Climate Change Vulnerability and Adaptation in the Blue Mountains Region

Editors
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Abstract

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The Blue Mountains Adaptation Partnership (BMAP) is a science-management partnership consisting of Malheur National Forest, Umatilla National Forest, Wallowa-Whitman National Forest, the U.S. Forest Service Pacific Northwest Research Station and Pacific Northwest Region, the University of Washington, and the Climate Impacts Research Consortium at Oregon State University. These organizations worked together over a period of two years to identify climate change issues relevant to resource management in the Blue Mountains region and to find solutions that can minimize negative effects of climate change and facilitate transition of diverse ecosystems to a warmer climate. The BMAP provided education, conducted a climate change vulnerability assessment, and developed adaptation options for federal agencies that manage 2.1 million hectares in northeast Oregon, southeast Washington, and a small portion of southwest Idaho.

Global climate models project that the current warming trend will continue throughout the 21st century in the Blue Mountains. Compared to observed historical temperature, average warming is projected to be 2.4-3.1 °C by 2050 and 3.2-6.3 °C by 2100, depending on greenhouse gas emissions. Precipitation may increase slightly in the winter, although the magnitude is uncertain.

The effects of climate change on hydrology in the Blue Mountains will be especially significant. Decreased snowpack and earlier snowmelt will shift the timing and magnitude of streamflow and decrease summer soil moisture; peak flows will be higher, and summer low flows will be lower. Pronounced changes in snow and streamflow will occur in headwater basins of the Wallowa Mountains, especially in high-elevation radial drainages out of the Eagle Cap Wilderness, with large changes occurring in the more northerly sections of the Umatilla and Wallowa-Whitman National Forests along the Oregon-Washington border. Mid-elevation areas where snow is currently not persistent (northern Blue Mountains, margins of Wallowa, Elkhorn, Greenhorn, and Strawberry Mountains) may become largely snow-free in the future.

Projected changes in climate and hydrology will have far-reaching effects on aquatic and terrestrial ecosystems, especially as frequency of extreme climate events (drought, low snowpack) and associated effects on ecological disturbance (streamflow, wildfire, insect outbreaks) increase. Vulnerability assessment and adaptation option development for the Blue Mountains conclude the following:

Water resources and infrastructure

- Effects: Decreasing snowpack and declining summer flows will alter timing and availability of water supply, affecting municipal and public uses downstream from and in national forests, and other forest uses including livestock, wildlife, recreation, firefighting, road maintenance, and in-stream fishery flows. Declining summer low flows will affect water availability during late summer, the period of peak demand (e.g., for irrigation and power supply). Increased magnitude of peak streamflows will damage roads near perennial streams, ranging from minor erosion to complete loss of the road prism, thus affecting public safety, access for resource management, water quality, and

aquatic habitat. Bridges, campgrounds, and national forest facilities near streams and floodplains will be especially vulnerable, reducing access by the public.

- Adaptation options: Primary adaptation strategies to address changing hydrology in the Blue Mountains include restoring the function of watersheds, connecting floodplains, reducing drainage efficiency, maximizing valley storage, and reducing fire hazard. Tactics include adding wood to streams, restoring beaver populations, modifying livestock management, and reducing surface fuels and forest stand densities. Primary strategies for infrastructure include increasing the resilience of stream crossings, culverts, and bridges to higher peak flows and facilitating response to higher peak flows by reducing the road system and disconnecting roads from streams. Tactics include completing geospatial databases of infrastructure (and drainage) components, installing higher capacity culverts, and decommissioning roads or converting them to alternative uses.

Fisheries

- Effects: Decreased snowpack will shift the timing of peak flows, decrease summer low flows, and in combination with higher air temperature, increase stream temperatures, all of which will reduce the vigor of cold-water fish species. Abundance and distribution of spring Chinook salmon, redband trout/steelhead, and especially bull trout will be greatly reduced, although effects will vary by location as a function of both stream temperature and competition from non-native fish species. Increased wildfire will add sediment to streams, increase peak flows and channel scouring, and raise stream temperature by removing vegetation.
- Adaptation options: Primary strategies to address climate change threats to cold-water fish species include maintaining or restoring natural flow regimes to buffer against future changes, decreasing fragmentation of stream networks so aquatic organisms can access similar habitats, and developing wildfire use plans that address sediment inputs and road failures. Tactics include using watershed analysis to develop integrated actions for vegetation and hydrology, protecting groundwater and springs, restoring riparian areas and beaver populations to maintain summer base flows, reconnecting and increasing off-channel habitat and refugia, identifying and improving stream crossings that impede fish movement, implement engineering solutions to improve stream structure and flow, decreasing road connectivity, and revegetating burned areas to store sediment and maintain channel geomorphology.

Upland vegetation

- Effects: Increasing air temperature, through its influence on soil moisture, is expected to cause gradual changes in the abundance and distribution of tree, shrub, and grass species throughout the Blue Mountains, with more drought tolerant species becoming more competitive. Ecological disturbance, including wildfire and insect outbreaks, will be the primary facilitator of vegetation change, and future forest landscapes may be dominated by younger age classes and smaller trees. High-elevation forest types will be especially vulnerable to disturbance. Increased abundance and distribution of non-native plant species will create additional competition for regeneration of native plant species.
- Adaptation options: Most strategies for conserving native tree, shrub, and grassland systems focus on increasing resilience to drought, low snowpack, and ecological

disturbance (wildfire, insects, non-native species). These strategies generally include managing landscapes to reduce the severity and patch size of disturbances, encouraging fire to play a more natural role, and protecting refugia. Tactics include using silvicultural prescriptions (especially stand density management) and fuel treatments to reduce fuel continuity, reducing populations of non-native species, potentially modifying seed zones for tree species, and revising grazing policies and practices. Rare and disjunct species and communities (e.g., whitebark pine, aspen, alpine communities) require adaptation strategies and tactics focused on encouraging regeneration, preventing damage from disturbance, and establishing refugia.

