilizer instead of urea; and
- the application of higher dosages by either larger rates or by repeated doses.

Nevertheless, Binkley et al. (1999) perspective is that even when higher concentrations of nutrients are achieved, the impact could be minimal with respect to degradation of water quality.

It is argued that the current criteria for stream nutrient concentrations are insufficient to evaluate fertilization's effects, particularly in the Cascade streams of the Pacific Northwest where the supply of nitrogen is the limit in primary production (Bothwell 1992; Anderson 2002). Nitrate concentrations resulting from forest fertilization very rarely exceed USEPA standard. Ammonia concentrations beyond prescribed limits have rarely been observed (Binkley et al. 1999). These standards are focused on protection of drinking water for human health, and they are not intended to prevent ecosystem degradation.

There are no drinking water standards for urea-N, as the compound is not toxic and does not represent a threat to human health (Binkley et al. 1999). However, there are standards for urea breakdown products, such as nitrate nitrogen (NO_3–N) and nitrite nitrogen (NO_2–N). Nitrate in drinking water can be a direct human health hazard when it is transformed to nitrite in the digestive system in quantities sufficient to reduce the oxygen-carrying capacity of red blood cells. This is mainly a concern for infants, pregnant women and nursing mothers. The EPA uses the 10 mg/L standard as the maximum contaminant level (MCL) for nitrate-N and 1 mg/L for nitrite-N for regulated public water systems (WQA 2013). Phosphorus in drinking water is generally not considered to pose serious or direct human health risks (Scatena 2000). Phosphorus is actually often added to municipal drinking water to reduce corrosion and leaching of lead and other toxins from water pipes. However, high phosphorous and nitrogen runoff can also create Harmful Algal Blooms (Gatz, 2018) as toxic blue-green algae called cyanobacteria (included in the Contaminant Candidate List [CCL]) (USEPA 2015).

In summary, while elevated N export often occurs after clearcut harvests and forest fertilization may sincrease dissolved N in some waterways, available scientific evidence suggests that these increases are usually temporary and do not seriously degrade drinking water quality in most cases. To date, nitrates have not been found to accumulate in drinking water as a sole result of forestry activities in quantities that exceed drinking water standards (Bisson et al. 1992; Binkley et al. 1999, Anderson 2002; Binkley et al. 2004). Perhaps of greater concern from a drinking water perspective are the cascading and cumulative ecological effects that elevated levels of nitrates and phosphates can have in lakes and rivers. This emerging issue is discussed in the following section.

6.3.2.1. Changes to Water Quality Due to Nutrients. Flint et al. (2008) and Poor and McDonnell (2007) conducted studies about fertilizers in the Pacific Northwest. Both studies pointed to changes in nutrient concentrations, with the largest contributor being the non-forest activities. Flint et al. (2008) suggested that human sewage is the main source of N, whereas Poor and McDonnell (2007) argued that agricultural catchments supplied N-concentrations larger than residential catchments. Both studies indicated that the smallest source of stream nitrogen associated with human activities is related to forest management. Nevertheless, the two studies used simple statistical analyses which did not provide evidence that the assumptions needed for valid inference were tested, which does not support generalization of their findings. Furthermore, Flint et al. (2008) hypothesized that “if fertilizer is applied on steeper slopes where surface flow is present, impacts on surface water quality could be greater” without, however, providing any experimental evidence to support this perspective.

In addition to these studies, a large body of research has been dedicated to the leakage of nutrients – mainly nitrogen and phosphorus – from the forest following harvesting operations. Almost all studies pointed to a change in water chemistry, sometimes even without the presence of a nearby harvest (Greathouse et al. 2014).

Several studies (i.e., Gravelle et al. (2009), Slesak et al. (2009), and Devine et al. (2012)) focused on nutrient dynamics were also of interest for the Pacific Northwest, even when no fertilization occurred. Gravelle et al. (2009) studied nutrient concentration dynamics before and after timber harvest in the Mica Creek Experimental Watershed in Idaho. Their study revealed a significant increase only in NO_3 + NO_2, but no change in total phosphorus, orthophosphate, and total nitrogen. These findings could be influenced by the possible inclusion of outliers in the analysis, clearly identified in Figure 9without a formal assessment of their impact. They also used an analytical framework not necessarily suitable for repeated measures, as the comparisons were executed using Student’s t-test.

Other studies developed models predicting nutrients concentration from environmental variables, such as flow, temperature or time of travel for a reach, for management or scientific decisions (Sigleo et al.

2010; Johnson et al. 2011; Wise and Johnson 2011; Devine et al. 2012). However, the absence of a formal and complete model development framework (Neter et al. 1996; Kitanidis 1997) suggests that the models are in essence a different perspective on hypothesis testing rather than an analytical tool. Without any assessment of the confidence in the results, some of these studies, such as Wise and Johnson (2011) or Johnson et al. (2011) serve predominantly an intuitional role rather than a decisional one.

The most common fertilizer used in the Pacific Northwest is urea. Binkley et al. (1999) argued that even though there were no detectable effects of forest fertilization on the composition and productivity of stream communities, more research was needed – “especially in relation to P fertilization”. The main effect of nitrogen and phosphorus is eutrophication, which leads to an explosive growth of plants that deprive oxygen to and ultimately suffocate other organisms. However, as evidence suggests, since BMPs became standard practice in forest management, the impact of forest fertilization on the addition of nitrogen to surface water is negligible in contrast with agriculture and residential activities (Binkley et al. 1999; Poor and McDonnell 2007; Flint et al. 2008). Therefore, assuming a proportional impact on eutrophication, one can infer that the main sources of eutrophication are related to actions occurring outside the managed forest. A similar conclusion was reached by Anderson (2002), which states that “biological responses may be minimal in small streams nearest to application because of light limitation, but may be elevated downstream where light is sufficient to allow algal growth”. He continues by saying that “algal response could be greatest in downstream reaches”.

6.3.3 Best Management Practices

According to Oregon Department of Forestry, the BMP are a set of practices, often voluntary, that reduce the non-point pollution to standards compatibale with water quality goals (Robben and Dent, 2002). Among those practices the presence of riparain buffers are recommended. In a comprehensive study of the impact of the most common herbicides used in the BMP practices, namely 2,4-D, glyphosate, haxazinone, imazapyr, metsufuron, sulfometuron and triclopyr, Michael (2004) concluded that single-stem injection and soil spot application with a 10 m buffer will lead to stream contamination of very small amounts, up to 0.04 mg/ L. While small, the 0.04 mg/l could represent “a level of contamination that cannot be eliminated by current methods of stream protection”, according to Michael (2004).

Current trends in water quality protection are focused on the effects of an increase in the riparian management area (RMA) width. Several studies revealed that herbicide application on ephemeral and intermittent streams without RMAs resulted in high level of stream contamination, sometimes up to 0.6 mg/ L on the day of application (Michael et al. 1999; Michael 2004). Michael (2004) argued that the increase of the buffer zone to protect the perennial streams have a beneficial effects of water quality, but an RMA beyond a 10m width will not lead to significantly different impacts on stream contamination with herbicides. Overall, the current results suggest that silvicultural herbicide applications implemented with contemporary BMPs are unlikely to result in chronic exposure of aquatic biota, and applications according to the BMP practices are unlikely to degrade surface waters (McBroom et al. 2013).

Current BMPs focus on keeping fertilizer applications well away from drinking water sources to reduce the chances of fertilizer being mistakenly applied directly into them. This is usually done by specifying retention of a buffer strip of vegetation adjacent to streams and water bodies (a streamside management zone, or SMZ) where fertilizer preparation and use is not allowed. The vegetated buffers also serve to help filter nutrients mobilized by harvesting and site preparation from subsurface flows before they enter waterways. Filtering effectiveness generally increases with increase in buffer width (Pike et al. 2010). Feller (2009) suggests that buffers greater than 100m remove essentially all excess nutrients, although effectiveness varies by watershed and with soil properties, topography, subsurface hydrology, vegetation type and other factors.

If properly implemented, BMPs to minimize nutrient flushing after forestry activities and the potential for fertilizers to get into waterways are generally considered to be effective (Cristan et al. 2016; Stednick 2008). However, rigorous studies of BMP effectiveness are still limited (Edwards et al. 2016) and most industrial forest owners apply fertilizers by helicopter (Hanley et al. 2006) which can be imprecise. Also, risks of dissolved nutrients in runoff affecting drinking water may be locally higher where the source watershed is less extensive, steeper and closer to the municipal water intake, contains a significant percentage of commercial timberland, or where tree plantations within the source watershed are fertilized multiple times.

6.4. Prevalence of Chemicals Found in Streams Related to Forest Management Activities

This section will describe the results of four monitoring studies conducted in watersheds that have active forest management as their primary land use. We will begin by providing U.S. Environmental Protection Agency (USEPA) standards and criteria for evaluating the presence of pesticides in drinking water, particularly their fates from application to breakdown.

6.4.1 Standards, Health Advisory, and Human Benchmarks for Forest Chemicals.

The U.S. Environmental Protection Agency (EPA) under the Safe Drinking Water Act (see Chapter 2) determines water quality standards for treated water through its National Primary Drinking Water Regulations (CFR Part 141). In addition to required water quality standards, the EPA also provides states with levels of pesticides to consider incorporating in their own procedures. Table 6-6 shows these different standards, and their relevant levels for chemicals commonly used in forest management. Because these standards and guides are for finished (i.e., treated) water, any levels exceeding them in raw water would require treatment.

Maximum Contaminant Level (MCL) are Federally-enforceable standards for finished (i.e., treated) drinking water allowed under the SDWA and developed under USEPA's regulatory authority (40 CFR §141.2). The criteria for determining whether to regulate is based on three criteria:

1. The contaminant may have an adverse effect on the health of persons;
2. The contaminant is known to occur or there is a high chance that the contaminant will occur in public water systems often enough and at levels of public health concern; and
3. Regulation of the contaminant presents a meaningful opportunity for health risk reductions for persons served by public water systems (https://www.epa.gov/dwregdev/how-epa-regulates-drinking-water-contaminants).

Table 6-6. USEPA 2018 Drinking Water Standards, Health Advisories, and Human Health Benchmarks for pesticides and nutrients (fertilizers).

Chemical	Trade Names	MCL (mg/L)	MCL Goal	Health Advisory Level (HAL) (mg./L)	Human Health Benchmark (HHBM) (µg./L)
2-4DP-p salts & esters	Hi-Dep; Weedar 64; Weed RHAP; Amine 4; AquaKleen (Amines)	0.07	0.07		230
Atrazine	Aatrex; Atratol; Fogard; Gesaprim; Griffex; Mebazine; Primatol A; Vectal	0.003	0.003	N/A	N/A
Aminopyralid	Milestone; Capstone; Opensight				3,000
Carbaryl	Sevin; Prokoz			1.0	N/A
Clopyralid	Stringer; Curtail; Transline; Redeem				960
Diflubenzuron	Dimilin				100
Glufosinate-ammonium	Liberty, Cheetah, Scout, others				40
Glyphosate	Roundup; Rodeo; Accord; Glyphomate	0.7	0.7	N/A	N/A
Hexazinone	Velpar; Pronone; 10G			3/2	N/A
Imazapyr	Arsenal; Chopper				16,000
Metsulfuron methyl	Opensight; Escort; Ally				1,600
Sulfometuron methyl	Oust				1,760
Triclopyr	Garlon 3A; Capstone; Redeem; Remedy				300
Nitrate (NO_3^-) Nitrogen	Urea Fertilizers	10	10	N/A	N/A
Nitrite (NO_2^-) Nitrogen	Urea Fertilizers	1	1	N/A	N/A

Sources:

HHBM: https://iaspub.epa.gov/apex/pesticides/f?p=HHBP:home

HAL (1-day, 10-day child exposure): https://www.epa.gov/sites/production/files/2018-03/documents/dwtable2018.pdf

MCL/MCL Goal: https://www.epa.gov/sites/production/files/2018-03/documents/dwtable2018.pdf

Trade names: http://pmep.cce.cornell.edu/profiles/extoxnet/TIB/tradename-index.html

Trade names: https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd496996.pdf

The list of contaminants subject to regulation with MCLs are listed in the Code of Federal Regulations (40 CFR §141.61) for both organic contaminants (§141.61(a) and synthetic organic contaminants (§141.61(c); and nitrate and nitrite nitrogen are addressed in §141.62(b)(7) and (8). When contaminant levels are above the MCLs, water utilities must apply additional measures to reduce their levels using “best available technology” (BAT) (40 CFR §). For organic contaminants, best available technologies are determined for each contaminant from within three types: granular activated carbon (GAC); packed tower aeration (PTA); or oxidation (OX). Depending upon the contaminant, there may be more than one acceptable treatment type (§141.61(b)). For inorganic contaminants (nitrate and nitrite nitrogen), BAT’s include ion exchange, reverse osmosis, and electrodialysis (nitrate only)(§141.62(c)).

Maximum Contaminant Level Goal (MCL Goal) is the maximum level of a contaminant in drinking water that has no human health effects, with the addition of a margin of safety to incorporate uncertainty. The MCL Goals are nonenforceable criteria (40 CFR §141.2).

Health Advisory Levels (HAL) are informal technical guidance for contaminants without enforceable standards but that may have human health effects (USEPA 2018). However, they may be used by states to set their own standards. Six health advisory (HA) levels are provided: (a) One-Day HA (child consuming 1 liter of water for one day); (b) Ten Day HA (up to 10 days of exposure, with child consuming 1 liter of water per day); (c) Lifetime HA (adult drinking 2 liters of water per day); (d) Reference Dose (RfD) that is likely to be without an appreciable risk during a lifetime, incorporating an order of magnitude of uncertainty, and based on a person's weight (mg/kg/day); and (e) Drinking Water Equivalent Level (DWEL) that is derived by multiplying the RfD by body weight and dividing this figure by daily water consumption. These HALs are based on noncarcinogenic effects. A sixth HA criteria is the level of the contaminant in water that would entail a lifetime cancer risk of 1 in 10,000 (USEPA 2018). In Table 6-6, we have reported the HAL using the both the 1-Day and 10-Day child exposure criteria for illustrative purposes; a complete list of contaminants and all standards and advisories can be found in USEPA (2018).

Human Health Benchmarks (HHBM) are defined as levels of pesticides "at or below which adverse health effects are not anticipated from one-day or lifetime exposures" (USEPA 2017). The HHBM were developed for those chemicals that USEPA has not set Health Advisory Levels or an enforceable federal drinking water standard (USEPA 2017). The HHBMs can be found at: https://iaspub.epa.gov/apex/pesticides/f?p=HHBP:home

Health-Based Screening Levels (HBSL) are developed by the USGS for contaminants that do not have USEPA Maximum Contaminant Levels (MCL) or Human Health Benchmarks for Pesticides (HHBPs) (Norman et al. 2018). These can be found at https://cida.usgs.gov/hbsl/apex/f?p=104:1::::::.

6.4.2 Pesticide Monitoring Techniques.

Studies of pesticides in water typically rely on two general types of sampling: passive sampling where the equipment remains in the stream for a certain duration, and is then taken to the laboratory for analysis; and grab samples that are taken at one time, stabilized, and analyzed in the laboratory. The benefit of passive samplers is that they integrate pesticide concentrations in the water column over a longer period of time, and are thus more likely to discover contaminants that are transitory or present in relation to rainfall or flow events. Two types of passive sampling equipment used: polar organic chemical integrative sampler (POCIS) and semipermeable membrane device (SPMD), often together since they target different classes of compounds; Alvarez (2010) reviews both POCIS and SPMD samplers. Both POCIS and SPMD samples can provide concentration and load determinations if adequate streamflow data is available. The National Environmental Methods Index (NEMI), a program of USEPA and USGS, provides specific information on available analysis techniques for contaminants through a searchable database (https://www.nemi.gov/home/).

Polar Organic Chemical Integrative Samplers (POCIS) are designed to sample water-soluble (polar or hydrophilic) organic chemicals from aqueous environments. The POCIS is an integrative sampler which provides time-weighted average concentrations of chemicals over deployment periods ranging from weeks to months. It consists of a solid material (sorbent) contained between two microporous polyethersulfone membranes. The membranes allow water and dissolved chemicals to pass through to the sorbent where chemicals are trapped. POCIS extracts are then analyzed by various instrumental techniques, including HPLC, GC, GC/MS and LC/MS (NEMI 2020a). POCIS are designed to sample the more water soluble organic chemicals with log K_{ow}s less than (<) 3. This includes most pharmaceuticals, illicit drugs, polar pesticides, phosphate flame retardants, surfactants, metabolites and degradation products (Alvarez 2010). The pesticide-POCIS uses a triphasic admixture of Isolute® ENV+ and Ambersorb® 1500 or 572 carbon dispersed on S-X3 BioBeads® (Alvarez 2010).

Semipermeable Membrane Devices (SPMD) are generally used for sampling neutral organic chemicals with a log octanol-water partition coefficient (K_{ow}) greater than 3. Polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), chlorinated pesticides, polybrominated diphenyl ethers (PBDEs), dioxins, and furans are all commonly measured using SPMDs (Alvarez 2010). Of particular importance is that quality control (QC) samples should represent 20 to 50-percent of the sample set and include SPMD-fabrication blanks, SPMD-process blanks, reagent blanks, field-blank SPMDs, permeability reference compound samples, SPMD spikes, and procedural spikes (NEMI 2020b).

6.4.3 Levels of Forest Chemicals Found in Streams Draining Pacific Northwest Forestlands.

Along with the studies used in the science review, we identified are four additional locations where water quality sampling has been (and is) conducted to determine pesticide levels likely related to forest management activities. These are: (1) the McKenzie River that provides the source for Eugene's water supply; (2) the South Yamhill River that provides the water supply for Sheridan, Oregon; (3) the Hood River watershed that is not used as surface source water; and (4) the Hoh River watershed in Washington State. Two of these areas (South Yamhill and Hood River) are part of the inter-agency network of Pesticide Stewardship Partnerships; while the Hoh River serves a similar function in Washington. The Eugene Water and Electric Board (EWEB) has a long history of studies related to water quality and land use in their source watershed, the McKenzie River. Other than EWEB's studies on the McKenzie, all the other studies were focused on evaluating the effects of pesticides on aquatic organisms. However, in none of these studies were the actual amounts of pesticides, their time of application, or location known. As a result, these studies are best characterized as reconnaissance level assessments of the prevalence of pesticides in streams draining forest lands.

6.4.3.1. Hood River PSP. The Hood River watershed has been the site of extensive pesticide monitoring since the late 1990s. The Hood Pesticide Stewardship Partnership was founded in 2000 to support this sampling through outreach to users. The Oregon Department of Environmental Quality began sampling for pesticides in the Hood River basin in 1999 (Temple and Johnson 2011), chlorpyrifos and azinphos-methyl were studied in 2002 and 2003 by Jenkins (2004), and the USGS supplemented the DEQ sampling in 2011-2012 (Hapke et al. 2016), and DEQ produced two reports on their grab samples and POCIS/SPMD sampling in 2014 (Masterson and Crown 2015, Crown et al. 2015). While the primary focus for sampling was pesticide use in orchards, and effluent from fruit packing operations, there were seven grab sample sites that have >85% forest <u>and</u> less than 5% agriculture or orchard land uses. (Temple and Johnson 2011, Appendix A). These sites are Dog (n = 41; 2001 – 2004); Hood, Middle Fork (n = 17; 1999 – 2000); Hood, West Fork, Mouth (n = 21; 2008 – 2009); Hood, West Fork, RM 2.5 (n = 6; 2008 – 2009);

Hood, West Fork, RM 4.7 (n = 20; 1999 – 2001); Neal Upper, Above Diversion (n = 113; 2001 – 2007) and Neal, Upper, Below Diversion (n = 97; 2003 – 2009). Most sampling took place during the months of March – June, with additional small numbers of samples in September and October depending upon the year and site (Temple and Johnson 2011, Appendix B). The range of pesticides included in the grab sampling changed over the period, with additional ones added in 2007, 2009, and 2010 (these latter ones were not included in Temple and Johnson, 2011). In addition to the grab samples, during 2011 and 2012 year round monitoring of pesticides was conducted using POCIS and SPMD passive samplers (Hapke et al. 2016). In both these studies, the focus was on effects of pesticides on aquatic life, particularly ESA-listed salmon.

The results from grab samples from 1999 to 2009 are reported in Temple and Johnson (2011). No detections were found for atrazine in 295 samples at the seven sites. There was one detection of the insecticide chlorpyrifos in 2008 at the upper Neal Creek below the diversion; the exact concentration was not reported, but it may have exceeded the 0.041 µg/L of the lowest aquatic life water quality standard. One detection of fluometuron, an herbicide only registered for cotton, was found in upper Neal Creek in April, 2009, but at a concentration four orders of magnitude below USEPA benchmarks. Finally, there were occurrences in 2009 of hexazinone at all sites in Neal Creek, likely as a result of forest operations; concentrations of between 0.04 and 0.10 µg/L were found in March through June, along with other occurrences at 0.04 µg/L in September and October. These concentrations of hexazinone are five to six orders of magnitude less than water quality benchmarks for aquatic life. Temple and Johnson (2011) also report detections of imazapyr in 2010 at sites below forest land uses, although this data is not incorporated into their analyses.

The USGS continuously monitored pesticides in the Hood River watershed from March 2011 to March 2012 using POCIS and SPMD samplers to determine time-weighted average water concentrations over each two-month deployment period (Hapke et al. 2016). Four sites were sampled: the mouth of Neal Creek (also sampled by DEQ); Rogers Creek (a tributary to the Middle Fork Hood River); Green Point Creek at its mouth; and the West Fork (W.F.) Hood River at its mouth. Based on the land use descriptions in Hapke et al. (2016), the Neal Creek site receives "pesticide-laden fruit processing facility wastewater discharge" (pg. 3); Rogers Creek has only 9% forest land use (compared to 14% agriculture with the remaining bare rock); while both the W. F. Hood River (DEQ sample location) and Green Point Creek (a tributary to the WF Hood River) deployments in the upper watershed were intentionally sited in cooperation with ODF because they would be harvested and sprayed during the deployment period in the fall, 2011.

The POCIS samplers detected four forestry-use herbicides at both the Green Point Creek and W.F. Hood River sites during their late August through October, 2011 deployment (Hapke et al 2016). These chemicals, and their concentrations at Green Point Creek and WF Hood River, respectively, were: triclopyr (0.170 µg/L., 0.250 µg/L); 2,4-D (0.170 µg/L, 0.250 µg/L); chlorsulfuron (0.027 µg/L, 0.026 µg/L); and at W.F. Hood River only, metsulfuron methyl (0.070 µg/L). Pyrethroids and chlorpyrifos were not detected at either site. Legacy pesticides, such as hexachlorobenzene (a fungicide) were found at Green Point Creek (0.000015 µg/L) and W.F. Hood River (0.000013 µg/L), and *o,p'* and *p,p'*-DDT at the W.F. Hood River site (0.000007 µg/L for each). None of these levels were sufficiently high to merit discussion.

Results from the 2014 DEQ grab samples and POCIS/SPMDs (Masterson and Crown 2015, Crown et al. 2015). One sample site, Upper Neal Creek downstream from the irrigation diversion, is the same site reported in the Temple and Johnson (2011) study. Low levels of hexazinone (Velpar) were found in four samples at the Upper Neal site; imazapyr was also found in four grab samples at this site in March and October at around 0.050 µg/L or lower. Both imazapyr and hexazinone concentrations were significantly lower in 2013 and 2014 than they were observed in 2009 – 2012 (Masterson and Crown 2015a). Diuron, an herbicide used to control annual and perennial broadleaf and grassy weeds in non-crop areas and fruits (NCBI 2020h) was found in once in March, twice in April, once in May, and once in October, 2014 at the Upper Neal Creek site at concentrations of 0.0111 µg/L., 0.0227 µg/L, 0.0402 µg/L, 0.0121 µg/L, and 0.0904 µg/L, respectively, much below its MRL of 4.29 µg/L. Propiconazole, a systematic foliar fungicide (ExToxNet 1993), was detected once in April and again in October, 2014 grab samples at concentrations of 0.0681 µg/L and 0.042 µg/L, again much below its MRL of 21.5 µg/L. Neither diuron nor propiconazle is currently approved for use in forestry. None of the POCIS/SPMD or sediment sample sites corresponded to a location having the majority of upstream land use in forestry (Crown et al 2015).

6.4.3.2. South Yamhill River PSP. The second Pesticide Stewardship Partnership (PSP) site that provides information on the effects of forest management on residues in stream water is a subset of the larger Yamhill PSP using three sites on tributaries to the South Fork Yamhill River in the Grand Ronde area (Cook et al. 2018). The South Fork Yamhill River is used as a surface water source for the City of Sheridan, especially during periods of high demand (MSA 2002). Forest land uses comprise 32% of the source watershed area for the City of Sheridan (DEQ 2018). The pesticide sample sites were at the mouths of Agency Creek, Gold Creek and Rogue River; with about 54 grab samples collected from October 2010 through October 2016. In addition, a POCIS sampler was installed at the Rogue River site for 28 days in October 2010. Other than standard ODF notifications for chemical activities, no additional detailed information on the timing, location, or amounts of pesticides applied during the study were available. Pesticide loadings into receiving streams were not determined because stream discharge data was not available.

Land uses in Agency Creek above the sampling site is 96.4% Forest, with 1.5% Urban and 2.1% Other. There is no Agriculture in the basin. For Gold Creek, 94.7% of land above the sample site is forest, with 3.3% urban and 1.8% "other". Only 0.2% is classified as agriculture. The Rogue River sub-watershed is more urban (8.6%), with other representing 4.4%, and agriculture 0.2%. The remaining 86.8% of land in the Rogue River sub-watershed is forest. Agricultural uses include grass and hay for livestock consumption, and small Christmas tree farms. The predominant herbicide active ingredients used by forest managers in the course of the study were glyphosate, imazapyr, atrazine, metsulfuron-methyl, sulfometuron methyl, and hexazinone. In addition to these chemicals, there were two degradates, AMPA from glyphosate and desethylatrazine from atrazine. Generally, herbicides were used during the spring (March 1 to May 30th) and fall (September 1 through October 31), based on ODF Notification start months (Cook et al. 2018). Table 6-7 shows a summary of the results using the U.S.E.P.A. aquatic life benchmarks.

As with Hood River, the primary focus for pesticide monitoring in the South Yamhill PSP study was the potential effects of pesticides on fish and other aquatic life. Of the herbicides, only imazapyr was detected in Agency Creek (0.126 **µg/L** in October 2010); while hexazinone (once), metsulfuron-methyl (twice), sulforneturon methyl (four times), and the glyphosate degradant AMPA (once) at the Rogue

River sampling location. The vast majority of pesticide detections were found in Gold Creek (note, Table 9 in Cook et al. [2018] incorrectly labels this as "Agency Creek"). Atrazine (7 times in 2011 and 2012, along with another 7 times for its degradate desethylatrazine in 2012) and hexazinone (twice in 2012) were found in Gold Creek. The aquatic herbicide fluridone, detected at Gold Creek in both April of 2012 and 2013, is not labelled for forestry use and it is unknown where and why it was used. DEET is an insect repellent that was detected twice in Gold Creek in September, 2012 and April, 2014, and once in Agency Creek in August, 2016. There is no expectation that the detection of DEET was related to a forest management activity. The 2010 POCIS deployment for 28 days in the Rogue River (at Highway 18) detected the presence of atrazine, hexazionone, and triclopyr in low levels; a grab sample taken once at the same site detected no herbicides.

The results from the South Yamhill study are similar to those found at Hood River. There are detections of herbicides typically used in forest management activities, but at levels significantly below those that are likely to cause harm to humans in either acute or chronic exposures (Cook et al. 2018). One limitation of the South Yamhill PSP study is the lack of sampling after the onset of rains in the fall.

Table 6-7. Water quality sampling results from the South Yamhill PSP, 2010 – 2016.

Herbicide	Samples (#)	Detections (#)	Detection Frequency (%)	Aquatic Life Benchmark (µg/L)	Benchmark Exceedences (#)
Atrazine	183	6	3.3%	1	0
AMPA	63	1	1.6%	249,500	0
DEET	168	3	1.8%	37,500	0
Desethylatrazine	153	8	5.2%	N/A	N/A
Fluridone	168	1	0.6%	480	0
Hexazinone	168	3	1.8%	7	0
Imazapyr	157	1	0.6%	24	0
Metsulfuron-methyl	57	2	3.5%	0.36	0
Sulfometuron methyl	153	4	2.6%	0.45	0

Source: Cook et al. 2018, Table 5.

6.4.3.3. EWEB McKenzie River Project. Forested lands represent 88% of the McKenzie watershed (Morgenstern et al. 2017). Over half the McKenzie watershed is managed by the Willamette NF; while the BLM manages another 1/6; the remaining one-third is a variety of industrial and non-industrial timberland owners. Most of these private owners are concentrated in the Mohawk, Gate Creek, Mill Creek and Quartz Creek basins. EWEB also owns the 900-acre Leaburg Forest, a patchwork of properties bordering the Leaburg Power Canal and Hydroelectric Plant. While one purpose of the forest is to generate revenue, improving forest condition will also improve water quality and increase habitat benefitting EWEB's Federal Energy Regulatory Commission (FERC) license requirements (EWEB 2017). Trout Mountain Forestry created a Stewardship Plan in 2016 intended to demonstrate best forest management practices to protect water quality and improve forest health; and the first thinnings in the forest were conducted in 2017 (Morgenstern 2017).

The EWEB developed its first drinking water source protection plan for EWEB was published in 2000 (Blair 2000). Forestry is identified as a potential source risk in the 2000 DWPP, but most concern related to sediment from forest roads and changes in peak flows from harvest activities. A new “10-Year Strategic Plan (2018-2028)” following the AWWA G-300 standards for implementing source water protection programs was finalized in 2017 (EWEB 2017). This plan emphasizes monitoring of storm events during first flush winter and spring storms. It also formalizes the “Pure Water Partnership” (www.purewaterpartners.org) to provide landowner incentives to protect source water quality. “Healthy Forests Clean Water” focuses in the middle and upper watershed to reduce wildfire risk, protect water quality, increase fish and wildlife habitat, and generate revenues through stewardship contracts on Willamette NF lands. Private industrial forestry activities are identified as a general focus area for the Middle McKenzie region of the watershed.

The EWEB conducted a baseline source water protection study from 2000 – 2009 (Morgenstern et al. 2011). One part of this project included an EWEB and USGS storm event monitoring program, begun in 2002 and extending to 2010, that focused on pesticides and other dissolved organic compounds. Industrial forestry was one of three primary land uses evaluated for the study; the others were urban runoff and agriculture (Kelly et al. 2012). Sub-basins likely to experience high amounts of chemical application were identified by ODF notifications. Sampling was conducted based on ODF notifications; however, there was no information on pesticide application rates, the exact chemical, or the volume applied. Grab samples were collected twice-yearly at 28 tributary and mainstem sites, resulting in 117 samples during 16 sampling events during storm runoff in the fall and spring. Fifteen of the sample sites were characterized as “forestry” with a total of 33 samples collected during the study. Of the 51 pesticide detections at the forestry sites, almost two-thirds occurred during spring storms (31), with the most of the others (18) occurring during fall storms, and non during non-storm sampling; there was a statistically significant relationship ($r=0.68$; $p<0.0001$) between pesticide detections and rainfall at the forestry sites. A total of 14 pesticide compounds were detected at forest sites, although using a minimum reporting level of 0.1 µg/L reduced this number to 3 compounds, of which 2 were unique to the forestry sites (imazapyr and nicosulfuron, with one detection each). Only one compound, triclopyr, exceeded 1.0 µg/L at a forestry site, but on further investigation was determined to be a recent homeowner application.

In addition, USGS began using POCIS/SPMD passive samplers in September-October, 2007 at three sites, and then expanding to six sites between March-October, 2010 (McCarthy et al. 2009). Results from the passive samplers are reported in McCarthy et al. (2012 [data] and 2014 [analyses]). Only one sampling site (E310, Camp Creek at Camp Creek Road) potentially shows the influence of active forest management; this site is 85% industrial forest, and 13.5% agriculture (pasture, Christmas trees, livestock, blueberries, etc.), with 1.5% rural residential land use. (Note, however, that Kelly et al. [2012] characterize this site as a “mixed” land use). During the 2007 POCIS sampling at Camp Creek, although 21 pesticides were detected, none were found at levels greater than the method detection level (MDL), generally less than one nanogram per liter (ng/L). In the 2007 SPMD sampling, diethyl phthalate, benzophenone, phenanthrene, flouranthene, and pyrene were detected at levels between the MDL and the method quantitation limit (MQL) (McCarthy et al. 2012). During the sampling at Camp Creek from 5/25 to 6/23/2010, hexachlorobenzene (HCB), lindane, o,p’-DDD, endosulfan-II, endosulfan sulfate and trans-permethrin were all found at, or greater than, the method quantification level (MQL) (McCarthy et al. 2012, Appendix 2, Table 4). In the 2011 sampling from 4/20 to 5/18 and 8/24-9/22, hexachlorobenzene (HCB) and pentachloroanisole (PCA) found at levels above MDL. None of these chemicals are typically used in forest management, although o,p'-DDD (Mitotane) may be present as a degradant of historic DDT applications (however, it's also used as a pharmaceutical to treat adrenocortical carcinoma and Cushing's syndrome in humans).

The EWEB Strategic Plan (Technical Appendix) and other communications are clear that they consider forested lands to produce higher quality water than from any other potential surface water source. They recognize, however, that forest management use of pesticides is a potential risk; but through their tracking and monitoring, they consider it comparatively low risk (Morgenstern et al. 2017). "The water quality data from samples collected downstream of industrial forest land uses indicates various pesticides being detected at low levels during significant rainfall events. Even though this data indicates forestry activities are a lower priority threat, EWEB continues to monitor water quality and work with forestry stakeholders to prevent and reduce wildfires, mitigate roads, increase riparian forest buffers, and reduce chemical use" (Morgenstern et al. 2017, 25). This perspective is shared by the USGS researchers who conducted the reconnaissance level monitoring: "… even though the data are limited, these results indicate that effects of forestry pesticide use are negligible at these locations in the river system … ." (Kelly et al. 2012, 30). "Although forest land use is predominant in the basin, and forestry pesticide use can be detected in small tributaries draining forested lands following application, these compounds rarely were detectable in the McKenzie River. Forestry pesticide use, therefore, probably is not a potential threat to drinking-water quality at the present time." (Kelly et al. 2012, 32). "… the majority of compounds that present a documented threat to drinking water quality, in terms of water-quality regulations or suspected endocrine disruption, are associated with agricultural and urban land use applications rather than forestry" (Kelly et al. 2012, 34).

6.4.3.4. WSDA Hoh River Study. Similarly to the South Yamhill PSP study in Oregon, the Washington State Department of Agriculture (WSDA) partnered with the Hoh Indian Tribe to evaluate pesticide occurrence in areas managed for commercial timber production on the Olympic Peninsula (Handcock 2018). For a pilot study, WSDA chose four tributary streams to the Hoh River that were expected to have timber harvests and reforestation activities during the study period. These streams are Elk Creek (3.74 $mi.^2$; 77% Washington Department of Natural Resources (DNR) land, 23% private timber, 0.4% non-profit); Winfield Creek (10.74 $mi.^2$; 54% DNR, 41% private timber, 5% non-profit); Lost Creek (2.11 $mi.^2$; 67% private timber, 2% state, 31% non-profit); and Nolan Creek (9.69 $mi.^2$; 49% state, 35% private timber; 17% non-profit). The non-profit lands are owned by The Nature Conservancy (TNC) and the Hoh River Trust and intended to create a 10,000 acre, 32-mile conservation corridor from Olympic National Park to the Pacific Ocean (TNC 2017); they have few timber harvests (Handcock 2018).

Each site was grab sampled six times during the summer of 2017: a mid-July background sample, and then weekly from August 9th through September 5th. Seven herbicides commonly used in forestry were analyzed: glyphosate: gludosinate-ammonia; aminomethylphosphonic acid (AMPA), a degradant of glyphosate; imazapyr; triclopyr; metsulfuron methyl; and sulfometuron methyl. Sample collection, storage, and analyses followed WSDA Standard Operating Procedures. Quality assurance methods for the pesticide analyzes included field replicates, field blanks, matrix spikes, matrix spike duplicates; laboratory quality control included laboratory control samples, laboratory control sample duplicates, surrogate spikes, and method blanks.

No herbicides were detected at any of the four sites during the background sampling on July 17th, or during the first sample event on August 16th. Nor were detections ever made during any sampling event in Winfield Creek. Herbicides were found at the other three sites sampled, with 13 positive detections found at these sites: glyphosate (4 detections), glufosinate-ammonium (7 detections), and AMPA (2 detections) (Table 6-8). The most detections occurred on August 23rd with five detections, and the only herbicide ever found in Lost Creek was glufosinate-ammonium on that date. At Nolan Creek, glufosinate-ammonium was found on 3 out of the 4 sample dates, glyphosate and its degradant AMPA on two occasions. At Elk Creek, glufosinate-ammonium was detected in the last three sample dates; as was glyphosate on 8/23 and 8/29, with its degradant AMPA on the last two sample dates. In no case were concentrations of herbicides detected above USEPA aquatic health benchmarks (Handcock 2018).

Table 6-8. Herbicide detections in the 2017 WSDA Hoh River study.

Date	Location	Herbicide	Concentration (µg/L)
8/9/2017	Nolan Creek	Glufosinate-ammonium	0.008
	Nolan Creek	Glyphosate	0.010
8/23/2017	Nolan Creek	Glufosinate-ammonium	0.058
	Nolan Creek	Glyphosate	0.032
	Lost Creek	Glufosinate-ammonium	0.008
	Elk Creek	Glufosinate-ammonium	0.010
	Elk Creek	Glyphosate	0.266
8/29/2017	Elk Creek	AMPA	0.015
	Elk Creek	Glufosinate-ammonium	0.003
	Elk Creek	Glyphosate	0.027
9/6/2017	Nolan Creek	Glufosinate-ammonium	0.008
	Elk Creek	AMPA	0.008
	Elk Creek	Glufosinate-ammonium	0.008

Data source: Handcock, 2018, Table 6.

Similar with the other studies discussed in this section, the exact timing, location, and formulation of herbicides in the tributary watersheds were unknown for this pilot study. It is possible that some of the detections may have resulted from activities other than forest management, such as vegetation control along roadsides. Specifically, during sampling at Elk Creek on 8/23, an aquatic noxious weed control spray team was working in the vicinity of the sample location.

6.4.4 Pesticide Fate Modelling Approaches.

The USEPA has a number of modeling approaches to evaluate contaminant risk. The Pesticide in Water Calculator (PWC) for both surface and groundwater based on percent cropped area in a watershed (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment#PWC). Additional USEPA models can be found at: https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment. The USDA Soil and Water Assessment Tool (SWAT) (https://swat.tamu.edu/) has been used in over 3,000 journal publications worldwide to model a wide range of pollutants. The uncalibrated model is based on a standard set of parameters, including digital elevation models (DEM); stream layers, in the U.S. typically the National Hydrographic Dataset (NHDPlus); landuse, generally the Cropland Data Layer (CDL) in the U.S.; soils, usually the Soil Survey Geographic (SURRGO) database; and weather (daily temperatures and precipitation). The SWAT model has been used to evaluate pesticide fates in the Coulonge drinking water catchment area of southwest France (Vernier et al. 2017), and also for spatiotemporal modeling of pesticide fates in agriculture (Wang 2019), but no references were found to it ever being used in forestry (Iowa State University SWAT literature database for peer-reviewed journal articles (https://www.card.iastate.edu/swat_articles/).

We could find only one example of pesticide fate modeling conducted in Oregon. A recent journal article presents the initial results of SWAT modeling in Zollner Creek in Pudding River sub-watershed of the Molalla River that is 91% agriculture (Janney and Jenkins 2019). Atrazine was the focus of the Zollner Creek monitoring, similar to other SWAT modeling (Winchell et al 2018). Janney and Jenkins used five different parameter optimization scenarios to evaluate modeling biases, considering ±25% of estimated stream discharges and atrazine concentration levels to be an acceptable model outcome. These scenarios sequentially added local knowledge and spatially explicit data to the standard SWAT model formulation. Scenarios that reduced bias were: (a) adding better precipitation data using NEXRAD, although some overestimate of flows remained; and (b) adding an estimated amount of tile drainage improved the model outcomes to "very good" using the Nash-Sutcliff modeling efficiency coefficient, a satisfactory percentage bias, and the best fit for the mean daily streamflow. Of the 29 parameters likely to influence the hydrologic simulation, streamflow was considered the most important (Janney and Jenkins 2019). However, according to the authors, the study was limited due to the lack of streamflow monitoring, information on the extent of tile drains, amounts and locations of atrazine applications, and the relative infrequency of pesticide sample collection (12 times).

In contrast to the Janney and Jenkins (2019) SWAT modeling where atrazine applications were unknown, Winchell et al. (2018) studied 27 watersheds in Illinois, Indiana, Ohio, and Texas, 25 of these as part of the Atrazine Ecological Monitoring Program (funded by Sygenta, an atrazine manufacturer) with the remaining two in Ohio in the Heidleberg University National Center for Water Quality Monitoring program. Atrazine data was provided through surveys at the crop reporting district that included total mass of atrazine applied, total area treated, and total crop area over several years. Temporal applications of atrazine within a probability distribution were estimated based on planting timing. Grab water samples were collected on an average of once every four days, as well as some rainfall-driven event-driven sampling to represent runoff. In general, Winchell et al. (2018) found that the uncalibrated model slightly over-predicted atrazine concentrations, but that the mean bias (observed/simulated) was less than one part per billion (0.93 ppb), and generally less than a factor of 2 in the concentration.

As an evolution of SWAT, the USEPA has developed a web-based, interactive pesticide fate model (https://epahawqs.tamu.edu/) at the 12-digit hydrologic unit code (HUC) scale that uses SWAT as its foundation (Yen et al. 2016). There seems to be potential to conduct these types of modeling exercises for primarily forested watersheds under active management to obtain better, and site-specific, pesticide fate information for community water supply watersheds.

6.5. Chemicals in Raw Water Supplies

As we've seen above, pesticides used in forest management find their way into streams, typically in very low concentrations and during either the application or first few storms in the winter. Yet, forest management is one of many activities where pesticides are applied. Various organizations sample to identify the types and amounts of pesticides in surface waters. Table 6-9 shows the results of sampling conducted statewide from 1995 to 2020 by the DEQ, ODA, USEPA and USGS for those pesticides currently labelled for forest use (there were no results identified for difubenzuron and glufosinate-ammonium). Almost 42,000 water sample results were available for these nine pesticides, and the table divides these into three categories: (a) the chemical wasn't detected; (b) the chemical was detected, but at concentrations below the ability of an instrument to quantify; and (c) concentrations that were sufficient to quantify. In the vast majority of cases, while a pesticide was found in surface waters, the concentrations were sufficiently low that in about 88% of the samples the level couldn't be quantified; and in those cases where it could, most were below water quality standards, in many cases by orders of magnitude. This is consistent with the results of studies that Oregon DEQ has done on toxics, both statewide (DEQ 2015) and specifically for the North Coast (DEQ 2019).

Table 6-9. Numbers of samples and detections of pesticides used in forestry in Oregon streams, 1995 – 2020.

Pesticide	Non-Detect	Quanti-fiable	% Detect	Total Samples
2,4-D	4,074	341	7.7%	4,415
Atrazine	7,730	2,553	24.8%	10,283
Carbaryl	5,445	442	7.5%	5,887
Glyphosate	802	421	34.4%	1,223
Hexazinone	4,620	152	3.2%	4,772
Imazapyr	4,869	219	4.3%	5,088
Metsulfuron-methyl	2,827	238	7.8%	3,065
Sulfometuron methyl	4,085	698	14.6%	4,783
Triclopyr	2,343	124	5.0%	2,467

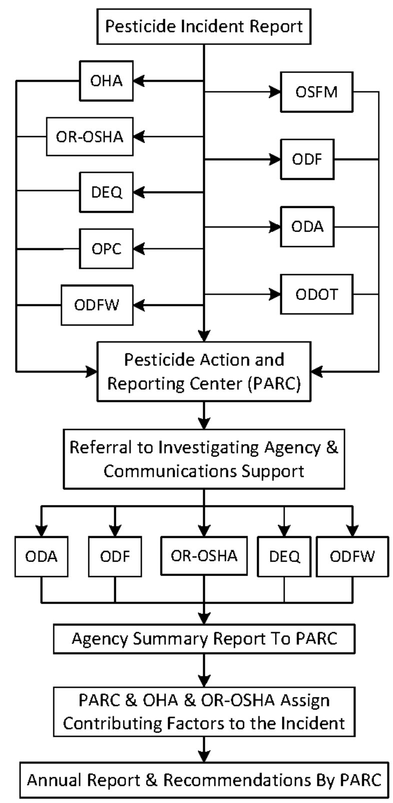
Source: DEQ, AWQMS Water Quality Monitoring Data, 1/1/1995 to 4/6/2020 for surface water samples collected by DEQ, ODA, USEPA and USGS (accessed 4/8/2020).

The SDWA requires public water suppliers to periodically test for toxic chemicals in both their raw water intakes and distribution systems. The Oregon Health Authority (OHA) reports these results as "Alerts" in their "Drinking Water Data Online" system. An "Alert" is issued if a water sample shows contaminant concentrations above one-half the Maximum Contaminant Level (MCL) for inorganic chemicals, and any detected value for synthetics and volatile organic groups. We queried this database for the same period as that covered by the results in Table 6-9, specifically looking for those drinking water systems that had Alerts for chemicals commonly used in forest management. From 1995 to April, 2020, there were 2,293 alerts for 26 organic chemicals found in drinking water. The data provided the sample location, which was either the entry point (EP) into the treatment plant, or from the distribution system (DIST). Many utilities have multiple sources for their raw water, and we were only interested in those from surface water, or groundwater under the influence of surface water (GU). Examining the records, we were able to cross-reference those entry points that were surface water from those from well water, ultimately finding 254 alerts for 74 public water supplies resulting from organic contaminants in surface waters at their raw water intakes.

Of these 254 alerts, only two are for chemicals used in forest management: 2,4-D and atrazine. There were four alerts for 2,4-D, two each for Salem and Lake Oswego. The Salem alerts occurred in August, 2016 and April, 2017 with concentrations of 0.12 µg/L and 0.11 µg/L, respectively, from its the North Santiam River raw water intake. The two Lake Oswego alerts occurred in late September, 2007 and June, 2013 with concentrations of 0.63 µg/L and 0.13 µg/L, respectively, from its Clackamas River raw water intake. The MCL for 2,4-D is 3.0 µg/L, so the levels of 2,4-D contamination found in these two systems ranged from one-fifth to one-three hundredth of the drinking water standard. The only other organic contaminant found at raw surface water intakes was atrazine, more commonly used in agriculture than forestry. It was found in the Canby Utility intake in February, 2015 at a concentration of 0.17 µg/L at its Mollala River source. Atrazine also has a 3.0 µg/L MCL, so the concentration found was one-twentieth of the standard.

6.6. Pesticide Application Violations

Pesticide use in Oregon is regulated by the State Pesticide Control Act (ORS Ch. 634) and enforced by the Oregon Department of Agriculture (ODA) under administrative rules (OAR Ch. 603, Div. 57 Pesticide Control). To be used in Oregon, pesticides are required to be registered with ODA, and as part of this regulation appropriate uses and controls are identified. The statute also preempts local pesticide regulations except on their own lands (ORS 634.057). Pesticide applicators must be licensed (ORS 634.122), and aerial applicators require an additional certificate (ORS 634.128). There are 22 categories of prohibited acts (ORS 634.372). The Act also established the Pesticide Analytical and Response Center (PARC) to receive and coordinate responses to pesticide incidents among state agencies (ORS 634.550). Figure 6-4 shows the process used to receive complaints through state agencies, investigate, and report findings (PARC 2018). The Oregon Department of Forestry has procedures for receiving pesticide-related complaints, and working with ODA and others on investigations (PARC 2018).



Source: PARC 2018.

Figure 6-4. Pesticide complaint process.

Pesticide incident reports are retained for five years (C. Higby, PARC, personal communication 2/24/2020). A query of ODA's Pesticide Program database on 2/21/2020 found 4,149 pesticide-related incidents from July, 2013 to August, 2019, of which 140 (3.4%) were related to forest use. In addition to a spreadsheet with summary data, each incident's Case Detail report was also reviewed. Figure 6-5 shows the number of reported incidents annually from 2013 through 2019. Note, however, that a single pesticide-related incident may have multiple complainants. Of the 140 incidents, almost 74% (104) are related to aerial applications of herbicides; 14% (19) are for ground applications in forest units; 5% (7) are for right-of-way spraying in forested areas; 4% (6) are applicator records and licensing reviews; and 3% (4) are general concerns not attributable to a specific application type. From the 140 incidents, there were 6 incidents where violations were found; and, in three cases, "Letters of Advisement" were sent as warnings.

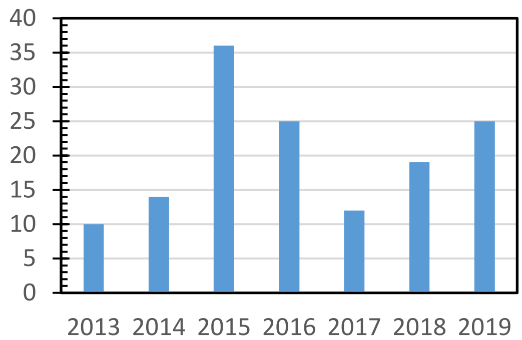


Figure 6-5. Annual pesticide-related incidents report to ODA.

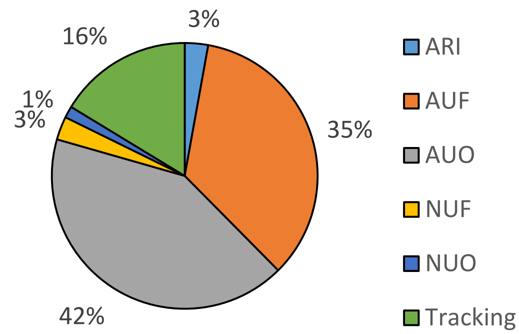


Figure 6-6. Types of pesticide-related incidents, 7/2013 – 8/2019.

The ODA classifies pesticide-related incidents into five different categories (Figure 6-6): (1) Agriculture Use Observation (AUO); (2) Agriculture Use Follow-up (AUF); (3) Non-agricultural Use Observation; (4) Non-agricultural Use Follow-up; (5) Applicator Records Inspection (ARI); and (6) Tracking. Observations are when an ODA Pesticide Investigator is on-site during an application, commonly accompanied by an ODF Stewardship Forester. As part of the Observation, the ODA Pesticide Investigator will also inspect the labels for all chemicals applied, determine whether the applicators are appropriately licensed, and in some cases return after a period of time to evaluate vegetation to determine if there are signs of drift. Typically observations are requested by landowners (timber companies) when they are conducting applications in sensitive areas such as adjacent to lakes and State Parks, where there are known concerns in the community (i.e., Triangle Lake), or past history with neighbors. Of the 61 observations (out of 140 incidents) during our analysis period, the vast majority (87%) are for aerial spraying, and 92% of those were at the request of the landowner. A similar proportion of ground application observations are initiated by landowners. Based on the case notes, ODA often encourages landowners to avail themselves of their availability to observe applications as a mechanism to reduce disputes. However, it is possible that ODA identifies a violation during these observations: it happened in 4 out of the 57 cases initiated by landowners, 3 times for aerial applications and once for ground.

The other major category of pesticide incidents are Complaints, comprising 38% (64) out of the 140 cases. These are classified by ODA as "Follow-ups" from either agricultural applications (i.e., forestry for our dataset); or non-agricultural use, which in our analysis corresponds to spray applications to rights-of-way in forested areas. Complaints generally result in an investigation by ODA (again, usually in conjunction with ODF Stewardship Foresters), that includes a site visit and discussion with the complainant, the applicator, the landowner. If a violation is suspected, samples of vegetation and/or soil may be taken for ODA laboratory analysis. A detailed case record of the incident in the form of an affidavit is prepared by the ODA Pesticide Investigator. Almost all of the forestry complaints regarding aerial applications are the result of concerns over drift. There were 5 cases of violations in the 41 (12%) cases where investigations led to a finding, mostly of ORS 634.372(4) for carelessness and negligence, and in one case ORS 634.372(9) for failure to follow label requirements. In one case, Cedar Valley near Gold Beach on the south coast, the aerial applicators license was revoked, although his $20,000 fine was suspended; and in another case (Applebee Aviation), had two violations (one of which resulted from spraying crew members) that resulted in their license suspended for a year and a total of $55,436 in fines. Complaints regarding ground-based applications and particularly rights-of-way generally involve spraying into live water, especially streams; or spraying onto adjacent properties (again, mostly for rights of way applications). There were 9 complaints over ground-applications: 4 were unresolved, usually due to either the operation hadn't yet been conducted, or the complained failed to cooperate; 4 other cases were investigated and found not to be violations; and in 1 case the violation was due to drift over a fenceline to adjacent property.

The third type of pesticide-related incidents involve ODA reviewing applicator records, i.e., Record Reviews. Whenever an observation or complaint is filed, the ODA Pesticide Investigator checks to determine whether the pesticide is appropriately labelled for forestry (or ROW) use and whether the concentration applied is within the label limits for that use. At the same time, the licenses of the operator and all applicators are confirmed, particularly whether they have the appropriate endorsements for forestry, ROW, and rotary-wing aerial (if applied by helicopter). For aerial operations, the GPS tracking records are also requested and reviewed. Beyond these standard operating procedures for observations and complaints, ODA may request three years of application records from operators and sample specific jobs to determine whether the paperwork contains the required elements (OAR 503-057-130). Pesticide dealers have similar record-keeping requirements (OAR 603-057-0140). There were 6 of these Records Inspections over the 2013 – 2019 period for operators involved in forestry applications. In two of these, it was determined that the operator was not involved in applications at that time. In three cases, the record keeping was determined inadequate and the operator received a Letter of Advisement (warning) to improve their practices. And, in one case the operator was a Letter of Advisement due to lacking appropriate licenses and endorsements for roadside spraying without a ROW endorsement.

The fourth type of pesticide-related incidents are designated as "Tracking" that is a general catch-all category. Tracking is used when there is too little information to initiate an investigation, a complainant fails to cooperate with the investigation, ODA (PARC) is contacted about a forthcoming application, or another agency is taking the lead on an investigation.

The ODA, often through PARC, receives about 700 pesticide-related referrals annually. Of these, about 3.5% are related to forestry; and for the forestry incidents, about 75% are concerned with aerial application of pesticides. Half these aerial-related referrals are requests by landowners for ODA to observe spraying; with the other half complaints about past or future applications. There are three "hotbeds" for complaints: the Triangle Lake/Noti area of the central Coast Range; Gold Beach on the south coast; and the Rogue/Applegate valley in southern Oregon. On average, there are only slightly over three pesticide-application violations of all types (application, record-keeping, licensing) per year in the forestry sector. In two cases, these violations resulted in the suspension or revocation of aerial application licenses. In context, according to ODF's FERNS notification system, there were likely around 7,000 applications per year over this period (Table 6-1), involving 454 applicators with Forestry license endorsements, including 115 applicators and operators with the aerial (helicopter) endorsement (Kachadoorian 2019).

6.7. Summary & Recommendations

The use of forest chemicals is a complex admixture of physics, biology, and social science. We addressed this by evaluating the extent and types of chemical uses and their effects on water quality, with a particular emphasis on effects at the raw water intake. We reviewed published, peer-reviewed, articles on effects of herbicide treatment; and evaluated a number of other studies conducted by agencies. We analyzed water quality data, both for streams as well as conditions at raw water intakes for community water supplies. Finally, we examined four years of forestry-related pesticide incidents to assist in understanding controversies related to forest chemical use. We will conclude the chapter by summarizing the findings from the information presented, and provide a set of recommendations for future efforts based on our analyses.

6.7.1 Summary

Chemicals play an integral role in the management of Oregon's forests. Based on an analysis of ODF's FERNS data, there are over 7,400 activities that involve chemical applications on potentially one million acres of Oregon forest land annually, with the vast majority of these herbicide applications to harvested units (Table 6-1). Applications range from herbicide spraying for site preparation prior to replanting, and competing vegetation control afterwards, animal and rodent repellants to protect seedlings, fertilization to increase growth rates after thinning, and for maintenance of rights-of-way for both travel and utility corridors. With the exception of rights-of-way, a defining characteristic of these chemical applications is that they occur infrequently over the 30 – 80 year typical harvest cycle (Figure 6-1). And while the public perception of chemical use in forests is amplified, pesticides applied to forest land represent only about from 2.8% (2007) to 4.2% (2008) of those used statewide according to data reported through the Oregon Pesticide Use Reporting System that was defunded in 2009 [ODA 2008, 2009]). Accordingly, it's relevant that only 3.5% of pesticide-related incidents involving forestry use of pesticides from the more recent ODA data.

In comparison to other sectors of Oregon's economy that use pesticides, those typically applied in forestry (Table 6-2) are less toxic to humans, move fairly rapidly through soil and water, and don't accumulate (Table 6-4). Most of these are herbicides that are not strongly absorbed (attached) to soil particles, are water soluble, have low volatility (i.e. evaporation and resuspension), and decay rapidly in both water and soil. This means that these herbicides don't tend to build up in the soil or bio-accumulate.

Contemporary best management practices, with a couple of additions, have the potential to protect areas off-site if followed. Extensive research (and accompanying models) have allowed a better understanding of the importance of droplet size distributions on reducing on pesticide drift, as have the development of adjuvants specifically tailored to mitigate drift. Helicopters have precise GPS and nozzle flow data loggers that accurately position the ship both in space and chemical delivery; some models can be preprogrammed to include flight plans that automatically buffer streams and sensitive areas. There is also substantial research from the agriculture community, and one paper reported here from forestry, on the value of wooded buffers to prevent drift into streams. Additions to the Forest Practice Act rules recently proposed through an industry-environmental collaborative process would extend forested buffers along non-fish streams.

This examination demonstrates that while pesticides are commonly detected in surface waters, in almost all cases they are found in concentrations below levels that can be accurately measured. When quantifiable detections are found, as we've seen from the forestry use studies, they tend to be transient and most likely to occur either during application or in early season storms. In particular, unless live water is directly sprayed (a label violation for herbicides used in forest silviculture), most herbicide runoff occurs during the first winter storms, in one report (Morgenstern 2014) this constituted 70% - 90% of the pesticide loadings, a finding that was confirmed by the Louch et al. (2017) and Caldwell and Courter (2019). A caveat here, again, is that the impact of forest chemicals on downstream raw source water supplies will depend on the size of the contributing watershed, the proportion annually subject to chemical applications, and other landuses in the basin.

6.7.2 Recommendations

1. **Pesticide use data needs to be reported**. It is difficult for the stakeholders and the affected public to understand the impacts, positive and negative, of forest chemicals without good reporting data. This is part of a larger concern over pesticide use relating to air and water quality in Oregon. At present, data on pesticide and chemical use is not routinely reported, even at the aggregate level. While ODF FERNS provides information on where and possibly when forest chemicals will be used, it allows multiple chemicals to be listed over long periods of time, with no subsequent reporting on what was actually applied unless a complaint was filed. In 1999 the Oregon Legislature created the Pesticide Use Reporting System (PURS), but it was never adequately funded and implemented. When its sunset provision was proposed for renewal during the 2019 Legislative Session in HB2980 there was broad support from across the political spectrum (Oregonians for Food and Shelter to the Farmworkers Union) for PURS to be extended and funded. This bill died in the Ways and Means Committee as the Legislature adjourned. A bill more specific to forestry was also introduced, HB4168 that implements the aerial application procedures and reporting requirements identified in the Memorandum of Understanding for the “Oregon Strategy” drafted by the timber industry and the conservation community (Governor’s Office 2020). This bill, too, died prior to passage in the House with adjournment. The Board of Forestry and ODF could by administrative rule change its notification system to require reporting and disclose chemicals used in management operations.

2. **Current water quality sampling efforts are insufficient**. A corollary to the lack of pesticide use information is the relative sparseness of data on potential pesticide loadings in surface waters, particularly at the raw water intakes for public water supplies. Most current sampling at raw water intakes is not correlated with times of likely chemical pulses, i.e., the early winter storms. Moreover, it’s clear from the silvicultural herbicide applications studies reviewed that detections and concentrations in receiving waters are highly variable even within a storm event. There is a similar constraint in the grab samples and automatic samplers that are commonly used: they provide detection and concentration information at point(s) of time, but not loads (i.e., the total mass of the substance transported in water over a given period of time) since stream discharge is usually not measured during the sampling (Meals et al 2013). Sampling and analysis techniques developed and applied by the U.S.G.S., such as POCIS and SPMD (see Section 6.4.2) have the capability to accurately integrate pesticide concentrations over longer time periods; and in conjunction with streamflow, the ability to estimate loads. These devices could be particularly beneficial at raw water intakes where there is concern over pesticide loadings and the quantity of water flowing into the intake is known.

3. **Study designs need improvement**. The majority of studies focused on assessing the impact of pesticides on water quality can be loosely characterized as “reconnaissance” or “case studies”

because of their study design and limited replicability. Most of the pesticide/herbicide peer-reviewed studies in the Pacific northwest, and other areas of the U.S. were conducted by industry or industry-supported organizations (NCASI) and tend to be short-term and locally-focused (Louch et al. 2017; Caldwell and Courter 2020). They have the advantage of knowing exactly when and what was applied, have more site-specific sampling, but are limited because the applicators know that they are being studied which may affect their behavior. In contrast, the PSP and USGS studies sampled over a longer period, but the PSP studies didn't have exact amounts and timing of application, and may have missed storm events; while the USGS studies using a sampling method that integrated pesticide concentrations over time, but was still limited because of unknown application amounts and timing. Improved study designs would incorporate random, applicator- and landowner-blind sampling of pesticide applications. This approach is critical for developing replicable scientific results.

4. **Pesticide fate modeling is a critical need**. Inference based on downstream measurements includes complex interactions between pesticide and environment, as well as assumptions on their spatial and temporal distribution, which still require significant research. A useful tool to answer many management questions is modeling. Complex hydrological models, such as the Soil and Water Assessment Tool (Wang 2019) could assist practitioners and regulators to understand the fate of silvicultural forest chemicals. The SWAT has been used for over 50 pesticide fate studies worldwide for agricultural practices, but not for pesticide fates in forest applications. While such process-based models have their limitations, they can provide a structured approach to evaluating herbicide movements at the watershed scale.

5. **Pesticide Stewardship Partnerships**. The PSPs are good outreach tools, but don't produce replicable science. The PSP doesn't collect pesticide application data and locations in its "partnerships,",and its sampling regimes aren't designed and implemented to catch episodic events (application, early winter storms) generally recognized to be when the highest concentrations are likely to be found. Additionally, the lack of streamflow data in these studies limits their ability to evaluate "loads" versus point concentrations. The benefits of the PSPs by involving landowners, applicators, and agency personnel could be further enhanced by better knowledge of pesticides applied and their timing, and better monitoring procedures as outline above.

6. **OSU Research Cooperatives provide a framework to support future studies**. Creating credible science in an arena as complex as forest chemical use requires long-term and intensive studies across the ownership landscape. One model to achieve this is the research cooperatives in the College of Forestry at Oregon State University. Since 1982 there has been an industry-agency-university cooperative studying forest revegetation that has a substantial record of accomplishments over its almost 40 year history, presently called the Vegetation Management Research Cooperative (http://vmrc.forestry.oregonstate.edu/). The VMRC has the partners and needed to successfully conduct the type of herbicide transport and fate studies and modeling described here.

7. **Wooded buffers prevent or reduce spray drift**. Both the Louch et al .(2017) and Caldwell and Courter (2020) studies demonstrated that non-buffered, small non-fish streams received spray during application. In contrast, the Thistle et al. (2009) study demonstrated the efficacy of wooded

buffers in capturing or deflecting fine spray drift. This finding is consistent with a number of studies on agricultural spray drift. The extension of wooded buffers to Small Non-fish streams under the Forest Practice Act and its rules would protect these streams from drift, and reduce potential loadings downstream. Extension of r spray exclusion zones along N streams is one of the proposals in the "Oregon Strategy" of the state, timber industry, and conservation groups (Governor's Office 2020); it's clear from the science that the effectiveness of these buffers would be improved if they were wooded.

6.8. Literature Cited

Allender, W. J. 1991. Movement of bromacil and hexazinone in a municipal site. Bull. of Environ. Contam. and Toxicol. 46: 284-291.

Alvarez, D.A., 2010, Guidelines for the use of the semipermeable membrane device (SPMD) and the polar organic chemical integrative sampler (POCIS) in environmental monitoring studies: U.S. Geological Survey, Techniques and Methods 1–D4, 28 pp.

Anderson, C.W., 2002, Ecological effects on streams from forest fertilization—Literature review and conceptual framework for future study in the western Cascades: U.S. Geological Survey Water-Resources Investigations Report 01–4047. 49 pp.

Arjo, W.M. and D.T. Bryson. 2007. Reducing non-target hazards of rodenticides in forest settings. Pp. 144 – 159 *In*: Nolte, D.L., W.M. Arjo, and D. Stalman (eds.), Proceedings of the 12th Wildlife Damage Management Conference, April 9-12, 2007, Corpus Christi, Texas. 542 pp.

ASTM (American Society of Testing and Materials). 2016. Standard Terminology Relating to Agricultural Tank Mix Adjuvants. ASTM E1519-16. https://www.astm.org/search/fullsite-search.html?query=E1519-16& accessed 4/16/2020.

Atwood, D. and C. Paisley-Jones. 2017. Pesticides Industry Sales and Usage: 2008 – 2012 Market Estimates. Biological and Economic Analysis Division, Office of Pesticide Programs, Office of Chemical Safety and Pollution Prevention, U.S. Environmental Protection Agency, Washington, D.C. pp. 32. https://www.epa.gov/sites/production/files/2017-01/documents/pesticides-industry-sales-usage-2016_0.pdf accessed 4/17/2020.

Bastardie, F., M. Cannavacciuolo, Y. Capowiez, J.R. de. Dreuzy, A. Bellido, and D. Cluzeau. 2002. A new simulation for modelling the topology of earthworm burrow systems and their effects on macropore flow in experimental soils. Biology and Fertility of Soils 36: 161-169.

Bernstein, L., Arkin, L., Lindberg, R., 2013. Oregon's Industrial Forests and Herbicide Use: A Case Study of Risk to People, Drinking Water and Salmon. Beyond Toxics, Eugene OR, p. 115. https://www.beyondtoxics.org/wp-content/uploads/2013/12/FINAL_Report_OregonIndustrialForest_and_HerbicideUse_12-17-13.pdf accessed 4/17/2020.

Beulke, S., C.D. Brown, C.J. Fryer, and W. van Beinum. 2004. Influence of kinetic sorption and diffusion on pesticide movement through aggregated soils. Chemosphere 57, 481-490.

Binkley, D., H. Burnham, and H. Lee. 1999. Water quality impacts of forest fertilization with nitrogen and phosphorus. Forest Ecology and Management 121: 191-213.

Binkley, D., J. Kay, M. Barry, and M.G. Ryan. 2004. First-rotation changes in soil carbon and nitrogen in a *Eucalyptus* plantation in Hawaii. Soil Sci. Soc. Am. J. 68: 1713-1719.

Bisson, P.A., G.G Ice, C.J. Perrin, and R.E. Bilby. 1992. Effects of forest fertilization on water quality and aquatic resources in the Douglas-fir region. Pp. 179–193 *In*: Chappell, H.N., G.F. Weetman, and R.E. Miller (eds.) Forest Fertilization: Sustaining and Improving Nutrition and Growth of Western Forests. University of Washington, Seattle, WA. 311pp.

Blair, S. 2000. Drinking Water Protection Plan, Source Protection Monitoring. Eugene Water and Electric Board, Eugene, OR. 99 pp.

Bothwell, M. 1992. Eutrophication of rivers by nutrients in kraft mill effluents. Water pollution research Journal of Canada 27: 447-472.

Buttle, J.M. and D.J. McDonald. 2002. Coupled vertical and lateral preferential flow on a forested slope. Water Resources Research 38: 18/1–18/17.

Büyüksönmez, F., R. Rynk, T.F. Hess, E. Bechinski. Occurrence, degradation and fate of pesticides during composting. Compost Science and Utilization 7(4): 66-82.

Caldwell, L.K. and L.A. Courter. 2020. Abiotic factors influence surface water herbicide concentrations following silvicultural aerial application in Oregon's north Coast Range. Int. Envir. Ass. and Mgmt. 16(1): 114-127.

Clark, L.A., G.J. Roloff, V.L. Tatum, L.L. Irwin. 2009. Forest herbicide effects on Pacific Northwest ecosystems: A literature review In. National Council for Air and Stream Improvement, Research Triangle Park, NC. pp. 181.

Cook, K.V., K. Masterson, and K. Abraham. 2018. Evaluation of South Yamhill Pesticide Stewardship Partnership Area. Oregon Water Quality Pesticide Management Team (OWEB, DEQ, ODF, ODA). 24 pp. https://www.oregon.gov/ODA/shared/Documents/Publications/PesticidesPARC/SouthYamhillPSPReport.pdf. Accessed 3/26/2019.

Cornejo-Oviedo, E., S. Boelker, D. Mainwaring, D. Maguire, F. Meinzer, and R. Brooks. Basal area growth, carbon isotope discrimination, and intrinsic water use efficiency after fertilization of Douglas-fir in the Oregon Coast Range. Forest Ecology and Management 389: 285-295.

(CPDA) Council of Producers & Distributors of Agrotechnology. 2019. Adjuvant certification program. https://cpda.com/adjuvant-certified-program/. Accessed 12/18/2019.

Cristan, R. W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E. Schilling. Effectiveness of forestry best management practices in the United States: Literature review. Forest Ecology and Management 360: 133-151.

Crown, J., K. Masterson, A. Hamel, S. Jetter, and L. Pillsbury. 2015. 2014 Hood River Water Quality Pesticide Monitoring Summary Report. Part II: POCIS/SPMD and Sediment Sample Monitoring Results.

Curran, W.S., and D.D. Lingenfelter, D.D., 1999. Adjuvants for Enhancing Herbicide Performance. Agronomy Facts 37. Penn State Cooperative Extension, College of Agricultural Sciences, Pennsylvania State University, University Park, PA. pp. 6.

Damalas, C.A. 2004. Herbicide Tank Mixtures: Common Interactions. Int. J. Agri. & Biol. 6(1):209-212.

Dent, L., and J. Robben. 2000. Aerial Pesticide Application Monitoring. Final Report. Forest Practices Monitoring Program, Oregon Department of Forestry, Salem, OR. 35pp.

(DHHS) U.S. Department Of Health And Human Services, 2007. NIOSH Pocket Guide to Chemical Hazards. DHHS (NIOSH) Publication No. 2005-149. Centers for Disease Control and Prevention, National Institute for Occupational Safety and Health, Cincinnati, OH. pp. 454.

DEQ (Oregon Department of Environmental Quality). 2018. Updated Source Water Assessment, City of Sheridan, PWS #4100811. Prepared for the City of Sheridan and Oregon Health Authority, May, 2018. 85 pp.

DEQ (Oregon Department of Environmental Quality). 2015. Statewide Water Quality Toxics Assessment Report. DEQ15-LAB-0065-TR. LAB-0035-TR. Laboratory and Environmental Assessment Division, Portland, OR. 56 pp.

DEQ (Oregon Department of Environmental Quality). 2019. North Coast Toxics Basin Summary. DEQ19-LAB-0035-TR. Laboratory and Environmental Assessment Division, Hillsboro, OR. 44 pp.

Devine, W.D., Footen, P.W., Strahm, B.D., Harrison, R.B., Terry, T.A., Harrington, T.B., 2012. Nitrogen leaching following whole-tree and bole-only harvests on two contrasting Pacific Northwest sites. Forest Ecology and Management 267, 7-17.

Dinger, E.J., Rose, R., 2010. Initial fall-spring vegetation management regimes improve moisture conditions and maximize third-year Douglas-fir seedling growth in a Pacific Northwest plantation. New Zealand Journal of Forestry Science 40: 90-108.

Dodds, W.K., V.H. Smith, and B. Zander. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: A case study of the Clark Fork River. Water Research 31: 1738-1750.

Durkin, P.R., 2011. Imazapyr: Human Health and Ecological Risk Assessment. Final Report. SERA TR-052-29-03a. Syracuse Environmental Research Associates for USDA-Forest Service, Southern Region. Manlis NY. 215 pp.

Durkin P and M. Follansbee. 2004. Clopyralid-revised human health and ecological risk assessment. Final Report. SERA TR-04-43-17-03c. Syracuse Environmental Research Associates for USDA-Forest Service, Southern Region. Manlis NY. http://www.fs.fed.us/foresthealth/pesticide/pdfs/120504_clopyralid.pdf accessed 4/17/2020.

Edwards, P.J. F. Wood, and R.L. Quinlivan, R.L. 2016. Effectiveness of best management practices that have application to forest roads: a literature synthesis. GTR-NRS-163. Northern Research Station, USDA-Forest Service, Newtown Square, PA, 2016; 171 pp.

Evans, J. G.H. Matschke, D.L. Campbell, P.L. Hegdal, and R.M. Engeman. 1990 Efficacy data for registration of strychnine grain baits to control pocket gophers (*Thomomys* spp.). Pp. 82 – 86 *In*: Davis, L.R. and R.E. Marsh (eds.). Proceedings of the 14th Vertebrate Pest Conference, March 6 – 8, 1990, Sacramento, CA. 349 pp.

EWEB (Eugene Water and Electric Board). 2017. EWEB's Drinking Water Source Protection 10-Year Strategic Plan (2018 – 2028). Eugene Water and Electric Board, Eugene, OR. 7 pp.

(ExToxNet) Extension Toxicology Network. 1993 Movement of pesticides in the environment. Toxicology Information Briefs. Revised 9/93. http://extoxnet.orst.edu/tibs/movement.htm accessed 4/2/2019.

Feller, M.C. 2009. Deforestation and nutrient loading to fresh waters. Pp. 29-44 *In*: Likens, G.E. (ed.) Encyclopedia of inland waters. Volume 1. Elsevier, Academic Press, New York, N.Y.

Feng, J.C. and S. Navratil. 1990. Sampling for zero-time hexazinone residues in forest soildissipation study. Can. J. For Res. 20: 1549-1552.

Fenner, K., S. Canonica, L.P. Wackett, and M. Elsner. 2013. Evaluating pesticide degradation in the environment: blind spots and emerging opportunities. Science, 341: 752-758.

Fishel, F. 2013. Pesticide Formulations. Publication #P1231. Agronomy Department, University of Florida, Institute of Food and Agricultural Sciences Extension. Gainesville, FL. 13 pp.

Flint, M.C., R. Harrison, B. Strahm, B.A. Adams. 2008. Nitrogen Leaching from Douglas-fir forests after urea fertilization. Journal of Environmental Quality 37: 1781-1788.

Fox, T.R., H.L. Allen, T.J. Albaugh, R. Rubilar, and C.A. Carlson. 2007. Forest fertilization and water quality in the United State. Better Crops 91: 7-9.

Gatz, L. 2019. Freshwater harmful algal blooms: Causes, challenges, and policy considerations. CRS Report R44871. Updated September 5, 2019. Congressional Research Service, Washington, D.C. 30 pp. https://crsreports.congress.gov/product/pdf/R/R44871 accessed 4/17/2020.

Gilliom, R.J., Barbash, J.E., Crawford, C.G., Hamilton, P.A., Martin, J.D., Nakagaki, N, Nowell, L.H., Scott, J.C., Stackelberg, P.E., Thelin, G.P., and Wolock, D.M., 2006, Pesticides in the nation's streams and ground water, 1992–2001: U.S. Geological Survey Circular 1291, The Quality of Our Nation's Waters, 184 p., https://pubs.er.usgs.gov/publication/cir1291.

Governor’s Office. 2020. Governor Kate Brown Announces Continued Agreement on Science-Based Forest Management. Press Release, March 31, 2020. https://www.oregon.gov/newsroom/Pages/NewsDetail.aspx?newsid=36287 accessed 4/21/2020.

Grandcoin, A., S. Piel and E. Baurès. 2017. AminoMethylPhosphonic acid (AMPA) in natural waters: Its sources, behavior and environmental fate. Water Research 117: 187-197.

Gravelle, J.A., G. Ice, T.E. Link, and D.L. Cook. 2009. Nutrient concentration dynamics in an inland Pacific Northwest watershed before and after timber harvest. Forest Ecology and Management 257: 1663-1675.

Greathouse, E.A., J.E. Compton, and J. Van Sickle. 2014. Linking landscape characteristics and high stream nitrogen in the Oregon Coast Range: Red alder complicates use of nutrient criteria. JAWRA Journal of the American Water Resources Association 50: 1383-1400.

Handcock, J. 2018. WSDA Report: Assessment of Herbicides in Surface Water in the Hoh River Watershed. AGR PUB 103-686 (N/6/18) Washington State Department of Agriculture, Olympia, WA. 28 pp. https://agr.wa.gov/FP/Pubs/docs/103-686_Hoh_Herbicide_Assessment.pdf. Accessed 4/12/2019.

Hanson, B.; Bond, C.; Buhl, K.; Stone, D. 2015. Pesticide Half-life Fact Sheet; National Pesticide Information Center, Oregon State University Extension Services. http://npic.orst.edu/factsheets/half-life.html. Accessed 4/2/2019.

Hapke W.B., Morace J.L., Nilsen E.B., Alvarez D.A., Masterson K. 2016. Year-Round Monitoring of Contaminants in Neal and Rogers Creeks, Hood River Basin, Oregon, 2011-12, and Assessment of Risks to Salmonids. PLoS ONE 11(6): e0158175. doi:10.1371/journal.pone.0158175

Hartzler, B., 2020. Role of spray adjuvants with postemergence herbicides. Extension and Outreach, Integrated Crop Management, Iowa State University, Ames IA.

https://crops.extension.iastate.edu/encyclopedia/role-spray-adjuvants-postemergence-herbicides accessed 4/17/2020.

Janney, P. and J. Jenkins. 2019. A systems approach to modeling watershed ecohydrology and pesticide transport. J. Env. Qual. 48(4): 1047-1056.

Jarvis, J.N., S. Almqvist, J. Stenström, E. Börjesson, E. Jonsson, and L. Torstensson. 2006. Modelling the leaching of imazapyr in a railway embankment. Pest management science 62: 940-946.

Jenkins, J. 2004. Environmental Monitoring of Chlorpyrifos and Azinphos-methyl Dissolved Residues in Hood River Tributaries for Years 2002 and 2003. Final Report to Hood River Soil and Water Conservation District. Agricultural Chemistry Research and Extension, Department of Environmenatl and Molecular Toxicology, Oregon State University, Corvallis, OR. 27 pp.

Johnson, H.M., J.L. Domagalski, and D.K. Saleh. 2011. Trends in pesticide Concentrations in streams of the western United States, 1993-2005. JAWRA Journal of the American Water Resources Association 47: 265-286.

Kachadoorian, R. 2019. Oregon Department of Agriculture (ODA) Pesticides Program. Presentation to the Pesticide Analytical and Response Center (PARC) Board Meeting September 18, 2019, Salem, OR. 21 pp. https://www.oregon.gov/ODA/programs/Pesticides/Documents/PARCMaterials/September2019/KachadoorianPresentation.pdf accessed 4/20/2020.

Karickhoff, S.W. 1981. Semi-empirical estimation of sorption of hydrophobic pollutants on natural sediments and soils. *Chemosphere* 10(8): 833-846.

Kelly, V.J., C.W. Anderson, and K. Morgenstern. 2012. Reconnaissance of land-use sources of pesticides in drinking water, McKenzie River, Oregon: U.S. Geological Survey Scientific Investigations Report 2012-5091. 46 pp.

Kelpsas, B. and C. Landgren. 2019. Forestry and Hybrid Cottonwood. *In*: Peachey, e. (ed.), *Pacific Northwest Weed Management Handbook* [online]. Revised March, 2019. Corvallis, OR: Oregon State University. https://pnwhandbooks.org/sites/pnwhandbooks/files/weed/chapterpdf/weed19-mforestry.pdf. Accessed 4/23/2019.

Kitanidis, P.K., 1997. Introduction to geostatistics: applications to hydrogeology. Cambridge University Press, Cambridge. 269 pp.

Kreutzweiser, D.P., S.B. Holmes, D.C. Eichenberg. 1994. Influence of exposure duration on the toxicity of triclopyr ester to fish and aquatic insects. Archives of Environmental Contamination and Toxicology 26: 124-129.

Lautenschlager, R.A., and T.P. Sullivan. 2004. Improving research into effects of forest herbicide use on biota in northern ecosystems. Wildlife Society Bulletin 32(4): 1061-1070.

Lee, C.H., P.C. Oloffs, and S.Y. Szeto. 1985, Persistence, degradation, and movement of triclopyr and its ethylene glycol butyl ether ester in a forest soil. J. Agric. Food Chem. 34: 1075-1079.

Lewis, K.A., Tzilivakis, J., Warner, D. and Green, A. (2016). An international database for pesticide risk assessments and management. Human and Ecological Risk Assessment: An International Journal, 22(4), 1050-1064. http://dx.doi.org/10.1080/10807039.2015.1133242.

Louch, J., V. Tatum, G. Allen, V.C. Hale, J. McDonnell, R.J. Danehy, and G. Ice. 2017. Potential risks to freshwater aquatic organisms following a silvicultural application of herbicides in Oregon's Coast Range. Integrated Environmental Assessment and Management 13: 396-409.

Lym, R.G. and O.R. Swenson. 1991. Sulfometuron persistence and movement in soil and water in North Dakota. J. Environ. Quality. 20: 209-215.

Masterson, K. and J. Crown. 2015. 2014 Hood River Water Quality Pesticide Monitoring Summary Report. Part I: Grab Sample Monitoring Results. Oregon Department of Environmental Quality, Portland, OR. 31 pp.

Matos, A.T., A.P. Pinho, L.M. Costa, and L.A. Morris. 2008. Streamside management zone (SMZ) efficiency in herbicide retention from simulated surface flow. Planta Daninha 26, 131-142.

McBroom, M.W., J. Louch, R.S. Beasley, M. Chang, G.G. Ice. 2013. Runoff of silvicultural herbicides applied using best management practices. Forest Science 59: 197-210.

McCarthy, K.A. and D.A. Alvarez. 2014. Time-integrated passive sampling as a complement to conventional point-in-time sampling for investigating drinking-water quality, McKenzie River Basin, Oregon, 2007 and 2010–11: U.S. Geological Survey Scientific Investigations Report 2013–5215, 14 pp.

McCarthy, K.A., D.A. Alvarez, and J.H. Goldman. 2012, Water-quality data from semipermeable-membrane devices and polar organic chemical integrative samplers deployed in the McKenzie River basin, Oregon: U.S. Geological Survey Data Series 692, 6 pp.

McCarthy, K.A., D. Alvarez, C.W. Anderson, W.L. Cranor, S.D. Perkins, and V. Schroeder. 2009. Evaluation of passive samplers for long-term monitoring of organic compounds in the untreated drinking water supply for the City of Eugene, Oregon, September–October 2007: U.S. Geological Survey Scientific Investigations Report 2009–5178, 20 pp.

Meals, D.W., R.P. Richards1, and S.A. Dressing. 2013. Pollutant load estimation for water quality monitoring projects. Tech Notes 8, April 2013. Developed for U.S. Environmental Protection Agency by Tetra Tech, Inc., Fairfax, VA, 21 p. https://www.epa.gov/polluted-runoff-nonpoint-source-pollution/nonpoint-source-monitoring-technical-notes accessed 4/21/2020.

Michael, J.L., 2003. Environmental fate and impacts of sulfometuron on watersheds in the southern United States Journal of Eenvironmental Quality 32: 456-465.

Michael, J.L., 2004. Best management practices for silvicultural chemicals and the science behind them. Water, Air, and Soil Pollution: Focus 4: 95-117.

Michael, J.L. and D.G. Neary. 1993. Herbicide dissipation studies in southern forest ecosystems. Environ. Toxicol. Chem. 12: 405-410.

Michael, J.L., D.P. Batzer, J.B. Fischer, and H.L. Gibbs. 2006. Fate of the herbicide sulfometuron methyl (Oust®) and effects on invertebrates in drainages of an intensively managed plantation. Canadian Journal of Forest Research 36: 2497-2504.

Michael, J.L., J.E.C. Webber, D.R. Bayne, J.B. Fischer, H.L. Gibbs, and W.C. Seesock. 1999. Hexazinone dissipation in forest ecosystems and impacts on aquatic communities. Canadian Journal of Forest Research 29, 1170-1181.

Morgenstern, K. 2017. Informational update on upcoming selective timber harvest activities. Memorandum from Karl Morgenstern to the EWEB Board, Leaburg Forest Management, June 23, 2017. Eugene Water and Electric Board, Eugene, OR. 4 pp

Morgenstern, K. 2014. Protecting the McKenzie Watershed: Eugene's Sole Source of Drinking Water. Presentation to the AWWA/PNWS Meeting, Eugene, OR. May, 2014.

Morgenstern, K., N. Toth, and D. Donahue. 2017. Strategic Planning Technical Report, Drinking Water Source Protection Program (2018 – 2028). Eugene Water and Electric Board (EWEB), Eugene, OR. 176 pp.

Morgenstern, K., D. Donahue, and N. Toth. 2011. McKenzie River Watershed Baseline Monitoring Report, 2000 to 2009. Eugene Water and Electric Board (EWEB), Eugene, OR. 114 pp.

MSA (Murry, Smith & Associates, Inc.). 2002. Water Source/Supply Facility Plan for the City of Sheridan. Including Addendum No. 1, July 2002. Murray, Smith & Associates, Inc. Engineers/Planners, Portland, OR. 14 pp.

Nason, G.E. and D.D. Myrold. 1992. Nitrogen fertilizers: Fates and environmental effects in forests. Pp. 67-81 in H.N. Chappell, G.F. Weetman, and R.E. Miller (eds.). Forest Fertilization: Sustaining and improving nutrition and growth of western forests. Contribution No. 73. Institute of Forest Resources, University of Washington, Seattle, WA.

(NCASI). National Council for Air and Stream Improvement, Inc. 2013. Measurement of glyphosate, imazapyr, sulfometuron methyl, and metsulfuron methyl in in Needle Branch streamwater. Special Report No. 13-01. Research Triangle Park, NC: National Council for Air and Stream Improvement. 130 pp.

(NCBI) National Center for Biotechnology Information. 2020. PubChem Database. 2,4-Dichlorophenoxyacetic acid, CID=1486, https://pubchem.ncbi.nlm.nih.gov/compound/2_4-Dichlorophenoxyacetic-acid (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020a. PubChem Database. Imazapyr, CID=54738, https://pubchem.ncbi.nlm.nih.gov/compound/Imazapyr (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020b. PubChem Database. Sulfometuron-methyl, CID=52997, https://pubchem.ncbi.nlm.nih.gov/compound/Sulfometuron-methyl (accessed on Apr. 17, 2020)

(NCBI) National Center for Biotechnology Information. 2020c. PubChem Database. Triclopyr, CID=41428, https://pubchem.ncbi.nlm.nih.gov/compound/Triclopyr (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020d. PubChem Database. Atrazine, CID=2256, https://pubchem.ncbi.nlm.nih.gov/compound/Atrazine (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2002e. PubChem Database. Glyphosate, CID=3496, https://pubchem.ncbi.nlm.nih.gov/compound/Glyphosate (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020f. PubChem Database. Hexazinone, CID=39965, https://pubchem.ncbi.nlm.nih.gov/compound/Hexazinone (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020g. PubChem Database. Metsulfuron-methyl, CID=52999, https://pubchem.ncbi.nlm.nih.gov/compound/Metsulfuron-methyl (accessed on Apr. 17, 2020).

(NCBI) National Center for Biotechnology Information. 2020h. PubChem Database. Propiconazole, CID=43234, https://pubchem.ncbi.nlm.nih.gov/compound/Propiconazole (accessed on Apr. 17, 2020).

(NPIC) National Pesticide Information Center, 2019. Active Ingredient Fact Sheets. Oregon State University, Corvallis, OR. ww.npic.orst.edu/ingred/specchem.html.

(NPIC) National Pesticide Information Center. 2020. Herbicide Properties Tool. Oregon State University, Corvallis, OR. http://npic.orst.edu/HPT/ accessed 4/16/2020.

Neter, J., M.H. Kutner, C.J. Nachtsheim, and W. Wasserman. 1996. Applied linear statistical models. WCB McGraw-Hill, Boston. 735 pp.

Norman, J.E., P.L. Toccalino, and S.A. Morman. 2018. Health-based screening levels for evaluating water-quality data (2d ed.). U.S. Geological Survey web page, accessible at https://water.usgs.gov/nawqa/HBSL, doi:10.5066/F71C1TWP.

(NWQMC) National Water Quality Monitoring Council. 2020a. Polar Organic Chemical Integrative Sampler. National Environmental Methods Index. https://www.nemi.gov/methods/method_summary/10622/ accessed 4/17/2020.

(NWQMC) National Water Quality Monitoring Council. 2020b. Semipermeable Membrane Device. National Environmental Methods Index. https://www.nemi.gov/methods/method_summary/9973/ accessed 4/17/2020.

(ODA) Oregon Department of Agriculture. 2008. Pesticide Use Reporting System 2007 Annual Report. July, 2008 Salem, OR. 46 pp.

(ODA) Oregon Department of Agriculture. 2009. Pesticide Use Reporting System 2008 Annual Report. June, 2008 Salem, OR. 37 pp.

(OHA) Oregon Health Authority. 2014. Public Health Assessment Highway 36 Corridor Exposure Investigation. Final. Environmental Health Assessment Program, Public Health Division. Salem OR. 138 pp.

Ongley, E.D. 1996. Control of water pollution from agriculture. FAO Irrigation and Drainage Paper 55, Chapter 4, Pesticides as water pollutants. U.N. Food and Agriculture Organization, Rome, Italy. http://www.fao.org/3/w2598e/w2598e07.htm. Accessed 4/1/2019.

PARC (Pesticide Analytical and Response Center). 2018. Coordinating Oregon's Response to Pesticide Incidents. Version 4.0, March, 2018. Prepared by the Pesticide Analytical and Response Center Board, Oregon Department of Agriculture, Salem, OR. 101 pp.

Paranjape, K., V. Gowariker, V.N. Krishnamurthy, and S. Gowariker. 2015. Sulfometuron methyl herbicide. *In*: Paranjape, K., V. Gowariker, V.N. Krishnamurthy, V.N., and S. Gowariker, S. (eds.). *The Pesticide Encyclopedia*. CABI: Croydon UK. 726 pp.

Penner, D. 2000. Activator adjuvants. Weed Technology 14: 785-791.

Perakis, S.S., and E.R. Sinkhorn. 2011. Biogeochemistry of a temperate forest nitrogen gradient. Ecology 92: 1481-1491.

Perry, S., and C. Randall (eds.). 2000. Forest Pest Management: A Guide for Commercial Applicators Category 2. Extension Bulletin E-2045. Integrated Pest Management, Michigan State University, East Lansing, MI. 111 pp. https://www.canr.msu.edu/resources/forest_pest_management accessed 4/17/2020.

Persad, A.S., J. Cowden, A.K. Hotchkiss, C. Keshava, J.S. Lee, A. Marcus, A. Rooney, and R. Sams. 2011. Toxicological review of urea. EPA/635/R-10/005F Integrated Risk Information System (IRIS), U.S. Environmental Protection Agency, Washington DC. 96 pp. https://cfpub.epa.gov/ncea/iris/iris_documents/documents/toxreviews/1022tr.pdf accessed 4/17/2020.

Pike, R.G., M.C. Feller, J.D. Stednick, K.J. Rieberger, and M. Carver. 2010. Water quality and forest management. Pp. 401-440 *In*: Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winker and K.D. Bladon (editors). Compendium of forest hydrology and geomorphology in British Columbia. B.C. Min. For. Range, For. Sci. Prog., Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Manag. Handb. 66. www.for.gov.bc.ca/hfd/pubs/Docs/Lmh/Lmh66.htm accessed 4/17/2020.

Pinho, A.P., L.A. Morris, C.R. Jackson, W.J. White, P.B. Bush, and A.T. Matos. 2008. Contaminant retention potential of forested filter strips established as SMZs in the Piedmont of Georgia. JAWRA Journal of the American Water Resources Association 44: 1564-1577.

Poor, C.J., and J.J. McDonnell. 2007. The effects of land use on stream nitrate dynamics. Journal of Hydrology 332: 54-68.

Popovska-Gorevski, M., M.L. Dubocovich, and R.V. Rajnarayanan. 2017. Carbamate insecticides Ttarget human melatonin receptors. Chemical Research in Toxicology 30: 574-582.

Putney, J. 2019. Assessing mechanisms of Douglas-fir growth response to operational nitrogen fertilization in western Oregon. MS Thesis. Department of Forest Engineering, Resources, and Management. Oregon State University, Corvallis, OR. 203 pp.

Rahman, A., T.K. James, and P. Sanders, P. 1993. Leaching and movement of imazapyr in different soil types. pp. 115–119 *In*: Proc. Forty-Sixth New Zealand Plant Protection Conference, Christchurch, New Zealand, 10–12 August 1993, New Zealand Plant Protection Society, Rotorua, New Zealand.

Roy, D. N., S.K. Konar, D.A. Charles, J.C. Feng, R. Prasad, and R.A. Campbell. 1989. Determination of persistence, movement, and degradation of hexazinone in selected Canadian boreal forest soils. J. Agric. Food Chem. 37: 443-447.

Scarbrough, S.L., C.R. Jackson, S. Marchman, G. Allen, J. Louch, and M. Miwa. 2015. Herbicide concentrations in first-order streams after routine application for competition control in establishing pine plantations. Forest Science 61: 604-612.

Scatena, F.N. 2000. Drinking water quality. Pp. 7-25 *In*: Dissmeyer, G.E. (ed.) Drinking water from forests and grasslands: a review of the scientific literature. Gen. Tech. Rep. SRS-39. USDA-Forest Service, Southeast Forest Experiment Station, Asheville, N.C. 255 pp.

Shipitalo, M.J., W.A. Dick, and W.M. Edwards. 2000. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. Soil and Tillage Research 53: 167-183.

Sigleo, A.C., W.E. Frick, and L. Prieto. 2010. Red alder (*Alnus rubra*) distribution influences nitrate discharge to coastal estuaries: Comparison of two Oregon watersheds. Northwest Science 84(4): 336-350.

Slesak, R.A., S.H. Schoenholtz, T.B. Harrington, and B.D. Strahm. 2009. Dissolved carbon and nitrogen leaching following variable logging-debris retention and competing-vegetation control in Douglas-fir plantations of western Oregon and Washington. Canadian Journal of Forest Research 39: 1484-1497.

Stednick J.D. (2008) Long-term Water Quality Changes Following Timber Harvesting. Pp. 157-170 *In*: Stednick J.D. (ed.) *Hydrological and Biological Responses to Forest Practices*. Ecological Studies, vol 199. Springer, New York, NY. 330pp.

Stephenson, G.R., K.R. Solomon, C.S. Bowhey, and K. Liber. 1990. Persistence, leachability, and lateral movement of triclopyr (Garlon) in selected Canadian forestry soils. J. Agric. Food Chem. 38: 584-588.

Sundaram, K.M.S., and S.Y. Szeto. 1987. Distribution and persistence of carbaryl in some terrestrial and aquatic components of a forest environment. Journal of Environmental Science and Health, Part B 22: 579-599.