Special habitats

- Effects: Riparian areas and wetlands will be especially vulnerable to higher air temperature, reduced snowpack, and altered hydrology. The primary effects will be decreased establishment, growth, and cover of species such as cottonwood, willow, and aspen, which may be displaced by upland forest species in some locations. However, species that propagate effectively following fire will be more resilient to climate change. Reduced groundwater discharge to groundwater-dependent ecosystems will reduce areas of saturated soil, convert perennial springs to ephemeral springs, eliminate some ephemeral springs, and alter local aquatic flora and fauna communities.
- Adaptation options: Primary strategies for increasing resilience of special habitats to changing climate include maintaining appropriate densities of native species, propagating drought tolerant native species, maintaining or restoring natural flow regimes to buffer against future changes, and reducing stresses such as conifer encroachment, livestock grazing, and ungulate browsing. Tactics include planting species with a broad range of moisture tolerance, controlling non-native species, implementing engineering solutions to maintain or restore flows, restoring beaver populations, reducing damage from livestock and native ungulates, and removing infrastructure (e.g., campsites, springhouses) where appropriate.

The BMAP facilitated one of the largest climate change adaptation efforts on federal lands to date, including participants from stakeholder organizations interested in a broad range of resource issues. It achieved specific goals of national climate change strategies for the U.S. Forest Service, providing a scientific foundation for resource management, planning, and ecological restoration in the Blue Mountains region. The large number of adaptation strategies and tactics, many of which are a component of current management practice, provide a pathway for slowing the rate of deleterious change in resource conditions. Rapid implementation of adaptation in sustainable resource management will help maintain critical structure and function of terrestrial and aquatic ecosystems in the Blue Mountains. Long-term monitoring will help detect potential climate change effects on natural resources, and evaluate the effectiveness of adaptation options that have been implemented.

Keywords: Access, adaptation, Blue Mountains, Blue Mountains Adaptation Partnership, climate change, fire, forest ecosystems, fisheries, hydrology, roads, science-management partnership, special habitats, vegetation, wildlife.

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Chapter 1: Introduction

Katherine Hoglund Wyatt[1]

The U.S. Forest Service Pacific Northwest Research Station, Pacific Northwest Region, and three national forests (Malheur, Umatilla, and Wallowa-Whitman) initiated the Blue Mountains Adaptation Partnership (BMAP) in 2013. The BMAP is a science-management collaboration with the goals of increasing climate change awareness, assessing vulnerability, and developing science-based adaptation strategies to reduce adverse effects of climate change and ease the transition to new climate states and conditions (see http://adaptationpartners.org/bmap). Developed in response to the proactive climate change strategies of the Forest Service (USDA FS 2008, 2010a,b), and building on previous efforts in national forests (Halofsky et al. 2011; Swanston et al. 2011; Littell et al. 2012; Rice et al. 2012; Swanston and Janowiak 2012; Raymond et al. 2013, 2014), the partnership brings together Forest Service scientists, University of Washington scientists, and Forest Service resource managers to plan for climate change in the Blue Mountains of northeastern Oregon and southeastern Washington.

Climate Change Response in the Forest Service

Climate change is an agency-wide priority for the Forest Service, which has issued direction to administrative units for responding to climate change (USDA FS 2008). In 2010, the Forest Service provided specific direction to the National Forest System in the form of the National Roadmap for Responding to Climate Change (USDA FS 2010a) and the Performance Scorecard for Implementing the Forest Service Climate Change Strategy (USDA FS 2010b). The goal of the Forest Service climate change strategy is to “ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources” (USDA FS 2010b). To achieve this goal, the performance scorecard contains 10 criteria grouped in four dimensions: (1) increasing organizational capacity; (2) partnerships, engagement, and education; (3) adaptation; and (4) mitigation and sustainable consumption. Progress towards accomplishing elements of the scorecard must be reported annually by each national forest and national grassland; all units are expected to accomplish 7 of 10 criteria by 2015, with at least one “yes” in each dimension. National forests in the Forest Service Pacific Northwest Region have also completed climate change action plans that indicate how they will comply with the scorecard elements by 2015.

The BMAP built on several existing efforts in ecosystem-based management and ecological restoration to address climate change and put these efforts in a broader regional context in the Blue Mountains region. There have been multiple restoration initiatives in the Blue Mountains over the last 20 years. Recently (in 2013), the Forest Service Blue Mountains Restoration Strategy Interdisciplinary Team was convened to coordinate restoration among the three Blue Mountains national forests, and this team works closely with five collaborative groups

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operating in the area. As of summer 2013, management priorities were dry forest restoration and strategic fuel treatments, both ecological priorities considered in Hessburg et al. (2005). These efforts are aimed at restoring fire-adapted forests and helping to reduce wildfire severity (USDA FS 2013). Other restoration activities are prioritized by national forests, with some coordination among forests (e.g., within river basins for aquatic restoration). The BMAP works in conjunction with these management priorities to access the best available science on climate change effects and implement climate change adaptation plans.