Tatum, V.L., 2004. Toxicity, transport, and fate of forest herbicides. Wildlife Society Bulletin 32, 1042-1048.

Tatum, V.L., C.R. Jackson, M.W. McBroom, B.R. Baillie, E.B. Schilling, and T.B. Wigley. 2017. Effectiveness of forestry best management practices (BMPs) for reducing the risk of forest herbicide use to aquatic organisms in streams. Forest Ecology and Management 404: 258-268.

Temple, W.B., and H.M. Johnson. 2011, Occurrence and distribution of pesticides in surface waters of the Hood River basin, Oregon, 1999–2009. U.S. Geological Survey Scientific Investigations Report 2011–5082. 84 pp.

Thistle, H.W., G.G. Ice, R.L. Karsky, A.J. Hewit, and G. Door. Deposition of aerially applied spray to a stream within a vegetative barrier. Transactions of the ASABE. 52(5): 1481-1490.

TNC (The Nature Conservancy). 2017. Hope on the Hoh: 10,000 acres for recreation and conservation. Press Release, May 11, 2017, The Nature Conservancy in Washington, Seattle, WA. http://www.washingtonnature.org/fieldnotes/hoh-announcement-nature-conservancy-hoh-river-trust. Accessed 4/12/2019.

Trent, A., D. Nolte, and K. Wagner. 2001. Comparison of commercial deer repellents. Tech Tip 0124-2331-MTDC. U.S.D.A.-Forest Service, Missoula Technology and Development Center, Missoula, MT. 6 pp.

Trumbo, J., and D. Waligora. 2009. The impact of the herbicides imazapyr and triclopyr triethylamine on bullfrog tadpoles. California Fish and Game 95: 122-127.

Tu, M., C. Hurd, and J.M. Randall. 2001. *Weed Control Methods Handbook*. Wildland Invasive Species Team, The Nature Conservancy, University of California, Davis, CA. 219 pp.

UC-IPM (University of California – Integrated Pest Management). 2019. Herbicide Symptoms Modes of Action [database]. University of California, Davis, Statewide Integrated Pest Management Program [database]. http://herbicidesymptoms.ipm.ucanr.edu/MOA/. Accessed 4/26/2019.

USEPA 1993. Reregistration Eligibility Decision (RED) Facts – Glyphosate. EPA-738-F-93-011. Office of Prevention, Pesticides, and Toxic Substances, U.S. Environmental Protection Agency, Washington, D.C. 8 pp.

(USEPA) U.S. Environmental Protection Agency. 1998. Reregistration Eligibility Decision - Triclopyr. EPA 738-R-98-011. Office of Prevention, Pesticides, and Toxic Substances, U.S. Environmental Protection Agency, Washington, D.C. March 31, 2004. 286pp. https://archive.epa.gov/pesticides/reregistration/web/pdf/2710red.pdf accessed 4/17/2020.

(USEPA) U.S. Environmental Protection Agency. 2004a. Carbaryl IRED Facts. Office of Prevention, Pesticides and Toxic Substances, Washington DC, p. 318. https://archive.epa.gov/pesticides/reregistration/web/pdf/carbaryl_ired.pdf accessed 4/17/2020.

(USEPA) U.S. Environmental Protection Agency, 2016. Carbaryl Hazard Summary 63-52-2 Updated January, 2000. In. Environmental Protection Agency, Washington DC, p. 4.

(USEPA) U.S. Environmental Protection Agency. 2004b. Pesticide Fact Sheet: Mesosulfuron-methyl Conditional Registration. Office of Prevention, Pesticides, and Toxic Substances, U.S. Environmental Protection Agency, Washington, D.C. March 31, 2004. https://www3.epa.gov/pesticides/chem_search/reg_actions/registration/fs_PC-122009_31-Mar-04.pdf accessed 4/17/2020.

(USEPA) U.S. Environmental Protection Agency. 2005. Pesticide Fact Sheet: Aminopyralid Conditional Registration. Office of Prevention, Pesticides, and Toxic Substances, U.S. Environmental Protection Agency, Washington, D.C. August 10, 2005. https://www3.epa.gov/pesticides/chem_search/reg_actions/registration/fs_PC-005100_10-Aug-05.pdf. Accessed 4/28/2019.

(USEPA) U.S. Environmental Protection Agency. 2014. Registration Review – Preliminary Problem Formulation for the Ecologycal Risk Assessment and Drinking Water Exposure Assessment to be Conducted for Imazapyr and Imazapyr Isopropylamine. Memorandum of March 10, 2014 from Environmental Fate and Effects Division to Pesticide Reevaluation Division.46 pp. http://npic.orst.edu/HPT/refs/imazapyr_epadoc1.pdf. Accessed 5/7/2019.

(USEPA) U.S. Environmental Protection Agency. 2015. Algal Toxin Risk Assessment and Management Strategic Plan for Drinking Water. Strategy Submitted to Congress to Meet the Requirements of P.L. 114-45. 810R04003. November, 2015. 81 pp. https://www.epa.gov/sites/production/files/2015-11/documents/algal-risk-assessment-strategic-plan-2015.pdf. Accessed 4/19/2019.

(USEPA) U.S. Environmental Protection Agency. 2017. Fact Sheet: Human Health Benchmarks for Pesticides – 2017 Update. EPA 822-F-17-001. Office of Water, U.S. Environmental Protection Agency, Washington, D.C. https://www.epa.gov/sites/production/files/2015-10/documents/hh-benchmarks-factsheet.pdf. Accessed 3/26/2019.

(USEPA) U.S. Environmental Protection Agency.2018. 2018 Edition of the Drinking Water Standards and Health Advisories. Office of Water, U.S. Environmental Protection Agency, Washington, D.C. https://www.epa.gov/sites/production/files/2018-03/documents/dwtable2018.pdf. Accessed 3/26/2019.

Vasilakoglou, I.B., I.G. Eleftherohorinos, and K.B. Dhima. 2001. Activity, adsorption and mobility of three acetanilide and two new amide herbicides. Weed Research 41, 535-546.

Vernier, F., O. Leccia-Phelpin, J.-M. Lescot, S. Minette, A. Miralles, D. Barberis, C. Scordia, V. Kuentz-Simonet, and J.-P. Tonneau. 2017. Integrated modeling of agricultural scenarios (IMAS) to support pesticide action plans: the case of the Coulonge drinking water catchment area (SW France). Environ. Sci. Pollut. Res. 24: 6923-6950.

Walker, A. and S.J. Welch. 1989. The relative movement and persistence in soil of chlorsulfuron, metsulfuron-methyl and triasulfuron. Weed Research 29: 375-383.

Wang, R., Y. Yuan, H. Yen. M. Grieneisen, J. Arnold, D. Wang, C. Wang, M. Zhang. 2019. A review of pesticide fate and transport simulation at watershed level using SWAT: Current status and research concerns. Science of the Total Environment 669: 512-526.

Wauchope, R.D., T.M. Buttler, A.G. Hornsby, P.W.M. Augustinjin-Beckers, and J.P. Burt. 1992. The SCS/ARS/CES pesticide properties database for environmental decision-making. Reviews of Environmental Contamination and Toxicology 123: 1-164.

Weber, J.B., R.L. Warren, L.R. Swain, and F.H. Yelverton. 2007. Physicochemical property effects of three herbicides and three soils on herbicide mobility in field lysimeters. Crop Protection 26: 299-311.

Winchell, M.F., N. Peranginangin, R. Srinivasan, and W. Chen. 2018. Soil and Water Assessment Tool Model Predictions of Annual Maximum Pesticide Concentrations in High Vulnerability Watersheds. Health & Ecological Risk Assessment 14(3): 358-368.

Wise, D.R., and H.M. Johnson. 2011. Surface-water nutrient conditions and sources in the United States Pacific Northwest. JAWRA Journal of the American Water Resources Association 47(5): 1110-1135.

Wu, H., J.S. Kimball, M.M. Elsner, N. Mantua, R.F. Adler, and J. Stanford. 2012. Projected climate change impacts on the hydrology and temperature of Pacific Northwest rivers. Water Resources Research 48(11): W11530. DOI: 10.1029/2012WR012082

(WQA) Water Quality Association. 2013. Nitrate/Nitrite Fact Sheet. International Headquarters & Laboratory, Lisle, IL. 6 pp. https://www.wqa.org/Portals/0/Technical/Technical%20Fact%20Sheets/2014_NitrateNitrite.pdf accessed 4/17/2020.

Yen, H., P. Daggupati, M.J. White, R. Srinivasan, A. Gossel, D. Wells, and J.G. Arnold. 2016. Application of large-scale, multi-resolution watershed modeling framework using thee hydrology and water quality system (HAWQS). Water 8(4): 164. https://doi.org/10.3390/w8040164

6.9. Appendix Table 6-A. Mentions of chemicals in ODF Notifications of Operations (NOAPs).

Chemical	Common Name	Brand Name	Additive	Carrier	Family
2,4-D	131				2,4-D
2,4-D & dicamba, formulation:amine	1				Mixture
2,4-D & dicamba, formulation:ester	37				Mixture
2,4-D & Picloram, formulation:amine	35				Mixture
2,4-D Amine 4, formulation:amine		103			2,4-D
2,4-D LV4, formulation:ester		102			2,4-D
2,4-D LV6, formulation:ester		851			2,4-D
2,4-D, formulation:acid	1,126				2,4-D
2,4-D, formulation:amine	1,207				2,4-D
2,4-D, formulation:choline	608				2,4-D
2,4-D, formulation:ester	4,609				2,4-D
Aatrex 4L		38			Atrazine
AAtrex Nine-O		1			Atrazine
Accord Concentrate		283			Glyphosate
Accord XRT II		1,703			Glyphosate
Activator 90			358		Surfactant
AD-Wet 90 CA			60		Surfactant
Agri-Dex			9		Surfactant
Alligare 90 (non-ionic surfactant)			62		Surfactant
Amine 4 2,4-D Weed Killer, formulation:amine		5			2,4-D
Aminopyralid	166				Aminopyralid
Aminopyralid and Metsulfuron Methyl	1,891				Picolinic Acid/Sulfonylurea
AMS Complete			9		Buffer
Amtide MSM 60 DF		4			Metsulfuron Methyl
Anti-Foam 20			4		Anti-foaming Agent
Aquamaster		97			Glyphosate
Aquaneat		116			Glyphosate
Arsenal Applicator's Concentrate		261			Imazapyr
Atrazine	5,366				Atrazine
Atrazine 4L		1,061			Atrazine
Atrazine 90 DF		30			Atrazine
Atrazine 90 WDG		1			Atrazine
Atrazine 90 WSP		3			Atrazine
Base Camp LV6, formulation:ester		11			2,4-D
Bronc Max			1		Buffer
Buccaneer		5			Glyphosate
Capstone, formulation:amine		48			Mixture

Chemical	Common Name	Brand Name	Additive	Carrier	Family
Cayuse Plus			1		Buffer
Chopper Gen2		18			Imazapyr
Clasp			1		Deposition Aid
Clean Slate		31			Clopyralid
Cleantraxx		655			Oxyfluorfen/Penoxsulam
CLIMB alkalinity agent			275		Buffer
Clopyralid	8,149				Clopyralid
Clopyralid 3		26			Clopyralid
Compadre			865		Deposition Aid Agent
Competitor Modified Vegetable Oil			286		Surfactant
Complete Compatibility			4		Emulsifier
Conquer			1,902		Surfactant
Cornerstone		8			Glyphosate
Cornerstone Plus		26			Glyphosate
Credit 41		18			Glyphosate
Crop Oil Concentrate			1,066		Surfactant
Crosshair			6,796		Deposition Aid Agent
Decimal			4		Buffer
Derigo (experimental use permit only)		1			Metsulfuron?
Dicamba	52				Dicamba
Diesel Oil				259	Carrier
DMA4 IVM, formulation:amine		4			2,4-D
Dyne-Amic			1,375		Surfactant
Element 3A, formulation:amine		240			Triclopyr
Element 4, formulation:ester		654			Triclopyr
Epoleon N-100			46		Deodorizer
Epoleon N-7C			12		Deodorizer
Escort XP		560			Metsulfuron methyl
Esplanade F		132			Indaziflam
Fighter F 10			175		Anti-foaming Agent
Finale VU		3			Clopyralid
Fire Zone-Vegetable Oil			113		Surfactant
Flumioxazin	134				Flumioxazin
Fluroxypyr	105				Fluroxypyr
Foam Buster			745		Anti-foaming Agent
Foam Buster Max			149		Anti-foaming Agent

Chemical	Common Name	Brand Name	Additive	Carrier	Family
Foam Fighter			19		Anti-foaming Agent
Forest Crop Oil				3,785	Surfactant
Forester's Nonselective		3			Glyphosate
Forestry Garlon XRT, formulation:ester		386			Triclopyr
Freelexx, formulation:choline		6			2,4-D
FTF Defoamer			4		Anti-foaming Agent
Garlon 3A, formulation:amine		534			Triclopyr
Garlon 4 Ultra, formulation:ester		719			Triclopyr
Garlon 4, formulation:ester		847			Triclopyr
Gly Star 5 Extra		2			Glyphosate
Gly Star 5 Original		29			Glyphosate
Gly Star Original		46			Glyphosate
Gly Star Plus		28			Glyphosate
Glyfos		5			Glyphosate
Glyfos X-TRA		16			Glyphosate
Glyphosate	14,759				Glyphosate
Glyphosate 4 Plus		141			Glyphosate
Glyphosate 5.4		334			Glyphosate
Glyphosate, Diquat Dibromide & Indizaflam	3				Mixture
Grounded			2,373		Surfactant
Grounded-W			11		Surfactant
Hardball, formulation:acid		427			2,4-D
Hasten			470		Surfactant
Hel-Fire			2		Buffer
Herbimax			163		Surfactant
Hexazinone	6,481				Triazine
Hexazinone & Sulfometuron methyl	1,043				Mixture
Hi-Light Blue			1,100		Spray Indicator
Imazapyr	11,765				Imazapyr
Imazapyr & Metsulfuron Methyl	270				Mixture
Imazapyr 2SL		89			Imazapyr
Imazapyr 4SL		289			Imazapyr
Indaziflam	934				Indaziflam
Induce			1,352		Surfactant
Inlet			44		Surfactant
In-Place			48		Drift Inhibitor
Insist 90			186		Surfactant
Insist 90 Plus			14		Surfactant

Chemical	Common Name	Brand Name	Additive	Carrier	Family
InterActive			267		Surfactant
IVM Marking Dye			173		Spray Indicator
Kinetic			17		Surfactant
LI 700			390		Surfactant
Liberate			676		Surfactant
LO-VOL 6 2,4-D, formulation:ester		224			2,4-D
Mad Dog		1			Glyphosate
Mad Dog Plus		26			Glyphosate
Marking Dye 1 (blue)			45		Spray Indicator
Metcel VMF		54			Metsulfuron-methyl
Metsulfuron Methyl	4,310				Metsulfuron Methyl
Milestone		18			Aminopyralid
Milestone VM		3			Aminopyralid
Milestone VM Plus, formulation:amine		211			Triclopyr
Mineral Oil, Alkylphenol Polyoxyethylene			451		Surfactant
Mor-Act			186		Surfactant
Mor-Act Crop Oil Concentrate			32		Surfactant
MSM 60		116			Metsulfuron Methyl
MSM 60 DF		31			Metsulfuron Methyl
MSO Concentrate			9,074		Surfactant
MSO Concentrate with Leci-Tech			1,291		Surfactant
MSO-100			15		Surfactant
No Foam			950		Anti-foaming Agent
NU-FILM 17			4		Surfactant
NU-FILM IR			4		Surfactant
NU-Film P			3		Surfactant
Odor Mask			47		Deodorizer
Opensight		677			Mixture
Oust Extra		1,522			Mixture
Oust XP		993			Sulfometuron-Methyl
Outpost 22K		16			Picloram
Oxyfluorfen and Penoxsulam	753				Mixture
Payload		35			Flumioxazin
Petroleum oil, emulsifier				467	Surfactant
Petroleum Oil, Polyoxyethylene Sorbitan fatty acid ester				502	Surfactant

Chemical	Common Name	Brand Name	Additive	Carrier	Family
Phase			326		Surfactant
Picloram	45				Picolinic Acid
Polaris AC Complete		914			Imazapyr
Polaris SP		437			Imazapyr
Power-Line Methoil (MSO)			14		Surfactant
Premium MSO			52		Surfactant
Pyraflufen Ethyl	28				Pyraflufen Ethyl
Quest			1		Buffer
R-11			106		Surfactant
Rainier-EA			1		Surfactant
Ranger Pro		1			Glyphosate
Razor		7			Glyphosate
Razor Pro		2			Glyphosate
Renegade 2.0			1		Surfactant
Renegade EA			6		Surfactant
Rodeo		679			Glyphosate
Rotary 2 SL		221			Imazapyr
Roundup Custom		31			Glyphosate
Roundup Pro		93			Glyphosate
Roundup Pro Concentrate		179			Glyphosate
Roundup ProMax		10			Glyphosate
RRSI NIS surfactant			89		Surfactant
Saflufenacil	142				Saflufenacil
SFM 75		129			Sulfometuron-Methyl
SFM Extra		102			Sulfometuron-Methyl
Silwet L-77			24		Surfactant
Solve 2,4-D, formulation:ester		3			2,4-D
Spray Indicator			602		Spray Indicator
Spyder		35			Sulfometuron-Methyl
Spyder Extra		47			Mixture
Sta-Put			25		Drift Inhibitor
Sta-Put Plus			12		Drift Inhibitor
Strike Zone DC			1		Drift Inhibitor
Sulfomet		291			Sulfometuron-Methyl
Sulfomet Extra		150			Mixture
Sulfomet XP		129			Sulfometuron-Methyl

Chemical	Common Name	Brand Name	Additive	Carrier	Family
Sulfometuron Max		37			Sulfometuron-Methyl
Sulfometuron Methyl	8,674				Sulfometuron Methyl
Sulfometuron Methyl & Metsulfuron Methyl	3,927				Mixture
Super Marking Dye 1 (blue)			74		Spray Indicator
Super Spread 90			16		Surfactant
Super Spread MSO			1,208		Surfactant
Surface (non-ionic surfactant)			90		Surfactant
Syl-Coat			132		Surfactant
Sylgard 309			556		Surfactant
Syl-Tac			2,039		Surfactant
Syl-Tac EA			28		Surfactant
Tahoe 3A, formulation:amine		1			Triclopyr
Tahoe 4E, formulation:ester		3			Triclopyr
Topramezone	188				Topramezone
Tordon 22K		46			Picloram
Tordon K		2			Picloram
Tordon RTU, formulation:amine		6			Mixture
Transline		2,239			Clopyralid:
Triclopyr	273				Triclopyr
Triclopyr & Aminopyralid, formulation:amine	393				Mixture
Triclopyr 3A, formulation:amine		132			Triclopyr
Triclopyr 4E, formulation:ester		88			Triclopyr
Triclopyr, formulation:acid	1,467				Triclopyr
Triclopyr, formulation:amine	2,494				Triclopyr
Triclopyr, formulation:ester	6,425				Triclopyr
Triclopyr, formulation:choline	612				Triclopyr
Trycera, formulation:acid		7			Triclopyr
Vanquish		4			Dicamba
Vastlan, formulation:choline		63			Triclopyr
Velossa		671			Hexazinone
Velpar DF		1,217			Hexazinone
Velpar L		268			Hexazinone
Vista		5			Fluroxypyr
Vista XRT		64			Fluroxypyr
W.E.B. Oil				2,757	Carrier
Weedmaster, formulation:amine		19			Mixture
Weedone LV4 EC, formulation:ester		3			2,4-D
Weedone LV6 EC, formulation:ester		53			2,4-D

Chemical	Common Name	Brand Name	Additive	Carrier	Family
Westar		191			Mixture
Widespread Max			3		Surfactant
Grand Total	**88,603**	**23,506**	**41,187**	**5,974**	

Chapter 7. Natural Organic Matter and Disinfectant Byproducts

Bogdan Strimbu, Jeff Behan, and Jon Souder

The relationship between natural organic matter (NOM) and disinfection byproducts (DPB) is important because two DPBs, total haloacetic acids (HAA5) and total trihalomethanes (TTHM), are regulated by the U.S.E.P.A. under the Safe Drinking Water Act. These DPBs are created when carbon in water comes into contact with the chlorine disinfectant that is required to remain as residual throughout a water utility's distribution system until the water comes out the tap. The carbon can be from natural sources, can result from human activities, may be added during water treatment, and may be formed through the disinfection process in the treatment plant. In this chapter we will focus on that fraction of carbon from natural sources, and additional inputs that may result from active forest management.

The chapter will begin with an overview of natural organic matter in water, its various sources, how it is distinguished between particulate organic matter (POM) which is defined as > 45 microns in size, and dissolved organic matter (DOM), and its effects on the water treatment process. The next section will focus on the chemical characteristics of NOM, how it is analyzed, the size and molecular structure of DOM, and the characteristics of POM. After this, we will provide an overview of forest management effects on NOM based on the scientific literature review, specifically discussing forest harvesting, forest roads, and natural disturbances such as wildfire and bark beetle infestations. The fourth section will cover the effects of NOM on potable water treatment. The chapter will conclude with an analysis of DBP detections in finished water to provide a context to better understand the relationship between NOM in source water and DBPs in finished, potable water.

7.1. Sources, classification, and treatment issues

Natural organic matter (NOM) is defined as non-living organic molecules found in the environment in soil, sediments and water. Natural organic matter is a product of mostly plant but also animal tissue decay and plays a pivotal role in the carbon cycle (Nebbioso and Piccolo, 2013). Living matter is mostly composed of well-defined molecules such as proteins, nucleic acids, lipids, sugars and cellulose. In contrast, NOM is mostly composed of molecules of unknown structure. There is an incipient paradigm shift in the soil science community, which includes the study of NOM, and is based on the observation that the deeper we look the more components we find, and therefore complexity increases to level that entities such as humus, humin, humic acid or fulvic acids become general terms rather than some specific forms of organized matter (Schmidt et al. 2011; Marín-Spiotta et al. 2014). Despite these challenges NOM has been extensively researched because of its ecological and geochemical importance and influences on pollutant fate and transport in the environment.

A key consideration for drinking water providers is identifying sources of, and reducing the quantity of NOM that arrives at their raw water intakes. Prior to the early 1970's, treatment of NOM in raw water focused on aesthetic issues such as color. Then, research demonstrated that NOM is a precursor constituent in the formation of hazardous disinfection byproducts (DBPs). NOM from forest detritus is a major precursor to DBPs in drinking water sources (Bhardwaj 2006). Thus, forest management activities that influence the quantity and mobility of this source of NOM in source waters can influence the potential for DBPs to form during water treatment (Majidzadeh et al. 2019).

Expanded understanding of linkages between NOM and DBPs continues to spur changes in drinking water treatment and regulation. (O'Melia 2006.)

Control of environmental processes defining the NOM transformation is achievable if knowledge about its molecular composition is available (Piccolo 2012). Molecular characterization of NOM is a primary research objective in environmental and ecological chemistry (Nebbioso and Piccolo 2011). Recent developments in material sciences and the IT industry, such as nuclear magnetic resonance spectroscopy, or X-ray spectroscopic methods, allow detection and characterization of single organic compounds and, sometimes, homogeneous mixtures. However, because of the heterogeneous composition of NOM its description is still a challenge.

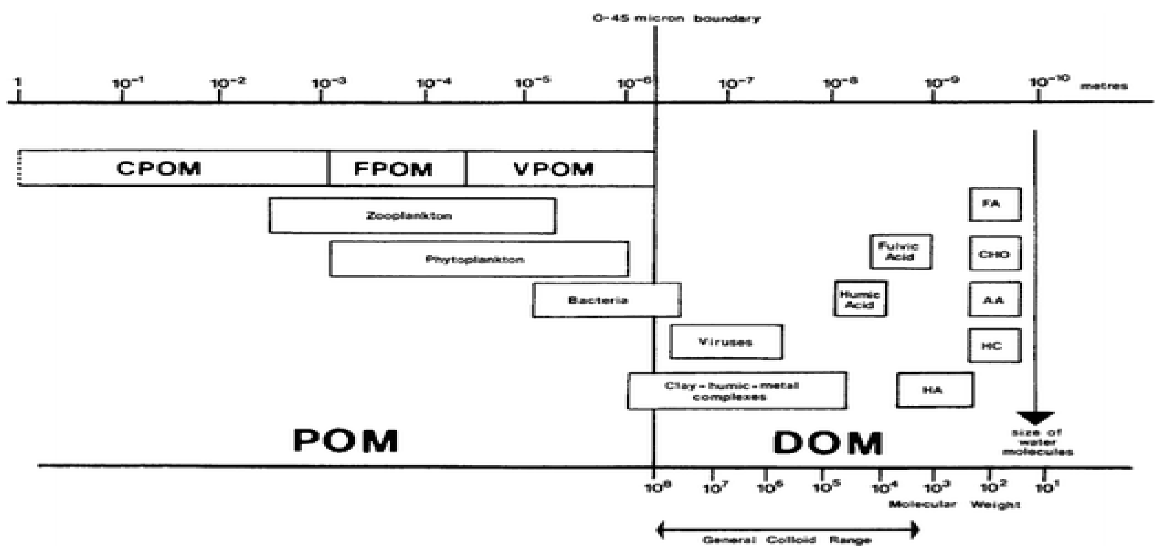
Natural organic matter in water is either formed in a water body, when it is called autochthonous, or is formed outside of the water body then transported into it when it is called allochthonous (Wershaw et al. 2005). For a particular water body, the composition of allochthonous NOM depends on the composition of the living matter compounds from which it originates, and on the natural diagenetic processes that alter the composition of the mixture of precursors (Wershaw 2004). As plant and animal tissue degrade in natural systems, soluble organic compounds are leached. These organic compounds are the source of the NOM. Intuitively, the chemical composition of NOM should reflect the composition of the plant and animal tissues from which it is derived (Leenheer et al. 2003; Wershaw et al. 2003; Leenheer et al. 2004). Autochthonous NOM is generally formed by microorganisms living in a water body. However, diagenetic processes (physical and chemical changes that alter the characteristics of sediment after deposition) in aquifers) can also alter the composition of NOM. The most common diagenetic process impacting NOM composition is sorption occurring on mineral surfaces and microbial degradation (Wershaw et al. 2005). Not surprisingly Wershaw et al. (1995; 1996a; 1996b) found that organic acids in NOM are strongly adsorbed by alumina (aluminum oxide). Furthermore, the NOM components forming complexes with alumina are preferentially adsorbed. Furthermore, besides sorption of NOM, the microorganisms metabolize plant-derived NOM and produce new types of compounds; such as the colloidal fraction of NOM from the Great Salt Lake (Leenheer et al. 2004).

Operationally, NOM is separated in two components: dissolved organic matter (DOM) and particulate organic matter (POM) (Thurman 1985). Because it is usually a fine colloidal suspension, DOM cannot be correctly regarded as a chemical solution. DOM and its composition are important in natural-water ecosystems because of the number of processes in which it becomes involved. The DOM acts as a strong chelating agent for metals, thus affecting their solubility, transport, and toxicity (Schnitzer 1972). Dissolved organic matter is fundamentally involved in the transport of organic pollutants (Carter and Suffet 1982), formation of colloidal particles (Tipping, 1986), aqueous photochemical reactions (Zafiriou et al. 1984), nutrient cycling and availability (Sanderman and Kramer 2013; Carlson and Hansell 2015; Wymore et al. 2015), and pH-buffering (Oliver et al. 1983). Dissolved organic matter and particulate organic matter are also important sources of energy in river-water ecosystems (Fisher and Likens 1973).

Leenheer has developed a procedure for the fractionation and description of NOM (Leenheer et al. 2000; Leenheer et al. 2004) into its DOM and POM components. The definition of DOM in this case is any organic matter that passes through a 0.45-µm filter pore whereas the POM does not (Figure 7-1). The DOM is further fractionated according to polarity. Leenheer's procedure allows isolation and characterization of hydrophilic and colloidal fractions that are usually absent in previous procedures (Leenheer et al. 2000). The identification of hydrophilic and colloidal fractions is important because NOM interacts with all chemical components of natural waters, which in turn alters the behavior of pollutants in surface and ground waters. For example, the solubilities of hydrophobic anthropogenic compounds are enhanced by NOM (Wershaw and Hayes 2001), and NOM forms complexes with metal ions that affect the bioavailability and toxicity of the metals to living organisms (Karlsson et al. 2005). Other types of reactions, such as hydrolysis of anthropogenic compounds, are probably also affected by NOM. The isolation and characterization of NOM fractions are of particular importance because each fraction has a unique set of properties. Vignati et al. (2005) showed that the toxicity of contaminants in natural waters is altered by interactions with the colloidal fraction of NOM.

Terrestrial DOM is the result of biological degradation and progressive concentration of organic compounds particularly resistant to degradation. Degradation of vascular plants supplies DOM with approximately 10% proteins, 30 - 50 % carbohydrates (mainly cellulose), some lipids concentrated in the roots and leaf cuticles (Killops and Killops 2004), 15 - 25 % lignin, and other bio-macromolecules. Moreover, evidence suggests a correlation between environmental conditions and type of terrestrial DOM derived from soil (Christ and David, 1996; Nguyen et al. 2019).

Lignin, an important tracer for terrestrial organic matter, consists of multiple phenylpropanoid units that are linked to each other by ether and carbon–carbon bonds that confer chemical stability to lignin, which is assumed to resist extensive microbial degradation (Verma et al. 2009). Proteins and



Source: Wershaw *et al*. (2005) AA, amino acids; CHO, carbohydrates; CPOM, coarse particulate organic matter; FA, fatty acids, FPOM, fine particulate organic matter; HA, hydrophilic acids; HC, hydrocarbon; VPOM, very fine particulate organic matter.

Figure 7-1. Size range of particulate (POM) and dissolved organic matter (DOM) and organic compounds in natural waters.

carbohydrates are, in contrast, biolabile compounds, because of the susceptibility of peptide and glycosidic bonds to hydrolysis by a variety of enzymes (Piontek et al. 2010).

There is increasing interest in DOM from other sources, for example atmospheric aerosols (Birdwell and Valsaraj 2010). Discovery, characterization and quantitative assessment of such alternative sources is critical for understanding the relevance of DOM in global carbon dynamics. This, however, is a challenging task because sampling and analysis of fog-water-derived DOM is more difficult than for surface or ground water. Interest in this subject is increasing. A thorough review of analytical methods for airborne DOM aerosols was published by Duarte and Duarte (2011). These authors emphasized the environmental significance of this underestimated source of organic carbon and discussed the use of nuclear magnetic resonance (NMR), infrared, and mass spectrometry (MS) methods for its analysis.

7.2. Chemical characteristics of NOM

As outlined in the previous section, NOM is a complex matrix of various organic chemicals found primarily in surface waters, but to a lesser extent in groundwater and fog. This section will provide more background on the chemistry of both DOM and POM, starting with how DOM is analyzed, then discussing its size, molecular architecture, and the special case of chromophoric DOM. Finally, POM will be characterized, particularly its relationship with stream discharge.

7.2.1 Analytical methods for DOM

Several challenges can complicate the analysis of NOM, including the difficulty of complete dissolution, lack of proper molecular separation (Schijf and Zoll 2011), extreme heterogeneity of samples, mutual interference from different classes of compound, and the tendency of association in complex superstructures (Piccolo 2001). Dissolved organic matter is no exception. However, the introduction of Fourier-transform ion cyclotron (FT-ICR) MS has resulted in substantial analytical improvement. This is the most advanced instrumentation available for detection of ionized organic compounds, because of its ultrahigh resolution and because it is usually coupled with non-destructive ion sources, for example electrospray ionization (ESI). The impact of FT-ICR MS on NOM analysis has been outstanding (Kujawinski et al. 2002), and it has rapidly become one of the first choice in DOM studies (Kim et al. 2003b). As a consequence of the introduction of FT-ICR MS, the number of masses characterized in DOM analysis has increased to such an extent that results can be efficiently reported only in simplified diagrams, as plots sorting m/z ratios by homologous series (such as Kendrick plots) and by O/C and H/C ratios (such as Van Krevelen diagram) (Kim et al. 2003a; Wu et al. 2004).

Fourier-transform ion cyclotron MS has substantially improved the capacity to identify DOM molecules. Two specific applications of FT-ICR MS are worth mentioning—detection of either hydrogen-deficient aromatic compounds or nitrogenous organic molecules. In fact, FT-ICR MS high resolution scanning is effective for detection of ions with H/C ratios, which imply a large number of double bond equivalents (DBE) and thus restricts possible structures to condensed aromatic rings; and ions with heteroatoms, such as nitrogen.

High-performance size-exclusion chromatography coupled with either the traditional UV detection or combined multi-detectors (Kawasaki et al. 2011) is also used for qualitative and quantitative evaluation of DOM. High pressure liquid chromatography has also been coupled with NMR spectrometers in on-line mode, thus enabling investigation of the structure of separated DOM molecules (Lam et al. 2007).

The MS methods have become conventional for DOM analysis because dissolved organic molecules are readily ionized, especially in negative-ion mode. However, many problems remain unsolved. First, non-ionizable compounds cannot be characterized by MS. Second, ionization of terrestrial HS or DOM is a complex phenomenon prone to irreproducible results, because of molecular interferences as a result of complex inhomogeneous, supramolecular associations (Peuravuori et al. 2007; Nebbioso et al. 2010; Wickramasekara et al. 2012). These limitations prevent reliance on MS methods alone to achieve structural identification of DOM molecules.

Nuclear magnetic resonance spectroscopy has become fundamental in complementing DOM characterization, because ionization is not required for the NMR excitation. Both solution and solid-state NMR spectroscopy are well established tools in the environmental sciences (Abdulla et al. 2010). Molecular structural information has been obtained from conventional mono, bi, and tri-dimensional NMR spectra (Simpson et al. 2003), and information about molecular diffusion properties and stacking arrangements of organic matter is obtained by use of diffusion ordered spectroscopy (Šmejkalová and Piccolo 2008).

7.2.2 Size and molecular architecture of DOM

A significant amount of fresh water DOM is derived from terrestrial soil organic matter (SOM) that underwent specific transformations that increased its affinity for an aqueous environment. Soil organic matter is traditionally and operationally divided in three pools: fulvic acids, humic acids, and humin, according to their solubility in acids and alkali. Contemporary understanding regards SOM as an aggregate of numerous heterogeneous molecules of relatively small molecular mass held together by weak non-covalent bonds. There is experimental evidence to show that DOM is also arranged in similar supramolecular associations (Peuravuori et al. 2007).

The composition of fresh water DOM is believed to depend on the transformation of plant and decomposed animal compounds into humic-like substances. Investigation of river and lake DOM composition with different techniques supports the hypothesis of plant genesis. In particular, NMR spectroscopy using two dimensional long-range correlation techniques has enabled characterization of compounds directly related to decay of terpenes, for example carboxyl-rich alicyclic molecules and material derived from linear terpenoids, in agreement with previous DOM literature (Leenheer et al. 2003). The same NMR techniques also enabled detection of hexopolysaccharides and aromatic structures, possibly of lignin origin.

Lignins are regarded as refractory substances and, in fact, their contribution to total DOM accumulated along riverine paths, owing to slower mineralization, and was found in the greatest concentration in oceans. In this scenario, the chemical properties of compounds are bound to affect their distribution in water. Although the contribution of lipids is expected to be limited because of their limited aqueous solubility they are nevertheless found in lacustrine DOM. This suggests that the supramolecular structure of DOM enhances the solubility of specific hydrophobic molecules by forming complex associations with them. The dynamics of hydrophobic compounds in natural bodies is complex, because they are also important components of POM. However, although it is well established that hydrophobic organic matter is an abundant component of POM and sedimentary matter, it is not yet clear whether hydrophobic DOM and POM are related to each other.

Because of the further complexity introduced by the tendency of DOM molecules in solution to associate, assessment of the size of DOM particles is not straightforward. Interestingly, whereas mass spectrometry of DOM indicates molecular masses lower than 1000 Da for most compounds, size-exclusion chromatography (SEC) profiles of the same sample suggests a much larger hydrodynamic volume. This discrepancy confirms that single molecules are prone to spontaneous association, but more evidence should be gathered on how this structure is organized.

Nebbioso and Piccolo (2013) argue that a plausible structure for fresh water DOM is an aggregation of spontaneous self-associated superstructures formed by plant-derived products of natural decay, such as lipids, amino sugars, sugars, terpene derivatives, aromatic condensed structures, and lignin-derived compounds.

7.2.3 Chromophoric DOM

Chromophoric dissolved organic matter (CDOM) is the light absorbing fraction of dissolved organic carbon (Rochelle-Newall and Fisher 2002). Interaction with solar radiation is a fundamental property of fresh water DOM and is very relevant in freshwater environmental interactions. The mechanism of formation of chromophoric DOM (CDOM) is still debated, but experimental evidence over the last decade suggests that organic matter derived from phytoplankton, initially colorless, is processed by microbial flora into fluorescent DOM. In fact, CDOM isolated after incubation of algae was found to grow concomitantly with microbial mass (Rochelle-Newall and Fisher 2002). Further evidence of the involvement of phytoplankton in the formation of lacustrine DOM came from quantitative assessment of average and daily rates of in-situ production (Zhang et al. 2009). It has also recently been reported that fluorescence absorption peaks for humic and fulvic acids increased proportionally with the amount of DOM. These acids are probably formed under terrestrial conditions and then transported in natural water bodies, thereby affecting fluorescence response (DePalma et al. 2011). The different chemical composition of autochthonous and humic DOM in fresh water necessitated more systematic description. Hence, the humification index and the index of recent autochthonous contribution were usefully introduced (Huguet et al. 2009).

The chemical origin of the colloidal properties of CDOM have also been investigated by flow-field flow fractionation (Stolpe et al. 2010), assuming differentiation between humic and/or fulvic-like and protein-like compounds. Whereas the origin of the latter was attributed to fresh-water autochthonous life, the sources of the former materials are believed to be terrestrial. Furthermore, it seems there is a strict correlation between the size fraction and the composition of the colloidal phase, with protein-like materials occurring primarily in the smaller size fraction and humic-type materials in the larger (Boehme and Wells 2006).

7.2.4 Characterization of DOM

The molecular composition of fresh water DOM has been studied less than that of marine DOM, probably owing to the greater effect of oceanic DOM on the geochemical carbon balance. Nevertheless, several studies have tried to remedy this and characterize fresh water DOM substances in detail. The NMR spectroscopy experiment of Lam et al. (2007) performed on lacustrine DOM showed the potential of this technique in recognizing and quantifying functional groups even in such a complex DOM system. Lam et al. (2007) succeeded in differentiating aliphatic, carbohydrate, aromatic, and carboxyl-rich alicyclic molecules as well as characterizing specific regions assignable to well-known organic species. They also differentiated terpene-derived carboxyl-rich alicyclic molecules from the material derived from linear terpenoids; therefore, allowing further analysis of terpene metabolism in DOM. However, the conventional technique for molecular investigation of DOM is, again, FT-ICR MS, because of its resolving power, which is capable of revealing hundreds of empirical formulae and furnishing plausible molecular structures for each unknown compound.

7.2.5 Characteristics of POM

To understand the movement of POM from small mountainous river systems to the ocean, several studies were focused on the concentration and composition of suspended particles from rivers draining into the Pacific Ocean (Hood et al. 2006; Hatten et al. 2012; Goñi et al. 2013). The studies investigated watersheds covered a wide range of sizes, from less than 1 km2 (Hood et al. 2006) to 10,000 km2 (Goñi et al. 2013) with a discharge rates from 42 m3/s to 208 m3/s and variability from 65 m3/s to 517 m3/s, respectively. Even that the basins were very different the findings were similar, in the sense that “concentrations of all measured constituents in both rivers increased as a function of discharge, resulting in their export being dominated by short-lived, wintertime high-discharge events” (Goñi et al. 2013). The same pattern was observed in Figure 7-2 for three small watersheds from the H.J. Andrews Long-Term Ecological Research (Hood et al. 2006).

Hatten et al. (2012) found significant differences in the particle compositions collected at low and high discharges, even though the watershed studied had similar 14C ages with other small mountainous river systems. The low flows contained primarily organic detritus from non-vegetation sources (e.g., algal cells) while particles with vegetation and soil-derived POM dominated the high flows. Biomarker compositions indicate that a significant portion of the POM originated from areas affected by shallow landslides and riparian zones, which could be caused by the low uplift rates in combination with high net primary production and relatively thick soils. According to Hood et al. (2006), during storms the DOC gained more humic material, which increased between 9 and 22 % in the study watersheds. The humic content of DOC decreased after the storm but was still elevated compared with the pre-storm samples.

Similar to Hood et al. (2006), Goñi et al. (2013) found that concentration of all constituents increased with discharge, indicating that mobilized materials comes from the watershed exhibiting shallow landslides. The composition of the discharge, namely the lignin phenols and cutin acids, suggest the source was conifer-dominated forest vegetation but with significant inputs of non-woody angiosperms. The results support previous findings that areas manifesting

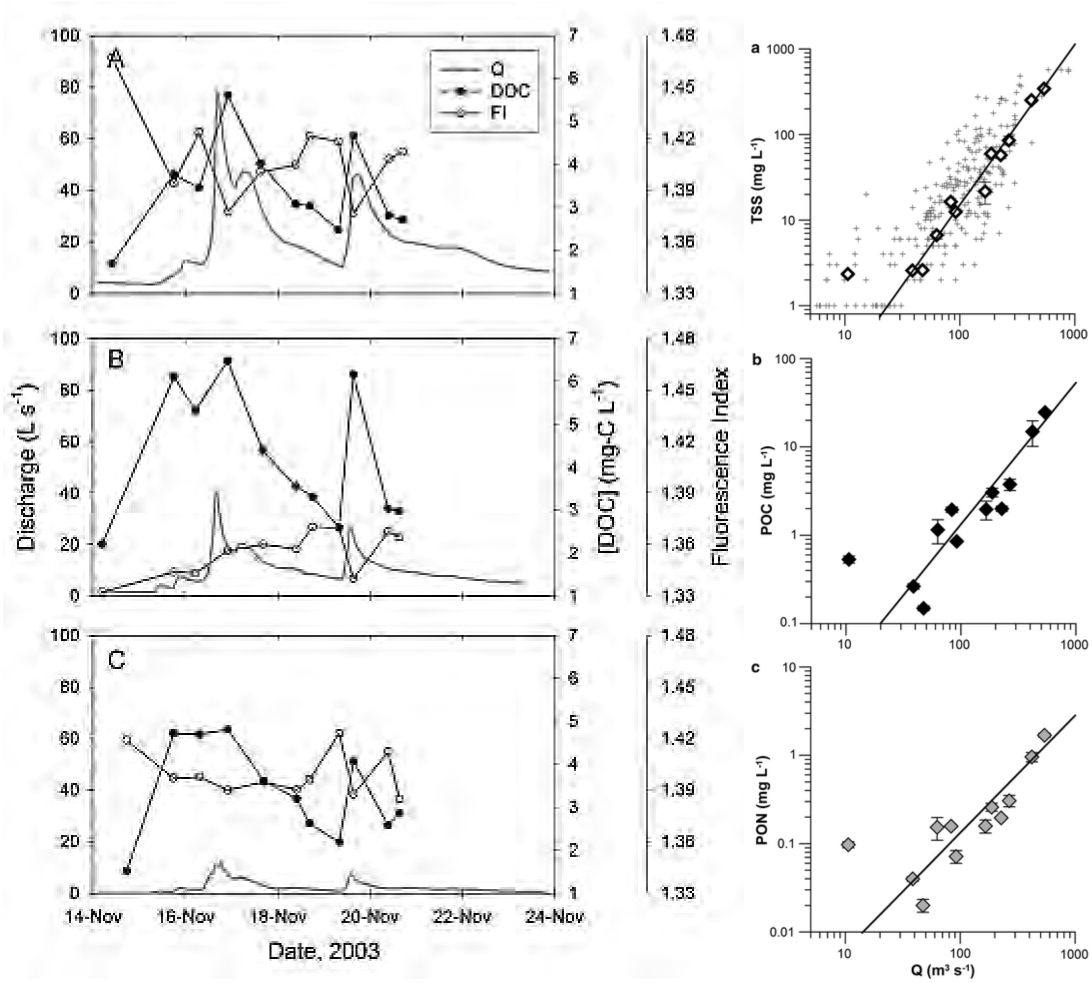


Figure 7-2. Time series of fluorescence index (FI), DOC concentrations, and discharge in (A) watershed 1, (B) watershed 2, and (C) watershed 10, as presented by Hood *et al*. (2006) and the concentrations of a total suspended sediment (TSS), b total particulate organic carbon (POC) and c total particulate organic nitrogen (PON) plotted versus discharge (Q, $m^3 s^{-1}$) from the study Hatten *et al*. (2012).

frequent landslides are commonly covered by a mixture of gymnosperms (e.g., Douglas-fir and hemlock) and angiosperms (bigleaf maple, red alder, vine maple) (Roering et al. 2003). Based on the δ13C and Δ14C signatures the biogenic POM seems to have a mean residence time of a few hundred years that characterizes the transit times of mobilized silts and clays from headwater valleys. It is possible that most of the POM transported during the high-discharge events of 2007 - 2009 from the Umpqua watershed came from soils mobilized during shallow landslides, as there was no evidence to support significant amount of materials from deeper soils and/or bedrock.

7.3. Forest management effects on NOM

The present review used more than 100 studies regarding with NOM, out of which 30 were pertinent to Oregon. The studies are either observational, which basically test some hypotheses, or modeling, which aim to predict stream behavior following various agents of change, natural or human triggered. The papers of Nieminen et al. (2017) and Leenheer (2009) are notable because they are review studies, with the limitation that the former is focused on the Scandinavian peninsula and the latter is addressed to researchers. Nevertheless, Leenheer (2009) sets the stage for NOM study, by stating that "obtaining pure NOM compounds that can be identified by conventional analyses is not yet possible, and the most homogeneous of NOM fractions still contain hundreds to thousands of compounds." He goes even further, by saying that "NOM structures derived from analytical data are models of average data sets, and these models are only approximations." While some studies aimed at understanding the dynamics of NOM in relatively undisturbed forests, understood as reduced active forest management in the last 40 years (Lee and Lajtha 2016), other were focused on the impact of natural events on NOM, such as fire (Abdelnour et al. 2013; Wang et al. 2015a, 2015b, 2016) or bark beetle (Kraus et al. 2010; Beggs and Summers 2011; Brouillard et al. 2016). Multiple studies were concentrated on impact of human activities on NOM, particularly forest harvesting (Nieminen et al. 2017), but only one was carried out in the Pacific Northwest in last two decades (Kelliher et al. 2004). An interesting series of three papers were dedicated to streams meandering within an urban watershed, Fanno Creek in Washington County, Oregon (Goldman et al. 2014; Keith et al. 2014; Sobieszczyk et al. 2014). The modeling studies were focused either on dissolved organic carbon and total mercury concentrations in small watersheds following clearcutting (Zhang et al. 2016) or on the effects of forest harvest on catchment carbon and nitrogen dynamics (Abdelnour et al. 2013).