Science-Management Partnerships

Previous efforts in the Pacific Northwest and beyond have demonstrated the success of science-management partnerships for increasing climate change awareness among resource managers and adaptation planning on federal lands. Olympic National Forest and Tahoe National Forest initiated the first science-management partnerships for developing adaptation options for individual national forests (Littell et al. 2012). The Olympic climate change study assessed resource vulnerabilities and developed adaptation options for Olympic National Forest and Olympic National Park on the Olympic Peninsula in Washington (Halofsky et al. 2011). Similar to efforts in the Olympics, the North Cascadia Adaptation Partnership assessed vulnerabilities and formulated adaptation options for two national forests and two national parks in Washington (Raymond et al. 2014). In collaboration with three management units in California—Tahoe National Forest, Inyo National Forest, and Devils Postpile National Monument—the Forest Service Pacific Southwest Research Station held climate change education workshops and developed the Climate Project Screening Tool in order to incorporate adaptation into project planning (Morelli et al. 2012). In response to requests from the Shoshone National Forest in northern Wyoming, the Forest Service Rocky Mountain Research Station synthesized information on past climate, future climate projections, and potential effects of climate change on the multiple ecosystems within the forest (Rice et al. 2012). In the largest effort to date in the eastern United States, the Forest Service Northern Research Station, in collaboration with the Chequamegon-Nicolet National Forest of northern Wisconsin and numerous other partners, conducted a vulnerability assessment for natural resources (Swanston et al. 2011) and developed adaptation options (Swanston and Janowiak 2012). Another joint national forest and Forest Service research vulnerability assessment effort focused on the vulnerability of watersheds to climate change (Furniss et al. 2013). The watershed vulnerability assessments, conducted on 11 national forests throughout the United States, were locally focused (at a national forest scale) and included water resource values, hydrologic reaction to climate change, watershed condition and landscape sensitivity. The assessments were intended to help national forest managers identify where limited resources could be best invested to increase watershed resilience to climate change.

The processes, products, and techniques used for several studies and other climate change efforts on national forests have been compiled in a guidebook for developing adaptation options for national forests (Peterson et al. 2011). The guidebook outlines four key steps to facilitate adaption in national forests: (1) become aware of basic climate change science and integrate that understanding with knowledge of local conditions and issues (review), (2) evaluate sensitivity of natural resources to climate change (rank), (3) develop and implement options for adapting resources to climate change (resolve), and (4) monitor the effectiveness of on-the-ground management (observe) and adjust as needed. The BMAP is focused on implementation of the principles and practices in the guidebook.

The Blue Mountains Adaptation Partnership Process

The BMAP is a science-management partnership focused on vulnerability assessment and adaptation planning for the Malheur, Umatilla, and Wallowa-Whitman National Forests, encompassing an area of 2.14 million hectares in Oregon and Washington (fig. 1.1). The BMAP process includes: (1) a vulnerability assessment of the effects of climate change on natural resources and infrastructure, (2) development of adaptation options that will help reduce negative effects of climate change and assist the transition of biological systems and management to a warmer and a changing climate, and (3) development of an enduring partnership to facilitate ongoing dialogue and activities related to climate change in the Blue Mountains region.

We assessed the vulnerability of natural resources and infrastructure and developed options for adapting resources and management to a changing climate. Based on their importance in the region and current management concerns and challenges, the BMAP focused on water resources, fisheries, and vegetation (upland; riparian, wetland, and groundwater dependent systems). These resources are similar to the resources that were the focus of the Olympic climate change case study (Halofsky et al. 2011) and North Cascadia Adaptation Partnership (Raymond et al. 2014), but reflect different regional priorities.

Vulnerability assessments typically involve exposure, sensitivity, and adaptive capacity (Parry et al. 2007), where exposure is the degree to which the system is exposed to changes in climate, sensitivity is an inherent quality of the system that indicates the degree to which it could be affected by climate change, and adaptive capacity is the ability of a system to respond and adjust to the exogenous influence of climate. Vulnerability assessments can be both qualitative and quantitative and focus on whole systems or individual species or resources (Glick et al. 2011). Several tools and databases are available for systematically assessing sensitivity (e.g., Lawler and Case 2010, Luce et al. 2014) and vulnerability of species (e.g., Potter and Crane 2010).

For the BMAP, we used scientific literature and expert knowledge to assess exposure, sensitivity and adaptive capacity to identify key vulnerabilities for water use and infrastructure, fisheries, and vegetation. The assessment process took place over approximately four months, and involved monthly to biweekly phone meetings for each of the three resource-specific assessment teams. Each assessment team refined key questions that the assessment needed to address, selected values to assess, and determined which climate change impact models best informed the assessment. In some cases, assessment teams conducted spatial analyses and/or ran and interpreted models, selected criteria in which to evaluate model outputs, and developed maps of model output and resource sensitivities. To the greatest extent possible, teams focused on effects and projections specific to the BMAP region and used the finest scale projections that are scientifically valid (Littell et al. 2011).