7.3.1 Timber harvesting

The impact of forest management on DOM was studied with PARAFAC model (Stedmon and Bro 2008) by Lee and Lajtha (2016) who found that the proportion of protein-like DOM, which is inversely related to DOC, increases during low-flow, whereas for shallow subsurface flow it decreases. Their study confirmed the importance of the antecedent soil moisture on DOC, and consequently on DOM (van Verseveld et al. 2009). Lee and Lajtha (2016) pointed towards a relatively reduced DOM source from microbial-processes. They predicted that basins on which younger stands are growing would have larger contribution of protein-like and microbial-like components in stream water than basins with old growth because it seems that DOC still has relatively lower values in streams from harvested watersheds even after half century (Lajtha and Jones 2018). They argue that the reason for reduced DOC, and consequently DOM, is the reduced amount of coarse woody debris, which was diminished during the harvesting and regeneration process by slash removal and/or burning, as well as site preparation. Their inference is supported by local evidence from the H.J. Andrews experimental forest as well as by other studies carried out in the Hubbard Brook Experimental Forest, New Hampshire (Cawley et al. 2014) or in the Coweeta Hydrologic Laboratory, North Carolina (Yamashita et al. 2011). Their main finding is that the impact of forest harvest is long-lasting, many decades after harvesting the metabolism of DOM is still being affected. One can argue that the complexity of the hydrological systems is another factor responsible for the presence of a change in DOM concentrations even after 50 years, in the sense that the system followed another path triggered by events significantly altering the species composition, such as harvesting, wind throw or fire (Prigogine 1997; Sprott 2003; Phillips 2004; Freire and DaCamara 2019). From the nonlinear perspective, it can be argued that any change would position the watershed behavior on a different trajectory from the DOM concentration perspective; therefore, changes in DOM would have occurred with or without harvesting.

An excellent review presenting the effects forest harvest on NOM (Nieminen et al. (2017) focused on peatlands. They argued that soil characteristics contribute to nutrient exports following harvesting, but the contribution of soil characteristics to the export of the other nutrients and DOC from harvested peatland forests is not necessarily well documented. The studies focused on sediment reduction and nutrient exports from drained peatland forests have assessed either the impacts of sedimentation ponds (Joensuu et al. 1999) or natural and restored wetland buffers (Vikman et al. 2010; O'Driscoll et al. 2014), which have shown that sedimentation ponds are effective only in retaining the exports of particles, whereas wetland buffers not only retain particles but also decrease the export of dissolved elements (Vikman et al. 2010; O'Driscoll et al. 2014).

Nieminen et al. (2017) found that the main factors affecting the export of elements after forest harvesting in drained peat wetlands are soil characteristics, nutrient uptake by vegetation, management of forest residues, and drainage and site preparation. Soil characteristics, particularly phosphorus (P) adsorption capacity and iron (Fe) content, may have a strong impact on the exports of DOC for high water table conditions following harvesting. Reduction of Fe in anoxic soils reduces the number of protons, increasing soil water pH, which results in a break-up of R-Fe(III)-R associations, leading to an increase in electronegativity of organic moieties (Grybos et al. 2009). In these conditions the exports of Fe and DOC may increase for soils with high Fe content (Nieminen et al. 2015; Kaila et al. 2016). Nitrogen (N) exports following forest harvest seems to be higher from minerotrophic (Kaila et al. 2015) than from ombrotrophic sites (Nieminen 2003); a possible reason being that organic substances contain organic N, which increase mobilization and leaching under anoxic conditions in Fe-rich minerotrophic peats.

It has been shown that vegetation may be a substantial sink for the nutrients released from soil and harvest residues after harvesting (Kaila et al. 2014). Three studies researched vegetation seeding as a means to mitigate P export from forested blanket peat catchments, where the recovery of natural vegetation was assessed as being too slow for an efficient retention of the nutrients released due to harvesting (Asam et al. [2012); O'Driscoll et al. [2011, 2014]). These studies showed that up to several kilograms of P per hectare may accumulate in vigorously growing vegetation. O'Driscoll et al. (2014) revealed that the uptake of P measured in vegetation supplied a corresponding reduction in the P exported through leaching. Therefore, vegetation can accumulate nutrients that would otherwise be leached.

Several studies suggested higher nutrient concentrations in soil and soil water under harvest residue piles than residue-free areas in harvested peatland forests (Rodgers et al. 2010; Asam et al. 2014), which suggest that whole-tree harvesting would be efficient in decreasing nutrient exports. Conversely, Kaila et al. (2014) and Kaila et al. (2015) found non-significant differences in N and P exports between whole-tree and stem-only harvested catchments in Scots pine dominated ombrotrophic peatlands in south-central Finland or Norway spruce in Finland. However, the studies by Asam et al. (2014) and O'Driscoll et al. (2014) found that whole-tree harvest could decrease P and N exports from blanket peat catchments.

Multiple studies have revealed that forests can be harvested without inducing significant erosion and export of suspended sediments (Nieminen 2003; Rodgers et al. 2011), particularly when harvest residues are used as mats to improve soil carrying capacity against heavy harvest machinery. However, the risk of erosion increases considerably during the cleaning of the existing ditch networks or during the excavation of new ditches after harvesting (Holden et al. 2007). Erosion becomes a serious problem when the ditches are excavated deep into the mineral soil, (Joensuu et al. 1999; Nieminen 2003). In drained peatlands evidence suggested that soil disturbance between ditches (by mounding or stump harvesting) may not increase erosion, but erosion cannot be avoided where ditches are cleaned or new ditches are excavated after harvesting Nieminen et al. (2017). If limited soil disturbance was observed in sensitive areas, such as peatlands, one can infer that regions less sensitive, such as Pacific Northwest, would exhibit at least the same behavior, particularly when best management practices are applied.

In the last two decades the impact of forest harvest on NOM has been studied using complex models, such as the Visualizing Ecosystems for Land Management Assessments, VELMA (Abdelnour et al. 2013). The VELMA is a spatially distributed ecohydrology model that simulates changes in soil water infiltration and redistribution, evapotranspiration, surface and subsurface runoff, carbon and nitrogen cycling in plants and soils, and the transport of dissolved carbon and nitrogen into streams. The model combines watershed level and soil column level frameworks. The multilayered soil column, which consists of n soil layers, is the fundamental hydrologic and ecological unit. The soil column framework is placed within the catchment framework using catchment topography, which is gridded into pixels, each pixel consisting of one coupled soil column. The neighboring soil columns communicate through downslope lateral transport of water and nutrients. The model computes the surface and subsurface runoff responsible for the lateral transport and feeds the uphill soil column to the surrounding downslope soil columns. Nutrients transported downslope from one soil column to another soil column are processed through sub-models that ensure the discharge of water and nutrients into the stream from all soil columns.

The watershed framework contains a sub-model for lateral transport of nutrients, the equations for which are detailed in Abdelnour et al. (2011). The model uses climatic data for the watershed 10 of the H.J. Andrews Long-Term Ecological Research from 1 January 1969 to 31 December 2008. Input data for the model are daily temperature, precipitation, atmospheric nitrogen deposition, daily streamflow, and NO3, NH4, DON and DOC losses to the stream, overlaid on a 30m resolution digital terrain model. Results supplied by the model theorize that losses of dissolved nutrients in the pre-harvest old-growth forest were generally low and contained primarily of organic nitrogen and carbon. However, after harvest the carbon and nitrogen losses from the terrestrial system to the stream and atmosphere increased, following the reduced plant nitrogen uptake, increased soil organic matter decomposition, and high soil moisture (Figure 7-3). Finally, the modeling exercise suggests that the rate of forest regrowth following harvest was lower than that after fire because post-clear-cut stocks and turnover of detritus nitrogen were substantially lower than after fire (Figure 7-3).

The soil column framework includes four sub-models: soil temperature model, soil plant model, nitrification, and denitrification (Abdelnour et al. 2013). The soil plant model is the only model that combines multiple sub-models, namely atmospheric nitrogen deposition, Michaelis - Menten functions, plant mortality, plant uptake, water stress function, biomass root function, vertical transport of nutrients, and soil organic carbon decomposition.

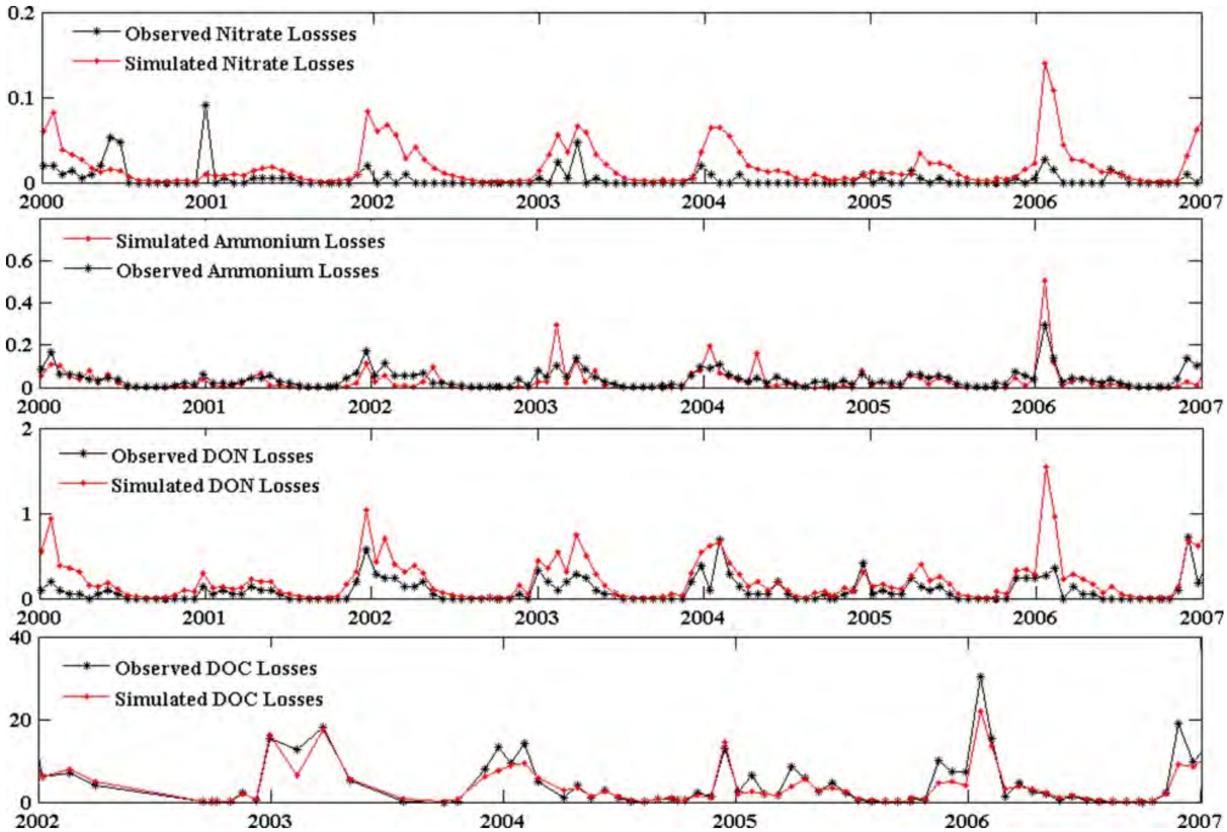


Figure 7-3. The simulated (red dots) versus observed (black dots) of NO3 (mg Nm−2), NH4 (mg Nm−2), DON (mg Nm^{-2}), and DOC losses (mg Cm^{-2}) to the stream after the 1975 clear-cut of WS10 in the H.J. Andrews, according to Abdelnour et al. (2013). The x axis represents the selected data between 2000 and 2007 for NO_3, NH_4, and DON losses and between 2002 and 2007 for DOC losses, while the y axis represents the amount of daily losses that reaches the stream.

A similar process –based model (Figure 7-4) was used by Zhang et al. (2016), to project the impact of forest management (clearcut, regeneration and growth) on DOC and total mercury export from the forest-dominated Pine Marten Brook and Moose Pit Brook watersheds of Kejimkujik National Park (KNP), Nova Scotia (NS), Canada. Zhang et al. (2016) suggest that during a forest rotation, DOC and total mercury concentrations decrease after clearcut to a minimum at approximately 15 years after regeneration, and then increase with age. They found that large debris pools left on site after clearcutting can provide significant pulses in DOC and within-watershed total mercury export during the first 2 - 3 years after harvest. The model suggests a sinusoidal variation of the DOC- concentration and total mercury concentration, with a maximum in autumn followed by a minimum in the spring, another maximum in June and the second minimum before leaf fall. The field data used to calibrate the model indicated that conifer species and wetland-dominated watersheds are prone to transferring more DOC and total mercury to aquatic ecosystems than deciduous and dryland-dominated watersheds.

7.3.2. Forest roads

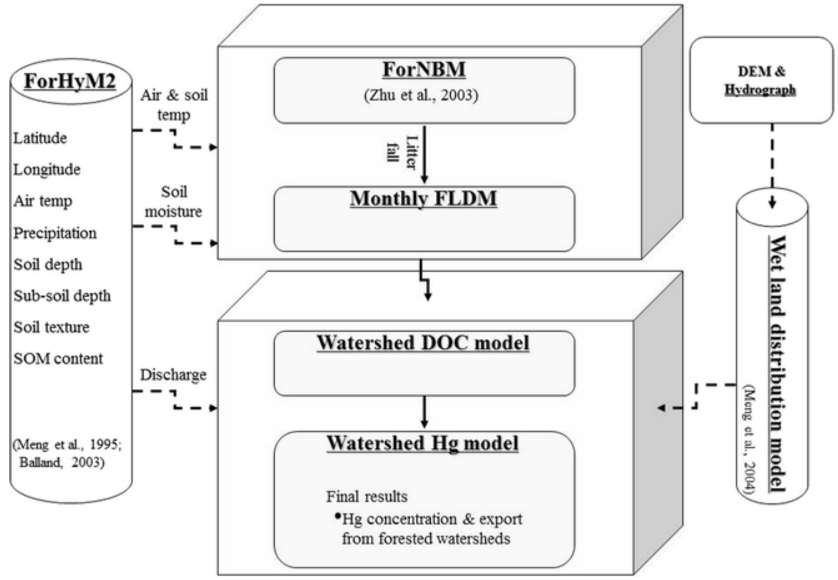


Figure 7-4. Structure of the in-stream DOC- and Hg-concentration model according to Zhang *et al*. (2016). ForHyM2 is used to simulate variation in soil temperature and moisture and stream discharge. ForNBM is used to simulate forest growth and litter production. Monthly FLDM is used to simulate monthly litter decomposition. Output from FLDM is used as input to the DOC production and export model component.

Based on the study of Yanai et al. (2003), Abdelnour et al. (2013) argued that it is difficult to separate the effects on NOM of plant biomass removal from the effects of roads. Furthermore, to reduce the impact on experimental results of the lack of delineating the effects of biomass removal from roads the most common approach is to use models (Yanai et al. 2003; Abdelnour et al. 2013; Zhang et al. 2016). To model such complex systems researchers rarely develop ad-hoc models, they most likely use either existing models or build one for preexisting modules that fit the objectives, data availability, and methodology (Yanai et al. 2003; Zhang et al. 2016). As such in the last 20 years, only few papers were dedicated to the relationship between NOM and forest roads (Deljouei et al. 2015; Abdi et al. 2018). The only study since 2000 pertinent to Pacific Northwest that we found is Abdi et al. (2018), which aimed at assessing the relationship between the amount of organic matter and the behavior of forest soil as road material, which is not the focus of this review. Evidence for the Pacific Northwest area was provided that the main export of NOM and disinfection byproducts (DBP) is triggered by the first major rain event occurring in the fall (Kraus et al. 2010).

7.3.3. Natural disturbances: wildfire and beetles

Whereas forest roads impact on NOM is difficult to quantify and separate from other sources, the story is different picture for natural catastrophic events, specifically wildfire and mountain pine beetles, on which there has been a significant amount of research (Beggs and Summers 2011; Wang et al. 2015a, 2015b, 2016; Brouillard et al. 2016).

Wildfires are increasing in frequency and severity in the United States, which is likely altering the chemistry and quantity of NOM and DBP traveling outside forested watersheds. Wang et al. (2015a) claim that we have mostly speculative understanding on the effects of the fire triangle (heat, oxygen, and fuel) on DOM alteration. A similar statement was made by Wang et al. (2015b) who assert that the effects of wildfire on drinking water quality are limited, especially in terms of NOM and NOM-associated formation of DBP. Considering that forest floor is a major source of terrestrial DOM, they investigated characteristics and DBP formation of water extractable organic matter from nonburned detritus and two types of burned detritus (i.e., black ash, suggesting moderate fire severity and white ash, suggesting high fire severity) associated with the 2013 Rim Fire in California. A similar laboratory study aiming to answer analogous questions on DOM and DBP was carried out using detritus from Pinus ponderosa and Abies concolor. Spectroscopic results suggest that burned-detritus extracts had lower molecular weight and divergent aromaticity depending on oxygen availability. The laboratory findings show that DBP precursors in fire-affected forest detritus are highly dependent on temperature and oxygen availability. The 2013 Rim Fire revealed that wildfire consumed a large portion of organic matter from the detritus layer, which led to lower yields of water extractable organic carbon and organic nitrogen. Therefore, the wildfire triggers an overall reduction in water extractable terrestrial DBP precursor yield from detritus (Wang et al. 2015b).

The last 15 years of bark beetle infestation had a significant impact on water quality as a result of increased organic carbon release and hydrologic shifts induced by the tree dieback. Brouillard et al. (2016) analyzed 10 years of municipal data, from 2004 to 2014, across six water treatment facilities in the Rocky Mountains which cover the extent of beetle impact. The study revealed a significant increasing trend in total organic carbon and total trihalomethane production within the beetle-infested watersheds, while no or insignificant trends were found in watersheds with lower impact (Figure 7-5). Alarmingly, the total trihalomethane concentration trend in the watersheds that experienced high bark beetle impact exceeded regulatory maximum contaminant levels during 2013 and 2014.

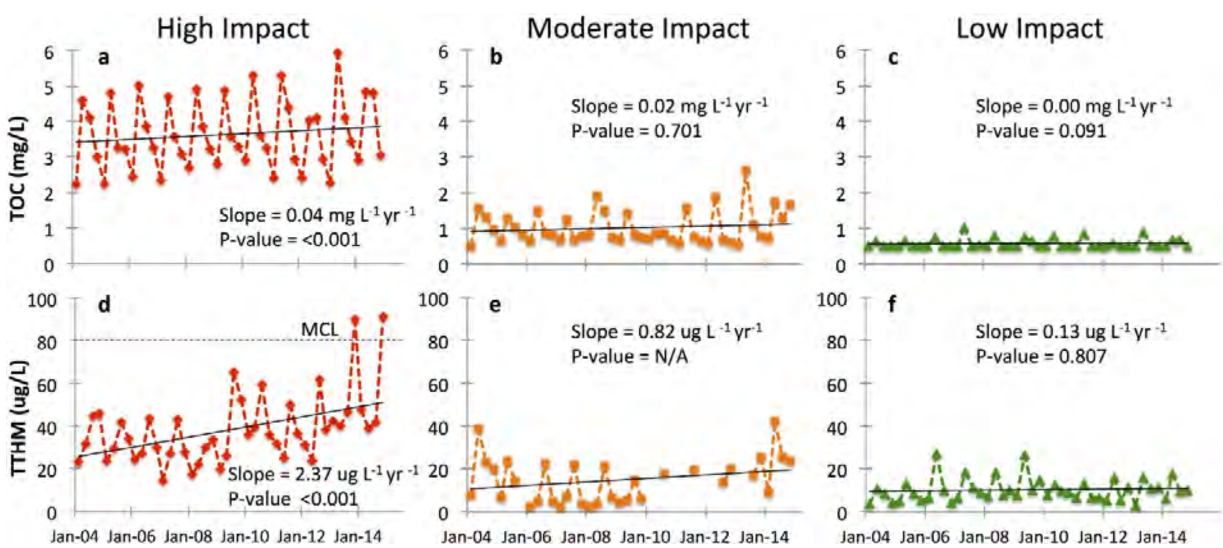


Figure 7-5. Analysis of compiled water quality data obtained from six water treatment facilities from Brouillard et al. (2016). The results shows an increasing trend in high impact watersheds compared to lower impacted watersheds. Panels depict water quality data from 2004 to 2014 where a–c. display binned total organic carbon (TOC) concentrations and d–f. portray binned total trihalomethane (TTHM) concentrations in high (red diamonds), moderate (orange squares), and low impact (green triangles) watersheds with a regulatory total trihalomethane maximum contaminant level (MCL) of 80 ppb.

Brouillard et al. (2016) found that surface water from high impact watersheds exhibited significantly higher total organic carbon, aromatic signatures, and DBP formation than watersheds with lower infestation levels. Furthermore, the spectroscopic analyses of surface water suggest that heightened DBP precursor levels are a function of both total organic carbon and aromatic character. The relationship between total organic carbon and aromatic character was heightened during precipitation and runoff events. In these situations the altered hydrologic flow paths resulting from tree mortality seem to mobilize organic carbon and elevate DBP formation potential for months after runoff. Brouillard et al. (2016) found that water quality is impacted nearly one decade after bark beetle infestation, but significant increases in total organic carbon mobilization and DBP precursors are limited to areas that experience massive tree mortality.

7.4. Effects of NOM on potable water treatment

Dissolved organic matter in the environment is found, with rare exceptions, at extremely low concentrations (0.5–1.0 mg L^{-1} in the oceans); inorganic salts exceed this value by several orders of magnitude (Wershaw et al. 2005). Therefore, specifically designed techniques are generally used to increase the concentration of DOM and to remove salts. Retention-based methods involving XAD resins or 18C stationary phases have been extensively investigated, only to reveal that a variable, but substantial, part of DOM is lost because of incomplete retention.

Ultrafiltration has been used to remove large volumes of water through membrane pores which are restrictive for DOM but not for water molecules and small ions. It is a formidable desalting method that is also used to purify water from excessive DOM content. Ultrafiltration is also affected by incomplete recovery of organic carbon, but to a lesser extent.

Reverse osmosis is an improvement of ultrafiltration, as it operates similarly by allowing water through membranes but restricting cut-off for DOM. However, in reverse osmosis (RO) the solution is forced by a pressure gradient to flow against osmotic flow (hence reverse osmosis). Different and more restrictive membranes are used for reverse osmosis than for ultrafiltration, resulting in greater retention of ions in DOM samples. Such retention consists mainly of ions derived from silicic acid (H4SiO4) and sulfuric acid (H2SO4). Development of methods such as RO coupled with electrodialysis and pulsed electrodialysis was, in fact, intended to minimize these inorganic impurities. These processes are now rapidly becoming conventional for DOM purification and are in constant development, optimization, and standardization.

Treatment of sediments to enable solid-state cross-polarization magic-angle spinning NMR spectroscopy of their labile organic matter is routinely based on acid washing and chemical digestion with hydrochloric acid (HCl) and/or hydrofluoric acid (HF). However, adoption of sensitive instrumental methods for DOM analysis, for example ultrahigh-resolution mass spectrometry, may prevent such sample pretreatment, thus limiting the formation of artefacts resulting from the processes of DOM concentration and pH modification. A promising application for DOM separation seems to be the development of carbon nanotubes as solid-phase extraction stationary phases which exploit the affinity of nano-structures for organic compounds in solution. Such stationary phases have, however, so far resulted in limited recovery that ranges between 30 and 80%, depending on DOM type and specific selectivity for low-molecular-weight DOM fractions.

7.5. Prevalence of standards exceedances

There are 63 chemicals where finished water sampling detected a contaminant that meets the "Alert" or an "Exceedance" of USEPA standards. The trigger value for Alerts varies according to the chemical, in some cases it's any presence, but for the ones we're concerned with an Alert is issued if the concentration is greater than or equal to 50% of the Maximum Contaminant Level (MCL). We're primarily concerned here with the two disinfection byproducts, Total Haloacetic Acids (HAA5), and Total Trihalomethanes (TTHM), but it is useful to understand how frequent these two contaminants are compared to others sampled.

The data records show Alerts or Exceedances for 63 chemicals, divided into eight categories: Inorganic chemicals; volatile organic chemicals; synthetic organic chemical; radiological agents; nitrate; nitrite; Arsenic; and Asbestos. There were a total of 5,813 "Detections" of these 63 chemicals from 697 public water supplies over the almost 20 year period. For two disinfection byproducts, HAA5 is the 4th most commonly detected (5.9%), while TTHM is the 5th most common (5.0%). In comparison, nitrate (nitrogen) is the most commonly detected contaminant (44.2%); arsenic is the second most common (16.3%); and tetrachloroethylene is the third (6.3%).

Based on an examination of the data, the DBPs are only found in water samples from the utilities' distribution system, and according to the incident reports, these samples are oftentimes taken at the end of long pipe runs. The vast majority of detections and exceedances of DBPs are from utilities that utilize surface water sources (including produced, purchased, and groundwater under the influence of surface water): for HAA5, it is 97%, and for TTHM it is 83%. If a DBP is found, most often it exceeds the MCL standard (89% for HAA5; 93% for TTHM) rather than just the lower detection level that triggers an alert. Figure 7-6 shows the wide variation in the numbers of yearly detections during the period 2004 to 2019 ranging from a high of 80 for HHA5 in 2004 to a low of 8 for TTHM in 2017. On average, there are about 30 detections of HAA5 state-wide, and 25 of TTHM, annually.

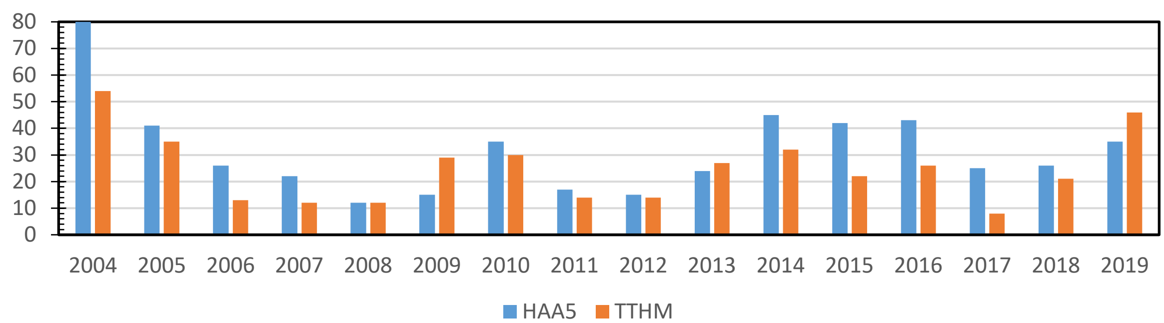


Figure 7-6. Yearly detections of total haloacetic acids (HAA5) and total trihalomethanes (TTHM).

One hundred forty two public water systems that had detections of DBPs from 2002 through 2020. There are three general patterns in these detections. First, the vast majority of these public water systems (77%) have occasional and infrequent detections, i.e., less than one every two years. The second pattern is that a particular utility will have a cluster of detections before resolving the problem, i.e., detections in fewer than 9 years from 2003 – 2019, but will have more than eight total detections; this pattern represents about 17%, or 24 of the utilities with detections. Finally, a very small number of utilities have chronic detections year-in and year-out; there are 7 utilities representing 5% of the utilities with detections, and less than 5% of all public water suppliers.

Most detections, and thus most DBP exceedances, are within 125% of the MCL (Figure 7-7). As mentioned above, most detections of HAA5 and TTHM exceed the MCL threshold, 91% and 93%, respectively. Almost 60% of HAA5 and TTHM detections are within 125% of the MCL, with the majority of the remainder within 150% of the MCL, 15% and 23% respectively. There are relatively few detections that are over 200% of the MCL, and the highest of these may be sampling errors.

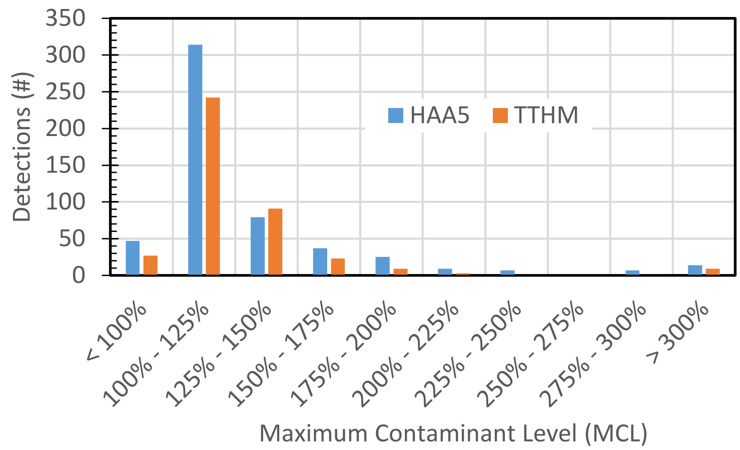


Figure 7-7. DBP detection sample concentrations compared to the maximum contaminant level (MCL).

The two regulated DBPs, total haloacetic acids (HAA5) and total trihalomethanes (TTHM), are respectively the fourth- and fifth-most frequent contaminant alerts and exceedances in the Oregon Health Authority's database. Disinfection byproduct detections in finished drinking water show that in the vast majority of cases the utility relies on surface water as their primary source. Most detections are isolated events, but a subset of water utilities (17%) have clusters of detections with absences in intervening years, while a smaller set (5%) have chronic, annual, detections of DBPs in their water systems. Further, most exceedances are within 150% of the maximum contaminant level.

7.6. References

Abdelnour, A., B. McKane, R., Stieglitz, M., Pan, F., Cheng, Y., 2013. Effects of harvest on carbon and nitrogen dynamics in a Pacific Northwest forest catchment. Water Resources Research 49, 1292-1313.

Abdelnour, A., Stieglitz, M., Pan, F., McKane, R., 2011. Catchment hydrological responses to forest harvest amount and spatial pattern. Water Resources Research 47.

Abdi, E., Babapour, S., Majnounian, B., Amiri, G.Z., Deljouei, A., 2018. How does organic matter affect the physical and mechanical properties of forest soil? Journal of Forestry Research 29, 657-662.

Abdulla, H.A.N., Minor, E.C., Dias, R.F., Hatcher, P.G., 2010. Changes in the compound classes of dissolved organic matter along an estuarine transect: A study using FTIR and 13C NMR. Geochimica et Cosmochimica Acta 74, 3815-3838.

Asam, Z.-u.-Z., Kaila, A., Nieminen, M., Sarkkola, S., O'Driscoll, C., O'Connor, M., Sana, A., Rodgers, M., Xiao, L., 2012. Assessment of phosphorus retention efficiency of blanket peat buffer areas using a laboratory flume approach. Ecological Engineering 49, 160-169.

Asam, Z.-u.-Z., Nieminen, M., O'Driscoll, C., O'Connor, M., Sarkkola, S., Kaila, A., Sana, A., Rodgers, M., Zhan, X., Xiao, L., 2014. Export of phosphorus and nitrogen from lodgepole pine (Pinus contorta) brash windrows on harvested blanket peat forests. Ecological Engineering 64, 161-170.

Beggs, K.M.H., Summers, R.S., 2011. Character and Chlorine Reactivity of Dissolved Organic Matter from a Mountain Pine Beetle Impacted Watershed. Environmental Science & Technology 45, 5717-5724.

Bhardwaj, V. 2006. Disinfection By-products. Journal of Environmental Health. Vol. 68, No. 10.

Birdwell, J.E., Valsaraj, K.T., 2010. Characterization of dissolved organic matter in fogwater by excitation–emission matrix fluorescence spectroscopy. Atmospheric Environment 44, 3246-3253.

Boehme, J., Wells, M., 2006. Fluorescence variability of marine and terrestrial colloids: Examining size fractions of chromophoric dissolved organic matter in the Damariscotta River estuary. Marine Chemistry 101, 95-103.

Brouillard, B.M., Dickenson, E.R.V., Mikkelson, K.M., Sharp, J.O., 2016. Water quality following extensive beetle-induced tree mortality: Interplay of aromatic carbon loading, disinfection byproducts, and hydrologic drivers. Science of The Total Environment 572, 649-659.

Carlson, C.A., Hansell, D.A., 2015. Chapter 3 - DOM Sources, Sinks, Reactivity, and Budgets. In: Hansell, D.A., Carlson, C.A. (Eds.), Biogeochemistry of Marine Dissolved Organic Matter (Second Edition). Academic Press, Boston, pp. 65-126.

Carter, C.W., Suffet, I.H., 1982. Binding of DDT to dissolved humic materials. Environmental Science & Technology 16, 735-740.

Cawley, K.M., Campbell, J., Zwilling, M., Jaffé, R., 2014. Evaluation of forest disturbance legacy effects on dissolved organic matter characteristics in streams at the Hubbard Brook Experimental Forest, New Hampshire. Aquatic Sciences 76, 611-622.

Christ, M.J., David, M.B., 1996. Temperature and moisture effects on the production of dissolved organic carbon in a Spodosol. Soil Biology and Biochemistry 28, 1191-1199.

Deljouei, A., Abdi, E., Babapour, S., 2015. The relation between organic matter content and CBRof road subgrade: Case study Hyrcanian forests, Iran. In: Kanzian, C., Erber, G., Kuhmaier, M. (Eds.), 48thFORMEC Symposium 2015. Forest Engineering Network, Linz, Austria, pp. 365-367.

DePalma, S.G.S., Arnold, W.R., McGeer, J.C., Dixon, D.G., Smith, D.S., 2011. Variability in dissolved organic matter fluorescence and reduced sulfur concentration in coastal marine and estuarine environments. Applied Geochemistry 26, 394-404.

Duarte, R.M.B.O., Duarte, A.C., 2011. A critical review of advanced analytical techniques for water-soluble organic matter from atmospheric aerosols. TrAC Trends in Analytical Chemistry 30, 1659-1671.

Fisher, S.G., Likens, G.E., 1973. Energy Flow in Bear Brook, New Hampshire: An Integrative Approach to Stream Ecosystem Metabolism. Ecological Monographs 43, 421-439.

Freire, J.G., DaCamara, C.C., 2019. Using cellular automata to simulate wildfire propagation and to assist in fire management. Nat. Hazards Earth Syst. Sci. 19, 169-179.

Goldman, J.H., Rounds, S.A., Keith, M.K., Sobieszczyk, S., 2014. Investigating organic matter in Fanno Creek, Oregon, Part 3 of 3: Identifying and quantifying sources of organic matter to an urban stream. Journal of Hydrology 519, 3028-3041.

Goñi, M.A., Hatten, J.A., Wheatcroft, R.A., Borgeld, J.C., 2013. Particulate organic matter export by two contrasting small mountainous rivers from the Pacific Northwest, U.S.A. Journal of Geophysical Research: Biogeosciences 118, 112-134.

Grybos, M., Davranche, M., Gruau, G., Petitjean, P., Pédrot, M., 2009. Increasing pH drives organic matter solubilization from wetland soils under reducing conditions. Geoderma 154, 13-19.

Hatten, J.A., Goñi, M.A., Wheatcroft, R.A., 2012. Chemical characteristics of particulate organic matter from a small, mountainous river system in the Oregon Coast Range, USA. Biogeochemistry 107, 43-66.

Holden, J., Gascoign, M., Bosanko, N.R., 2007. Erosion and natural revegetation associated with surface land drains in upland peatlands. Earth Surface Processes and Landforms 32, 1547-1557.

Hood, E., Gooseff, M.N., Johnson, S.L., 2006. Changes in the character of stream water dissolved organic carbon during flushing in three small watersheds, Oregon. Journal of Geophysical Research: Biogeosciences 111.

Huguet, A., Vacher, L., Relexans, S., Saubusse, S., Froidefond, J.M., Parlanti, E., 2009. Properties of fluorescent dissolved organic matter in the Gironde Estuary. Organic Geochemistry 40, 706-719.

Joensuu, S., Ahti, E., Vuollekoski, M., 1999. The effects of peatland forest ditch maintenance on suspended solids in runoff. BOREAL ENVIRONMENT RESEARCH 4, 343-355.

Kaila, A., Asam, Z., Koskinen, M., Uusitalo, R., Smolander, A., Kiikkilä, O., Sarkkola, S., O'Driscoll, C., Kitunen, V., Fritze, H., Nousiainen, H., Tervahauta, A., Xiao, L., Nieminen, M., 2016. Impact of Re-wetting of Forestry-Drained Peatlands on Water Quality—a Laboratory Approach Assessing the Release of P, N, Fe, and Dissolved Organic Carbon. Water, Air, & Soil Pollution 227, 292.

Kaila, A., Lauren, A., Srakkola, S., Kuivusalo, H., Ukonmaanho, L., Xiao, L., Asam, Z., O'Driscoll, C., Nieminen, M., 2015. The effect of clear-felling and harvest residue removal on nitrogen and phosphorus export from drained Norway spruce mires in southern Finland. BOREAL ENVIRONMENT RESEARCH 20, 693-706.

Kaila, A., Sarkkola, S., Laurén, A., Ukonmaanaho, L., Koivusalo, H., Xiao, L., O'Driscoll, C., Asam, Z.-U.-Z., Tervahauta, A., Nieminen, M., 2014. Phosphorus export from drained Scots pine mires after clear-felling and bioenergy harvesting. Forest Ecology and Management 325, 99-107.

Karlsson, T., Persson, P., Skyllberg, U., 2005. Extended X-ray Absorption Fine Structure Spectroscopy Evidence for the Complexation of Cadmium by Reduced Sulfur Groups in Natural Organic Matter. Environmental Science & Technology 39, 3048-3055.

Kawasaki, N., Matsushige, K., Komatsu, K., Kohzu, A., Nara, F.W., Ogishi, F., Yahata, M., Mikami, H., Goto, T., Imai, A., 2011. Fast and precise method for HPLC–size exclusion chromatography with UV and TOC (NDIR) detection: Importance of multiple detectors to evaluate the characteristics of dissolved organic matter. Water Research 45, 6240-6248.

Keith, M.K., Sobieszczyk, S., Goldman, J.H., Rounds, S.A., 2014. Investigating organic matter in Fanno Creek, Oregon, Part 2 of 3: Sources, sinks, and transport of organic matter with fine sediment. Journal of Hydrology 519, 3010-3027.

Kelliher, F.M., Ross, D.J., Law, B.E., Baldocchi, D.D., Rodda, N.J., 2004. Limitations to carbon mineralization in litter and mineral soil of young and old ponderosa pine forests. Forest Ecology and Management 191, 201-213.

Killops, S., Killops, V., 2004. ntroduction to Organic Geochemistry. Blackwell Publishing, MAlden MA.

Kim, S., Kramer, R.W., Hatcher, P.G., 2003a. Graphical Method for Analysis of Ultrahigh-Resolution Broadband Mass Spectra of Natural Organic Matter, the Van Krevelen Diagram. Analytical Chemistry 75, 5336-5344.

Kim, S., Simpson, A.J., Kujawinski, E.B., Freitas, M.A., Hatcher, P.G., 2003b. High resolution electrospray ionization mass spectrometry and 2D solution NMR for the analysis of DOM extracted by C18 solid phase disk. Organic Geochemistry 34, 1325-1335.

Kraus, T.E.C., Anderson, C.A., Morgenstern, K., Downing, B.D., Pellerin, B.A., Bergamaschi, B.A., 2010. Determining Sources of Dissolved Organic Carbon and Disinfection Byproduct Precursors to the McKenzie River, Oregon. Journal of environmental quality 39, 2100-2112.

Kujawinski, E.B., Freitas, M.A., Zang, X., Hatcher, P.G., Green-Church, K.B., Jones, R.B., 2002. The application of electrospray ionization mass spectrometry (ESI MS) to the structural characterization of natural organic matter. Organic Geochemistry 33, 171-180.

Lajtha, K., Jones, J., 2018. Forest harvest legacies control dissolved organic carbon export in small watersheds, western Oregon. Biogeochemistry 140.

Lam, B., Baer, A., Alaee, M., Lefebvre, B., Moser, A., Williams, A., Simpson, A.J., 2007. Major Structural Components in Freshwater Dissolved Organic Matter. Environmental Science & Technology 41, 8240-8247.

Lee, B.S., Lajtha, K., 2016. Hydrologic and forest management controls on dissolved organic matter characteristics in headwater streams of old-growth forests in the Oregon Cascades. Forest Ecology and Management 380, 11-22.

Leenheer, J.A., 2009. Systematic approaches to comprehensive analyses of natural organic matter. Annals of Environmental Science 3, 1-130.

Leenheer, J.A., Croué, J.-P., Benjamin, M., Korshin, G.V., Hwang, C.J., Bruchet, A., Aiken, G.R., 2000. Comprehensive Isolation of Natural Organic Matter from Water for Spectral Characterizations and Reactivity Testing. In, Natural Organic Matter and Disinfection By-Products. American Chemical Society, pp. 68-83.

Leenheer, J.A., Nanny, M.A., McIntyre, C., 2003. Terpenoids as major precursors of dissolved organic matter in landfill leachates, surface water, and groundwater. Environmental Science & Technology 37, 2323-2331.

Leenheer, J.A., Noyes, T.I., Rostad, C.E., Davisson, M.L., 2004. Characterization and origin of polar dissolved organic matter from the Great Salt Lake. Biogeochemistry 69, 125-141.

Majidzadeh, H., Chen, H., Coates, T.A., Tsai, K.P., Olivares, C.I., Trettin, C., Uzun, H., Karanfil, T. and Chow, A.T., 2019. Long-term watershed management is an effective strategy to reduce organic matter export and disinfection by-product precursors in source water. International Journal of Wildland Fire, 28(10), pp.804-813.

Marín-Spiotta, E., Gruley, K.E., Crawford, J., Atkinson, E.E., Miesel, J.R., Greene, S., Cardona-Correa, C., Spencer, R.G.M., 2014. Paradigm shifts in soil organic matter research affect interpretations of aquatic carbon cycling: transcending disciplinary and ecosystem boundaries. Biogeochemistry 117, 279-297.

Nebbioso, A., Piccolo, A., 2011. Basis of a Humeomics Science: Chemical Fractionation and Molecular Characterization of Humic Biosuprastructures. Biomolecules 12, 1187-1199.

Nebbioso, A., Piccolo, A., 2013. Molecular characterization of dissolved organic matter (DOM): a critical review. Analytical and Bioanalytical Chemistry 405, 109-124.

Nebbioso, A., Piccolo, A., Spiteller, M., 2010. Limitations of electrospray ionization in the analysis of a heterogeneous mixture of naturally occurring hydrophilic and hydrophobic compounds. Rapid Communications in Mass Spectrometry 24, 3163-3170.

Nguyen, M.L., Goldfarb, J.L., Plante, A.F., Lau, B.L.T., Hockaday, W.C., 2019. Sorption temperature and the stability of iron-bound soil organic matter. Geoderma 341, 93-99.

Nieminen, M., 2003. Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland. BOREAL ENVIRONMENT RESEARCH 8, 53-59.

Nieminen, M., Koskinen, M., Sarkkola, S., Laurén, A., Kaila, A., Kiikkilä, O., Nieminen, T.M., Ukonmaanaho, L., 2015. Dissolved Organic Carbon Export from Harvested Peatland Forests with Differing Site Characteristics. Water, Air, & Soil Pollution 226, 181.

Nieminen, M., Sarkkola, S., Laurén, A., 2017. Impacts of forest harvesting on nutrient, sediment and dissolved organic carbon exports from drained peatlands: A literature review, synthesis and suggestions for the future. Forest Ecology and Management 392, 13-20.

O'Driscoll, C., O'Connor, M., Asam, Z.-u.-Z., de Eyto, E., Rodgers, M., Xiao, L., 2014. Creation and functioning of a buffer zone in a blanket peat forested catchment. Ecological Engineering 62, 83-92.

O'Driscoll, C., Rodgers, M., O'Connor, M., Asam, Z.-u.-Z., de Eyto, E., Poole, R., Xiao, L., 2011. A Potential Solution to Mitigate Phosphorus Release Following Clearfelling in Peatland Forest Catchments. Water, Air, & Soil Pollution 221, 1-11.

Oliver, B.G., Thurman, E.M., Malcolm, R.L., 1983. The contribution of humic substances to the acidity of colored natural waters. Geochimica et Cosmochimica Acta 47, 2031-2035.

O'Melia, C.R. 2006. Fundamentals of particle stability. Chapter 18 in Interface Science and Technology Water Treatment. Interface Science and Technology, Volume 10, 2006, Pages 317-362. 7.1. Natural organic matter, NOM

Peuravuori, J., Bursáková, P., Pihlaja, K., 2007. ESI-MS analyses of lake dissolved organic matter in light of supramolecular assembly. Analytical and Bioanalytical Chemistry 389, 1559-1568.

Phillips, J.D., 2004. Divergence, Sensitivity, and Nonequilibrium in Ecosystems. Geographical Analysis 36, 369-383.

Piccolo, A., 2001. The supramolecular structure of humic substances. Soil Science 166, 810-832.

Piccolo, A., 2012. The nature of soil organic matter and innovative soil managements to fight global changes and maintain agricultural productivit. In: Piccolo, A. (Ed.), Carbon Sequestration in Agricultural Soils. Springer Verlag, Heidelberg, pp. 1-20.

Piontek, J., Lunau, M., Händel, N., Borchard, C., Wurst, M., Engel, A., 2010. Acidification increases microbial polysaccharide degradation in the ocean. Biogeosciences 7, 1615-1624.

Prigogine, I., 1997. The end of certainty. The Free Press, New York.

Rochelle-Newall, E.J., Fisher, T.R., 2002. Chromophoric dissolved organic matter and dissolved organic carbon in Chesapeake Bay. Marine Chemistry 77, 23-41.

Rodgers, M., O'Connor, M., Robinson, M., Muller, M., Poole, R., Xiao, L., 2011. Suspended solid yield from forest harvesting on upland blanket peat. Hydrological Processes 25, 207-216.

Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z.-u.-Z., Nieminen, M., Poole, R., Müller, M., Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. Forest Ecology and Management 260, 2241-2248.

Roering, J.J., Schmidt, K.M., Stock, J.D., Dietrich, W.E., Montgomery, D.R., 2003. Shallow landsliding, root reinforcement, and the spatial distribution of trees in the Oregon Coast Range. Canadian Geotechnical Journal 40, 237-253.

Sanderman, J., Kramer, M.G., 2013. Differential production yet chemical similarity of dissolved organic matter across a chronosequence with contrasting nutrient availability in Hawaii. Biogeochemistry 113, 259-269.

Schijf, J., Zoll, A.M., 2011. When dissolved is not truly dissolved—The importance of colloids in studies of metal sorption on organic matter. Journal of Colloid and Interface Science 361, 137-147.

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property. Nature 478, 49-56.

Schnitzer, M., 1972. Humic substances in the environment. Bekker, New York.

Simpson, A.J., Kingery, W.L., Hatcher, P.G., 2003. The Identification of Plant Derived Structures in Humic Materials Using Three-Dimensional NMR Spectroscopy. Environmental Science & Technology 37, 337-342.

Šmejkalová, D., Piccolo, A., 2008. Aggregation and Disaggregation of Humic Supramolecular Assemblies by NMR Diffusion Ordered Spectroscopy (DOSY-NMR). Environmental Science & Technology 42, 699-706.

Sobieszczyk, S., Keith, M.K., Rounds, S.A., Goldman, J.H., 2014. Investigating organic matter in Fanno Creek, Oregon, Part 1 of 3: Estimating annual foliar biomass for a deciduous-dominant urban riparian corridor. Journal of Hydrology 519, 3001-3009.

Sprott, J.C., 2003. Chaos and time-series analysis. Oxford University Press, Oxford.

Stedmon, C.A., Bro, R., 2008. Characterizing dissolved organic matter fluorescence with parallel factor analysis: a tutorial. Limnology and Oceanography: Methods 6, 572-579.

Stolpe, B., Guo, L., Shiller, A.M., Hassellöv, M., 2010. Size and composition of colloidal organic matter and trace elements in the Mississippi River, Pearl River and the northern Gulf of Mexico, as characterized by flow field-flow fractionation. Marine Chemistry 118, 119-128.

Thurman, E.N., 1985. Organic Geochemistry of Natural Waters. Martinus NijhofflDr W. Junk Publishers, Dordrecht.

Tipping, E., 1986. Some aspects of the interactions between particulate oxides and aquatic humic substances. Marine Chemistry 18, 161-169.

van Verseveld, W.J., McDonnell, J.J., Lajtha, K., 2009. The role of hillslope hydrology in controlling nutrient loss. Journal of Hydrology 367, 177-187.

Verma, P., Junga, U., Militz, H., Mai, C., 2009. Protection mechanisms of DMDHEU treated wood against white and brown rot fungi. In, Holzforschung, p. 371.

Vignati, D.A.L., Dworak, T., Ferrari, B., Koukal, B., Loizeau, J.-L., Minouflet, M., Camusso, M.I., Polesello, S., Dominik, J., 2005. Assessment of the Geochemical Role of Colloids and Their Impact on Contaminant Toxicity in Freshwaters: An Example from the Lambro−Po System (Italy). Environmental Science & Technology 39, 489-497.

Vikman, A., Sarkkola, S., Koivusalo, H., Sallantaus, T., Laine, J., Silvan, N., Nousiainen, H., Nieminen, M., 2010. Nitrogen retention by peatland buffer areas at six forested catchments in southern and central Finland. Hydrobiologia 641, 171-183.

Wang, J.-J., Dahlgren, R.A., Chow, A.T., 2015a. Controlled Burning of Forest Detritus Altering Spectroscopic Characteristics and Chlorine Reactivity of Dissolved Organic Matter: Effects of Temperature and Oxygen Availability. Environmental Science & Technology 49, 14019-14027.

Wang, J.-J., Dahlgren, R.A., Erşan, M.S., Karanfil, T., Chow, A.T., 2015b. Wildfire Altering Terrestrial Precursors of Disinfection Byproducts in Forest Detritus. Environmental Science & Technology 49, 5921-5929.

Wang, J.-J., Dahlgren, R.A., Erşan, M.S., Karanfil, T., Chow, A.T., 2016. Temporal variations of disinfection byproduct precursors in wildfire detritus. Water Research 99, 66-73.

Wershaw, R.L., 2004. Evaluation of conceptual models of natural organic matter (humus) from a consideration of the chemical and biochemical processes of humification. In. US Geological Survey, Reston VA, p. 44.

Wershaw, R.L., Hayes, T.M., 2001. Solubilization of Anthropogenic Compounds by Humic Substances. In: Clapp, C.E., Hayes, M.H.B., Senesi, N., Bloom, P.R., Jardine, P.M. (Eds.), Humic Substances and Chemical Contaminants. Soil Science Society of America, Madison, WI, pp. 165-176.

Wershaw, R.L., Leenheer, J.A., Cox, L.G., 2005. Characterization of dissolved and particulate natural organic matter (NOM) in Neversink Reservoir, New York. In. U.S. Geological Survey, Reston VA, p. 24.

Wershaw, R.L., Rutherford, D.W., Leenheer, J.A., Kennedy, K.R., Cox, L.G., Koci, D.R., 2003. Biogeochemical processes that produce dissolved organic matter from wheat straw. In. U.S. Geological Survey, Reston VA, p. 14.

Wershaw, R.L., Llaguno, E.C., Leenheer, J.A., 1996a. Mechanism of formation of humus coatings on mineral surfaces 3. Composition of adsorbed organic acids from compost leachate on alumina by solid-state 13C NMR. Colloids and Surfaces A: Physicochemical and Engineering Aspects 108, 213-223.

Wershaw, R.L., Llaguno, E.C., Leenheer, J.A., Sperline, R.P., Song, Y., 1996b. Mechanism of formation of humus coatings on mineral surfaces 2. Attenuated total reflectance spectra of hydrophobic and hydrophilic fractions of organic acids from compost leachate on alumina. Colloids and Surfaces A: Physicochemical and Engineering Aspects 108, 199-211.

Wershaw, R.L., Leenheer, J.A., Sperline, R.P., Song, Y., Noll, L.A., Melvin, R.L., Rigatti, G.P., 1995. Mechanism of formation of humus coatings on mineral surfaces 1. Evidence for multidentate binding of organic acids from compost leachate on alumina. Colloids and Surfaces A: Physicochemical and Engineering Aspects 96, 93-104.

Wickramasekara, S., Hernández-Ruiz, S., Abrell, L., Arnold, R., Chorover, J., 2012. Natural dissolved organic matter affects electrospray ionization during analysis of emerging contaminants by mass spectrometry. Analytica Chimica Acta 717, 77-84.

Wu, Z., Rodgers, R.P., Marshall, A.G., 2004. Two- and Three-Dimensional van Krevelen Diagrams: A Graphical Analysis Complementary to the Kendrick Mass Plot for Sorting Elemental Compositions of Complex Organic Mixtures Based on Ultrahigh-Resolution Broadband Fourier Transform Ion Cyclotron Resonance Mass Measurements. Analytical Chemistry 76, 2511-2516.

Wymore, A.S., Rodríguez-Cardona, B., McDowell, W.H., 2015. Direct response of dissolved organic nitrogen to nitrate availability in headwater streams. Biogeochemistry 126, 1-10.

Yamashita, Y., Kloeppel, B.D., Knoepp, J., Zausen, G.L., Jaffé, R., 2011. Effects of Watershed History on Dissolved Organic Matter Characteristics in Headwater Streams. Ecosystems 14, 1110-1122.

Yanai, R.D., Currie, W.S., Goodale, C.L., 2003. Soil Carbon Dynamics after Forest Harvest: An Ecosystem Paradigm Reconsidered. Ecosystems 6, 197-212.

Zafiriou, O.C., Joussot-Dubien, J., Zepp, R.G., Zika, R.G., 1984. Photochemistry of natural waters. Environmental Science & Technology 18, 358A-371A.

Zhang, C., Jamieson, R.C., Meng, F.-R., Gordon, R., Bourque, C.P.-A., 2016. Projecting In-stream Dissolved Organic Carbon and Total Mercury Concentrations in Small Watersheds Following Forest Growth and Clearcutting. Water, Air, & Soil Pollution 227, 323.

Zhang, Y., van Dijk, M.A., Liu, M., Zhu, G., Qin, B., 2009. The contribution of phytoplankton degradation to chromophoric dissolved organic matter (CDOM) in eutrophic shallow lakes: Field and experimental evidence. Water Research 43, 4685-4697.

THIS PAGE INTENTIONALLY LEFT BLANK

CHAPTER 8. ASSESSMENT OF WILDFIRE EXPOSURE TO PUBLIC WATER SUPPLY AREAS IN OREGON

Michelle A. Day, Chris Ringo, and Alan A. Ager

8.1 Introduction

The cause of recent wildfire catastrophes can be traced to multiple factors including the expanding urban footprint (Radeloff et al. 2018), human ignitions (Nagy et al. 2018), droughts (Littell et al. 2016), and high-wind events (Abatzoglou et al. 2018). In 2018 alone, over 58,000 wildfires burned 8.8 million acres in the western US (NIFC 2018). As wildfire frequency and intensity increases (Westerling 2016, Abatzoglou et al. 2017), understanding the impacts of high-severity wildfire on ecosystem function is critical, particularly the negative effects on soils (Certini 2005) and drinking water source areas (Robinne et al. 2019).

Wildfires remove litter, duff and vegetative cover leading to the creation or enhancement of hydrophobic soil layers, increasing surface runoff and erosion potential (Neary et al. 2005, Larsen et al. 2009, Robichaud and Ashmun 2013). Large and severe wildfires can occur at the watershed scale and affect hydrologic processes including changes in stream flow, flood frequency, erosion and sedimentation (Neary et al. 2005, Smith et al. 2011). Post-fire changes in water chemistry, and sediment transport can increase pollutant loads, with related significant consequences for human health, safety and aquatic habitats (Morrison and Kolden 2015, Nunes et al. 2018, Rust et al. 2018, Hohner et al. 2019). In 2017 the Eagle Creek Fire east of Portland started in the Columbia River Gorge, burned over 48,000 acres and took three months to contain. It burned within one mile of the Bull Run Watershed that supplies drinking water to 1 million people within the Portland metropolitan area. The intensity of these effects are in turn related to burn severity, soil characteristics, topography, fuel type and post-fire weather conditions (Certini 2005, Shakesby and Doerr 2006).

The growing awareness of the expanding scale of wildfire risk to communities and watersheds and water supplies in the US, has led to a wide range of research focused on fuel treatments to reduce post-fire impacts to watersheds and drinking water. At the same time watershed investment programs are being initiated in the western United States to address wildfire risk to municipal water (City of Ashland 2019, FWPP 2019). Researchers are using wildfire simulation models to test hypothetical treatment scenarios and estimate the potential reduction in risk as measured by metrics that measure adverse impacts including soil erosion (Elliot et al. 2016, Jones et al. 2017) and change in water yield (Srivastava et al. 2018). Typically soil burn severity is quantified using gridded flame length outputs from fire models (Elliot 2016). These can be cross-walked to erosion predicted models like the watershed erosion prediction project (WEPP) and exiting geospatial data on potential fire effects (Miller et al. 2011, Flanagan et al. 2013). Financial analyses that compare the cost of fire mitigation to water supplies have shown both positive (Jones et al. 2017) and negative (Gannon et al. 2019) rates of return from fuel management programs depending on assumptions about fire occurrence. Wildfires are relatively rare, and using risk frameworks that incorporate probabilistic expected impacts (versus conditional that a fire occurs) undermines the cost benefit analyses unless other values can be included in rate of return investment schemes including avoided suppression costs, wildlife habitat, ecological restoration, recreation, and public safety (Gannon et al. 2019). Typically the fuel treatment studies that examine water issues are restricted to a watershed but now can be scaled up over large areas of the west using geospatial data on potential post-fire erosion rates for forests and shrublands (Miller et al. 2011). This latter work was completed by the Disturbed WEPP project using the GeoWEPP model.

Several new forest management authorities are being implemented that motivate increasing the scale of activities that span jurisdictions and landowner boundaries (USDA Forest Service 2018). These include authorizing legislation such as the good neighbor authority (2015) and the 2014 Farm Bill and the recent shared stewardship program (USDA Forest Service 2018). In turn, the growing emphasis on cross-boundary management of wildfire issues has motivated the research community to expand risk frameworks that are fine tuned to meet the needs of new authorizing legislation (Ager et al. 2018, Ager et al. 2019a, Ager et al. 2019b). ***For instance, existing risk assessment technologies and frameworks do not explicitly examine the cross-boundary problem intrinsic to wildfire risk from large public wildlands*** (WWWRA 2013, Dillon et al. 2014). Most risk assessments simply measure in situ risk, without a linkage to the source of large fires that typically start in wildlands long distances from developed areas and the sources of water they are dependent on (Robinne et al. 2018, Robinne et al. 2019). Clearly, in an era where the scale of risk is rapidly expanding with larger and larger fires, it is important to understand topological properties of cross-boundary fire on landscapes fragmented by ownership and jurisdictions.

In this report, we first summarize methods used to assess wildfire exposure and transmission and then provide a detailed assessment of cross-boundary wildfire exposure in Oregon between major land tenures (private, public, state, and federal) and drinking water source areas. The goal of the work is to provide decision support information to public and private fire mitigation programs. The outputs from this study can be specifically used to prioritize cross-boundary, shared stewardship projects aimed at reducing fire exposure to drinking water.

8.2 Methods

8.2.1. Wildfire Risk versus Exposure

Wildfire risk concerns the estimation of expected loss, calculated as the product of the likelihood of a fire at a given intensity and the consequence(s). By contrast, wildfire exposure concerns the juxtaposition of threatened values in relation to predicted fire occurrence and intensity, without estimating potential loss (SRA 2006). In this assessment we focus on wildfire exposure to reduce complexity and not bias the results with assumed loss functions that have high levels of uncertainty in terms of fire effects on public water supply areas (PWSA).

8.2.2 Study Area and Land Tenure Assignment

The study area included more than 150 land tenures in Oregon, grouped in 15 major classes derived from the Protected Areas Database of the United States (USGS 2016), and updated with private land tenure information from Pacific Northwest timberland ownership (Atterbury Consultants 2017) (Figure 8-1). The three largest major land tenures were the US Forest Service administered land (FS) (6.3 million ha), the Bureau of Land Management (BLM) (6.4 million ha) and private (non-industrial) land (8.2 million ha).

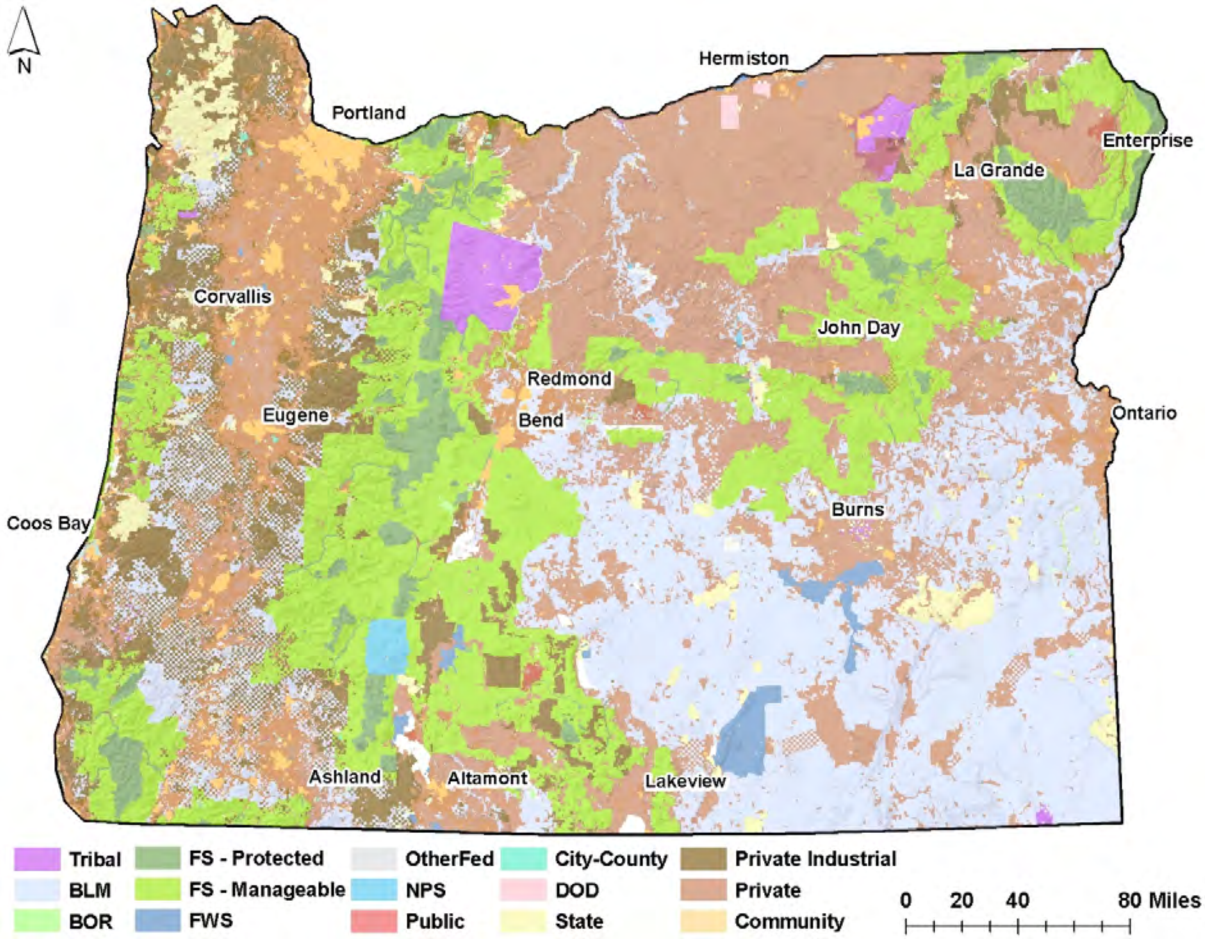


Figure 8-1. Land tenures in Oregon based on the Protected Areas Database (USGS 2016) and Pacific Northwest timberland ownership (Atterbury Consultants 2017). Figure from Day et al. (2018).

To assess cross-boundary exposure to PWSAs we divided the study area into public water supply reporting regions including the Cascades, Coastal, Northeast Oregon, Southwest Oregon and Willamette/Umpqua (Figure 8-2). We included 159 PWSAs in the analysis, although 19 experienced virtually no wildfire.

8.2.3 Wildfire Simulations

Wildfire simulation data from FSim (Finney et al. 2011; version 2016) were used to predict wildfire exposure within and among the land tenures and transmission into PWSAs. FSim generates daily wildfire scenarios for a large number of wildfire seasons using relationships between historical Energy Release Component (ERC) (Bradshaw et al. 1983) and historical fire occurrence. Wildfires are simulated with the minimum travel time (MTT) (Finney 2002) algorithm under weather conditions derived from time series analysis of historical weather. Weather data were derived from the network of remote automated weather stations located throughout the US (Zachariassen et al. 2003). Fuel models (Figure 8-3), canopy cover and canopy fuel layers were derived from LANDFIRE (2014). FSim outputs include the ignition location of each fire, fire perimeters, and grids of burn probability (Figure 8-4) and conditional

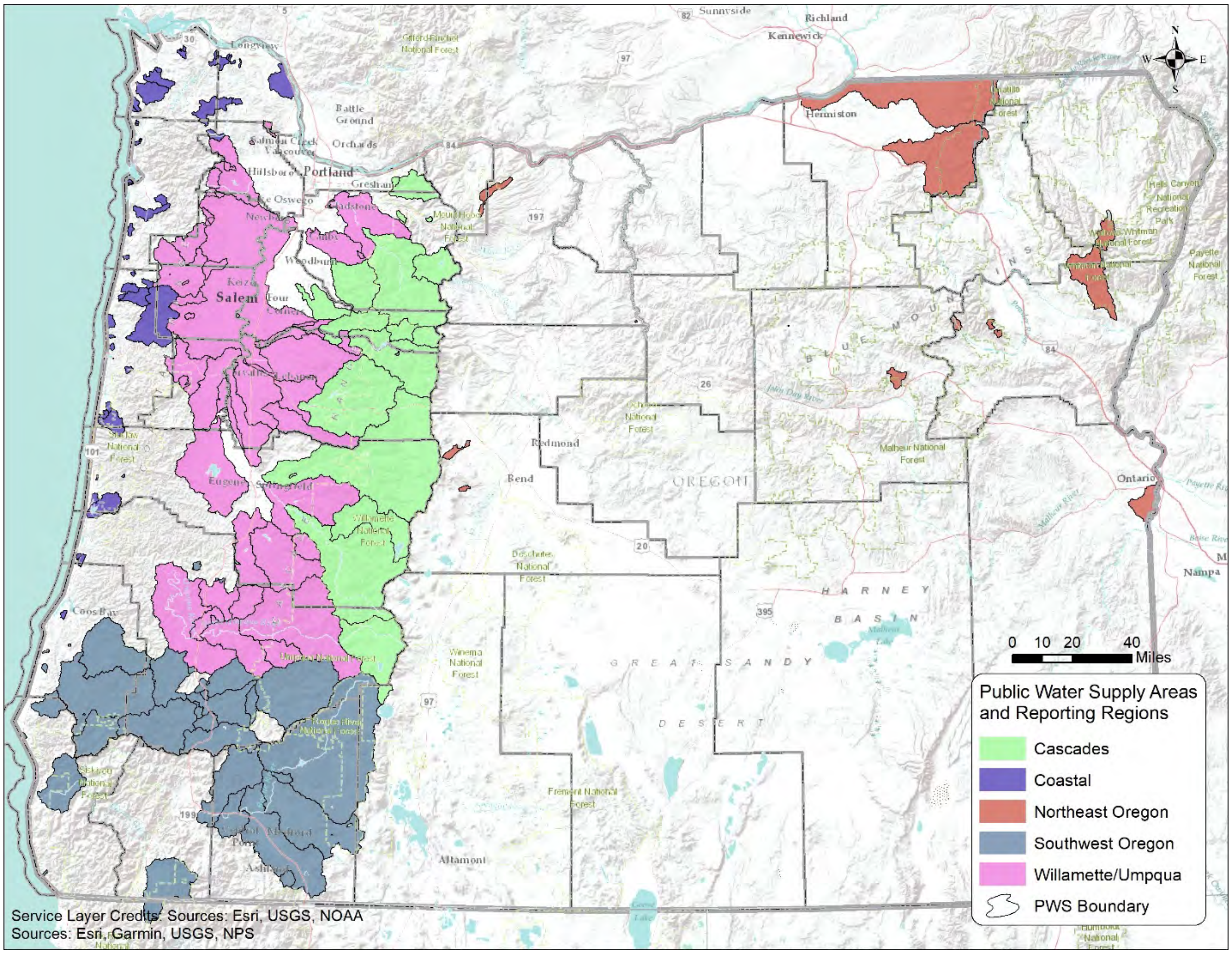


Figure 8-2. Public water supply areas and reporting regions used in data analysis.

probabilities by flame length category (Figure 8-5). The data used consisted of 1,430,417 ignitions simulated inside and within ca. 5 mi buffer around the PWSA layer, representing 10,000 fire season replicates depending on the region (Finney et al. 2011).

We also calculated and mapped two wildfire exposure metrics to illustrate the spatial scale and complexity of wildfire exposure in relation to the geography of land tenure across the state. Each metric was calculated at 500 × 500 m pixel resolution. The metrics describe both the scale and composition of fire effects that ignite elsewhere and arrive at a given pixel. The fire size potential index was the average fire size (ha) that was generated by an ignition in each pixel. Here, each simulated fire was attributed to the ignition point and the points smoothed to create a continuous raster coverage. The fire size arrival index measured the average fire size (ha) that burned each pixel.

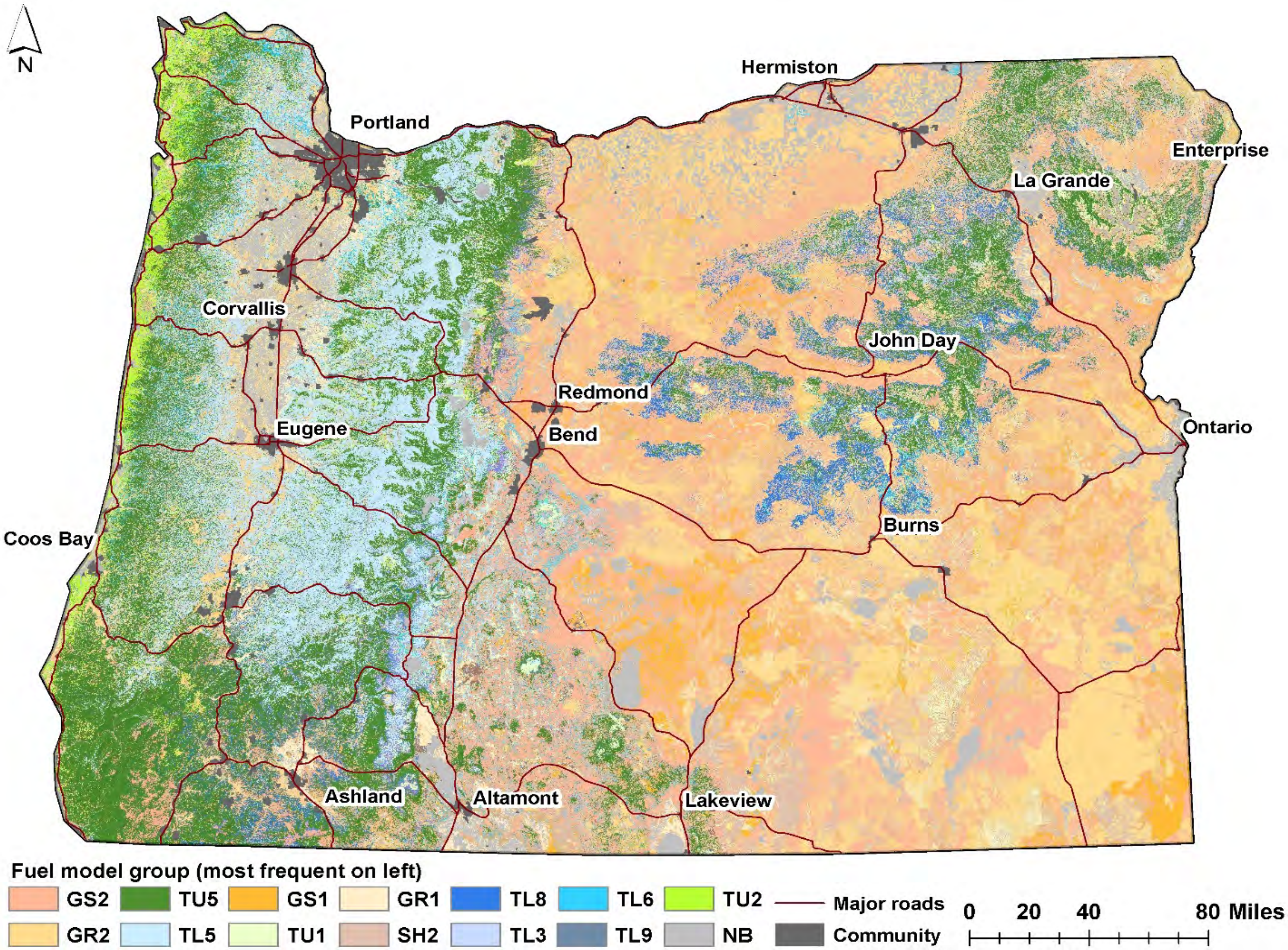


Figure 8-3. Fuel models used in the wildfire transmission analysis were derived from LANDFIRE (2014) based on Scott and Burgan (2005 (Day et al. 2018). Note the LANDFIRE version used in the FSim analysis was 2012.

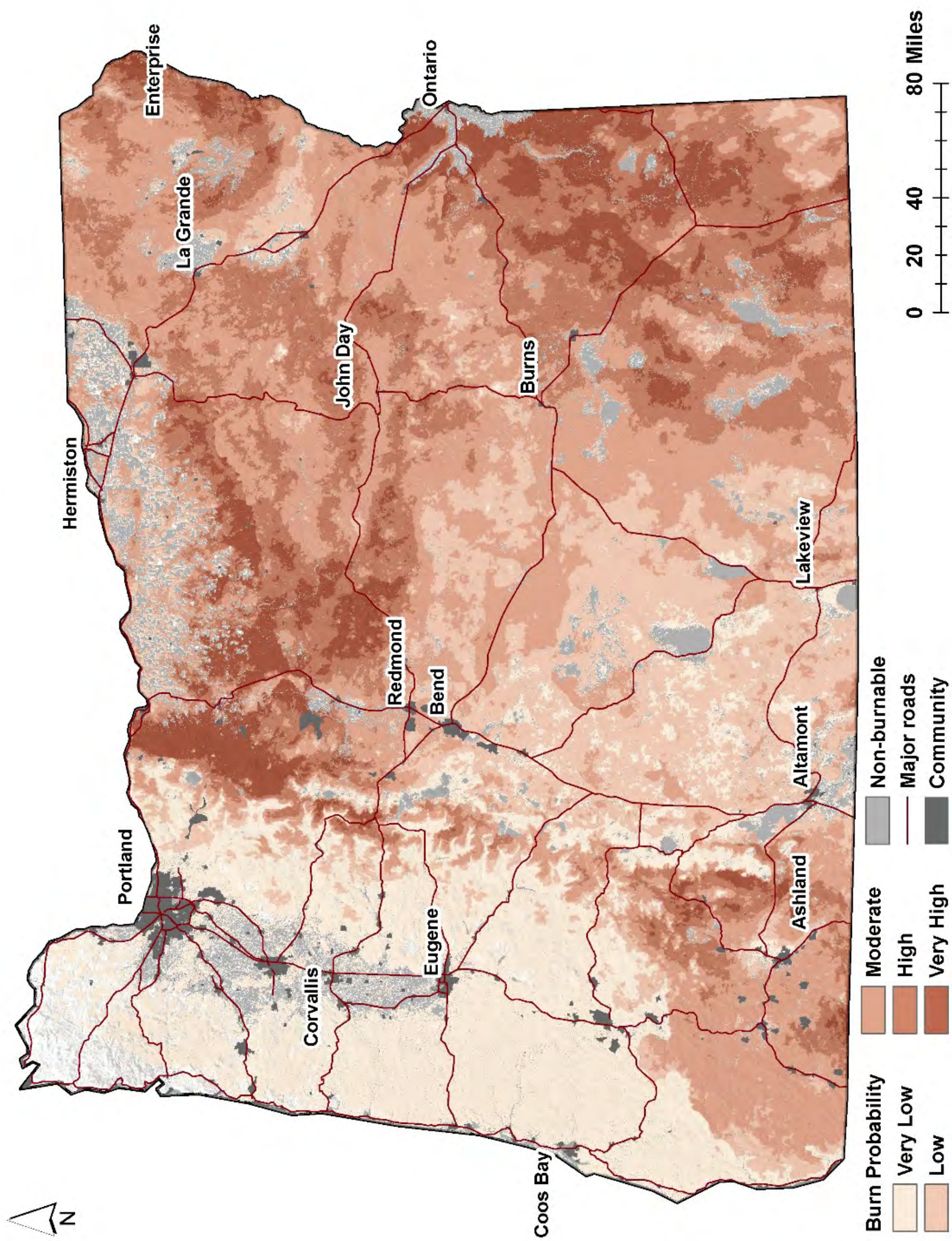


Figure 8-4. Annual burn probability estimated from simulation modeling (Day et al. 2018).

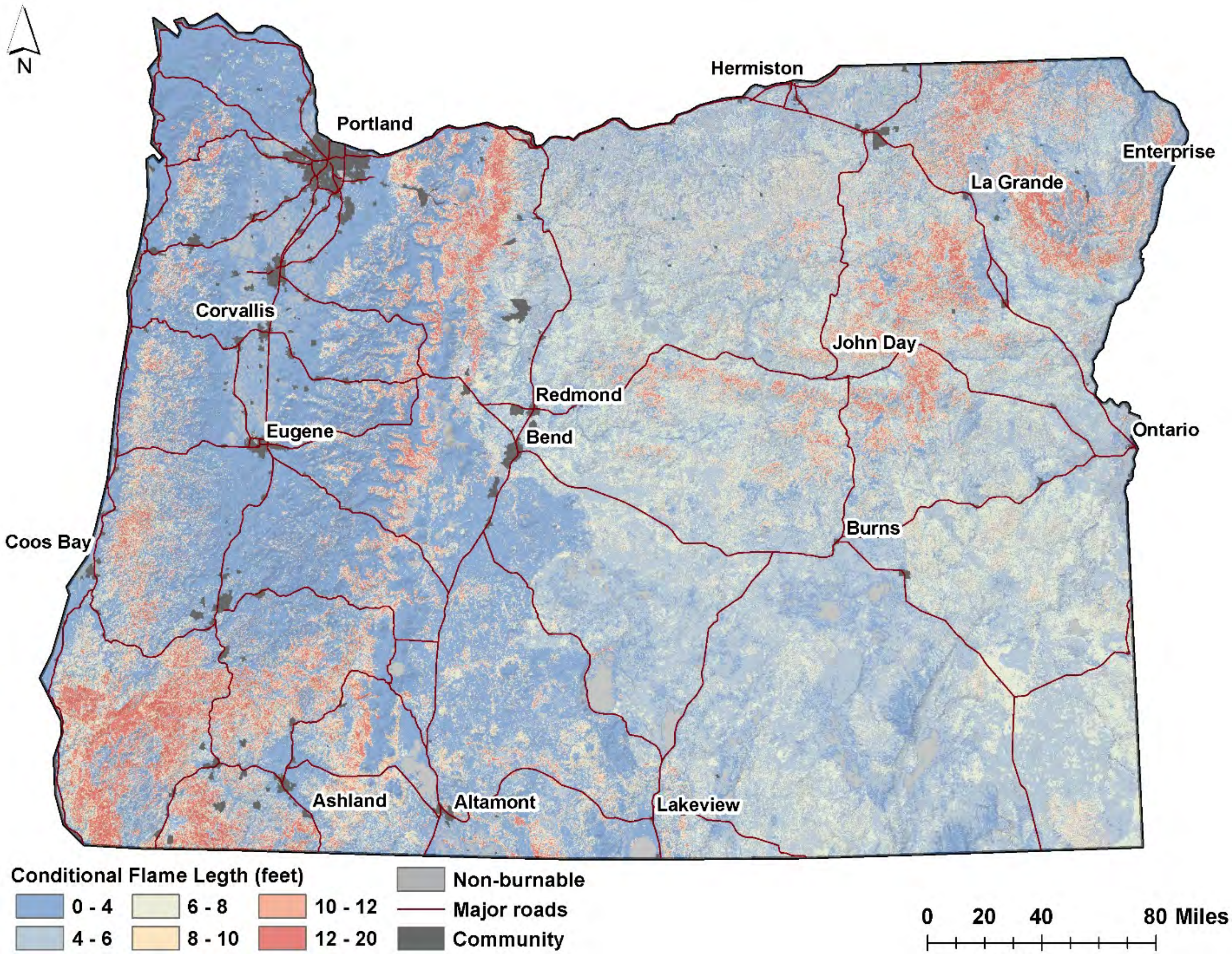


Figure 8-5. Mean flame length or predicted wildfire intensity estimated from simulation modeling (Day et al. 2018). Higher values represent higher potential for crown fire activity.

8.2.4 Quantifying Cross-Boundary Wildfire

Analysis of wildfire transmission was conducted at the PWSA and PWS region scales, similar to the methods described in Ager et al. (2017a) and Ager et al. (2018). Cross-boundary wildfire was quantified by intersecting wildfire perimeters with major land tenures and PWSAs in Oregon (Figures 8-1 and 8-2). Polygons were dissolved by the major land tenure to avoid a false fragmentation within the same agency/land owner. The origin of each wildfire was assigned based on the point of ignition. Total burned area within each PWSA was aggregated by incoming fire (Incoming, the area burned of all fires ignited outside the PWSA and entering each particular PWSA), outgoing fire (Outgoing, the area burned of all fires ignited in a PWSA that escapes its boundaries) and self-burning or non-transmitted (the area burned within a PWSA by ignitions in the same PWSA)(Figure 8-6). We analyzed wildfire transmission to 1) delineate the areas that send fire into each PWSA, 2) quantify the ownership breakdown of that contributing area, and 3) assess the fire intensity, frequency, and size of the fires burning into each PWSA. PWS regional results are presented here and results by individual PWSA will be presented in an online atlas.

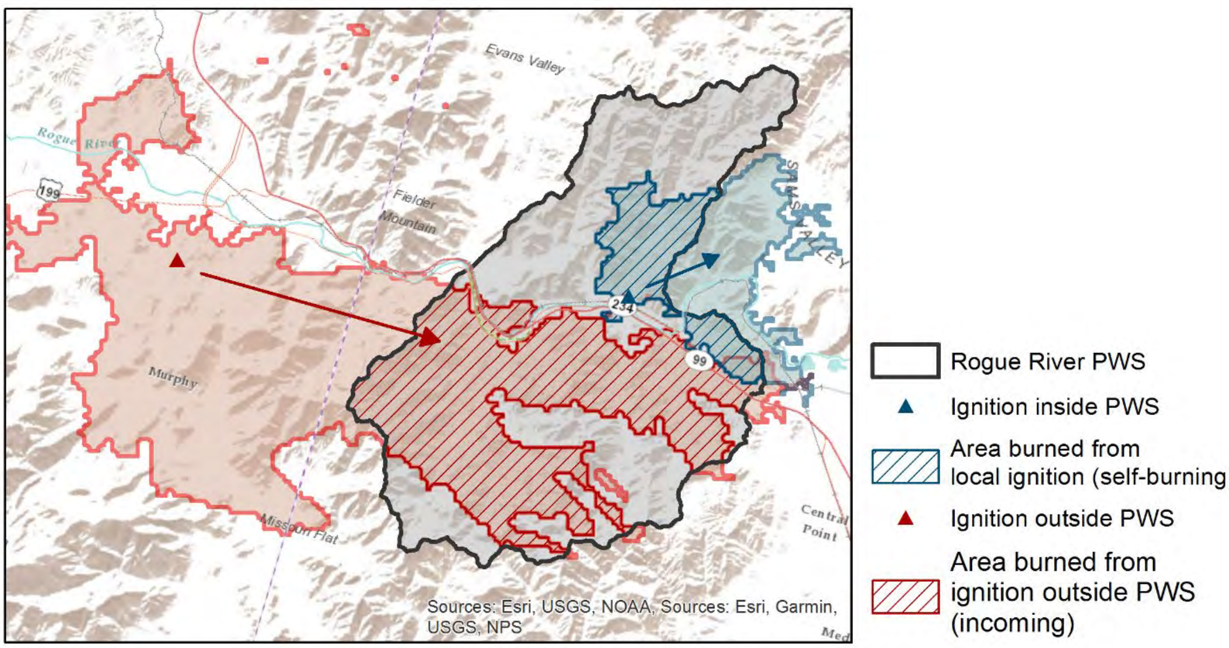


Figure 8-6. Cross-boundary fire components for an example public water supply area (PWSA) and example simulated fires. Cross-boundary exposure to PWSAs was calculated by intersecting simulated wildfire perimeters with PWSA boundaries and attributing wildfire exposure to the source land parcel (red triangle represents ignition outside of a PWSA). Arrows indicate direction of fire spread. Wildfires ignited locally are considered self-burning; wildfires ignited outside of the PWS and burning inside are considered incoming.

8.2.5 Fireshed Mapping

We used wildfire simulation results to identify the areas where large fires are likely to ignite and exposure PWSAs. These “firesheds” define the biophysical risk containers in and around PWSAs and the sources of risk in terms of ownership. Firesheds can be further characterized by fire regime and management capability although this was beyond the scope of the current work. We mapped PWSA firesheds by creating a continuous smoothed surface of predicted wildfire exposure from all FSim ignitions that resulted in fires that intersected PWSA polygons. We used inverse distance weighting geostatistical interpolation, implemented through the ArcGIS geostatistical analyst module (ESRI 2013), using a 5 km fixed search radius. In addition to a state-wide fireshed map, firesheds were developed individually for each PWSA and will be available in an online atlas.

8.3 Results

9.3.1 Exposure Metrics

We quantified and mapped the scale of wildfire exposure in the study area with the two exposure metrics as described above. The fire size potential index (Figure 8-7) identified locations that generated the largest fires, with the highest values observed for southwestern Oregon and parts of northeastern and eastern Oregon. The fire size arrival index (Figure 8-8) estimated the average size of the fire that burned each pixel, with the highest values again in southwest Oregon but also large areas of central and southeast Oregon.

8.3.2 Predicted Wildfire Exposure by Public Water Supply Area and Region

Predicted area burned in 100 years was highest for PWSAs in the eastern Cascades, southwest Oregon and in eastern Oregon (Figures 8-10 and 8-11; Table 8-1). Mean fire size, total annual area burned and the number of simulated fires that exposed PWSAs also varied substantially across the regions (Table 1) with the largest fires and the highest area burned occurring in southwestern Oregon. The individual PWSA with the highest exposure on a percentage basis was the City of The Dalles, although 16% of the PWSAs had no fire exposure and 64% were exposed on less than 1% of their total area (Table 2). Although these numbers are small, wildfire risk often comes from extreme but rare events. The average size of the largest fire over 10,000 simulated fires season that burned PWSAs was 121,314 acres and values ranged from 40 acres for the City of Veronia to 654,013 acres for the City of Gold Hill (Table 8-2).

Table 8-1. Wildfire exposure to public water supply area (PWSA) regions in Oregon.

PWSA Region	Number of simulated fires	Mean fire size (acre)	Total annual area burned (acre)	Percentage burned in 100 yrs
Southwest	222,652	2,769	12,552	2.0 (0.01-6.4)
Cascades	198,033	1,073	6,593	1.7 (0.02-6.6)
Northeast	59,654	2,717	4,138	5.1 (0.9-17.3)
Willamette/Umpqua	290,574	268	806	0.1 (0-0.88)
Coastal	27,181	440	6	0.01 (0-0.07)

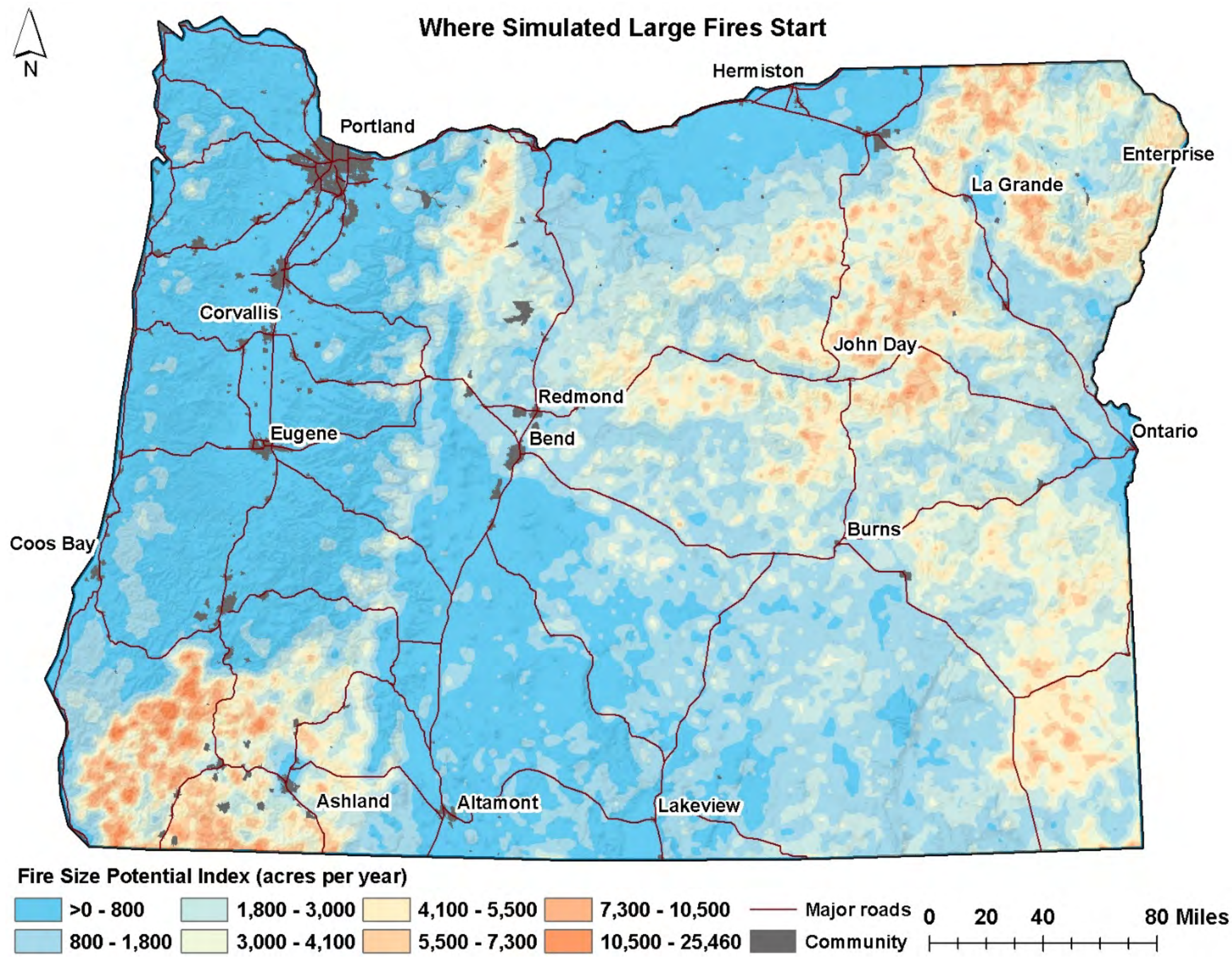


Figure 8-7. Fire size potential index is the average fire size that was generated by a simulated ignition in a given place and shows the potential fire sources. Figure from Day et al. (2018).

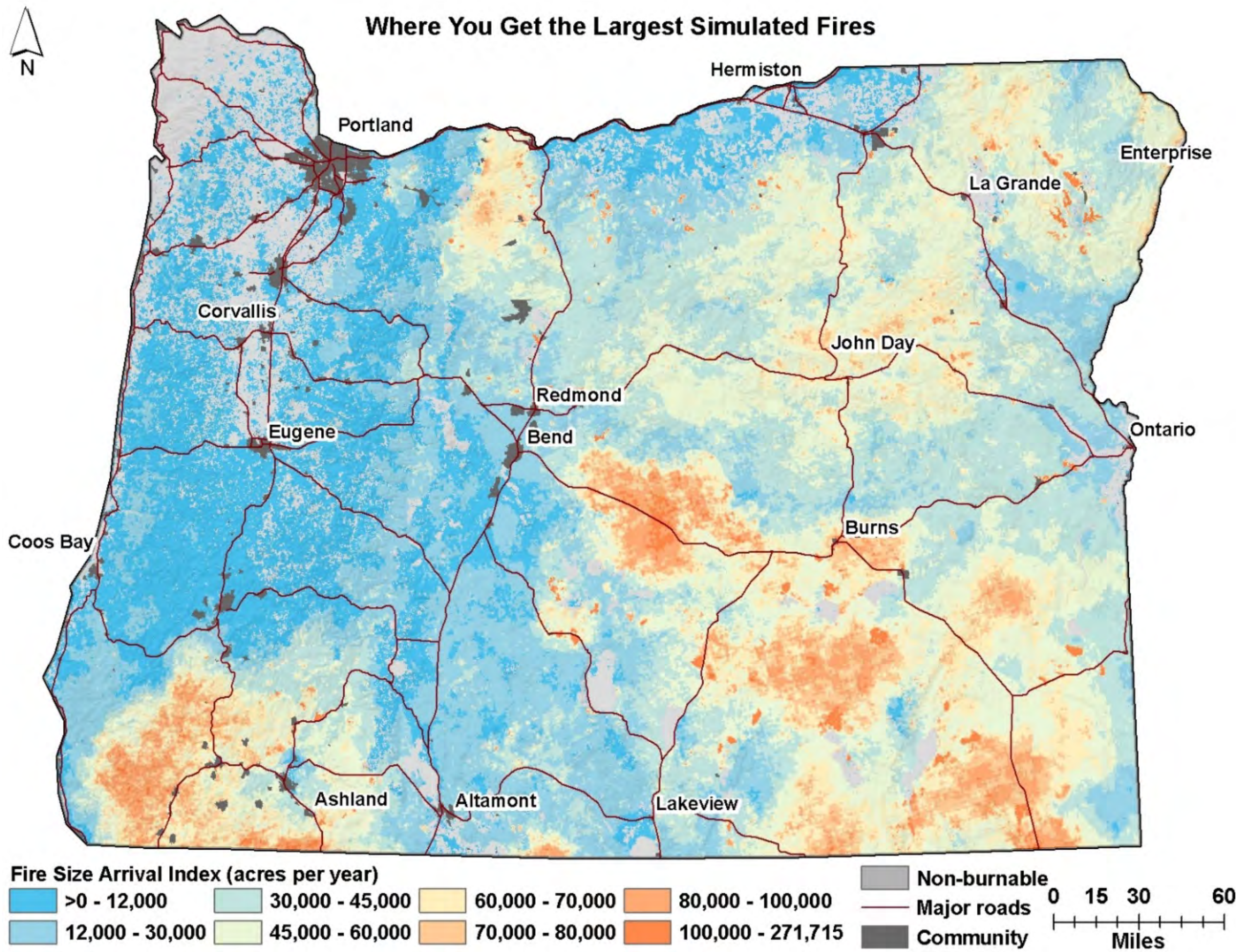


Figure 8-8. Fire size arrival index is the average fire size of all simulated fires that arrived at a given place and shows the areas at risk from large fires (Day et al. 2018).

Table 8-2. Wildfire exposure as measured by percentage area burned in 100 years for the top 25 public water supply areas (PWSA) in Oregon.