By working collaboratively with scientists and resource managers and focusing on a specific region, the goal of BMAP was to go beyond general concepts to identify adaptation options that can be implemented into projects and plans (Peterson et al. 2011; Raymond et al. 2013, 2014; Swanston and Janowiak 2012). After identifying key vulnerabilities for each resource sector, a workshop was convened in La Grande, Oregon in April 2014 to present and discuss the vulnerability assessment, and to elicit potential adaptation options from resource managers. For each resource sector, participants identified strategies (general approaches) and tactics (on-the-ground actions) for adapting resources and management practices to climate change. Participants also identified opportunities and barriers for implementing these strategies and tactics into current projects, management plans, partnerships, regulations, and policies.

Participants generally focused on adaptation options that can be implemented given our current scientific understanding of climate change effects, but they also identified research and monitoring that would benefit future efforts to assess vulnerability and adapt management practices. Facilitators captured information generated during the workshops with a set of spreadsheets adapted from Swanston and Janowiak (2012). Initial results from the workshops were augmented with continued dialogue with Forest Service resource specialists.

This publication contains a chapter on expected climatological and hydrological changes in the Blue Mountains, and one chapter for each of the resource sectors covered in the vulnerability assessment (water resources, fisheries, and upland vegetation, riparian wetland, and groundwater dependent systems). Each of the latter chapters includes a review of climate change effects, sensitivities, and current management practices (collectively the vulnerability assessment) and results of the adaptation planning discussions. Resource managers and other decision makers can use this publication in several ways. First, the synthesis of projected changes in climate and hydrology, and potential effects on water resources, fisheries, and vegetation is a state-of-science reference for addressing climate change in planning documents and projects. The publication is not a comprehensive synthesis of all literature on climate change effects in the region, but it emphasizes the biggest challenges for these resource sectors that are known at this time. Second, land managers can draw from the adaptation options presented in this report as they begin to implement actions in response to changes in climate and hydrology. We expect that over time, and as needs and funding align, that appropriate adaptation options will be incorporated into plans and programs of national forests and possibly other agencies.

Adaptation planning is an ongoing and iterative process. Implementation may occur at critical times in the planning process, such as when managers revise Forest Service land management plans and other planning documents, or after the occurrence of extreme events and ecological disturbances (e.g., wildfire). We focus on adaptation options for the Forest Service, but this publication provides information that can be used by other land management agencies as well. Furthermore, the BMAP process can be emulated by national forests, national parks, and other organizations in the Pacific Northwest and beyond, thus propagating climate-smart management across ever larger landscapes.

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Chapter 2: Ecological, Biogeographical, and Historical Context of the Blue Mountains

Katherine Hoglund Wyatt[2]

The Blue Mountains Adaptation Partnership (BMAP) includes the Wallowa-Whitman, Umatilla, and Malheur National Forests, which occupy 2.14 million hectares in the Blue Mountains and their subranges (referred to hereafter as the Blue Mountains) of primarily northeastern Oregon and a small portion of southeastern Washington (fig. 2.1). The area is climatologically and ecologically diverse, and although each national forest has a specific cultural and political context, similarities in geological, cultural, and ecological histories unite the Blue Mountains region. For example, following Euro-American settlement, the Blue Mountains experienced extensive sheep and cattle grazing, and later a strong timber economy (Oliver et al. 1994, Wissmar et al. 1994). Periodic and extensive outbreaks of mountain pine beetle, (*Dendroctonus ponderosae* Hopkins), western spruce budworm (*Choristoneura freemani* Razowski), and Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough) occurred throughout the region in the 1900s (Rainville et al. 2008). As a result of this historical and biological legacy, management across the three national forests has sought to respond to ecological disturbances and restore fire-prone ecosystems. Establishment of forest reserves for watershed protection in the early 1900s, development of ranching and agriculture industry through the 20th century, and growing concerns for water quality and fisheries in the later 20th century have refocused attention on restoring watershed and aquatic ecosystems. Congressional designation of seven wilderness areas within the three national forests (fig. 2.1) has further coalesced management objectives. In addition, the revised land management plan for the Blue Mountains National Forests, currently in draft form, unites the three national forests in a joint planning framework (USDA FS 2014). This common historical and ecological context formed the basis of joint discussion on vulnerability to climate change and enabled the BMAP to identify common adaptation strategies that are relevant to the region as a whole.

Ecological Setting

The complex geological history of the Blue Mountains is the foundation for the ecological diversity of the area. Oceanic subduction under the North American plate during the late Triassic and late Jurassic, followed by terrestrial sedimentation and volcanic material deposition, formed the basis of the Blue Mountains (Brooks 1979, White et al. 1992, Wilson and Cox 1980). The region was glaciated 20,000-14,000 BP (Johnson et al. 1994). Glacial deposition and volcanic ash from Glacier Peak 12,000 years ago and Mt. Mazama 6,900 years ago was generally redistributed on north-facing slopes and broad basins (Alt and Hyndman 1995); this variation in deposition continues to contribute to broad differences in soil productivity (Jaindl et al. 1996, Johnson et al. 1994, Simpson 2007) and vegetation composition (Kelly et al. 2005). Mudstone,

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sandstone, siltstone, and shale are common throughout the western Blue Mountains, whereas schist, slate, argillite, tuff, chert, and intrusive granite are more common in the eastern Blue Mountains; limestone and Columbia River Basalt are prevalent throughout the region (Jaindl et al. 1996, Orr and Orr 1999).