PWSA Name	Rank	PSWA Area (acres)	Area burned in 100 yrs (acres)	(%)	Number of land tenures contributing to exposure[1]	Largest fire (acres)[2]
City of The Dalles	1	20,560	3,555	17.3	7	290,345
Young Life Wash Family Ranch	2	242	22	9.2	3	185,016
USFS Timber Lake JCC	3	83,672	5,511	6.6	2	248,166
Prairie City	4	15,499	1,006	6.5	4	528,367
Richland, City Of	5	113,940	7,284	6.4	4	311,610
City of Cave Junction	6	148,775	9,490	6.4	7	490,695
City of Sumpter	7	6,723	411	6.1	4	528,367
Baker City	8	6,843	411	6.0	5	232,739
Country View MH Estates	9	734,026	42,930	5.8	7	312,358
Ashland Water Department	10	12,736	644	5.1	5	598,588
City of Grants Pass	11	170,960	8,340	4.9	7	521,502
Breitenbush Hot Springs	12	35,722	1,701	4.8	4	145,679
City of Pendleton	13	283,054	13,182	4.7	6	338,577
Medford Water Commission	14	289,951	13,282	4.6	7	312,358
City of Rogue River	15	69,007	3,022	4.4	6	521,502
City of Glendale	16	119,381	5,137	4.3	7	521,502
City of Canyonville	17	22,657	930	4.1	6	521,502
USFS Tiller Ranger Station	18	288,523	11,675	4.0	4	237,655
Angler's Cove/SCHWC	19	10,703	375	3.5	4	296,475
City of Gates	20	240,452	7,722	3.2	7	141,331
City of Gold Hill	21	284,023	9,020	3.2	7	654,013
City of Riddle	22	192,494	5,395	2.8	7	521,502
City of Hermiston	23	390,040	10,675	2.7	7	338,577
PP&L-Toketee Village	24	224,206	5,763	2.6	2	139,507
City of Ontario	25	44,355	1,057	2.4	3	196,307

[1] Number of land tenures where fires ignite and burn into the PWSA

[2] Largest fire to exposure the PWSA

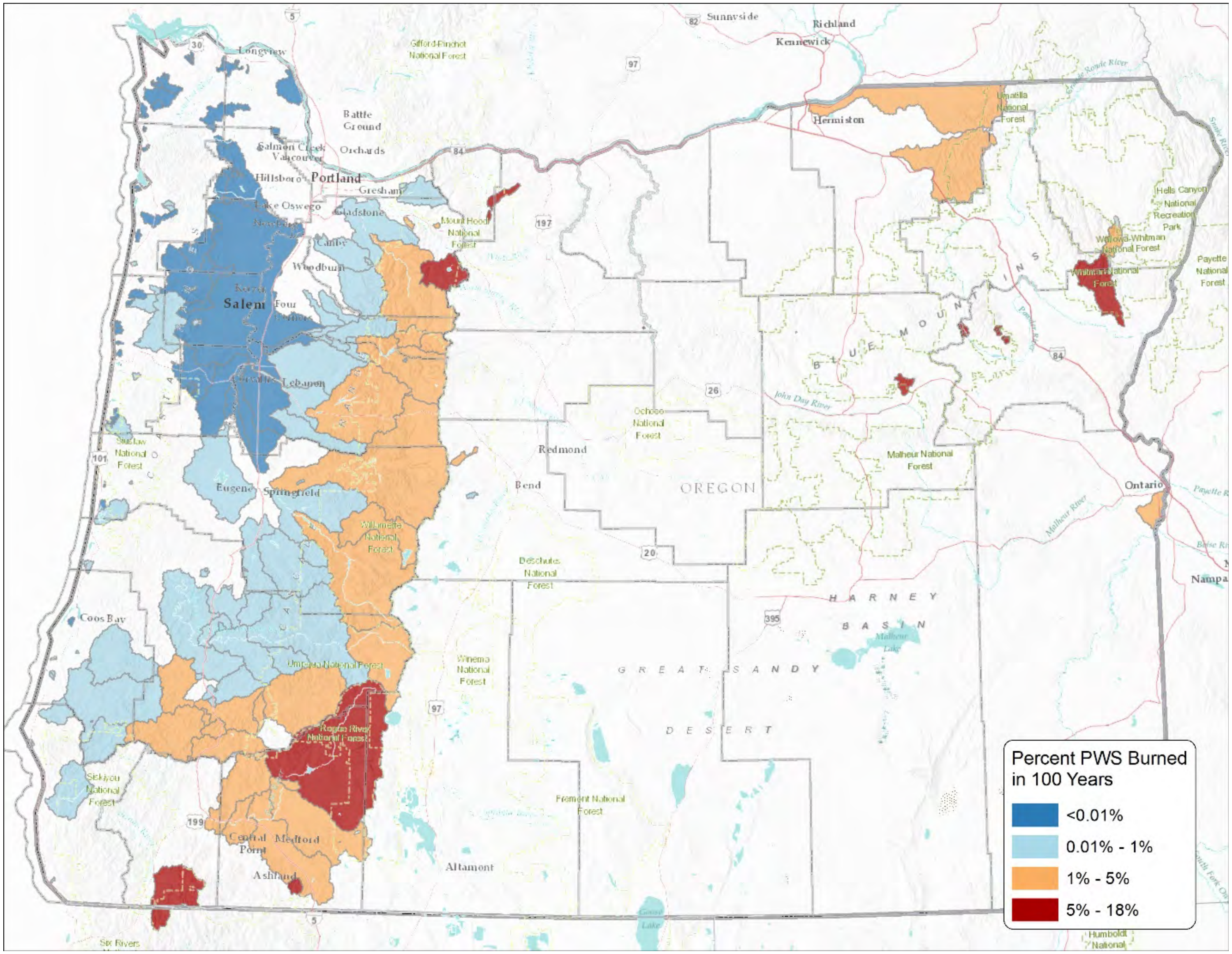


Figure 8-9. Predicted percentage area burned in 100 years of public water supply areas (PWSA) showing the relative differences in PSWA exposure across the state.

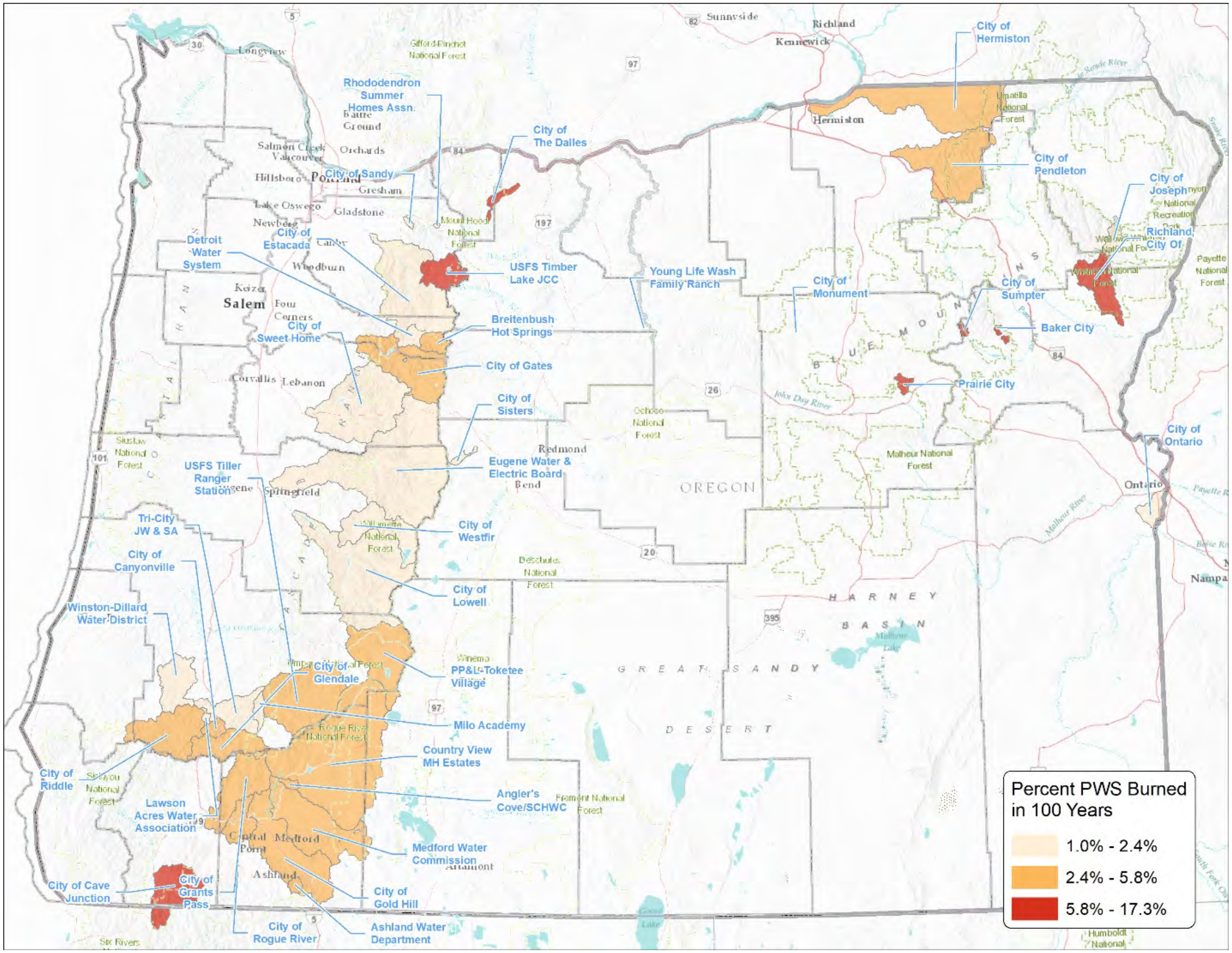


Figure 8-10. Predicted percentage area burned in 100 years for the 40 public water supply areas (PWSA) with highest exposure to wildfires.

8.3.3 Predicted Wildfire Transmission by Land Tenure

There was high variability among the major land tenures and their contribution to PSWA wildfire exposure within and among PWSA regions (Figure 8-11). The US Forest Service (Federal-FS) was the leading contributor to area burned in all but the Coastal region where private industrial lands were the largest contributor.

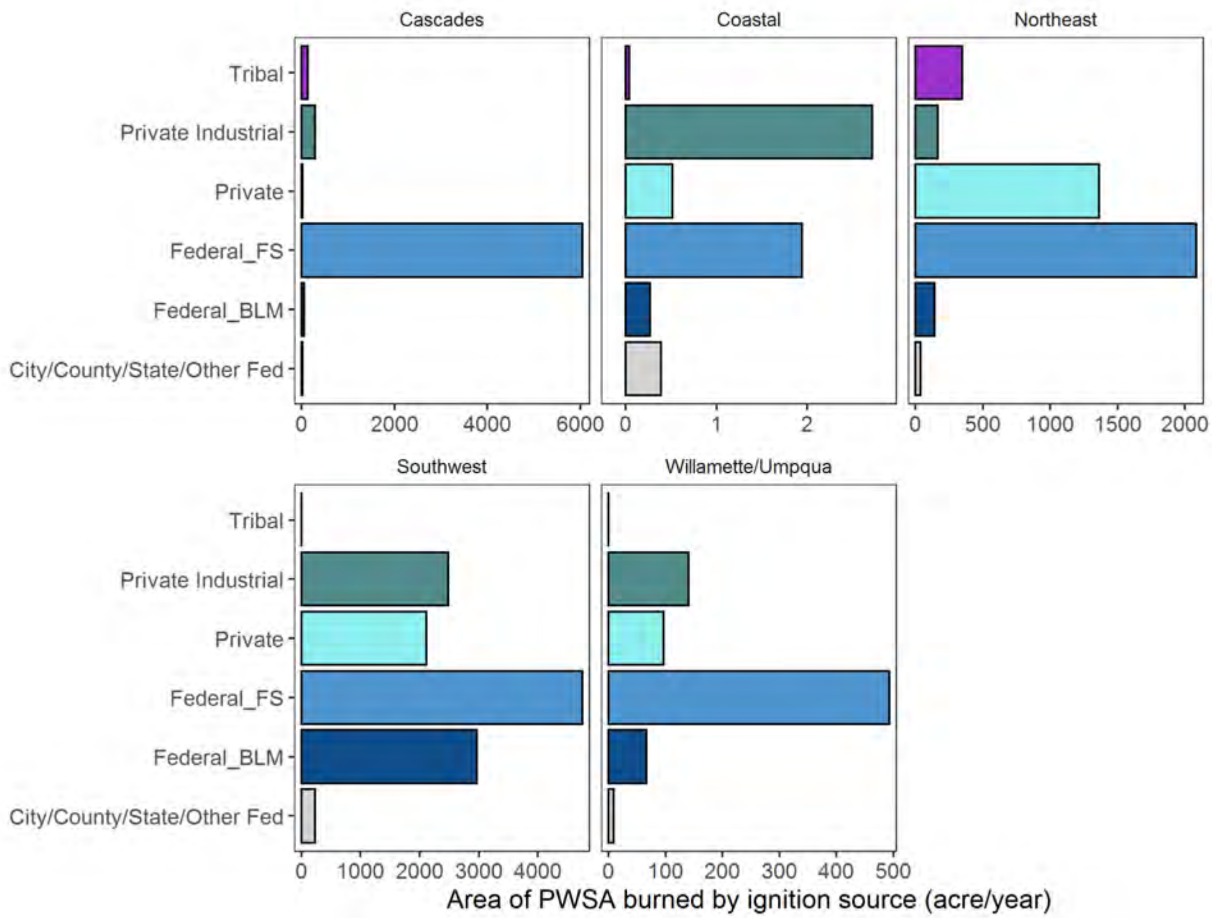


Figure 8-11. Predicted annual area burned by ignition source in public water supply area (PWSA) regions. Note the differences in the scale of the x-axis panels.

8.3.4 Public Water Supply Area Firesheds

Firesheds were generated for each of the 140 PWSAs that experienced wildfire in our simulations. Firesheds represent the biophysical risk containers in and around PWSAs and the sources of risk in terms of ownership; they represent areas surrounding each PWSA that can ignite and transmit large wildfires that expose an individual PWSA. Fireshed boundaries are often magnitudes larger than the administrative boundary of the PWSA and can represent a mosaic of land tenures. As an example, the fireshed of the City of Rogue River PWSA is 12 times larger (830 thousand acres) than the PWSA itself with four land tenures as the major sources of exposure (Figure 8-12). Mitigation of wildfire exposure in this example would require the collaborative planning by one federal agency, city/county managers, and representative of the private and private industrial communities. In contrast, the fireshed of The Dalles is 22 times larger than The Dalles PWSA and would require collaborative planning with six land tenures, although 77% of the exposure comes from national forest lands (Figure 8-13).

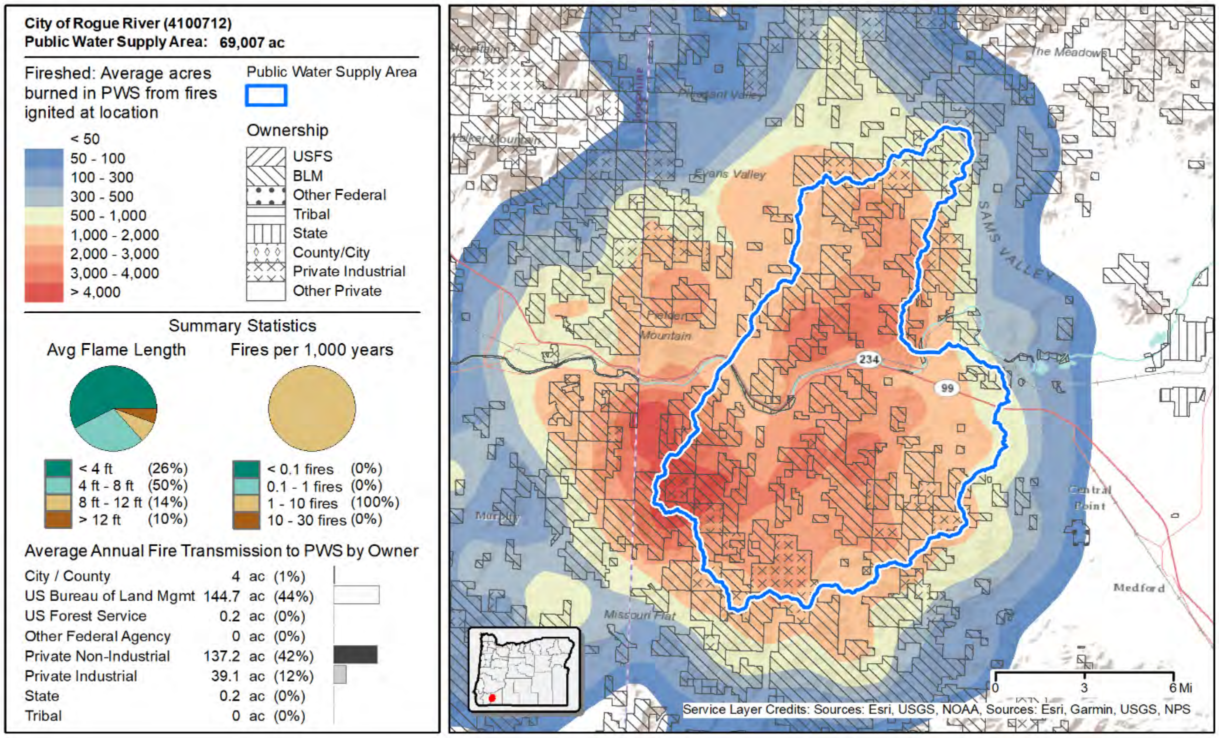


Figure 8-12. Example cross-boundary wildfire analysis results for an individual public water supply area (PWSA), City of Rogue River. Results for all PWSAs can be found in an online atlas.

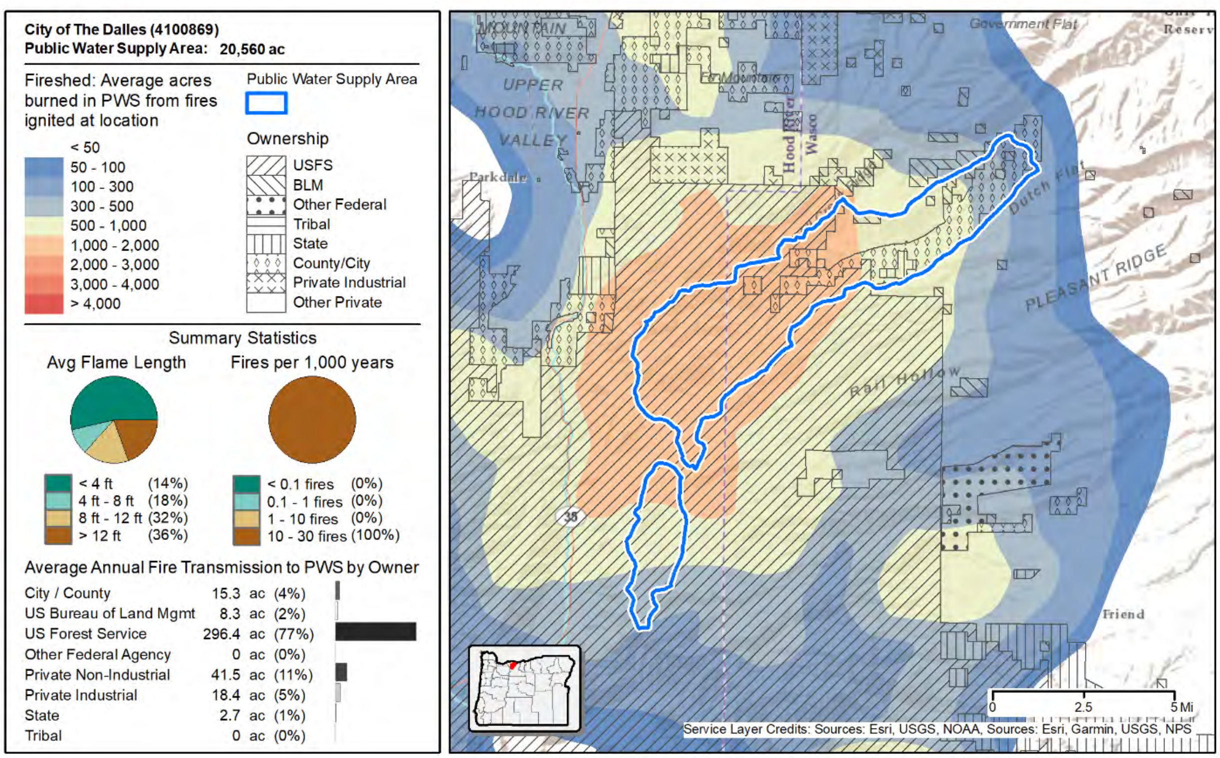


Figure 8-13. Example cross-boundary wildfire analysis results for an individual public water supply area (PWSA), City of The Dalles. Results for all PWSAs can be found in an online atlas.

8.4 Discussion

The juxtaposition of fire prone forests in and around critical municipal watersheds intermixed with a high number of homes and infrastructure, and in close proximity to dense urban areas under a changing climate, creates a complex fuel management problem. Our analysis showed that, while rare, large and severe fire events will continue to occur, especially in the southwest, eastern cascades and eastern portions of the state exposing public water supply areas. Our analysis also showed that if forest management has the potential to reduce fuels and restore ecological resiliency, the scale of the risk will required a coordinated, multi-agency, multi land owner collaborative response. Thus, coordinated and targeted fuel management and forest restoration activities that minimize the risk of fire exposure to public water supply areas, maximize landscape resilience to wildfire, and expand decision space for beneficial wildfire management will be needed (Stephens et al. 2016).

Translating the findings in this report to prioritize fuel management activities is straightforward. Maps of fire transmission to PWSAs can be used as priorities in scenario planning models (Ager et al. 2011, Ager et al. 2017b) to design and sequence project areas and treatment units within them. Including potential treatment costs and revenues associated with harvesting and fuels treatments into planning makes it possible to examine economic costs and benefits associated with forest management to protect water. Optimization models can also be used to locate treatments to address multiple values and risk, including wildfire transmission to the WUI, forest health, and wildfire risk to other values. Since cost benefit analyses generally do not show benefits from forest management to water supplies (Gannon et al. 2019), identifying the manifold effects of treatments can at help expand the treatment footprints. Novel tax funding mechanisms used in cities like Ashland and Flagstaff (City of Ashland 2019, FWPP 2019) to fund fuel treatments should take advantage of assessments like that reported here to strategically treat high transmission areas.

Our Fireshed maps are also useful for identifying the scale of risk to PWSAs and determining the relative contribution from different landowners. The scale of risk is typically underestimated in risk reduction planning efforts, and as fires grow larger under a changing climate the scale of risk continues to increase. Newer initiatives like shared stewardship (USDA Forest Service 2018) recognizes that the increasing scale of risk requires cross-boundary prioritization and action to treat at the appropriate scale. The core idea in this initiative is to expand land treatments across boundaries to reduce the scale mismatch between wildfire risk and the current forest management footprint. However, the process will require spatial planning to co-prioritize projects, meaning that respective federal and state assessments on land conditions (threats and opportunities) will require a multi-criteria approach to integrate the respective priorities identified in agency and state assessments and understand trade-offs (Ager et al. 2018). Assessments of cross-boundary risk, such as the work presented here, can be integrated into this process and used as a management objective to target forest management where wildfires are predicted to spread across federal and state boundaries and expose drinking water or other highly valued resources.

8.5 Literature Cited

Abatzoglou, J. T., J. K. Balch, B. A. Bradley, and C. A. Kolden. 2018. Human-related ignitions concurrent with high winds promote large wildfires across the USA. International Journal of Wildland Fire 27:377-386.

Abatzoglou, J. T., C. A. Kolden, A. P. Williams, J. A. Lutz, and A. M. S. Smith. 2017. Climatic influences on interannual variability in regional burn severity across western US forests. International Journal of Wildland Fire 26:269.

Ager, A. A., M. A. Day, P. Palaiologou, R. Houtman, and C. Ringo. 2019a. Cross-boundary wildfire and community exposure: A framework and application in the western US. Gen. Tech. Rep. RMRS-GTR-392, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Ager, A. A., C. R. Evers, M. A. Day, H. K. Preisler, A. M. Barros, and M. Nielsen-Pincus. 2017a. Network analysis of wildfire transmission and implications for risk governance. PLoS ONE 12: e0172867. doi: 10.1371/journal.pone.0172867 http://dx.doi.org/10.1371/journal.pone.0172867.

Ager, A. A., P. Palaiologou, C. Evers, M. A. Day, C. Ringo, and K. C. Short. 2019b. Wildfire exposure to the wildland urban interface in the western US. Applied Geography 111: 102059. doi: 10.1016/j.apgeog.2019.102059 https://doi.org/10.1016/j.apgeog.2019.102059.

Ager, A. A., P. Palaiologou, C. R. Evers, M. A. Day, and A. M. G. Barros. 2018. Assessing transboundary wildfire exposure in the southwestern United States. Risk Analysis 38:2105-2127.

Ager, A. A., N. M. Vaillant, and M. A. Finney. 2011. Integrating fire behavior models and geospatial analysis for wildland fire risk assessment and fuel management planning. Journal of Combustion 572452: 19. doi: 10.1155/2011/572452 http://dx.doi.org/10.1155/2011/572452.

Ager, A. A., K. C. Vogler, M. A. Day, and J. D. Bailey. 2017b. Economic opportunities and trade-offs in collaborative forest landscape restoration. Ecological Economics 136:226-239.

Atterbury Consultants. 2017. Pacific Northwest Timberland Ownership Maps.

Bradshaw, L. S., J. E. Deeming, R. E. Burgan, and J. D. Cohen. 1983. The 1978 National Fire-Danger Rating System: Technical documentation. Page 44. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. Oecologia 143:1-10.

City of Ashland. 2019. Ashland Forest Resiliency Stewardship Project. https://www.ashland.or.us/SectionIndex.asp?SectionID=503&utm_source=watershed&utm_medium=web&utm_campaign=watershed.

Day, M. A., A. A. Ager, P. Palaiologou, and C. Ringo. 2018. Cross-boundary wildfire transmission in Oregon -- A quantitative transmission analysis. Final report to the Oregon Department of Forestry, USDA Forest Service, Rocky Mountain Research Station, National Fire Decision Support Center.

Dillon, G. K., J. Menakis, and F. Fay. 2014. Wildland fire potential: a tool for assessing wildfire risk and fuel management needs. Pages 60-76 *in* R. E. Keane, M. Jolly, R. Parsons, and K. Riley, editors. Large wildland fire conference, RMRS-P-73. USDA Forest Service, Rocky Mountain Research Station, Missoula, MT.

Elliot, W. J., M. E. Miller, and N. Enstice. 2016. Targeting forest management through fire and erosion modelling. International Journal of Wildland Fire.

ESRI. 2013. ArcGIS Desktop: Release 10.2. Environmental Systems Research Institute, Redlands, CA.

Finney, M. A. 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research 32:1420-1424.

Finney, M. A., C. W. McHugh, I. C. Grenfell, K. L. Riley, and K. C. Short. 2011. A simulation of probabilistic wildfire risk components for the continental United States. Stochastic Environmental Research and Risk Assessment 25:973–1000. http://www.treesearch.fs.fed.us/pubs/39312.

Flanagan, D. C., J. R. Frankenberger, T. A. Cochrane, C. S. Renschler, and W. J. Elliot. 2013. Geospatial application of the water erosion prediction project WEPP model. American Society of Agricultural and Biological Engineers 56:591-601.

FWPP. 2019. Flagstaff Watershed Protection Project. http://flagstaffwatershedprotection.org/.

Gannon, B. M., Y. Wei, L. H. MacDonald, S. K. Kampf, K. W. Jones, J. B. Cannon, B. H. Wolk, A. S. Cheng, R. N. Addington, and M. P. Thompson. 2019. Prioritising fuels reduction for water supply protection. International Journal of Wildland Fire 28:785-803.

Hohner, A. K., C. C. Rhoades, P. Wilkerson, and F. L. Rosario-Ortiz. 2019. Wildfires alter forest watersheds and threaten drinking water quality. Accounts of chemical research 52:1234-1244.

Jones, K. W., J. B. Cannon, F. A. Saavedra, S. K. Kampf, R. N. Addington, A. S. Cheng, L. H. MacDonald, C. Wilson, and B. Wolk. 2017. Return on investment from fuel treatments to reduce severe wildfire and erosion in a watershed investment program in Colorado. Journal of Environmental Management 198:66-77. https://www.ncbi.nlm.nih.gov/pubmed/28501609.

Larsen, I. J., L. H. MacDonald, E. Brown, D. Rough, M. J. Welsh, J. H. Pietraszek, Z. Libohova, J. de Dios Benavides-Solorio, and K. Schaffrath. 2009. Causes of post-fire runoff and erosion: water repellency, cover, or soil sealing? Soil Science Society of America Journal 73:1393-1407.

Littell, J. S., D. L. Peterson, K. L. Riley, Y. Q. Liu, and C. H. Luce. 2016. A review of the relationships between drought and forest fire in the United States. Global Change Biology 22:2353-2369. <Go to ISI>://WOS:000378722000006.

Miller, M. E., L. H. MacDonald, P. R. Robichaud, and W. J. Elliot. 2011. Predicting post-fire hillslope erosion in forest lands of the western United States. International Journal of Wildland Fire 20:982-999.

Morrison, K. D. and C. A. Kolden. 2015. Modeling the impacts of wildfire on runoff and pollutant transport from coastal watersheds to the nearshore environment. J Environ Manage 151:113-123. https://www.ncbi.nlm.nih.gov/pubmed/25549866.

Nagy, R., E. Fusco, B. Bradley, J. T. Abatzoglou, and J. Balch. 2018. Human-related ignitions increase the number of large wildfires across U.S. ecoregions. Fire 1: 4. doi: https://doi.org/10.3390/fire1010004 https://doi.org/10.3390/fire1010004.

Neary, D. G., K. C. Ryan, and L. F. DeBano. 2005. Wildland fire in ecosystems: effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-vol. 4., USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

NIFC. 2018. Wildland fire summary and statistics annual report. National Interagency Coordination Center, Boise, Idaho.

Nunes, J. P., S. H. Doerr, G. Sheridan, J. Neris, C. Santín Nuño, M. B. Emelko, U. Silins, P. R. Robichaud, W. J. Elliot, and J. Keizer. 2018. Assessing water contamination risk from vegetation fires: Challenges, opportunities and a framework for progress. Hydrological Processes 32:687–694.

Radeloff, V. C., D. P. Helmers, H. A. Kramer, M. H. Mockrin, P. M. Alexandre, A. Bar-Massada, V. Butsic, T. J. Hawbaker, S. Martinuzzi, and A. D. Syphard. 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences 115:3314-3319.

Robichaud, P. R. and L. E. Ashmun. 2013. Tools to aid post-wildfire assessment and erosion-mitigation treatment decisions. International Journal of Wildland Fire 22:95-105.

Robinne, F.-N., K. D. Bladon, U. Silins, M. B. Emelko, M. D. Flannigan, M.-A. Parisien, X. Wang, S. W. Kienzle, and D. P. Dupont. 2019. A regional-scale index for assessing the exposure of drinking-water sources to wildfires. Forests 10:384.

Robinne, F. N., K. D. Bladon, C. Miller, M. A. Parisien, J. Mathieu, and M. D. Flannigan. 2018. A spatial evaluation of global wildfire-water risks to human and natural systems. Sci Total Environ 610-611:1193-1206. https://doi.org/10.1016/j.scitotenv.2017.08.112.

Rust, A. J., T. S. Hogue, S. Saxe, and J. McCray. 2018. Post-fire water-quality response in the western United States. International Journal of Wildland Fire 27:203.

Shakesby, R. and S. Doerr. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74:269-307.

Smith, H. G., G. J. Sheridan, P. N. J. Lane, P. Nyman, and S. Haydon. 2011. Wildfire effects on water quality in forest catchments: A review with implications for water supply. Journal of Hydrology 396:170-192.

SRA. 2006. Society for Risk Analysis. http://www.sra.org/.

Srivastava, A., J. Q. Wu, W. J. Elliot, E. S. Brooks, and D. C. Flanagan. 2018. A simulation study to estimate effects of wildfire and forest management on hydrology and sediment in a forested watershed, northwestern U.S. Transactions of the ASABE 61:1579-1601.

Stephens, S. L., B. M. Collins, E. Biber, and P. Z. Fulé. 2016. US federal fire and forest policy: emphasizing resilience in dry forests. Ecosphere 7: e01584. doi: 10.1002/ecs2.1584 https://doi.org/10.1002/ecs2.1584.

USDA Forest Service. 2018. Towards shared stewardship across landscapes: An outcome-based investment strategy. FS-118, USDA Forest Service, Washington, DC.

USGS. 2016. Protected Areas Database of the United States (PAD-US). USGS Gap Analysis Program (GAP). https://gapanalysis.usgs.gov/padus/data/metadata/. Accessed: 14 March 2016.

Westerling, A. L. 2016. Increasing Western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B Biological Sciences 371: 20150178. doi: 10.1098/rstb.2015.0178 http://dx.doi.org/10.1098/rstb.2015.0178.

WWWRA. 2013. West Wide Wildfire Risk Assessment - Final Report. Final Report, Oregon Department of Forestry, Council of Western State Foresters and the Western Forestry Leadership Coalition.

Zachariassen, J., K. F. Zeller, N. Nikolov, and T. McClelland. 2003. A review of the Forest Service remote automated weather station (RAWS) network. Gen. Tech. Rep. RMRS-GTR-119, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Chapter 9: Case Studies of Community Water Systems

Emily Jane Davis

9.1 Background

We conducted three case studies to delve deeper into how managers of forested drinking water supply watersheds identify and address management concerns that have affected/could affect source water. This includes how they collaborate with other landowners and managers to identify, monitor, and respond to these concerns. Case studies followed the following procedures.

9.1.1. Case study selection

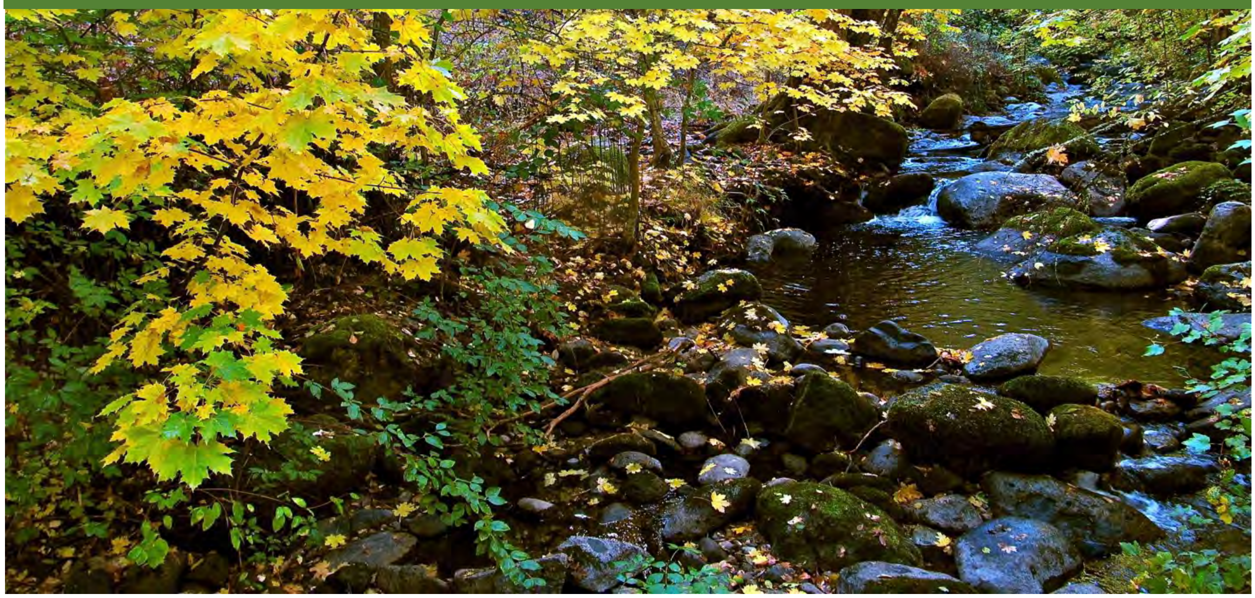
Survey respondents were stratified by location (Coast, Dryside, or Valley), primary landownerships in source watershed(s), and size of systems. We then purposively chose three case studies, one from each location. Cases were also selected to represent a range of relevant contexts and issues: 1) a private industrial forestland context and a small system (Oceanside), 2) a public lands context with a proximate wildland-urban interface and extensive collaboration on source watershed management (Ashland), and 3) a public lands context with less proximity, collaboration, and public engagement (Baker City).

9.1.2. Case study data collection

In each case, documentation was gathered and reviewed, including survey responses, source water assessments, forest management plans and information if available, and any other relevant documents found online. We used this information to develop a draft profile of each case. We then contacted representatives involved in the management of each drinking water system and the source watershed(s). Four interviews were conducted in each study location. Interview questions focused on verifying draft profiles and obtaining additional insights into forest management concerns and any collaboration to address them. Detailed notes were taken during each interview. Tours were also conducted of drinking water supply facilities including plants, intakes, and any applicable sites where past/current/future effects to drinking water could be observed. Following data collection, draft profiles were updated and content verified by all interviewees.

9.2 Ashland Water Department

Communities served: Ashland
Population served: 21,505
Source watersheds: Ashland Creek**,** Rogue River subbasin
Source water area size: 19.9 sq. miles/12,735 acres
Stream miles in drinking water source area: 82.88
Land ownership: 98% federal (Rogue River-Siskiyou National Forest); 2% local government
Public access: Open to public except for water treatment facility and reservoir areas
PWS #: 4100047



9.2.1. About the Ashland Water Department

- Organized as a municipal department with 14 full time staff.
- Primary source is Ashland Creek; backup sources for late summer are Talent Irrigation District (Howard Prairie and Hyatt Lakes) and City of Medford (Big Butte Springs or Lost Creek Lake). Ashland Creek is 303(d) listed for sediment above the dam.
- Treatment system: piped from Hosler Dam on Reeder Reservoir (Figure 9-1) to a flocculation basin and sand filters. One treatment plant is located in watershed.
- Winter daily production is 1.75 million gallons/day; summer is five million gallons/day; total storage capacity for the entire system is approximately six million gallons.
- Has a Source Water Assessment updated in 2018; does not currently have a Drinking Water Source Protection Plan.
- Conducts monitoring of algal species and toxins by collecting

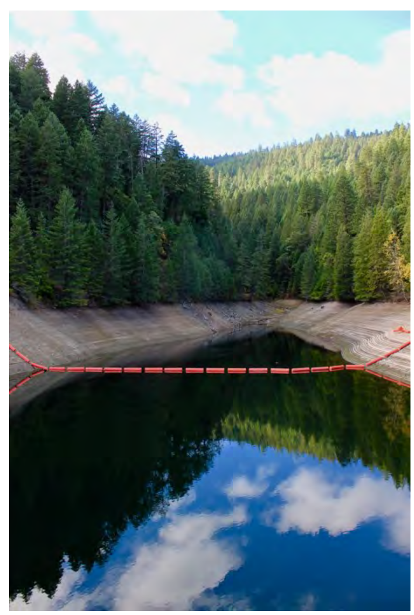


Figure 9-1. Reeder Reservoir in November 2018.

samples prior to treatment for bloom, and collects physical data with a sonde.

- Contributes a ratepayer fee to the Ashland Forest Resiliency (AFR) project for fuels and forest management in Ashland Creek watershed. AFR is a multi-partner effort to restore characteristic fire regimes and forest health in the watershed and adjacent areas.

9.2.2. Management Concerns

- Blue-green algae; although the reservoir's elevation (approximately 2800 feet) and cold winters help, there is concern about growth from warmer water temperatures and sunlight exposure.
- Erosion and debris flows, given the soil composition of decomposed granite, number of stream miles in erodible soils (62.45), percentage of soil erosion potential (75%), and steep slopes. Sediment is accumulating at the bottom of Reeder Reservoir, although two small reservoirs above it provide some containment. Winter storm events can exacerbate sedimentation. Suspended sediment has not been a major issue and has been manageable through treatments.

Potential pollution sources identified in Source Water Assessment, 2018

- Forest fire hazard
- Unstable soils
- TV tower underground storage tank
- Septic system at Mt. Ashland ski area
- Sedimentation
- Alkalinity, ammonia, chloride, dissolved oxygen, pH, and phosphate phosphorus
- Stream crossings
- Road density

- The risk of wildfire given the forest types and hazardous fuels conditions in the watershed, and regional tendency to have lightning-caused fires; and concern for suppression and post-fire impacts including erosion and multiple years of sedimentation, use of retardant in large quantities, loss of tree cover, and impacts to water treatment infrastructure.
- Public access and use; as the majority of the watershed is public land, it is open to the public. There are few roads, but numerous trails that can contribute to erosion. Dispersed camping can contribute to elevated E. coli levels downstream.
- Future water quantity, as Hosler Dam is not large enough to capture more water and the infrastructure costs to change this are currently prohibitive.
- Multiple seismic, landslide, and wildfire vulnerabilities at current treatment plant site.
- Flooding, particularly after rain-on-snow events, that can affect the treatment plant and sedimentation.

9.2.3. Addressing Concerns

9.4.9. Algae Monitoring and Treatment

The Ashland Water Department (AWD) visually inspects daily and tests as needed for various algal species, obtaining results about type and enumeration from a certified lab. They typically treat reservoir water two to three times a year in most summers by broadcasting a "Green Clean" hydrogen peroxide pellet product by boat. They also monitor sediment and nutrients that can encourage algal growth.

9.4.10. Sedimentation

The AWD monitors amount and extent of sediment deposits on the floor of Reeder Reservoir (Figure 9-1), but there is no easy way to remove these. Historical sluicing and some catching in two small reservoirs upstream from Reeder help (Figure 9-2), but sediment deposition is increasing with time.

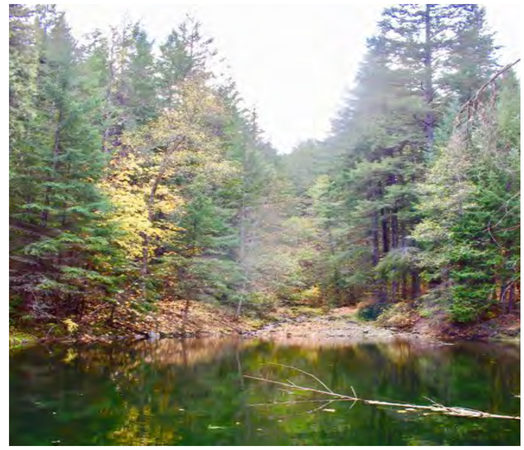


Figure 9-2. Small reservoir above Reeder.

This intersects with concerns about erosion and increased sedimentation from wildfire. Sediment levels are routinely monitored and will be addressed when they begin to affect water quality.

9.4.11. Ashland Forest Resiliency

The City of Ashland (led by the Fire Department) participates in the Ashland Forest Resiliency project (AFR), a multi-stakeholder effort to restore forest health and reduce the risk of uncharacteristic wildfires on the Forest Service lands that comprise the Ashland Creek watershed (Figure 9-3). The AFR was preceded by many years of community interest in forest management activities, beginning with a cooperative agreement between the City and the Forest Service that was signed in 1929 to codify a need for community consultation on any actions in the watershed. Protests over a planned timber sale to fund fuel breaks in the 1990s spurred the development of the Ashland Watershed Protection

Project with input from community members through the Ashland Watershed Stewardship Alliance, then the creation of a larger, landscape-level plan for the watershed. The City of Ashland and partners worked to develop a “community alternative” for that plan, which became the AFR project. The AFR decision, signed in 2009, authorized 7,600 acres of the watershed for treatments including hand and mechanical thinning, and prescribed fire. Its goals include the reduction of wildfire risk, particularly to prevent fires from moving from lower to higher elevations; and the enhancement of large trees and wildlife habitat.


Figure 9-3. Madrone and oak area in the Ashland watershed.

To implement AFR, the Rogue River-Siskiyou National Forest entered into a ten-year Master Stewardship Agreement (MSA) with the City of Ashland, The Nature Conservancy, and Lomakatsi Restoration Project. The MSA is based on mutual benefit and allows these partners central roles in accomplishing the treatments. Lomakatsi provides the implementation workforce through its own crews and contracts with additional entities, The Nature Conservancy leads an extensive collaborative monitoring program to understand ecological and other impacts of the work, and the City provides funding through a ratepayer fee and manages community engagement. Some monitoring related to water quality has been supported, such as macroinvertebrate and substrate monitoring. AFR has attracted additional investment from the American Recovery and Reinvestment Act, Joint Chiefs Landscape Restoration Partnerships program, Forest Service's Hazardous Fuels program, and Oregon Watershed Enhancement Board's Focused Investment Partnerships program for management of the watershed and adjacent areas of public and private lands. Given the steep slopes and high costs of treating this landscape, as well as the potential transmission of fire risk outside of the watershed, these resources have been essential. Future opportunities and challenges include the need to treat more acres in strategic areas, and to utilize treatments that can more effectively reduce fuels, which will require additional Forest Service analysis and collaboration.

9.4.12. Public Use and Management

Public activity in the watershed primarily occurs below the dam in Lithia Park, but trail networks still allow access upstream. The Forest Service has mapped trails in the watershed and partnered with the Ashland Woodlands and Trails Association (AWTA) to engage all trail user groups to develop the Master Trails Plan for the Ashland Watershed (Figure 9-4). The AWTA raised necessary funding for a third-party environmental analysis process to implement the Master Trails Plan, which can help control and direct public use of the watershed. The AWD also monitors E. coli levels and will close Ashland Creek to swimming and access if they become unsafe.

Figure 9-4. Trail networks exist throughout

9.4.13. Diversification of Sources

The size of Hosler Dam (Figure 9-5) and the substantial costs of a new dam currently limit the ability of this system to capture more water. The AWD has diversified to other backup drinking water sources that are typically used in late summer: Talent Irrigation District (since 1970s; Howard Prairie and Hyatt Lakes) and City of Medford (since 2013; Big Butte Springs or Lost Creek Lake). Talent Irrigation District water is pumped from a ditch to the Ashland plant, while Medford water is transferred via a pipeline.


Figure 9-5. Hosler Dam.

9.4.14. New Treatment Plant

The AWD treatment plant is currently located in a narrow, steep-sided canyon of Ashland Creek, where it is threatened by potential landslides, seismic activity, and wildfire (Figure 9-6). Prior major flood events have submerged the facility. A new plant has been planned and designed for a safer site, and slated to become operational in 2021-2022.

Figure 9-6. Flocculation basin at treatment facility.

9.2.4. Key Takeaways

- A multi-partner effort like the AFR project is necessary to incorporate the diverse social, economic, and ecological desires that the community of Ashland holds for the management of its watershed. This is particularly essential in the public lands ownership context, where the Forest Service must consider diverse public values in its decisions. Development of scientifically-sound monitoring and robust community plans helps address questions and foster adaptation.
- Activities necessary to reduce hazardous fuels and wildfire risk can be costly in areas with steep slopes and complex forest types. The partnership's strengths and ability to seek multiple authorities and programs to accomplish this work within and adjacent to the watershed is necessary; and expands outcomes beyond what the Forest Service alone could fund or accomplish (Figure 9-7).

Figure 9-7. Forest workers perform hand thinning.

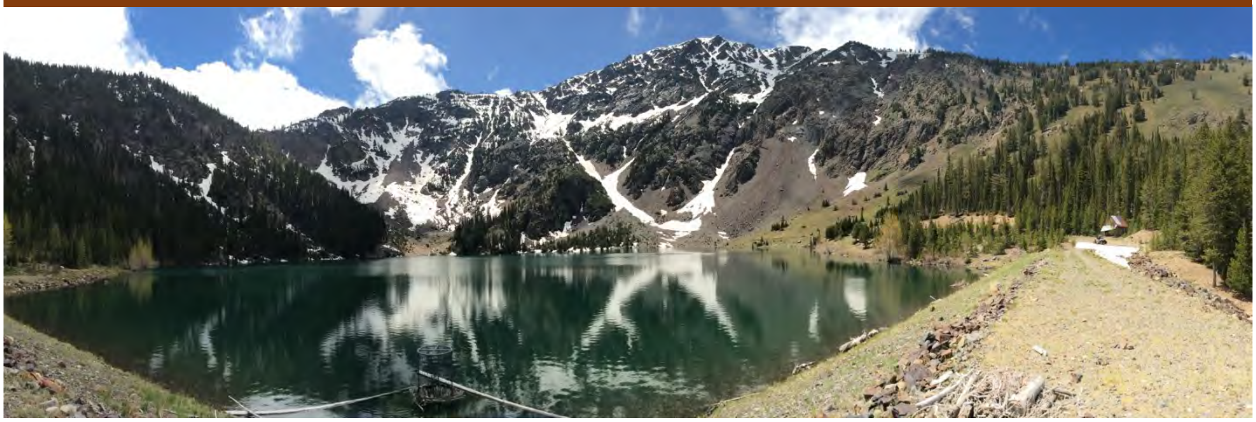
- The City of Ashland has been proactive in articulating its interest in the watershed and using formalized structures and tools (MOU, community alternative, Master Stewardship Agreement, ratepayer fee) to participate in active forest management. Its investment in forestry staff and the fire department provides the human capacity necessary to be part of collaborative efforts.