The Blue Mountains are a collection of small mountain ranges, with the highest elevations in the Eagle Caps, and the smaller Elkhorn, Greenhorn, Strawberry, Wenaha, and Aldrich mountain ranges. Elevation in the region ranges from 267 to 3,000 m with high points throughout the Wallowa-Whitman (Sacajawea Peak, 3,001 m), Malheur (Strawberry Mountain, 2,756 m), and Umatilla (Vinegar Hill Northeast, 2,147 m) National Forests. Climatic differences, created in part by complex topography, further contribute to diversity in the Blue Mountains. The southern portion of the Blue Mountains, including the Strawberry subrange, is in the rain shadow of the Cascade Range and is most prominently affected by Great Basin climatic patterns. The result is warmer and drier conditions; winter minimum temperatures range from -5 to 5°C, summer high temperatures from 5.5 to 18 °C, and precipitation from 20 to 100 cm annually (PRISM Climate Group). In the northern Blue Mountains, maritime air flows through the Columbia River Gorge, resulting in higher precipitation (40-200 cm annually) and less seasonally varied temperatures (Caraher et al. 1992, Heyerdahl et al. 2001, Mock 1996, Simpson 2007, Wells 2006).

Elevation gradients and ecological disturbance further define the ecological associations of the Blue Mountains. Historically, a low severity fire regime at low elevations promoted ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson) and dry mixed conifer forest dominated by ponderosa pine; these communities historically composed 20-50 percent of the overall landscape and 40-75 percent of the forested landscape (Rainville et al. 2008). In the central and southern Blue Mountains, woodlands composed of western juniper (*Juniperus occidentalis* Hook.), Idaho fescue (*Festuca idahoensis* Elmer), and bitterbrush (*Purshia tridentata* [Pursh] DC.), as well as shrublands composed of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* [Rydb.] Beetle), scabland sagebrush (*Artemisia rigida* [Nutt.] A. Gray), and curl-leaf mountain-mahogany *(Cercocarpus ledifolius* Nutt.) are also common at low elevation (Jaindl et al. 1996, Johnson et al. 1994). At mid elevation, a mixed-severity fire regime, in conjunction with moderate environmental conditions, supports lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. ex S. Watson), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and western larch (*Larix occidentalis* Nutt.). Although it only occurs as a scattered component in other stands, these mid-elevation stands often include western white pine (*Pinus monticola* Douglas ex D. Don). At high elevations, subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) and whitebark pine (*Pinus albicaulis* Engelm.) transition to alpine meadows of sedge and fescue (Jaindl et al. 1996, Johnson et al. 1994). In the northern Blue Mountains, the woodland zone supports tall shrublands (with, for example, western serviceberry (*Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roem.), black hawthorn (*Crataegus douglasii* Lindl.), and western chokecherry (*Prunus virginiana* var. *demissa* (Nutt.) Torr.)) rather than juniper and sagebrush.

Decades of fire exclusion, livestock grazing, and timber harvest have heavily altered historical vegetation, in turn increasing the likelihood of severe fire and insect outbreaks (Hessburg et al. 2005, Hessburg and Agee 2003, Langston 1995, Lehmkuhl et al. 1994, Mutch et al. 1993, Rainville et al. 2008); many of the most profound changes have been in lowland ponderosa pine stands, which have increased in density and true fir composition (Harrod et al. 1999, Hessburg et al. 2005). Beyond fire disturbance, severe wind events, insect outbreaks, drought, flooding, and landslides interact to affect vegetation. These disturbance agents, in conjunction with variation in soils and microclimate, create mosaics and heterogeneity throughout all dominant vegetation associations (Jaindl et al. 1996, Johnson 1994, Johnson et al. 1994).

The diversity of vegetation in the Blue Mountains supports abundant and diverse fauna. Mule deer (*Odocoileus hemionus* Rafinesque), pronghorn (*Antilocarpa Americana* Ord), whitetail deer (*Odocoileus virginianus* Zimmermann), Rocky Mountain elk (*Cervus elaphus* L.), bighorn sheep (*Orvis canadensis* Shaw), Rocky Mountain goat (*Oreamnos americanus* Blainville), black bear (*Ursus americanus* Pallas), wolverine (*Gulo gulo* L.), badger (*Taxidea taxus* Schreber), mountain lion (*Felis concolor* L.), coyote (*Canis latrans* Say), pine martin (*Martes martes* L.), mink *(Neovison vison* Schreber), and American beaver (*Castor canadensis* Kuhl) are all found within the region (Jaindl et al. 1996, Tiedemann et al. 1998). Avian diversity is even greater (263 species total) and includes ruffed grouse (*Bonasa umbellus* L.), blue grouse (*Dendragapus obscurus* Say), pileated woodpecker (*Dryocopus pileatus* L.), sharp-shinned hawk (*Accipiter striatus* Vieillot), and American bald eagle (*Haliaeetus leucocephalus* L.) (Jaindl et al. 1996, Tiedemann et al. 1998).

The Blue Mountains encompass over 16,000 km of perennial streams, almost 2000 lakes and ponds, and thousands of springs. Twenty-two fish species are found in the area, including Chinook salmon (*Oncorhynchus tshawytscha* Walbaum in Artedi), steelhead (*O. mykiss* Walbaum), interior redband trout (*O. m. gibsii* Walbaum), bull trout (*Salvelinus confluentus* Suckley) and Dolly Varden (*S. malma* Walbaum in Artedi) (Jaindl et al. 1996).

Cultural History of the Blue Mountains

Native Americans, including the Nez Perce, Cayuse, Walla Walla, Shoshone, Bannocks, Wasco, Burns Paiute, Umatilla, and Warm Springs tribes, have long inhabited the Blue Mountains region (Robbins and Wolf 1994, Heyerdahl et al. 2001). Currently, the Confederated Tribes of the Umatilla Indian Reservation (Cayuse, Walla Walla, and Umatilla tribes), Confederated Tribe of Warm Springs (Wasco, Paiute, and Warm Springs), and the Burns Paiute Reservations are in the vicinity of the Blue Mountains. Many of these tribes continue to have relationships with the national forests in the Blue Mountains.

Native Americans used the Blue Mountains for hunting and gathering. The Eagle Cap Wilderness was the summer home of the Joseph Band of the Nez Perce, who hunted bighorn sheep and deer throughout the area (USDA FS and NPS 1982). Hunting, as well as berry and root collection, likely occurred throughout the Blue Mountains region (Robbins and Wolf 1994, Heyerdahl et al. 2001, Richards and Alexander 2006). Fire was used to promote desired plant species as well as improve rangeland and hunting grounds (Johnson 1994, Robbins and Wolf 1994, Heyerdahl et al. 2001).

Lewis and Clark came through the Blue Mountains between 1804 and 1806, and were followed in the subsequent three decades by trappers, missionaries, naturalists, and government scientists. Oregon Trail emigrants settled in the area beginning in 1843, spurring conflict and war with the Cayuse Indians (Oliver et al. 1994). From 1850 to 1890, the Euro-American population grew from 13,000 to 357,000 (Robbins and Wolf 1994). The discovery of gold in the

John Day area in 1862, the Homestead Act of 1862, and 1880s railroad construction further increased settlement and settler-Indian conflict (Robbins and Wolf 1994, Wissmar et al. 1994).

Following initial settlement, sheep and then cattle grazing shaped the Blue Mountains landscape. As the last two states to implement grazing regulations, Oregon and Washington faced overgrazing through the late 1800s. Sheep grazing began in 1860, increased steadily from 1890 to 1920, and then gradually declined; at its peak there were 225,000 sheep and lambs in the Umatilla National Forest alone (Oliver et al. 1994). Cattle grazing was most prolific from 1940 to 1980, reaching almost 350,000 cattle and calves in the Blue Mountains region (Irwin et al. 1994). Although increased rangeland regulation and management through the 1970s reduced overgrazing, the effects of a century of grazing persists, especially on vegetation, fuels, and riparian areas (Camp 1999, Irwin et al. 1994, Oliver et al. 1994).

Following the opening of the first mill in John Day, Oregon in 1862, the cultural and ecological importance of logging steadily increased. Starting in 1908, schools and public roads received 25 percent of timber receipts, increasing incentives for logging. Following World War II, increased vehicle capacity and improved chainsaws spurred increasing logging activity. The maximum annual volume was 92,000 m^3 in the 1940s and 210,000 m^3 in the 1950s, increasing to 520,000 m^3 in 1973 (Oliver et al. 1994, Wissmar et al. 1994). Across Baker, Grant, Harney, Union, Wallowa, and Wheeler Counties, production peaked between 1985 and 1992, with an annual output of over 1.4 million m^3 (Rainville et al. 2008).

Starting in the 1970's restoration efforts began to repair damage to watersheds and fisheries, and restoration efforts expanded in the 1990s with state and federal listing of fish species. Recent logging efforts have focused on forest restoration rather than large-scale timber sales. Annual output has been below 240,000 m^3 since 1997 (for Baker, Grant, Harney, Union, Wallowa, and Wheeler Counties combined). From 1988 to 1995, 60 percent of the timber area consisted of clearcuts and seed-tree harvests, decreasing to less than 30 percent after 1997 and less than 10 percent after 2000. By 2001, over 50 percent of the timber area consisted of commercial thinning projects. Of land administered by the Forest Service in the Blue Mountains, 29 percent is open to active forestry (Rainville et al. 2008). Other restoration efforts began in the 1970s to repair damage to watersheds and fisheries, expanded in the 1990s with state and federal listing of water quality and fish species

Geography, History, and Management

National Forests and Wilderness Areas

Wallowa-Whitman National Forest covers 968,214 ha in the northeast corner of Oregon and a sliver of Idaho, and includes four wilderness areas totaling 237,024 ha: the Eagle Cap, Hells Canyon, North Fork John Day, and Monument Rock (fig. 2.1). The North Fork John Day Wilderness is composed of four separate units, one of which is on the Wallowa-Whitman National Forest and three of which are on the Umatilla National Forest. Similarly, Monument Rock Wilderness is shared with the Malheur National Forest. Protected for scenery, recreation, fisheries, wildlife, and historical cultural resources, 10 rivers within the Wallowa-Whitman National Forest are designated in the Wild and Scenic Act of 1988 (fig. 2.1).

Umatilla National Forest, located to the west of the Wallowa-Whitman National Forest, covers 566,560 ha in northeastern Oregon and southeastern Washington (approximately 78% is in Oregon and 22% is in Washington), and it includes three wilderness areas totaling 128,858 ha: the Wenaha-Tucannon, North Fork Umatilla, and North Fork John Day (three of four units; fig. 2.1). Wild and scenic rivers cover 93 km, protecting anadromous fish runs, especially steelhead trout and Chinook salmon, and migratory bull trout.