9.2.5. About the Ashland case study

Information from this study came from several sources, including Ashland's 2018 Source Water Assessment, a survey completed in summer 2018; and interviews with representatives from the Ashland Water Department, Ashland Fire Department, and Rogue-River Siskiyou National Forest. One tour of the district's reservoir and treatment plant was also conducted. We wish to thank the interviewees for their generous time in providing information and the tour. The final case study report was reviewed by participants for accuracy.

9.3 Baker City Water Department

Communities served: Baker City
Population served: 9,880
Source watersheds: Powder Basin (Goodrich, Elk, Salmon, Little Salmon, Mill, Little Mill, and Marble Creeks). Elk Creek is 303(d) listed for temperature.
Source water area size: 9,746 acres
Stream miles in drinking water source area: 11.9 miles
Land ownership*:* 99.8% federal (Wallowa-Whitman National Forest); .2% private
Public access: Not open to the public except for Marble Creek Road; seasonal hunting access by permit
PWS #: 4100073



9.3.1. About Baker City Water Department (BCWD)

- Organized as a municipal department with 5 full time and 20 part time staff.
- Additional water sources are an aquifer storage and recovery well. Watershed groundwater provides approximately 88 -98 percent of municipal water supply.
- Treatment system: Water travels from 12 diversions across seven creeks into two pipelines that feed one plant in Baker City with a chlorine contact chamber and UV treatment system.
- Winter daily production is 1 million gallons/day; summer is 5.5 to 6 million gallons/day; total storage capacity for the entire system is approximately 200 million gallons.
- Watershed was designated as municipal watershed in 1912 and is classified as two inventoried roadless areas (IRAs)
- Has a Source Water Assessment performed in 2003; has a 2014 Watershed Management Plan following Environmental Protection Agency guidance; does not currently have a Drinking Water Source Protection Plan but will complete one in 2019 through support from the Natural Resources Conservation Program’s National Water Quality Initiative and state agencies.
- Partners with the Wallowa-Whitman National Forest (NF) through a Memorandum of Understanding signed in 1991.

- The Face of the Elkhorns was defined as a wildland-urban interface (WUI) area in the Baker County Community Wildfire Protection Plan (CWPP).

- Source water monitoring requirements follow the Surface Water Treatment Rule for surface systems without filtration.

9.3.2. Management Concerns

> ***Potential pollution sources identified***
>
> - Cutting and yarding of trees leading to increased erosion, turbidity, and chemical changes
> - Reservoir contributions to prolonged turbidity
> - Erosion (near Goodrich Creek intake)

- Wildfire risk given the hazardous fuels conditions in the watershed. The forest is composed of ponderosa pine and mixed conifer stands, some of which are overstocked and dense. Recent large fires in the Baker City area (not in the watershed) such as the Cornet-Windy Ridge complex in 2015 contribute to this concern. Fires to date in the watershed have been small (under 10 acres) and quickly contained. In addition to a fire start inside the watershed, there is concern for starts outside the watershed, particularly to the south-southwest, that could move into the watershed. There is a regional tendency for lightning-caused fires. Of the 12 diversions, those on Salmon, Marble, Mill, and Goodrich Creeks may be most vulnerable; Salmon due to locally- continuous heavy fuels and limited access, and the others to the threat of fires moving from private lands up in elevation to the watershed.

- Post-fire impacts such as sedimentation and its effects on water treatment infrastructure would pose issues. The BCWD UV system does not provide sediment filtration, and the water department would be forced to switch to the backup groundwater source that likely provides only one month of supply. The location of intakes across multiple sub-watersheds helps reduce vulnerability, but a large fire event could cover the entire watershed under certain fire conditions. In addition, the loss of riparian tree cover through wildfire is a concern for its potential contribution to ongoing erosion and sedimentation. The majority of slopes in the watershed exceed a gradient of 30 percent, and many are considered “very steep” at over 60 percent; although the well-drained soils reduce risk of landslides.

- Ability of watershed to meet supply demands based on capacity of sources. Although the population of Baker City has not grown, there has been an increase in demand for water and a need to balance multiple users including households, irrigators, and municipal properties. Allowing enough water for agricultural producers is important given the economic significance of that sector to Baker County’s economy. Years of drought and reduced snowpack in the Elkhorns can lower water quantity from the diversions and the amount of water in the Goodrich Reservoir. This challenge would be particularly severe if combined with the shutdown of intakes due to a wildfire.

- Biological contamination. Livestock and wildlife can contribute biological contaminants. Although livestock are not allowed on the public lands of the watershed, straying and fence breaks can occur. A Cryptosporidium outbreak occurred in the summer of 2013, sickening a number of local residents. Pathogen levels are monitored and minimal excepting this outbreak, but the experience has elevated BCWD and community concern about water quality.

9.3.3. Addressing Concerns

Hazardous Fuels Reduction

Figure 9-8. Ponderosa pine stand in the watershed.

There has been limited forest management and fuels reduction activity within the watershed itself (Figure 9-8).

Challenges to accomplishing fuels reduction and forest restoration there include road access and condition for machinery, regulations and limitations to management options in Inventoried Roadless Areas, and smoke management limitations to the application of prescribed fire near the community. However, inclusion of the watershed as a WUI area indicates that it is at high risk and that there is a high priority for action there. As it is national forest land, the watershed is subject to National Environmental Policy Act (NEPA) analysis requirements. Two NEPA projects that have occurred are the Washington/Watershed Project (Environmental Impact Statement, decision signed in 1995) and the Foothills Fuels Reduction Project (Categorical Exclusion, decision signed in 2004). Management actions under these decisions have included commercial thinning, pre-commercial thinning, whip felling, mechanized slash treatment, hand piling, pile burning, and prescribed fire treatments. Thinning has been performed by helicopter and hand. An Environmental Analysis was also completed in 2016 and work has begun to improve the pipeline and road, burying the pipe more deeply and improving the road atop for safer passage of vehicles and equipment (Figure 9-9).

Figure 9-9. Area of unimproved pipeline road.

There is desire from both the BCWD and the Wallowa-Whitman NF for further activity. The watershed is on the Forest Service's work plan for 2019, meaning that funding for a new NEPA process has been allocated. Work is slated to begin with the formation of an interdisciplinary team and initial data collection in summer 2019. During the NEPA process, the BCWD will be a major project proponent, and it is anticipated that other area stakeholders such as adjacent landowners and the Powder Basin Watershed Council will participate. The Wallowa-Whitman Forest Collaborative group may also have interest in collaborating on this project. Data analysis and stakeholder interests will shape the specifics of the approach, but the planning area will include the watershed and adjacent lands. Treatments that may be considered could include strategically-located fuel breaks to prevent fire transmission at the private-public land interface around the watershed and pipeline road, and on ridges to reduce fire spread within the watershed. Areas adjacent to the road, particularly as it is improved, and in more pine-dominant stands offer more options for treatment.

Outside of the watershed, there have been several completed NEPA decisions to the south and southwest that have led to multiple years of fuels reduction and forest health restoration activities including landscape-level prescribed burning that is now in the maintenance phase. These activities are still underway and may help reduce the risk of fire transmission from these areas into the watershed.

Future Water Supply

The BCWD is building system redundancy and additional capacity from available groundwater sources to help address concerns with future water supply from the source watershed (Figure 9-10). Excess water is available in winter for storage in the aquifer storage and recovery well, but there are challenges in balancing the City's water right for beneficial use of water during injection season with the needs of surrounding properties. The BCWD is in a pre-design stage for a new groundwater well to be developed, and has requested a modification to their existing groundwater rights in order to combine them into one right to use more effectively where they own land.

Figure 9-10. Water storage infrastructure.

Biological Contamination

Management of sources of potential biological pollution includes monitoring, fence maintenance, and UV treatment (Figure 9-11). The 2014 Watershed Management Plan states that "increased monitoring, treatment, and preventative measures will be identified to reduce pathogen-inducing conditions. The key is to focus on prevention and reduction of turbidity, organics, and pathogens." Monitoring must occur as required by the Surface Water Treatment Rule for surface systems without filtration. Access to the watershed for routine sampling is difficult given restrictions, but downstream sampling also provides data. The BCWD is also using aerial observation and cameras to monitor containment of cattle off the watershed, and any concentrations of ungulate populations. The BCWD has also taken infrastructure improvement steps including the burying of a previously-exposed settlement area and repair to fencing at the Elk Creek diversion. They have obtained a grant for ongoing maintenance and repair of fences to attempt to prevent future breaches, and partner with the adjacent allotment holders to monitor fence condition during the grazing season. In addition, the recent acquisition of support from NRCS and state agencies through the National Water Quality Initiative will allow the BCWD to develop a watershed assessment and outreach strategy to address agriculture-related water impacts and become eligible for future Farm Bill funding.

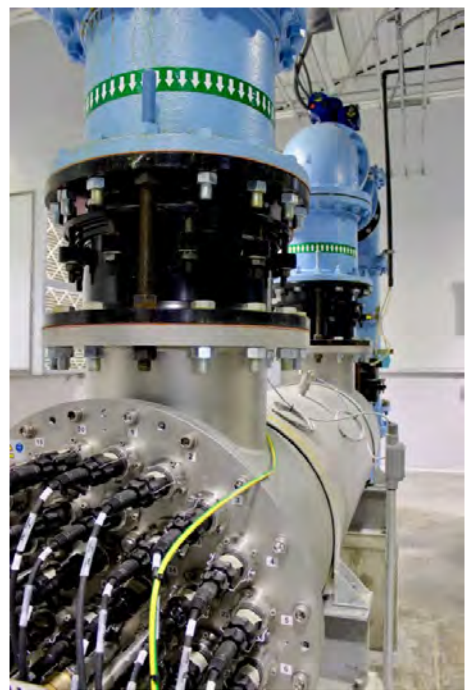


Figure 9-11. A component of the UV treatment system.

9.3.4. Key Takeaways

- Regular, such as quarterly, communication between the Forest Service and a municipality with source watersheds on national forest land assists in maintenance of relationships and proactive capacity for identifying issues and opportunities. This helps keep drinking water source protection issues on the table when both partners are also busy with other responsibilities and projects.

- Field tours and opportunities to view the watershed and potential management issues together in person help increase mutual understanding of conditions, challenges, and opportunities.

- Written documentation of agreements and meetings can assist in the creation of agreements and institutional memory, which is important in a context with the frequent personnel turnover that can occur in both the Forest Service and city management.

- There can be city and community frustration with the time and other requirements of the NEPA process for management actions on federal land. Increased experience and exposure can help build mutual understanding through the process. The pending NEPA process for the watershed should provide concrete opportunities to address concerns and plan new projects; which necessitated the Forest Service prioritizing the watershed area and obtaining funding to do so.

- Municipalities and other partners may aid federal partners in managing source watersheds by building political support and obtaining grant funding from sources not accessible to federal agencies.

- Having multiple intakes/diversions in several locations across a source water drinking water area requires management effort and cost, but also offers diversity of options; for example, by reducing vulnerability to wildfires or other effects.

9.3.5. About the Baker City case study

Information from this study came from several sources, including Baker City's 2003 Source Water Assessment, 2014 Watershed Management Plan, a survey completed in summer 2018; and interviews with representatives from the Baker City Water Department and Wallowa-Whitman National Forest. One tour of the district's reservoir and treatment plant was also conducted. We wish to thank the interviewees for their generous time in providing information and the tour. The final case study report was reviewed by participants for accuracy. Photos by the Wallowa-Whitman National Forest, except for Baker City water infrastructure and UV treatment by Emily Jane Davis.

9.4 Oceanside Water District

Communities served: Oceanside and Cape Meares
Population served: 650; 541 connections
Source watersheds: Short Creek (Oceanside) and Coleman Creek (Cape Meares), in the Netarts Bay/Sand Lake/Neskowin Creek Watershed in the Wilson-Trask-Nestucca Sub-Basin of the Northern Oregon Coast Basin. Short Creek is fish-bearing and Coleman Creek is not.
Source water area size: 2.04 square miles
Land ownership: 99.9% private industrial timberland; Stimson Lumber Company and Green Crow Corporation
Public access: None
PWS #: 4100585

9.4.1. About Oceanside Water District

- Organized as a special district under ORS Chapter 198. Has four staff (three full time and one part time) and a board of commissioners.

- Short Creek is the source watershed for Oceanside, and Coleman Creek is the source for Cape Meares. Recently obtained access to Baughman Creek, where an intake may be established for future backup use.

- Have two treatment plants. The Cape Meares plant can be fed from the Short Creek plant in case of emergency in Coleman Creek. The Oceanside plant has recently carried out $7.2 million in major system upgrades. Raw water treatment consists of an initial intake through a fish screen, then passage through a membrane filtration system.

- Winter daily water production is 50-60,000 gallons/day; summer is 130-140,000 gallons/day; total storage capacity for the entire system is approximately 750,000 gallons.

- Has a Source Water Assessment completed in 2003 and used to identify potential areas of risk to the two creeks. Does not currently have a Drinking Water Source Protection Plan.

- Communicates with a small local committee of citizens, the Oceanside Clean Water Committee, a subcommittee to the Oceanside Neighbors Association, an officially recognized Citizens Advisory Committee.

9.4.2. *Management* Concerns

- Application of forest chemicals to plantations and roadsides. Forest managers use herbicides to enhance plantation productivity by reducing competition facing tree seedlings, and to control noxious weeds and maintain roads. Spraying of herbicides is typically done on the ground by backpack or truck and not near open water in accordance with the Oregon Forest Practices Act; and GPS tracks and visual marking of buffers guide application. The Oceanside Water District (OWD) is concerned about spray spread by rainfall or aerial vapor drift, or spills that could place herbicides in creeks.

> ***Potential pollution sources identified in Source Water Assessment, 2003***
>
> Cutting and yarding of trees may contribute to increased erosion, resulting in turbidity and chemical changes.
>
> Overapplication or improper application of pesticides or fertilizers.
>
> Spills, leaks, or improper handling of chemicals during transportation, use, storage, and disposal.

- Currently, the only governmental monitoring of source water is an SOC (Synthetic Organic Contaminant) test carried out per state mandate once every three years at a randomly-selected time. The District performs the required tests every three years on Short Creek, but quarterly on Coleman Creek as the creek was just recently brought on line.

- Turbidity following forest operations and from forest roads. Clearcut harvests have not occurred for approximately 40 years in the drinking water source area, but are pending in the next two years and road systems are currently being improved in preparation. There are 10.6 miles of Stimson roads and .49 miles of Green Crow roads within the Short Creek watershed, and .3 miles of Stimson roads within the Coleman Creek watershed.


Figure 9-12. Improvement of forest road on Stimson timberland for future harvest operations.

- There are two locations where roads on Stimson lands cross perennial tributaries of Short Creek and are of major concern from the perspective of the OWD (Figure 9-12).

- Runoff after winter storms, which can be significant in this coastal region. Sediment in Short Creek during extreme rainfalls has caused temporary shutdown of the Oceanside treatment plant in past events. Intake relocation and upgrades to this plant have helped reduce this challenge somewhat, but it is still necessary to close the raw water intake following an extremely heavy downpour. During this time, the OWD operates off water stored in several storage tanks throughout the town.

- Point source pollution from gravel quarries through discharge or runoff of holding ponds combined with rainfall affected Short Creek and caused a multi-day plant shutdown in 2007 (Figure 9-13). Followup inspection from the Department of Geology and Mineral Industries reported no runoff and this pit is now in the process of reclamation.

Figure 9-13. Portions of a prior harvest unit and the vicinity of a rock quarry, common activities that may occur in coastal watersheds. Area shown is adjacent to but not within the headwaters of Short Creek.

- Landslides on Short Creek given the steepness of its canyon, and instability of sandy soils atop basalt bedrock on Cape Meares.
- Future water quantity from Coleman Creek for the Cape Meares community.
- Potential wildfire risk; although there is not an immediate history of wildfires in the area, post-fire erosion and reduced tree cover should a fire occur is a significant concern.
- A major new potential issue is the re-routing of a county road in the Coleman Creek watershed. At present, the intake for Coleman Creek is located upstream of the existing road, but the road will be rerouted up Cape Meares along Coleman Creek due to a landslide (Figure 9-14). This will result in the existing intake point being downstream of the road where it would become susceptible to transportation- related spills.

Figure 9-14. Current intake on Coleman Creek for Cape Meares treatment plant.

9.4.3. *Addressing* Concerns

The OWD, private landowners, and partner agencies are working on addressing issues of management concern in the source watersheds through *proactive communication, mitigation, and planned monitoring projects.*

Planned Forest Operations And Herbicide Application

First, there is advance communication and information about planned operations. OWD staff and one of its board commissioners subscribe to and run queries in FERNS to obtain information about planned forest operations. They are able to view notifications of planned operations one year in advance, when these are placed in the system.

Stimson also uses an internal communication checklist to ensure that all drinking water suppliers with intakes on their properties have timely communication about planned operations in accordance with Oregon's Forest Practices Act. Stimson notifies all water masters:

1) a minimum of 15 days prior to application;

2) on the planned date of application;

3) one day prior to the actual application day; and

4) on the day of application, prior to starting the application and when it is completed.

Second, the OWD, Stimson, and Green Crow have also communicated proactively about potential source water management concerns. The OWD has expressed their desire to gather data about potential effects of herbicide spraying. Stimson and Green Crow have agreed to further notify the OWD one week prior to a planned herbicide treatment so that the OWD may take precautions and prepare its supply. Following this notification, the watermaster will charge all reservoirs to their maximum. The companies will notify the OWD again on the day of spraying, and the intake to the water processing plant will be closed. Then, with funds from the Oregon Health Authority and Department of Environmental Quality, the OWD will conduct an experiment. They will take water samples from the intake synchronously with the spraying, using grab samples and POCIS measurements for an extended period after the spraying. This experiment has not yet taken place, but is anticipated to occur at the time of the next herbicide treatment. The companies will also notify the OWD when they are preparing roads for future harvest, which may involve regrading, rocking, and replacing culverts.

Turbidity

Oceanside drinking water supply area. When the turbidity reaches certain levels, sediment has Major seasonal rain events have historically caused high turbidity in both creeks in the clogged the holding pond and intake, and the plant has shut down. The OWD has relocated its Short Creek intake to the center of the creek, which has greatly reduced this issue. Improvements to the Oceanside treatment plant have also increased the capacity of the system to filter sediment (Figure 9-15). However, slope instability and potential landslides near this intake still pose a concern (9-16).

Figure 9-15. Membrane filtration system in the Oceanside treatment plant.

Transportation Planning

Another concern for the OWD is potential contamination of the Coleman Creek supply from a road. A paved county road connecting Oceanside to Tillamook by Cape Meares has been closed due to a landslide. Tillamook County Roads and Transportation is currently conducting feasibility analyses and planning to relocate this road around the landslide area. The eventual location of this road would be upstream from the current diversion point for Coleman Creek. There is concern about the potential for vehicle-related accidents, and hazardous material spills, trash, and public access as a result. The OWD is working with the County to evaluate relocation of this diversion point to above the new road route.

Figure 9-16. Intake on Short Creek for Oceanside treatment plant. Stability of bank slopes in this area is a management concern.

Diversifying Drinking Water Sources

The OWD is working to diversify and increase its future supply by developing a new intake on Baughman Creek (Figure 9-17). Rights to this creek were recently deeded to the OWD by the Rosenburg family. There is a historic access point and intake site on this creek. The OWD will be restoring road access to this site by clearing the road footprint, and investigating the necessary steps and costs to install a new intake. This would allow them to draw drinking water from three different creeks on different parts of Cape Meares.

Figure 9-17. Site of potential future improved intake on Baughman Creek.

9.4.4. Key *Takeaways*

- More consistent and proactive communication between the OWD and landowners (Stimson and Green Crow) has enhanced cooperation. Communication has historically been intermittent as it has been solely based around issues with quarry operations or planned forest operations. Establishing a schedule of regular meetings, such as quarterly, may be useful.
- Stimson's use of a process communication checklist is intended to help ensure that the OWD and other water providers are notified beyond what is required by Oregon's Forest Practices Act.
- Opportunities to learn more about each other's goals and processes may have increased mutual understanding. Foresters for Stimson and Green Crow have toured the Oceanside treatment plant, and OWD commissioners and the watermaster have toured parts of the watershed in the past.
- In this spatially-smaller landscape with a limited number of landowners, individuals particularly matter. The interests and actions of the OWD staff and board, and company foresters, have made cooperation possible.
- The ability to develop a monitoring project and obtain data is anticipated to help improve a cooperative relationship by addressing uncertainties, providing scientific information, and giving the OWD and Stimson opportunities to communicate and learn together. The financial support from Oregon state agencies for this project is also necessary.

- The future development of a Drinking Water Source Protection Plan for the OWD may help codify these monitoring and cooperative efforts.

9.4.5. About the Oceanside case study

Information from this study came from several sources, including Oceanside's 2003 Source Water Assessment, a survey completed in summer 2018; and interviews with representatives from the Oceanside Water District, Stimson Lumber Company, and Green Crow Corporation. Two tours of the forested watershed and one tour of the district's intakes and treatment plant were also conducted. We wish to thank the interviewees for their generous time in providing information and tours, and OWD Commissioner Paul Newman for providing information and arrangements. The final case study report was reviewed by participants for accuracy.

9.5 Lessons Learned

Although the case studies were conducted in three different contexts, there were lessons learned from each case as well as common themes across cases that may offer broader insights.

9.5.1. Landownership frames the opportunities and challenges for managing source watersheds.

The laws and regulations that govern different types of forestland ownerships set the stage for what management activities are permitted, how they are to be conducted, and any public involvement. For example, Oregon's Forest Practices Act provides standards for the establishment, management, and/or harvest of trees on private industrial and nonindustrial forest lands. Public lands managed by federal agencies such as the US Forest Service or the Bureau of Land Management are subject to an array of laws and policies, as well as land use designations and requirements for public participation in management decisions. Drinking water providers who seek to interact and collaborate with their source forestland managers must do so with understanding of these existing frameworks, and the time and effort that it may take to engage.

9.5.2. Regular communication provides a foundation for relationships.

Regular communication between drinking water providers and source watershed land managers may assist the maintenance of relationships and proactive capacity for identifying issues and opportunities. This helps keep drinking water source protection issues on the table when both partners are also busy with other responsibilities and projects. Field tours and opportunities to view the watershed and potential management issues together in person may help increase mutual understanding of conditions, challenges, and opportunities. The scope and scale of this communication may necessarily vary by context. For example, it may be more informal and involve far fewer parties in areas where source watersheds are spatially small and systems serve smaller populations. Regardless, the need for both land managers and drinking water providers to be intentional and proactive about communication with each other remains.

Written documentation of agreements and meetings can assist in the creation of agreements and institutional memory, which is important when there is personnel turnover with any organization.

9.5.3. Specific projects offer opportunity to collaborate.

Planning forest management activities, a source water protect plan, or a monitoring effort can offer concrete ways for drinking water providers to engage with source watershed managers. Depending on the ownership of the source watershed, providers may be able to provide project design input, develop community plans, or create monitoring protocols. This may involve additional partners such as local nonprofits, government agencies, and community leadership. The opportunity to participate directly may improve understanding of source watershed conditions and needs, particularly though monitoring that could address uncertainties with scientific information. It can also bring leveraged funds from other sources that help support monitoring or management activities.

Chapter 10. Findings and Recommendations

Jon A. Souder and Jeff Behan

10.1 Introduction, overview, purpose.

Western forests are managed for many diverse purposes, including wood products, recreation, and wildlife habitat. By filtering rain and snowfall and delivering it to streams or aquifers, forests also produce the highest quality and most sustainable sources of fresh water on earth, arguably their most important ecosystem service. The public values water produced from forests very highly, and continues to rank water quality and quantity as primary concerns with forest management. Our extensive and diverse forests generally produce very high quality water and supply the majority of states community water systems. Forest practices designed to minimize impacts to water quality have improved significantly in recent decades. At the same time, demand for all forest ecosystem services continues to rise, against a backdrop of a changing climate and uncertain implications for water derived from forests. Together, these trends point to the importance of maintaining and expanding public awareness of current science knowledge regarding the complex relationships between forest hydrology and forest management.

With support from the Oregon Forest Resources Institute, our group at Oregon State University has spent the last two and a half years evaluating the effects of active forest management on source water quality for community water systems in Oregon. This evaluation included a science review focused on four topic areas: (1) water quantity; (2) sediment and turbidity; (3) forest chemicals; and (4) natural organic matter and disinfection by-products. The 156 community water suppliers in Oregon who rely on surface water as their primary source were surveyed, and three representing different geographic regions (coast, interior valleys, and semi-arid regions) had more in-depth case studies. Additionally, we examined Oregon forest operations notifications for the past four years (about 65,000), paying particular attention to use of forest chemicals, and reviewed incidents regarding chemical applications over the same time period.

In this chapter we pull from the preceding work to summarize our results, and in some cases provide recommendations for policy makers. In the interest of readability, we have chosen not to include citations of research to support each finding. For these citations and details, readers are referred to the chapters specific to each topic and section here.

10.2 Policy-related findings and recommendations

The Oregon Forest Practices Act (FPA) is the state's primary regulatory framework for addressing the environmental impacts of forest operations on state and private forest lands. The FPA sets standards for all commercial activities involving the establishment, management, or harvest of trees in the state. When passed in 1971, the FPA was the first legislation of its kind in the USA. The FPA's first rules were implemented in 1972 and emphasized BMPs, which have since been revised repeatedly in response to emerging environmental concerns and science findings.

The Safe Drinking Water Act (SDWA) was enacted in 1974, and significantly expanded in 1996, specifically to protect drinking water quality. The SDWA focuses on all U.S. surface water or groundwater sources actually or potentially used for drinking, and requires USEPA to establish and enforce standards to protect tap water. The USEPA National Primary Drinking Water Regulations (NPDWR) are legally enforceable standards, treatment techniques and water-testing schedules that apply to public water systems. The SDWA allows individual states to set and enforce their own drinking water standards if the standards are at a minimum as stringent as USEPA's national standards. The Oregon Health Authority (OHA) regulates the treatment and distribution of potable water under the Federal Safe Drinking Water Act, while the DEQ has regulatory authority under the Federal Clean Water Act (CWA) for point and non-point sources of pollution.

In the past, the CWA and SDWA had mostly separate goals and functions. The CWA focused on environmental protection and maintaining “fishable/swimmable” waters, primarily by identifying and regulating sources of pollution in waterways. In contrast, the SDWA focused on municipal water treatment standards and providing clean drinking water at the tap. Coordination across the CWA and SDWA is motivated by potential synergisms among goals and outcomes of these policies, recognizing that preventing contamination is much more cost effective at providing safe drinking water than removing contaminants or finding alternative water sources after the fact. In 1996, Congress significantly expanded the SDWA to facilitate prevention of contamination through an increased focus on drinking water source protection by requiring states to develop USEPA-approved programs to carry out Source Water Assessments (SWAs) for all public water systems in the state. The DEQ provides reports, general information and technical assistance regarding surface water systems, while the Oregon Health Authority (OHA) supplies these services for groundwater systems. Updated Source Water Assessments (USWAs) with more detailed data, maps, and technical information were completed for roughly 50% of these systems in 2016-2017.

Much of the existing knowledge regarding the effects of active forest management, in particular water and sediment interactions, comes from paired watershed studies conducted from the 1960s-1990s. Funding for long-term, paired watershed studies has declined, so knowledge regarding effects of current practices is more limited. Long-term studies on forestry/sediment/water quality relationships are expensive, time-consuming and thus relatively uncommon. However, major storms and associated peak flows are often a significant or even dominant driver of sediment movement, so whether or not one or more such storms occur during the duration of study can significantly affect results of studies that span only a few years.

- Most studies we reviewed were focused on the effects of forest management on water quality, but few were specific to drinking water quality. We were able to infer effects on source water quality in many cases, but the cause-and-effect linkages were not as direct as we would have preferred.

- Similarly, most of the studies were conducted in the upper parts of watersheds while raw water intakes are located at various and often substantial distances downstream. In addition to forest management, intervening land uses and contaminant sources may also affect water before it reaches an intake. The size of the source watershed, and its mixture of land uses and management actions, often confound the ability to isolate forest management effects.

- Research has identified general patterns for several aspects of forest management effects on water, but findings are often based primarily on a relatively small number of studies and locations. In many ways, how forestry may affect a particular source watershed represents a unique combination of size, geology, topography, ecology, land use history and also variability in present and future climate.

- Over time, changes related to climate warming are expected to result in significant increases in peak flow frequencies and magnitudes in the Pacific Northwest, especially in snow-dominated watersheds as more winter precipitation falls as rain. This suggests that any effects that forestry activities have on peak flows will intertwine with climate in increasingly complex ways.

- Harmful algal blooms of cyanobacteria (cyanoHABs) are a growing concern because they produce cyanotoxins that can cause sickness and death in humans and are predicted to increase as climate change progresses. Sources of phosphorus and nitrogen that exacerbate cyanoHABs from septic systems, fertilizers, agricultural runoff, and urban and forestry runoff are all likely to come under increasing scrutiny.

- Since 2013, FPA rule compliance monitoring has been conducted by ODF for BMPs related to road construction and maintenance, timber harvesting, some riparian management area measures, measures for small wetlands, and rules for operations near waters of the state. Audits through 2016 indicate generally high compliance rates, e.g. 97% overall compliance for 2016.

- Nonetheless, existing FPA rules are insufficient to protect some water quality attributes. Multiple studies have shown that existing riparian buffers do not meet the “protect cold water” standard. As we’ll see in the Forest Chemicals section, wooded buffer areas on non-fish bearing streams can prevent or reduce pesticide drift. And, as of June 2019, the FPA does not have any water quality-related landslide-prone area rules (although the rules related to landslide hazards to humans and infrastructure provide protection to some areas).

Policy-related recommendations:

1. ***Targeted research needed***. Additional research is needed to evaluate the effects of all types of land uses, and particularly forest management, on source water quality. Understanding the connections, and cause-and-effect linkages, between land management activities and source water quality can be improved with targeted studies in the many areas outlined in this report.

2. ***Information preservation***. Records retention policies constrained our ability to evaluate longer-term trends for both harvests and pesticide incidents. Most state records (in Oregon and elsewhere) are destroyed after five years. Retention of these records in State Archives would enable researchers to conduct more robust analysis and prediction.

3. ***Cooperative planning***. Drinking water protection plans (DWPP) provide a structure and venue for land managers and water utilities to cooperate on maintaining source water quality and quantity in the face of potential changes. The State and other entities (such as NRCS) should continue to provide support and funding for local groups to prepare these plans. Oregon State University can play a supporting role by providing information through its Oregon Explorer web-based service, and expertise in modeling and analysis.

4. ***Rules revisions***. The Governor's 2020 "Oregon Strategy" of state, timber industry, and conservation groups will likely improve water quality to the benefit of community water sources within those areas covered by the agreement. If the Legislature fails to act according to the MOU, the Board of Forestry should entertain rulemaking consistent with the agreement.

10.3 Findings and recommendations related to Community Water Suppliers

In Oregon, 238 source watersheds feed into 157 water treatment plants operated by 156 community water systems (CWSs) that utilize surface water, and shallow wells influenced by surface water, to provide the raw water source for almost 3 million Oregonians. Most (about 75%) of Oregon's population obtains drinking water from large (serving 10,001 - 100,000 people) or very large CWSs (serving more than 100,000 people), but most (about 80%) of the systems themselves are very small (29% of the 156 total; serving less than 500 people), small (34%; serving 501-3300 people), or medium (17%; serving 3301-10,000 people). Forty-one percent of survey respondents have drinking water primary source watersheds of 10 square miles or less in size. Almost two-thirds of the community water providers dependent on surface water serve small (35% of 156 total) or very small (29%) populations. Their small size limits the human, financial and infrastructure capacity of these providers. Compared to larger CWSs, smaller systems usually face higher costs per unit of finished water delivered, have smaller budgets, and operate with fewer dedicated staff, with some of the smallest systems being staffed by volunteers only. Fifty-eight percent of the Oregon CWSs that responded to our survey operate on a budget of $500,000 per year or less; 24% operate on a budget of $100,000 per year or less.

Our survey of CWS showed that the top three general areas of concern among survey respondents were forest harvest and management, stormwater runoff, and ability of the watershed to meet supply demands. Water providers—especially those serving smaller communities—often feel they have little control over activities in their source watersheds that affect the quality of their source water, including: water temperatures, nutrient levels, landslides, riparian buffer blowdown, wildfire risk and effects, forest chemicals, future water quantity, and sediment and turbidity. Large majorities (exceeding 70%) felt they had no control at all over multiple issues. For every issue affecting their source watersheds listed in the survey, respondents' level of concern over the issue was greater than their perceived control over it, especially wildfire impacts, forest chemicals, floods and sediment, and water temperatures and quantity.

Respondents' key "lessons learned" via experiences managing source watersheds fell roughly into three categories: the importance of 1) maintaining lines of communication with forest landowners; 2) being proactive and prepared rather than reactive in the face of events and challenges, and 3) actively managing for forest health. Specifically:

- Water provider survey respondents stressed the importance of knowing and communicating regularly with landowners and their agents in source watersheds, including logging crews who were on the ground, to have real-time discussions about forest operations as they occur.

- Respondents stressed the importance of proactively preparing for a range of possible events and situations via regular examination of the source watershed, knowing who to call in the event of problems, practicing response scenarios, stocking supplies such as filter bags, updating assessments and plans, and having all necessary documentation.

- Some respondents indicated that hands-on, fully-engaged management for forest health, with proactive planning, inventory, monitoring, and activities such as invasive species control and stand improvement, is necessary to maintain source water quality.

- Respondents indicated that their most important partners in managing their drinking water source watershed were private forestland owners (likely because they own many of the drinking water source areas for providers we surveyed) followed by watershed councils and SWCDs.

10.4 Water quantity findings and recommendations

Relationships between forest cover and type, forest management, and the quantity and timing of water produced by forested watersheds have been studied for at least 100 years. Understanding of these relationships has been significantly enhanced by research, especially long-term, paired watershed studies. We reviewed evidence regarding changes in (a) annual flow, (b) changes in peak flows and flooding, (c) changes in low (base) flows, and (d) changes in the timing of water delivery. Throughout, we noted the difficulty in trying to extrapolate from studies that typically took place in higher elevation, small watersheds to effects on downstream drinking water supplies. There is often considerable variability in results, with some studies finding large effects and others none at all. Effects that have been quantified at smaller scales may potentially "scale up" to larger watershed scales, but these larger scale effects are rarely studied and thus remain generally speculative. Lastly, conditions in many watersheds reflect the cumulative effects of actions conducted over the span of many decades of evolving forest management practices.

A substantial body of evidence has nevertheless accumulated, from an increasingly diverse array of research perspectives and methodologies:

- We know with considerable certainty that the percent area of the watershed harvested is the predominant factor affecting changes in stream flow volumes.
- Timber harvesting temporarily increases annual water production, especially in the first few years after harvest, with these increases declining in following years, as vegetation, including planted commercial timber species, establishes and starts growing vigorously.
- By volume, these changes often peak in the fall and early winter. By percentage, the largest changes in low flows often occur in late summer.

Peak flows and floods have implications for community water suppliers in terms of increased sediment transport, turbidity, and mobilization of pollutants, as well as potential damage to water treatment infrastructure. The generally accepted scientific understanding is that:

- Peak flow increases are most prominent for smaller, more frequent peak storm flow events, and these increases tend to decline as peak flow size and basin size increase.
- Snowpack changes related to climate warming are likely to result in large increases in peak flow magnitudes in mountainous areas such as the Cascades and Blue Mountains due to a greater

frequency and magnitude of extreme precipitation events, and a growing proportion of winter precipitation falling as rain instead of snow.

Seasonal low flows are of particular interest because they generally coincide in late summer with the period of greatest demand for drinking and irrigation water:

- Along with rising temperatures, dry years are increasing, low flows are declining and the annual low flow period is lengthening in duration.
- Stands of conifers established after clearcut harvests can, once they are 15 – 20 years old and growing quickly, significantly and persistently reduce summer low flows in comparison to the older stands they replaced.

In summary, the weight of available evidence indicates that forest management can and probably does affect the volume and timing of water delivered from managed watersheds and by extension, community water systems that are hydrologically connected downstream. The limitations on existing knowledge make it difficult to specify these effects for a particular area. However, linkages between water supplies and forest management (e.g., harvesting a significant percentage of the watershed) can be more readily established in smaller systems that are closer to the source watershed than in larger systems that are further away, with more intervening land uses. Finally, climate change and associated shifts in snowpack levels and timing, and in the frequency and severity of extreme weather events, will further complicate an already complex set of factors that influence the amount and timing of raw water provided in actively managed drinking water source watersheds.

10.5 Sediment/turbidity findings and recommendations

Linkages between active forest management and increased sediment loading in streams have been studied extensively and are well-established in broad terms. There is also an expanding body of evidence indicating that modern practices such as improved road building methods and stream buffers have significantly reduced sediment production from forest management activities, and the chances that this sediment will enter waterways. But these effects and findings are highly variable due to the complexity of interactions among factors such as site-specific ecology, geology and geomorphology, management prescriptions and land use histories. The specific sources of mobilized sediment within an actively managed area are also often not clear. Considerable uncertainty remains in predicting precisely how a particular set of forest management actions will affect sediment production in specific cases. Further, there is a paucity of research focused on linkages between sediment inputs related to timber harvesting and associated activities in headwater areas of watersheds and increases in suspended sediment or turbidity in water withdrawn downstream for domestic uses.

A range of potential contributing factors may help explain the lack of research focused on forestry and drinking water linkages. As watershed size and distance from forest management activities increase, it becomes progressively more challenging to isolate and quantify the effects of particular actions. There are usually cumulative effects resulting from forest management in larger watersheds, partly due to variability in forestry activities (e.g. road building and use, harvesting, site preparation) and timing of their impacts on stream sediment, with some actions having immediate effects and others taking years to become apparent. Timber has been harvested for a century or more in many Oregon watersheds, historically without BMPs in place, with a legacy of sediment production and sediment transfer downstream in many watersheds. Over time, affects accumulate in complex patterns across forestlands managed through multiple harvests and rotations. Distinguishing effects of modern forest practices from those used earlier, and whether increased sediment and turbidity originates primarily from remobilized natural or anthropogenic sediments within streams, streambank erosion, or sources external to the waterway is difficult and complex. Climate variability, the generally episodic nature of sediment movement, and the outsize influence of stochastic events such as infrequent large storms can introduce additional uncertainty into research findings. Finally, in larger watersheds, forest management is often not the only land use or potential source of sediments.

For these reasons, it is difficult to make specific, firm conclusions regarding how, where and the extent to which sediment produced by active forest management in a headwater area affects water quality at a drinking water intake downstream. There is, however, an extensive body of evidence accumulated through forestry and sediment-focused research conducted in upper watersheds that is highly relevant to drinking water quality. Reasoned inferences can be drawn from this evidence base regarding effects on drinking water sources because hillslopes, headwaters, and larger downstream waterways are all elements of fundamentally connected and integrated hydrological systems. Headwater streams comprise about 60-80% of total stream length in a typical river drainage and generate most of the streamflow in downstream areas, and these first and second-order streams cumulatively contribute to, and can profoundly affect water quality downstream.

Headwater streamflow is usually routed efficiently downstream, meaning that management-induced changes in streamflow parameters will accumulate downstream. Because turbidity and fine sediment can be readily transported downstream, changes in headwater inputs of these constituents may be directly linked to downstream conditions. In contrast, linkages between upstream inputs and downstream fluxes for coarse sediment and large woody debris are considerably weaker. It is also important to note the substantial variation in distances between actively managed forests and drinking water intakes across the range of different municipal water suppliers in Oregon. Studies that show forest management activities or forest roads increase sediment production and reduce stream water quality in headwaters can be more reliably extrapolated to i drinking water quality effects where intakes are in relatively closer proximity to these management activities and have fewer intervening land uses.

In general, due primarily to the complex interplay of factors outlined above and difficulties in isolating and quantifying the sources and fates of mobilized sediment, we found little direct evidence that forestry activities and forest roads impact community drinking water in Oregon. But there is considerable indirect evidence that forestry can have such affects, and likely continues to have effects in certain cases, inferred from (1) extensive findings regarding linkages between forestry activities and mass wasting in upper watersheds; (2) cumulative and legacy effects of harvesting, site preparation and forest roads dating from periods when BMPs were not as robust; (3) inevitable variability in BMP implementation and effectiveness; (4) the ability of fine sediment to be carried considerable distances, especially during peak flow events; (5) the inherent connectivity of hillslopes, headwaters and larger downstream waterways; and (6) the lack of provisions to protect small, non-fish bearing, ephemeral and intermittent streams during harvesting, and lack of water quality protection provisions for operations in landslide-prone areas.

Key findings are:

- A large body of evidence links forest management activities to increases in sediment production. Most of this evidence comes from research conducted in smaller first- and second-order watersheds, mainly to avoid the confounding effects of other land uses.

- Most available evidence suggests that forest roads, skid trails, log landings and slash burning are more likely to increase sediment mobilization than timber harvesting itself, but considerable knowledge gaps remain regarding the sources of increased sediment loads in streams in specific cases, e.g. roads, general harvest areas, or sources within the stream channel. Soil tracers and sediment "fingerprinting" show promise as research tools to provide insight on the specific sources of sediment associated with forest management.

- In steep terrain, landslides and debris flows have been identified as the primary sources of sediment inputs into streams and have been consistently shown to significantly increase in response to forest harvesting and forest roads in such terrain.

It is generally accepted that modern "best management practices" (BMPs), primarily improvements in road location, construction and use, and riparian management areas (RMAs) with buffers strips of forest vegetation along larger streams, have substantially reduced external sources of sediment into streams resulting from active forest management. However, forestry activities have occurred on a significant scale in Oregon for well over a century, mostly without modern BMPs, leaving a legacy of old forest roads in many watersheds, and unknown but potentially significant amounts of historic "legacy" sediment stored in Oregon waterways.

- Oregon forest practices for activities in landslide-prone terrain and for protection of smaller, non-fish bearing streams have not evolved to the same degree as for activities in other areas; scientific evidence regarding forest management effects on sediment and water quality must be interpreted in this context.

- There is growing recognition of the role and importance of forest harvesting effects on hydrologic regimes as drivers of sediment movement, e.g. the potential for increases in water yields and peak flows after harvesting to remobilize sediment stored in a stream, increasing suspended sediment and turbidity even in the absence of increased sediment inputs from sources external to the stream.

- Variability in research findings across different studies regarding sediment production from active forest management may be explained in some cases or to some degree by differences in geology (soil and rock type) and geomorphology (e.g. slope) and how these factors affect erodibility of sediments.

- The limited evidence available regarding larger, catchment-scale effects of forest operations and roads indicates that suspended sediment increases in the downstream direction as the size of the waterway increases.

In summary, the potential for forest operations to affect sediment mobilization and movement through drinking water source watersheds is higher for operations in steep, landslide-prone terrain, in areas with relatively more erodible soil and rock types, areas with a significant areal extent of unbuffered small streams, or where previous operations have left significant amounts of bare mineral soil or sediment stored in streams. Linkages between forest management and sediment production will increasingly be complicated (and potentially exacerbated) by predicted shifts in weather patterns associated with anthropogenic climate change, including increases in storm frequency and intensity, and in the proportion of winter precipitation falling as rainfall vs snowfall.

10.6 Forest chemicals findings and recommendations

Chemicals play an integral role in the management of Oregon's forests. Based on an analysis of ODF's FERNS data, there are over 7,400 activities that involve chemical applications on potentially one million acres of Oregon forest land annually, with the vast majority of these being herbicide applications to harvested units. Applications range from herbicide spraying for site preparation prior to replanting, and competing vegetation control afterwards, animal and rodent repellants to protect seedlings, fertilization to increase growth rates after thinning, and for maintenance of rights-of-way for both travel and utility corridors. With the exception of rights-of-way, a defining characteristic of these chemical applications is that they occur infrequently over the 30 – 80 year typical harvest cycle (Figure 6-1). And while the public perceives chemical use in forests as significant, pesticides applied to forest land represent only about from 2.8% (2007) to 4.2% (2008) of those used statewide according to data reported through the Oregon Pesticide Use Reporting System that was defunded in 2009. Accordingly, it's relevant that only 3.5% of pesticide-related incidents from the more recent ODA data involve forestry use of pesticides, and that about half of these are requests for staff to observe applications.

In comparison to other sectors of Oregon's economy that use pesticides, those typically applied in forestry are less toxic to humans, move fairly rapidly through soil and water, and don't accumulate. Most of these are herbicides that are not strongly absorbed (attached) to soil particles, are water soluble, have low volatility (i.e. evaporation and resuspension), and decay rapidly in both water and soil. This means that these herbicides don't tend to build up in the soil or bio-accumulate.

Contemporary best management practices, with a couple of additions, have the potential to protect areas off-site from the pesticide application if followed. Extensive research (and accompanying models) have allowed a better understanding of the importance of droplet size distributions on reducing pesticide drift, as has the development of adjuvants specifically tailored to mitigate drift. Helicopters have precise GPS and nozzle flow data loggers that accurately position the ship both in space and chemical delivery; some models can be preprogrammed to include flight plans that automatically buffer streams and sensitive areas. There is also substantial research from the agriculture community, and one paper reported here from forestry, on the value of wooded buffers to prevent drift into streams. Additions to the Forest Practice Act rules recently proposed through an industry-environmental collaborative process would extend forested buffers along non-fish streams.

The evidence we examined demonstrates that while pesticides are commonly detected in surface waters, in almost all cases they are found in concentrations below levels that can be accurately measured. When quantifiable detections are found, as we've seen from the forestry use studies, they tend to be transient and most likely to occur either during application or in early season storms. In particular, unless live water is directly sprayed (a label violation for herbicides used in forest silviculture), most herbicide runoff occurs during the first winter storms. In one report this constituted 70% - 90% of the pesticide loadings, a finding that was confirmed by two other studies.

A caveat here, again, is that the impact of forest chemicals on downstream raw source water supplies will depend on the size of the contributing watershed, the proportion annually subject to chemical applications, and other land uses in the basin. There are substantial knowledge gaps regarding the exact timing, locations, areas, amounts and formulations of forestry pesticides applied and also the effectiveness of BMPs for their use. These knowledge gaps can be at least partially addressed via more rigorous monitoring and reporting. If chemicals are to continue to be an acceptable tool in forest management from a public perspective, there is the need for investments in understanding their fates at the watershed/catchment scale. Also, most studies on the effects of silvicultural chemicals to investigate their safety prior to being authorized for public sale and use were conducted on the active ingredient only. In actual use, these chemicals are just about always mixed with other active ingredients and/or adjuvants. The effects of these “tank mixes” are often unknown.