Malheur National Forest covers 607,028 ha and includes four specially designated areas. Two wilderness areas, Monument Rock (shared with the Wallowa-Whitman National Forest) and Strawberry Mountain cover 35,742 ha (fig. 2.1). Vinegar Hill Indian Rock Scenic Area contains the highest point in the forest, and the Cedar Grove Botanical area has the only stand of Alaska cedar (*Callitropsis nootkatensis* [D. Don] D.P. Little) east of the Cascade Range. The Malheur River and North Fork Malheur River are protected as wild and scenic for their scenic value, fisheries, geology, and wildlife.

History and Management of the Blue Mountains

The Wallowa-Whitman, Umatilla, and Malheur National Forests are separate administrative units but share a common management history. The Blue Mountains Forest Reserve was informally removed from the public domain in 1902 and formally established in 1906. In 1908, all three national forests were designated to protect water, timber resources and rangeland (USDA FS 1997). The Wilderness Act of 1964, National Environmental Policy Act of 1969, Endangered Species Act of 1973, National Forest Management Act of 1976, and Oregon Omnibus Wild and Scenic Rivers Act of 1988 guide the three units. The Blue Mountains draft revised land management plan (USDA FS 2014) will provide overarching guidance for all resource management activities, succeeding existing land management plans for the national forests.

A shared history of periodic and extensive insect outbreaks further unites the three administrative units. To various degrees, insect outbreaks have been attributed to fire exclusion and subsequent high stand density and conversion from pine to fir (Rainville et al. 2008, Wickman 1992). Mountain pine beetle outbreaks occurred in 1905, 1932, and throughout the 1940s, 1950s, and 1960s, affecting 263,045 ha from 1955 to 1966 (Burke and Wickman 1990, Oliver et al. 1994). Western pine beetle (*Dendroctonus brevicomis* LeConte) outbreaks from 1953 to 1980 affected 4,000-35,000 ha annually (Rainville et al. 2008). From 1944 to 1958, western spruce budworm contributed to tree mortality on 364,217 ha in the Umatilla National Forest alone. Additional budworm outbreaks occurred from 1980 to 1992 (Oliver et al. 1994). Douglas-fir tussock moth outbreaks (1946-1948, 1963-1965, 1971-1975, 1991-1995) were the most prolific and damaging, with the 1970s outbreak affecting 255,000 ha (Mason et al. 1998, Rainville et al. 2008). Restoration efforts continue to address forest conditions affected by historical and potential future insect outbreaks.

The Blue Mountains are currently managed for a wide range of ecosystem services, including timber, water, livestock grazing, and recreation. Whereas timber, grazing, and mining have been historically prominent, recreation and tourism are increasingly important in terms of number of users and economic value. The Wallowa-Whitman National Forest has 4,667 km of trails and five scenic byways, and 20,000 hunters annually visit the Umatilla National Forest. In counties where employment was historically dominated by timber jobs, diverse management objectives including small-scale timber work have alleviated high unemployment experienced in the 1990s (Jaindl et al. 1996, Rainville et al. 2008).

Current restoration efforts in the Blue Mountains focus on improving the vigor of low-elevation dry forests, reducing fire hazard, restoring functional fish passages, improving habitat for several animal species, and improving riparian and stream conditions. The national Watershed Condition Framework (Potyondy and Geier 2011) and regional aquatic restoration strategies guide forest watershed and aquatic restoration programs, which include controlling invasives and restoring native plant communities. Addressing a long legacy of fire exclusion and timber practices that have created densely-stocked stands, mostly at lower elevations, forest restoration seeks to improve forest vigor and strategically reduce fuel loads. These efforts also seek to limit insect outbreaks, reduce wildfire severity, and encourage prescribed fire use (Rainville et al. 2008, USDA FS 2013). Whitebark pine is being protected in the face of multiple stressors, and quaking aspen (*Populus tremuloides* Michx.) regeneration is being augmented where possible. Stewardship contracting, which allows timber receipts to stay within the forest to fund unprofitable restoration efforts, has been widely used in the area (Rainville et al. 2008).

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Chapter 3: Climate Change and Hydrology in the Blue Mountains

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Current and Historical Climate in the Blue Mountains

The dominant influences on climatic patterns in the Pacific Northwest are the Pacific Ocean and the Cascade Mountain Range. The diurnal temperature range is higher east of the Cascade crest, further inland from the Pacific Ocean. More precipitation falls west of the Cascade Mountains crest, and a strong rain shadow greatly reduces precipitation east of the crest. The southern portion of the Blue Mountains, including the Strawberry subrange, is in the rain shadow of the Cascade Mountains and is predominantly influenced by Great Basin climatic patterns, resulting in warmer and drier conditions. In the northern Blue Mountains, maritime air flows through the Columbia River Gorge, resulting in higher precipitation and more moderate temperature variations (Western Regional Climate Center 2015).

It is important to establish a baseline of historical climate in the Blue Mountains before considering future change. The Blue Mountains area aligns closely with the National Climatic Data Center's (NCDC) Northeast Oregon climate division (Oregon Climate Division 8), which is the area for which we consider current and historical climate. It should be noted that the NCDC information consists of low-elevation climate data, and high-elevation climate patterns may differ from those at low elevations (Luce et al. 2013). The topography of the Blue Mountains results in orographically enhanced local precipitation totals despite being in the lee of the Cascade Range. The regional annual average precipitation is 44 cm (20th century average), with greater amounts in higher elevation areas in the region. The surrounding Columbia River Plateau and High Desert see less precipitation on an annual basis. The temperatures in the Blue

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Mountains are slightly cooler than those of the entire region; regionally averaged mean annual temperature is about 7.5 ˚C (1901-2000 average), with colder temperatures at higher elevations.