Recommendations related to forest chemicals:

1. ***Pesticide use data needs to be reported***. It is difficult for the stakeholders and the affected public to understand the impacts, positive and negative, of forest chemicals without good reporting data. This is part of a larger concern over pesticide use relating to air and water quality in Oregon. At present, data on pesticide and chemical use is not routinely reported, even at the aggregate level. While ODF FERNS provides information on where and possibly when forest chemicals will be used, it allows multiple chemicals to be listed over long periods of time, with no subsequent reporting on what was actually applied unless a complaint was filed. In 1999 the Oregon Legislature created the Pesticide Use Reporting System (PURS), but it was never adequately funded and implemented. When its sunset provision was proposed for renewal during the 2019 Legislative Session in HB2980 there was broad support from across the political spectrum (Oregonians for Food and Shelter to the Farmworkers Union) for PURS to be extended and funded. This bill died in the Ways and Means Committee as the Legislature adjourned. A bill more specific to forestry was also introduced, HB4168 that implements the aerial application procedures and reporting requirements identified in the Memorandum of Understanding for the “Oregon Strategy” drafted by the timber industry and the conservation community. This bill, too, died prior to passage in the House with adjournment. The Board of Forestry and ODF could by administrative rule change its notification system to require reporting and disclose chemicals used in management operations.

2. ***Current water quality sampling efforts are insufficient***. A corollary to the lack of pesticide use information is the relative sparseness of data on potential pesticide loadings in surface waters, particularly at the raw water intakes for public water supplies. Most current sampling at raw water intakes is not correlated with times of likely chemical pulses, i.e., the early winter storms. Moreover, it’s clear from the silvicultural herbicide applications studies reviewed that detections and concentrations in receiving waters are highly variable even within a storm event. There is a similar constraint in the grab samples and automatic samplers that are commonly used: they provide detection and concentration information at point(s) of time, but not loads (i.e., the total mass of the substance transported in water over a given period of time) since stream discharge is usually not measured during the sampling. Sampling and analysis techniques developed and applied by the

U.S.G.S., such as POCIS and SPMD have the capability to accurately integrate pesticide concentrations over longer time periods and, in conjunction with streamflow, the ability to estimate loads. These devices could be particularly beneficial at raw water intakes where there is concern over pesticide loadings and the quantity of water flowing into the intake is known.

3. ***Study designs need improvement***. The majority of studies focused on assessing the impact of pesticides on water quality can be loosely characterized as "reconnaissance" or "case studies" because of their study design and limited replicability. Most of the pesticide/herbicide peer-reviewed studies in the Pacific northwest, and other areas of the U.S. were conducted by industry or industry-supported organizations (NCASI) and tend to be short-term and locally-focused. They have the advantage of knowing exactly when and what was applied, have more site-specific sampling, but are limited because the applicators know that they are being studied which may affect their behavior. In contrast, the PSP and USGS studies sampled over a longer period, but the PSP studies didn't have exact amounts and timing of application, and may have missed storm events; while the USGS studies using a sampling method that integrated pesticide concentrations over time, but was still limited because of unknown application amounts and timing. Improved study designs would incorporate random, applicator- and landowner-blind sampling of pesticide applications. This approach is critical for developing replicable and reliable scientific results.

4. ***Pesticide fate modeling is a critical need***. Inference based on downstream measurements includes complex interactions between pesticide and environment, as well as assumptions on their spatial and temporal distribution, which still require significant research. A useful tool to answer many management questions is modeling. Complex hydrological models, such as the Soil and Water Assessment Tool (SWAT) could assist practitioners and regulators to understand the fate of silvicultural forest chemicals. The SWAT has been used for over 50 pesticide fate studies worldwide for agricultural practices, but not for pesticide fates in forest applications. While such process-based models have their limitations, they can provide a structured approach to evaluating herbicide movements at the watershed scale.

5. ***Pesticide Stewardship Partnerships***. The PSPs are good outreach tools, but don't produce replicable science. The PSP doesn't collect pesticide application data and locations in its "partnerships" and its sampling regimes aren't designed and implemented to catch episodic events (application, early winter storms) generally recognized to be when the highest concentrations are likely to be found. Additionally, the lack of streamflow data in these studies limits their ability to evaluate "loads" versus point concentrations. The benefits of the PSPs by involving landowners, applicators, and agency personnel could be further enhanced by better knowledge of pesticides applied and their timing, and better monitoring procedures as outlined above.

6. ***OSU Research Cooperatives provide a framework to support future studies***. Creating credible science in an arena as complex as forest chemical use requires long-term and intensive studies across the ownership landscape. One model to achieve this is the research cooperatives in the College of Forestry at Oregon State University. Since 1982 there has been an industry-agency-university cooperative studying forest revegetation that has a substantial record of accomplishments over its almost 40 year history, presently called the Vegetation Management Research Cooperative (http://vmrc.forestry.oregonstate.edu/). The VMRC has the partners and and

can bring the expertise needed to successfully conduct the type of herbicide transport and fate studies and modeling described here.

7. ***Wooded buffers prevent or reduce spray drift***. Directly spraying into live water is a label violation for most herbicides used in forest management. However, some small streams can be hard to detect and therefore may be inadvertently sprayed during aerial applications, resulting in herbicide detections downstream. Both pesticide fate studies from coastal Oregon demonstrated that non-buffered, small non-fish streams received spray during application. In contrast, another study demonstrated the efficacy of wooded buffers in capturing or deflecting fine spray drift. This finding is consistent with a number of studies on agricultural spray drift. The extension of wooded buffers to Small Non-fish (Type N) streams under the Forest Practice Act and its rules would protect these streams from drift, and reduce potential loadings downstream. Extension of spray exclusion zones along Type N streams is one of the proposals in the “Oregon Strategy” of state, timber industry, and conservation groups (Governor’s Office 2020). It is clear from the science that the effectiveness of these no-spray buffers would be improved if they were wooded.

10.7 Natural organic matter/disinfection byproducts findings and recommendations

The relationship between natural organic matter (NOM) and disinfection byproducts (DPB) is important because two DPBs, total haloacetic acids (HAA5) and total trihalomethanes (TTHM), are regulated by the U.S.E.P.A. under the Safe Drinking Water Act. These DPBs are created when carbon in water comes into contact with the chlorine disinfectant that is required to remain as residual throughout a water utility’s distribution system until the water comes out the tap. The carbon can be from natural sources, can result from human activities, may be added during water treatment, and may be formed through the disinfection process in the treatment plant.

The two regulated DBPs, HAA5 and TTHM, are respectively the fourth- and fifth-most frequent contaminant alerts and exceedances in the Oregon Health Authority’s database. Disinfection byproduct detections in finished drinking water show that in the vast majority of cases the utility relies on surface water as their primary source, and these samples are oftentimes taken at the end of long pipe runs. Most detections are isolated events, but a subset of water utilities (17%) have clusters of detections with absences in intervening years, while a smaller set (5%) have chronic, annual, detections of DBPs in their water systems. Further, most exceedances are within 150% of the maximum contaminant level.

Today, NOM is the raw water constituent that most often influences the design, operation, and performance of water treatment systems. In addition to its role in the formation of DBPs, NOM can overwhelm activated carbon beds used in water treatment and reduce their ability to remove organic micropollutants. NOM also contributes significantly to the fouling of membranes in all membrane technologies used in water treatment, and can promote microbial fouling and regrowth in water distribution systems.

Operationally, NOM is separated in two components: dissolved organic matter (DOM) and particulate organic matter (POM). A significant amount of fresh water DOM is derived from terrestrial soil organic matter (SOM) that underwent specific transformations that increased its affinity for an aqueous environment. The composition of fresh water DOM is believed to depend on the transformation of plant and decomposed animal compounds into humic-like substances. Freshwater DOM is an aggregation of spontaneous self-associated superstructures formed by plant-derived products of natural decay, such as lipids, amino sugars, sugars, terpene derivatives, aromatic condensed structures, and lignin-derived compounds.

Concentrations of constituents increase as a function of stream discharge, with their export being dominated by short-lived, wintertime high-discharge events. Low flows contain primarily organic detritus from non-vegetation sources (e.g., algal cells) while particles with vegetation and soil-derived POM dominated the high flows.

- Modelling indicates that many decades after harvesting the metabolism of DOM is still being affected. This is because carbon and nitrogen losses from the terrestrial system to waterways and the atmosphere increase due to reduced plant nitrogen uptake, increased SOM decomposition, and high soil moisture.

- During and after harvesting, if slash is removed and/or burned, dissolved organic carbon (DOC) and DOM are reduced due to the diminished amount of coarse woody debris remaining.

- Evidence for the Pacific Northwest area indicates that the main export of NOM and disinfection byproducts (DBP) is triggered by the first major rain event occurring in the fall.

- Wildfires are increasing in frequency and severity in the United States, which is likely altering the chemistry and quantity of NOM and DBP traveling outside forested watersheds. Wildfires consume a large portion of organic matter from the detritus layer, which leads to lower yields of water extractable organic carbon and organic nitrogen. Therefore, wildfires appear to trigger an overall reduction in water extractable terrestrial DBP precursor yield from detritus.

- The last 15 years of bark beetle infestation had a significant impact on water quality as a result of increased organic carbon release and hydrologic shifts induced by the tree dieback. Water quality is impacted nearly one decade after bark beetle infestation, but significant increases in total organic carbon mobilization and DBP precursors are limited to areas that experience massive tree mortality.

10.8 Fire risk findings and recommendations

The cause of recent wildfire catastrophes can be traced to multiple factors including the expanding urban footprint, human ignitions, droughts, and high-wind events. Wildfires remove litter, duff and vegetative cover leading to the creation or enhancement of hydrophobic soil layers, increasing surface runoff and erosion potential. Post-fire changes in water chemistry and sediment transport can increase pollutant loads.

Growing awareness of the expanding scale of wildfire risk to communities and watersheds and water supplies in the US has led to a wide range of research focused on fuel treatments to reduce post-fire impacts to watersheds and drinking water. Researchers are using wildfire simulation models to test hypothetical treatment scenarios and estimate the potential reduction in risk, identified by metrics that quantify adverse impacts including soil erosion and change in water yield.

Existing risk assessment technologies and frameworks do not explicitly examine the cross-boundary problem intrinsic to wildfire risk from large public wildlands. Wildfire risk concerns the estimation of expected loss, calculated as the product of the likelihood of a fire at a given intensity and the consequence(s). By contrast, wildfire exposure concerns the juxtaposition of threatened values in relation to predicted fire occurrence and intensity, without estimating potential loss. Methods used to assess wildfire exposure and transmission were summarized; then a detailed assessment of cross-boundary wildfire exposure in Oregon between major land tenures (private, public, state, and federal) and drinking water source areas was provided. These latter results for each community water supply will be included in an accompanying on-line atlas.

Predicted area burned in 100 years was highest for public water supply areas (PWSA) in the eastern Cascades, southwest Oregon, and eastern Oregon regions. Mean fire size, total annual area burned and the number of simulated fires that exposed PWSAs also varied substantially across the regions, with the largest fires and the highest area burned occurring in southwestern Oregon. There was high variability among the major land tenures and their contribution to PSWA wildfire exposure within and among PWSA regions (Fig. 11). The US Forest Service (Federal-FS) was the leading contributor to area burned in all but the Coastal region where private industrial lands were the largest contributor.

Firesheds were generated for each of the 140 PWSAs that experienced wildfire in our simulations. Firesheds represent the biophysical risk in and around PWSAs and the sources of risk in terms of ownership; and, they represent areas surrounding each PWSA that can ignite and transmit large wildfires that expose an individual PWSA. Fireshed boundaries are often magnitudes larger than the administrative boundary of the PWSA and can represent a mosaic of land tenures.

The juxtaposition of fire prone forests in and around critical municipal watersheds intermixed with a high number of homes and infrastructure, and in close proximity to dense urban areas under a changing climate, creates a complex fuel management problem. Forest management has the potential to reduce fuels and restore ecological resiliency; however, the scale of the risk will required a coordinated, multi-agency, multi land owner collaborative response. This will require coordinated and targeted fuel management and forest restoration activities that minimize the risk of fire exposure to public water supply areas, maximize landscape resilience to wildfire, and allow for beneficial wildfire management.

Translating the findings in this report to prioritize fuel management activities is straightforward. Maps of fire transmission to PWSAs can be used as priorities in scenario planning models to design and sequence project areas and treatment units within them. Including potential treatment costs and revenues associated with harvesting and fuels treatments into planning makes it possible to examine economic costs and benefits associated with forest management to protect water. The Fireshed maps are also useful for identifying the scale of risk to PWSAs and determining the relative contribution from different landowners. Newer initiatives like shared stewardship recognize that the increasing scale of risk requires cross-boundary prioritization and action to treat at the appropriate scale. Assessments of cross-boundary risk can be integrated into this process and used as a management objective to target forest management where wildfires are predicted to spread across federal and state boundaries and expose drinking water or other highly valued resources.

10.9 Findings and recommendations from the community water systems case studies

We conducted three case studies to delve deeper into how managers of forested drinking water supply watersheds identify and address management concerns that have affected/could affect source water. This includes how they collaborate with other landowners and managers to identify, monitor, and respond to these concerns. Water provider survey respondents were stratified by location (Coast, Dryside, or Valley), primary landownerships in source watershed(s), and size of systems. We then purposively chose three case studies, one from each geographic region. Cases were also selected to represent a range of relevant contexts and issues: 1) a public lands context with a proximate wildland-urban interface and extensive collaboration on source watershed management (Ashland); 2) a public lands context with less proximity, collaboration, and public engagement (Baker City); and 3) a private industrial forestland context and a small system (Oceanside). Key takeaways from these studies are presented below.

From the Ashland Case Study:

- A multi-partner effort like the Ashland Forest Restoration (AFR) project is necessary to incorporate the diverse social, economic, and ecological desires that the community of Ashland holds for the management of its watershed. This is particularly essential in the public lands ownership context, where the Forest Service must consider diverse public values in its decisions. Development of scientifically-sound monitoring and robust community plans helps address questions and foster adaptation.

- Activities necessary to reduce hazardous fuels and wildfire risk can be costly in areas with steep slopes and complex forest types. The AFR's strengths and ability to seek multiple authorities and programs to accomplish this work within and adjacent to the watershed is necessary; and expands outcomes beyond what the Forest Service alone could fund or accomplish.

- The City of Ashland has been proactive in articulating its interest in the watershed and using formalized structures and tools (MOU, community alternative, Master Stewardship Agreement, ratepayer fee) to participate in active forest management. Its investment in forestry staff and the fire department provides the human capacity necessary to be part of collaborative efforts.

From the Baker City Case Study:

- Regular, such as quarterly, communication between the Forest Service and a municipality with source watersheds on national forest land assists in maintenance of relationships and proactive capacity for identifying issues and opportunities. Field tours and opportunities to view the watershed and potential management issues together in person help increase mutual understanding of conditions, challenges, and opportunities. This helps keep drinking water source protection issues on the table when both partners are also busy with other responsibilities and projects.

- There can be city and community frustration with the time and other requirements of the NEPA process for management actions on federal land. Increased experience and exposure can help build mutual understanding through the process. Written documentation of agreements and meetings can assist in the creation of agreements and institutional memory, which is important in a context with the frequent personnel turnover that can occur in both the Forest Service and city management.

- Municipalities and other partners may aid federal partners in managing source watersheds by building political support and obtaining grant funding from sources not accessible to federal agencies.

From the Oceanside Case Study:

- More consistent and proactive communication between the Water District and private industrial timberland owners has enhanced cooperation. Communication has historically been intermittent as it has been solely based around issues with quarry operations or planned forest operations. Opportunities to learn more about each other's goals and processes may have increased mutual understanding. Foresters have toured the Oceanside treatment plant, and Water District commissioners and the watermaster have toured proposed forest operations.

- One industrial timberland owner's use of a process communication checklist is intended to help ensure that the Water District and other water providers are notified beyond what is required by Oregon's Forest Practices Act.

- In small rural landscapes with a limited number of landowners, individuals particularly matter. The interests and actions of the Water District staff and board, and company foresters, have made cooperation possible.

Although the case studies were conducted in three different contexts, there were common lessons learned from each case as well as common themes across cases that may offer broader insights.

1. ***Landownership frames the opportunities and challenges for managing source watersheds.*** The laws and regulations that govern different types of forestland ownerships set the stage for what management activities are permitted, how they are to be conducted, and any public involvement. For example, Oregon's Forest Practices Act provides standards for the establishment, management, and/or harvest of trees on private industrial and nonindustrial forest lands. Public lands managed by federal agencies such as the US Forest Service or the Bureau of Land Management are subject to an array of laws and policies, as well as land use designations and requirements for public participation in management decisions. Drinking water providers who seek to interact and collaborate with their source forestland managers must do so with understanding of these existing frameworks, and the time and effort that it may take to engage.

2. ***Regular communication provides a foundation for relationships.*** Regular communication between drinking water providers and source watershed land managers may assist the maintenance of relationships and proactive capacity for identifying issues and opportunities. This helps keep drinking water source protection issues on the table when both partners are also busy with other responsibilities and projects. Field tours and opportunities to view the watershed and potential management issues together in person may help increase mutual understanding of conditions, challenges, and opportunities. The scope and scale of this communication may necessarily vary by

context. For example, it may be more informal and involve far fewer parties in areas where source watersheds are spatially small and systems serve smaller populations. Regardless, the need for both land managers and drinking water providers to be intentional and proactive about communication with each other remains. Written documentation of agreements and meetings can assist in the creation of agreements and institutional memory, which is important when there is personnel turnover with any organization.

3. ***Specific projects offer opportunity to collaborate*****.** Planning forest management activities, a source water protect plan, or a monitoring effort can offer concrete ways for drinking water providers to engage with source watershed managers. Depending on the ownership of the source watershed, providers may be able to provide project design input, develop community plans, or create monitoring protocols. This may involve additional partners such as local nonprofits, government agencies, and community leadership. The opportunity to participate directly may improve understanding of source watershed conditions and needs, particularly though monitoring that could address uncertainties with scientific information. It can also bring leveraged funds from other sources that help support monitoring or management activities.

10.10 Final thoughts

The body of work here, and found in the supporting chapters, represents a substantial contribution towards understanding the effects of active forest management on drinking water source quality. The project's Steering Committee provided important perspectives and clarified priorities during our formative stage; and provided substantive reviews and comments as we crafted this report. Throughout, we have made every effort to be careful and critical in our reviews. We do not realistically expect that this report will resolve the many debates over forest management. However, we do hope that it will provide a common reference on current science and the policy context. If that is the case, then we will be satisfied.

THIS PAGE INTENTIONALLY LEFT BLANK

APPENDIX: METHODS

Jeff Behan and Emily Jane Davis

2.1. Drinking water provider survey methods

2.1.1. Question development and survey administration

Survey questions were developed through review of Adams and Taratoot (2001), relevant literature about drinking water utilities, and two rounds of input from the Trees to Tap Steering Committee (for survey instrument, see Appendix 1). Survey questions and methods were reviewed by OSU's Institutional Review Board and determined not to be human subjects-regulated activities. The questionnaire was finalized, tested, and administered using Qualtrics, an internet survey program for which OSU has an institutional license. The survey could be taken online or downloaded and returned via email or mail. The survey asked respondents to identify their community water system and their primary source watershed for drinking water, and then to respond to survey with reference to that primary source watershed only.

2.1.2 Recruitment and response rate

The survey recruitment process took place in several steps. A press release about the survey was published in the Spring 2018 issue of *H_2Oregon* (Vol. 40, No. 2), the quarterly newsletter of the Oregon Association of Water Utilities, of which most community water suppliers are members. Then, an email notifying the entire identified population of 156 drinking water providers was sent to inform them of the pending survey, then an invitation was sent in mid-May 2018. Finally, five follow-up email reminders were provided at weekly intervals, and several phone calls were taken to address participant questions.

By mid-June, the response rate remained below satisfactory. This was likely due to the summer season and concurrent source water issues in some areas of the state regarding algae. The research team consulted with the Steering Committee on a preferred approach for targeted follow up. Based on their interest, we identified the 20 systems with highest percentage of industrial private timberland, 20 with highest percentage of publicly-owned land, and ten owned by local governments. Forty-three phone calls with follow up emails were placed to all on these lists who had not yet taken the survey, eliciting 13 additional responses. Of these, five were completed while on the phone with the researcher, who entered the data. Final response totaled 54 systems, or 35 percent of the 156 systems in Oregon.

2.1.3. Analysis and review

Survey data were downloaded from Qualtrics, cleaned, and analyzed with Microsoft Excel using basic descriptive statistical approaches. The draft survey report was reviewed by Jon Souder and by the project Steering Committee.

2.1.4. Potential limitations

Although every identified system in Oregon had the opportunity to complete the survey, those with particular interest in the topics covered or the project may have been more inclined to take it.

Perspectives of potential respondents who were too busy, had disinterest or distrust in the project, or otherwise did not complete it are therefore not as well represented.

2.2. Science literature search and review methods

2.2.1. Scope of literature search

The Trees to Tap project included a literature search and review of science knowledge regarding the effects of active forest management on community water. The scope of the literature search and review was guided by Steering Committee discussions regarding the various ways in which active forest management and community water supplies intersect, and the level of public and agency interest in each. After these deliberations, the Steering Committee directed the reviewers to address four topic categories; in priority order, they are:

1. How active forest management may alter patterns and amounts of *sediment* delivered into waterways that serve as drinking water sources, and changes in sediment loads and turbidity at drinking water intakes;
2. How active forest management may alter the *quantity* of water delivered from forested watersheds that serve as drinking water sources, and temporal patterns of this delivery;
3. How the silvicultural use of *herbicides, pesticides* and *fertilizers* may affect drinking water; and
4. How active forest management may alter the patterns and amounts of *natural organic matter* (NOM) in community water sources, which in turn may affect the formation of *disinfection byproducts* (DBPs) in finished drinking water

Together, these topics encompass multiple processes and interactions that vary with ecosystem type, topography, geomorphology, scale and specific forest practices. *The effects of forestry activities on sediment production, water quantity and timing of water delivery, and forestry pesticides and fertilizers have been researched extensively. However, most of this work has been conducted in upper watersheds, with a focus on hydrological and ecological effects in that context. Very little research has addressed or quantified direct linkages to drinking water.* Issues related to NOM and DPBs have come into prominence more recently. To varying degrees, each topic encompasses a range of potentially relevant literature. For example, research on forestry activities and sediment input into streams has focused on issues ranging from harvesting practices and stream buffers to forest road location, construction, maintenance and use, to landslides, mass wasting, cumulative effects, and other factors.

The Steering Committee and review team discussed tradeoffs inherent in choosing between a wider scope encompassing four topics versus a deeper, more comprehensive examination of a single topic. The Steering Committee concluded that despite these tradeoffs, each of the four topic areas warranted attention. Criteria for this decision included level of public interest, relevance to provision of clean, safe drinking water, and potential availability of relevant scientific literature. This wider scope precluded a systematic literature search and review, which comprehensively examines a single narrow, targeted science question or set of closely related questions. Addressing four relatively complex topics also constrained the amount of effort that could be dedicated to searching for and compiling literature on each individual topic.

2.2.2. Literature search methods

The primary goal of the literature search was to find articles and studies assessing direct linkages between active forest management practices and effects on drinking water at the community water intake. It is generally recognized that forest practices in source watersheds can affect drinking water, but the science reviewers surmised and subsequent searching confirmed that relatively few studies have directly addressed these linkages in a scientifically rigorous manner. Instead, relevant science information is mostly distributed more diffusely across studies addressing effects of forest practices on soil and water quality higher in the watershed. Rather than a body of easily identifiable, focused and highly relevant studies, initial searching indicated the availability of a much larger volume of partially relevant science from which selected findings and inferences could be drawn. Thus, our overall approach was to cast a wide net for potentially relevant literature, using multiple sources, and to adopt relatively generous literature inclusion criteria. This allowed us to generate a broad base of potentially relevant literature from which the science reviewers could select a subset of references with the most reliable and useful information.

Primary literature searches

Electronic literature searches were conducted primarily using GoogleScholar, and also the Web of Science, 1Search, CAB Abstracts, ProQuest and TreeSearch academic literature databases. *A reference librarian was consulted to discuss search strategies and to help ensure that all potentially useful literature databases and sources were identified.* Searching was also done via “related articles” tools in GoogleScholar and other academic databases, using particularly relevant or highly cited references as the root. Additional searches were conducted within the online archives of some particularly relevant journals, such as the Journal of the American Water Resources Association and Journal of Soil & Water Conservation.

Table A-1 shows over 200 keyword strings that were searched. Most of these are unique but some strings were searched in more than one database. For each set of search returns, the first 50 “hits” were assessed for relevance by looking at the title, then abstract or summary, then full text as necessary on a case-by-case basis. The assumption here was that using a wider diversity of search terms was likely to be more productive than digging deeper than 50 results for each search string. If a search produced fewer than 50 returns, all were assessed. Most searches were time delimited “since 2000” so the compiled literature is mostly from this time period, but some older, highly cited, seminal or otherwise distinctive or particularly relevant references were compiled as well.

Table A-1: Databases and keywords searched

Search Terms - Google Scholar	**Search Date**	**Results-Anytime**	**Results-Time Limited**
"forest management" "drinking water"	2/5/2018	22,600	16,100 since 2000
"forest practices" "community water systems"	2/5/2018	57	
"forest practices" "drinking water systems"	2/5/2018	52	
"forest practices" "drinking water"	2/7/2018	1850	1470 since 2000
"forest practices" "drinking water" "pacific northwest"	2/8/2018	720	531 since 2000
"forest practices" "community water supply" "pacific northwest"	2/20/2018	33	
"forest practices" "municipal water systems"	2/5/2018	43	

Search Terms - Google Scholar	Search Date	Results-Anytime	Results-Time Limited
"forest practices" "municipal water"	2/5/2018	499	355 since 2000
"forest practices" "municipal watersheds"	2/28/2018	140	75 since 2000
"forest practices" "water supply"	2/12/2018	3,180	2,280 since 2000
"forest practices" "municipal water supply"	2/15/2018	157	104 since 2000
"forest practices" "turbidity" "municipal water"	2/15/2018	183	
"forest practices" "water treatment"	2/13/2018	674	497 since 2000
"forestry" "water supply" "Oregon"	2/12/2018	17,500	12,000 since 2000
"sediment" "logging" "drinking water"	2/6/2018	13,500	11,100 since 2000
"sediment" "timber harvesting" "drinking water"	2/6/2018	2,020	1,630 since 2000
"turbidity" "timber harvesting" "drinking water"	2/6/2018	927	727 since 2000
"turbidity" "coast range" "drinking water"	2/15/2018	487	374 since 2000
"timber harvesting" "drinking water"	2/6/2018	4,190	3,390 since 2000
"timber harvesting" "municipal water"	2/6/2018	962	698 since 2000
"timber harvesting" "water treatment"	2/13/2018	1,620	1,310 since 2000
"timber harvesting" "community water supply"	2/20/2018	110	
"timber harvesting" "community water supply" "pacific northwest"	2/20/2018	36	
"sediment" "forests" "drinking water"	2/6/2018	25,600	17,000 since 2000
"forest practices" "Oregon" "drinking water"	2/6/2018	928	699 since 2000
"forest practices" "water supply" "Oregon"	2/6/2018	1730	1200 since 2000
"forest practices" "water supply" "pacific northwest"	2/20/18	1,320	928 since 2000
"forest practices" "herbicides" "municipal water"	2/12/2018	114	
"forest herbicides" "municipal water"	2/12/2018	2	
"forest herbicides" "drinking water"	2/15/2018	44	
"forest herbicide*" "drinking water"	5/13/2018	69	40 since 2000
"forest herbicide*" "water" "oregon"	5/13/2018		145 since 2000
"forest regeneration" "herbicides" "municipal water"	2/12/2018	48	
"forest regeneration" "herbicides" "drinking water"	2/12/2018	304	228 since 2000
"Coast range" "water treatment"	2/13/2018	769	535 since 2000
"Coast range" "forest practices" "water treatment"	2/13/2018	103	
"logging roads" "water treatment"	2/14/2018	465	345 since 2000
"water quantity" "timber harvest" "coastal rainforest"	2/15/2018	7	
"timber harvesting" "stream volume"	2/15/2018	60	
"timber harvesting" "water production"	2/16/2018	533	342 since 2000
"timber harvesting" "water yield"	2/16/2018	2,510	1,660 since 2000
"timber harvesting" "water yield" "pacific northwest"	2/16/2018	921	619 since 2000
"forest practices" "water yield" "pacific northwest"	2/16/2018	657	445 since 2000
"forest management" "municipal water"	2/21/2018	2,700	2,120 since 2000
"forest management" "municipal water" "pacific northwest"	2/22/2018	698	519 since 2000
"forest management" "drinking water"	2/22/2018	22,800	15,900 since 2000
"forest management" "drinking water" "Oregon"	2/22/2018	3,340	2,620 since 2000
"forest watersheds" "impacts" "municipal water"	2/22/2018	142	
"forest watersheds" "impacts" "drinking water"	2/22/2018	542	
"timber harvest" "drinking water" "Oregon"	2/23/2018	1,500	1, 160 since 2000
"turbidity" "community watersheds"	2/28/2018	112	
"forest practices" "community watersheds"	2/28/2018	245	184 since 2000
"forestry" "municipal watersheds"	3/01/2018	616	395 since 2000
"forest roads" "municipal watersheds"	3/01/2018	138	
"municipal watersheds" "Oregon coast"	3/01/2018	74	

Search Terms - Google Scholar	**Search Date**	**Results-Anytime**	**Results-Time Limited**
"municipal watersheds" "turbidity" "Oregon"	3/01/2018	107	
"carbaryl" "forestry" "drinking water"	3/08/2018	1,050	816 since 2000
"carbaryl" "forest practices" "drinking water"	3/08/2018	35	
"atrazine" "forestry" "drinking water"	3/09/2018	2,940	2,360 since 2000
"atrazine" "forestry" "municipal water"	3/09/2018	264	
"forest harvesting" "drinking water" "Oregon"	3/15/2018	574	462 since 2000
"forest harvesting" "community water"	3/20/2018	164	123 since 2000
"forest harvesting" "municipal water"	3/20/2018	294	229 since 2000
"best management practices" "municipal water" “forestry”	3/21/2018	1,340	
"forestry best management practices" "municipal water"	3/21/2018	72	
"best management practices" "municipal water" "Oregon"	3/21/2018	1,120	953 since 2000
"forestry best management practices" "drinking water" "oregon"	3/21/2018		107 since 2000
"best management practices" "coast range"	5/10/2018	1,400	1,150 since 2000
"disinfection byproducts" "timber harvesting"	4/02/2018	40	
"disinfection byproducts" "forest harvesting"	4/03/2018	25	
"disinfection byproducts" "forests"	4/02/2018	556	
"disinfection byproducts" "forest roads"	4/03/2018	5	
"disinfection byproducts" "forestry"	4/03/2018	499	472 since 2000
"disinfection byproducts" "clearcutting"	4/03/2018	14	
"disinfection byproducts" "coast range"	4/03/2018	40	
"disinfection byproducts" "silviculture"	4/03/2018	55	
"disinfection byproducts" "logging"	4/03/2018	290	
"dissolved organic carbon" "timber harvesting" "Oregon"	4/03/2018	368	297 since 2000
"water treatment" "forest practices"	4/4/2018	683	504 since 2000
"dissolved organic carbon" "timber harvesting"	4/6/2018	841	706 since 2000
"dissolved organic carbon" "forest harvesting" "municipal water"	4/9/2018	37	
"dissolved organic carbon" "forest harvesting" "community water"	4/9/2018	11	
"dissolved organic carbon" "forestry" "community water"	4/9/2018	123	
"dissolved organic carbon" "forest practices" "oregon"	4/12/2018	219	173 since 2000
"water treatment" "forest practices" "Oregon"	4/8/2018	354	249 since 2000
"forest harvest" "drinking water" "oregon"	4/12/2018	379	312 since 2000
"forest operations" "drinking water"	4/19/2018	809	673 since 2000
"landslides" "drinking water" "forest harvest*"	4/23/2018	155	124 since 2000
"landslide*" "turbidity" "Oregon"	4/25/2018	2,410	1,820 since 2000
"forest engineering" "drinking water" "oregon"	4/26/2018		134 since 2000
"forest engineering" "municipal water" "oregon"	4/27/2018		29 since 2000
"trihalomethane*" "forest*" "Oregon"	5/2/2018	184	138 since 2000
"conifer release" "drinking water" "oregon"	5/3/2018	36	
"conifer release" "drinking water"	5/3/2018	58	
"conifer release" "municipal water"	5/3/2018	15	
"conifer release" "oregon" "coast range"	5/3/2018		83 since 2000
"conifer release" "water quality"	5/3/2018		125 since 2000
"forest roads" "herbicide*" "drinking water"	5/7/2018	217	175 since 2000
"aerial herbicide spraying" "forests"	5/10/2018		122 since 2000
"vegetation control" "douglas fir" "water quality"	5/14/2018	420	340 since 2000
haloacetic acid* forest* "water quality"	5/14/2018	916	864 since 2000
haloacetic acid* "forest management" "water quality"	5/14/2018		48 since 2000
"disinfection byproducts" "forest harvest" "water treatment"	5/14/2018	6	

Search Terms - Google Scholar	Search Date	Results-Anytime	Results-Time Limited
"dissolved organic matter" "forest management" "drinking water"	5/16/2018		293 since 2000
"dissolved organic matter" "forest harvest" "drinking water"	5/16/2018		61 since 2000
"dissolved organic matter" "forest operations" "oregon"	5/18/2018		34 since 2000

Search Terms - 1Search, OSU Valley Library	Search Date	Results-Anytime	Results-Time Limited
"forest practices" "drinking water" [material type: all sources]	2/26/2018	294	
"forest practices" "drinking water" [material type: articles]	2/26/2018	131	
"forest practices" "municipal water" [material type: articles]	2/26/2018	19	
"timber harvesting" "drinking water" [material type: articles]	2/26/2018	341	
"logging" "drinking water" [material type: articles]	2/27/2018	6,614	2,839 since 2000
"logging" "drinking water" "Oregon" [material type: articles]	2/27/2018	369	248 since 2000
"logging" "municipal water" [material type: articles]	2/27/2018		339 since 2000

Search Terms – Proquest, OSU Valley Library	Search Date	Results-Anytime	Results-Time Limited
"municipal water" "forest harvesting"	4/5/2018	55	
"municipal water" "timber harvesting"	4/5/2018	448	438 since 2000; 27 filtered for "scholarly journals"
"community water" "timber harvesting"	4/5/2018	127	126 Since 2000; 10 filtered for "scholarly journals"
"drinking water" "timber harvesting"	4/5/2018		135 since 2000, filtered for "scholarly journals"

Search Terms - CAB Abstracts, OSU Valley Library	Search Date	Results-Anytime	Results-Time Limited
("timber harvesting" OR "forest harvesting" OR "logging") AND ("municipal water" OR "community water" OR "drinking water")	4/10/2018		106 since 2000 within USA
(forestry) AND ("municipal water" OR "community water" OR "drinking water") yr:[2000 TO 2017]	4/10/2018		32 since 2000 within USA
(BMP) AND ("municipal water" OR "community water" OR "drinking water") yr:[2000 TO 2017]	4/10/2018		20 since 2000
("forest roads") AND ("municipal water" OR "community water" OR "drinking water") yr:[2000 TO 2017]	4/10/2018		2 since 2000
"forest practices" AND "disinfection byproducts"	4/10/2018		No results found
"forests" AND "disinfection byproducts"	4/10/2018		16
"forests" AND "atrazine" AND "water"	4/10/2018		44 within USA
"forests" AND "carbaryl" AND "water"	4/10/2018		27 within USA
"forest" AND "disinfection byproducts" AND "water"	4/10/2018		13 within USA
"forest management" AND "municipal water"	4/10/2018		8
"forest management" AND "drinking water"	4/10/2018		14 within USA
"forest harvest" AND "dissolved organic carbon" AND "drinking water"	4/12/2018	No matches	
"forest harvest" AND "dissolved organic carbon"	4/12/2018	25	
"timber harvest" AND "dissolved organic carbon"	4/13/2018	39	
"logging" AND "dissolved organic carbon"	4/13/2018	237	36 within USA
"forestry" AND "dissolved organic carbon"	4/13/2018	256	55 within USA
"forest" "atrazine" "drinking water"	4/16/2018	3	
"forest" "atrazine" "municipal water"	4/16/2018	0	

Search Terms - CAB Abstracts, OSU Valley Library	Search Date	Results-Anytime	Results-Time Limited
"forest" "herbicide" "municipal water"	4/16/2018	1	
"forest" "herbicide" "drinking water"	4/16/2018	27	
"tree regeneration" "herbicide" "water"	4/16/2018	1	
"conifer" "herbicide" "water"	4/16/2018		94 within USA 4 since 2000
"forest" "water treatment" yr:[2000 TO 2017]	4/16/2018		94 within USA since 2000
"Coast range" "water treatment"	4/16/2018	0	
"forest" "drinking water" "Oregon"	4/16/2018	8	
"forest harvest" "municipal water"	4/16/2018	0	
"forest harvest" "drinking water"	4/16/2018	3	
"water quality" "forest roads" "oregon"	4/16/2018	4	
"water quality" "timber harvest" "oregon"	4/16/2018	24	
"water quality" "forest roads"	4/16/2018	102	28 within USA since 2000
"water quality" "logging roads"	4/17/2018	17	
"water quality" "forest management"	4/17/2018	605	159 within USA since 2000

Search terms - Web of Science, Valley Library	Search Date	Results-Anytime	Results-Time Limited
"forest harvest" AND "water"	4/20/2018		88 since 2000
"forest* harvest*" AND "water"	4/20/2018		284 since 2000
atrazine AND "drinking water" AND forest*	4/20/2018	5	
atrazine AND "water" AND forest*	4/20/2018	78	
"disinfection byproducts" "forest*"	4/23/2018	7	
"disinfection byproduct" "forest*"	4/23/2018	14	
"disinfection byproducts" "logging"	5/10/2018	0	
"dissolved organic carbon" "forest*" "drinking water"	4/23/2018	51	
"dissolved organic carbon" "forest* harvest*"	4/23/2018	42	
"dissolved organic carbon" "logging"	5/10/2018	56	
"forest operations" "water" "oregon"	4/23/2018	1	
"forest operations" "water" "pacific northwest"	4/23/2018	1	
"forest harvest*" "water" "pacific northwest"	4/23/2018	23	
"forest engineering" "drinking water"	4/23/2018	0	
"forest engineering" "municipal water"	4/23/2018	0	
"forest engineering" "water"	4/23/2018	3	
"landslide*" "forest*" "water"	4/23/2018		57 within USA since 2000
"drinking water" "oregon"	4/25/2018		32 since 2000
"municipal water" "oregon"	4/25/2018		7 since 2000
“forest herbicides” “water”	4/25/2018	1	1 since 2000
"forest herbicides" "oregon"	5/3/2018	1	
"forest herbicides" "coast range"	5/3/2018	2	
"forest herbicide*" "pacific northwest"	5/3/2018	1	
"forest* herbicide*" "water"	5/2/2018	5	
"atrazine" "forest*" "water"	4/25/2018		39 within USA
"carbaryl" "forest*" "water"	4/25/2018	9	
"culvert*" "forest*" "turbidity"	4/25/2018	5	
"herbicide*" "forest*" "water"	5/2/2018		134 within USA

Search terms - Web of Science, Valley Library	Search Date	Results-Anytime	Results-Time Limited
"drinking water" "forest*" "oregon"	5/3/2018	2	
"drinking water" "forest*" "pacific northwest"	5/3/2018	1	
"municipal water" "forest*" "pesticides"	5/3/2018	0	
"water" "forest*" "pesticides"	5/3/2018		93 within USA
"aerial spraying" "forest*"	5/3/2018	48	
"aerial spray*" "timber*"	5/3/2018	0	
"aerial spray*" "regeneration"	5/3/2018	7	
"aerial spray*" "water"	5/10/2018	71	
"aerial spray" "oregon"	5/10/2018	1	
"herbicide*" "regeneration"	5/3/2018		117 since 2000 in topic "FORESTRY"
"best management practices" "oregon"	5/10/2018	34	
"streamside management zone*" "oregon"	5/10/2018	7	
"turbidity" "oregon"	5/10/2018	58	
"conifer release" "Oregon"	5/10/2018	5	
"conifer release" "water"	5/10/2018	3	
"conifer release" "herbicide"	5/10/2018	48	
"clearcut" "regeneration" "water"	5/11/2018	26	
"forest" "vegetation control" "water"	5/14/2018	54	
"understory vegetation control" "water quality"	5/14/2018	0	
"understory" "vegetation control" "water quality"	5/14/2018	0	
"vegetation control" "douglas fir" "water quality"	5/14/2018	0	
trihalomethane* forest*	5/14/2018	30	
haloacetic acid* forest*	5/14/2018	18	
"dissolved organic matter" "forest* harvest*" "water"	5/17/2018	8	

Journal searches:

Title or database	Search terms	Search Date	Returns
Springer Link	drinking water timber harvesting (since 2000)	3/22/2018	637
Springer Link	atrazine drinking water forestry (since 2000)	3/22/2018	63
Springer Link	herbicides drinking water forestry (since 2000)	3/22/2018	225
Springer Link	forestry AND drinking water AND Oregon (since 2000)	3/22/2018	129
Springer Link	timber harvesting AND drinking water AND Oregon (since 2000)	3/22/2018	39
Journal of Soil & Water Conservation	"forest*" "drinking water" [2000-2018]	5/18/2018	
Journal of Soil & Water Conservation	"forest" "harvest" "water" [2000-2018]	5/18/2018	195
Journal of Water & Health	"forestry"	5/18/2018	30 since 2000

TreeSearch (US Forest Service database):
https://www.fs.usda.gov/treesearch/
"drinking water" [since 2000] 49 results (reviewed 49)
"municipal water" [since 2000) 17 results (reviewed 17)
"community water" [since 2000) 1 result (reviewed 1)
"water quality" [since 2000] 558 results (1st 50 reviewed)

Medline
https://www.ncbi.nlm.nih.gov/pubmed?holding=orosulib
Search string suggested by Reference Librarian, searched 4/12/2018
("forests"[MeSH Terms] OR "forests"[All Fields] OR "forest"[All Fields]) AND ("pesticides"[Pharmacological Action] OR "pesticides"[MeSH Terms] OR "pesticides"[All Fields]) AND ("drinking water"[MeSH Terms] OR ("drinking"[All Fields] AND "water"[All Fields]) OR "drinking water"[All Fields]) *10 results, none relevant*
"forestry" "drinking water" *32 results none relevant*

Supplementary literature sources

Assessing a reference for potential relevance often required scanning the text or searching it for keywords, which sometimes revealed additional potentially relevant references cited in the text. In other cases, bibliographies were scanned directly. Publication lists on the websites of several researchers working on pertinent topics in the Pacific Northwest were scanned for relevant publications. Steering Committee members were also solicited for relevant literature. Some members submitted references.

During preparation of individual report chapters, many references in the spreadsheet were read more closely and additional topical searches were conducted to clarify certain points or follow leads. The science reviewers cited some additional literature from their own professional libraries, bibliographic databases and supplementary searches. Commenters on draft chapters sent out for review sometimes suggested additional references. New references identified via these processes were added to the spreadsheet on an ongoing basis. As of March, 2019, the literature spreadsheet contained approximately 800 references.

Compiling the literature

Literature that appeared to contain potentially relevant information was compiled in an Excel spreadsheet, with fields for:

- Person searching
- Date of search/date entered
- Database
- Keywords used
- Full citation
- URL
- Publication type (e.g. peer-reviewed, book chapter, agency report, NGO report, etc.)
- Publication date
- Geographic location (of study, if applicable)
- Abstract or summary
- Notes or pertinent details, i.e. where found if not via systematic search

Topic categories

References were organized into categories in the spreadsheet. The most appropriate category for each reference was not always clear. Where appropriate, some were entered under more than one category. The categories were as follows:

1. Synthesis, general, background, context
2. Water yield, peak flows
3. Forest roads
4. Stream & riparian buffers, streamside management zones
5. Best Management Practices
6. Skid trails, harvesting equipment
7. Turbidity, sediment – general
8. Geology, geomorphology, groundwater hydrology
9. Landslides & debris flows
10. Forest herbicides & pesticides
11. Mercury
12. Dissolved Organic Matter, dissolved organic carbon, disinfection byproducts, etc.
13. Forest fertilization
14. Nutrients: Nitrogen, nitrates, phosphates, potassium, etc
15. Forest management and water chemistry-general
16. Wildfire, fuels reduction, salvage logging and community water
17. Climate change and community water
18. Policy, social science, economics, case studies
19. Older literature (pre-1990)

Types of references

Within each category, references were further subdivided based on type of reference, including these:

- Peer reviewed journal publication
- Peer-reviewed agency reports (such as USFS and USGS reports)
- Proceedings paper
- Workshop report
- NGO report
- Book
- Book chapter
- Thesis/dissertation

Within each subcategory, references were listed by publication date, starting with the most recent first.

2.2.3. Limitations of the literature search

As explained above, the literature search was relatively broad in that it addressed several different ways that active forest management may affect community water. However, the search was also limited in that was not designed to encompass other values associated with water quality, e.g. impacts that affect salmonid fish. Similarly, while the effects of forest practices on forest soil parameters can and do affect water quality in a number of ways, these effects were not the primary focus of the searches. In order to increase the likelihood of finding relevant literature, and limit the results of individual searches to manageable numbers, most search strings were specifically focused on detecting literature linking forest practices and *drinking* water.

The ways that scientific research is published and catalogued may have affected our ability to find potentially relevant literature. Authors of scientific manuscripts are usually asked to suggest a set of keywords for their papers. In other instances, reference librarians or database managers may choose keywords. In both cases, keyword selection is a somewhat subjective matter, and varies depending on the particular journal the research is being published in, how the author chooses to frame and present their research, the ways reference librarians and database managers choose to catalog the work, and other factors. The result is that keywords such as "water" or "drinking water" may not be selected, even though the research may have significant implications for drinking water.

In short, research on the effects of forest practices on forest soils, fish habitat and other ecological aspects of water may also be relevant to drinking water; these effects are usually not mutually exclusive. But relationships between forest practices and drinking water are often not made explicit in the literature. In light of the complex ecological inter-relationships among forest practices and effects on water, and the diverse manner in which research on relationships between forestry and water is published and catalogued, it is inevitable that we did not identify all potentially relevant research. Because research that explicitly makes these linkages is limited, the search results include a wide range of tangentially relevant research.

Lastly and as noted above, the tradeoffs between a wider scope encompassing four topics versus a deeper, more comprehensive examination of a single topic resulted in inevitable limitations to the amount of effort that could be applied to searching for and reviewing literature for each topic.

2.2.4. Use of literature search results

As explained above, relatively generous literature inclusion criteria were adopted during the literature search, in order to demonstrate the breadth of information available and compile as much potentially useful literature as possible. In addition to peer-reviewed primary research literature, a wide range of other literature was compiled, such as reviews and syntheses, gray literature including agency reports such as USFS General Technical Reports (GTRs), non-governmental organization (NGO) reports, conference proceeding papers, book chapters, and graduate theses and dissertations. Selected literature discussing linkages between climate change, forest practices, and drinking water; wildfire, fuels reduction and drinking water; and policy-oriented literature was also compiled.

A subset of the hundreds of compiled references was reviewed in depth for the science reviews in chapters 5, 6, 7 and 8. The Steering Committee discussed how to winnow the extensive literature base and prioritize literature for inclusion in the science review. There was general agreement that results from studies in Pacific Northwest conifer forests published since 2000 would provide the most relevant science information, with papers from peer reviewed science journals being the most reliable. References such as USFS General Technical Reports (GTRs), USGS water reports, EPA reports, and water agency reports are often subject to some level of peer review and were also deemed potentially useful. The review scope included all forest types in Oregon with a particular focus on wetter, west side Cascade and Coast Range forests.

Reviewers focused primarily on peer-reviewed literature identified during the literature search, augmented with additional literature from their own professional libraries, bibliographic databases and supplementary searches. A broader range of resources compiled in the spreadsheet was used in various places in the report to provide background and context for the science review chapters.