Human influence on the climate is clear (IPCC 2013), and changes in the climate are already being realized across the Pacific Northwest, where temperatures have warmed by a statistically significant amount. Mean annual temperature in Northeast Oregon increased by 0.06 ˚C per decade between 1895 and 2013, consistent with the overall temperature trend of the entire Pacific Northwest (NOAA National Centers for Environmental Information 2015) (fig. 3.1). Only three years have been below the 20th century annual average temperature of 7.5 ˚C since 1990 (fig. 3.1). Precipitation in the Pacific Northwest is still dominated by interannual variability, such as the El Niño Southern Oscillation (ENSO) (Mote et al. 2013). However, the Blue Mountains region does not exhibit a clear precipitation signal in terms of ENSO phase in the winter months (NOAA Climate Prediction Center 2015). Although there is no detectable or significant precipitation trend in the region, the last 30 years were generally drier than the 20th century average, but with a few very wet years in the mid-late 1990s; the preceding decades (1940-1980) were much wetter than recent years (fig 3.2).

Future Climate Projections for the Pacific Northwest

Complex global climate models (GCMs) begin to answer questions about future climate. Climate modeling is mostly conducted at global to regional scales because of the computational power required to run GCMs. The disparity between the scale of GCM output and information needs for regional to sub-regional climate change planning presents some challenges. However, we consider the projections for the Pacific Northwest region relevant for planning in the Blue Mountains; variations in monthly and annual temperature are highly correlated across the Pacific Northwest region.

A number of modeling groups around the world have developed and run GCM simulations, which project future global climate under different future scenarios. The Coupled Model Intercomparison Project (CMIP) is a coordinated experiment involving many of these modeling groups worldwide, offering many simulations for scientists to assess the range of future climate projections for the globe. The latest CMIP experiment is the fifth phase of the project, referred to as CMIP5. Simulations of future climate are driven by Representative Concentration Pathways (RCPs), a departure from the last CMIP experiment, somewhat confusingly titled CMIP3. CMIP3 relied on the Special Report on Emissions Scenarios to drive model projections of future climate (Nakićenović and Swart 2000). The RCPs do not define greenhouse gas emissions, but instead define future concentrations of greenhouse gases, aerosols, and chemically active gases. RCPs encompass the range of current estimates regarding the evolution of radiative forcing, or the assumed rate of extra energy entering the climate system throughout the 21st century and beyond (van Vuuren et al. 2011). Although the models are run at the global scale, the following projections are for the Pacific Northwest (generally Oregon, Washington, Idaho, and western Montana) and are based on the analysis described in Mote et al. (2013). More information on CMIP can be found at http://cmip-pcmdi.llnl.gov/index.html.

For the Blue Mountains vulnerability assessment, following Mote et al. (2013), we considered two of the CMIP5 scenarios: RCP 4.5 (significant reduction in global greenhouse gases and climate stabilization by year 2100) and RCP 8.5 (increasing greenhouse gases to the end of the 21st century). For the Pacific Northwest, every GCM shows an increase in temperatures in the future, with differences depending on global greenhouse gas emissions (fig. 3.3). There is no plausible future climate scenario from any GCM in which the Pacific Northwest cools in future decades. For the 2041-70 period, models project warming of 1.1 °C to 4.7 °C compared to 1970-99, with the lower end possible only if global greenhouse gas emissions are significantly reduced (RCP 4.5 scenario). Through about 2040, both RCP 4.5 and 8.5 show a similar amount of warming; regional temperatures beyond 2040 depend on future global greenhouse gas emissions. Similar to annual temperature, all models are in agreement that each season will be warmer in the future, with the largest amount of warming occurring in the summer (table 3.1). In each season, RCP 8.5 projects warmer temperatures than RCP 4.5 (Dalton et al. 2013).

Projections for future annual precipitation do not display as clear of a signal as those for temperature; annual precipitation projections range from wetter to drier, and projections for future annual precipitation indicated small trends compared to natural year-to-year variability. Averaging all the model outputs for annual precipitation, the projected future precipitation is close to no change from historical, with a wide range of projections. There is some indication and greater model agreement that summers will be drier in the future, although summers in the Pacific Northwest are already quite dry (table 3.2) (Mote et al. 2013).

Hydrologic Processes in the Blue Mountains

Climate change will likely affect physical hydrological processes and resource values influenced by hydrological processes, including water use, infrastructure, and fish. Specifically, climate change will affect the amount, timing, and type of precipitation, and timing and rate of snowmelt (Luce et al. 2012, 2013; Safeeq et al. 2013), which will affect snowpack volumes (Hamlet et al. 2005), streamflows (Hidalgo et al. 2009, Mantua et al. 2010), and stream temperatures (Isaak et al. 2012, Luce et al. 2014b). Changes in the amount and timing of precipitation will also affect vegetation (chapters 6 and 7), which will further alter water supplies (Adams et al. 2011). Though climate change effects on vegetation will likely be important, they are not considered in the hydrological projections in this chapter. Here we describe hydrologic processes and regimes in the Blue Mountains, historical trends in hydrologic parameters (snowpack, peak streamflow, low streamflow, and stream temperatures), and projected effects of climate change on those hydrologic parameters (box 3.1).

Some of the streamflow simulations shown in this report were generated by the Variable Infiltration Capacity (VIC) model (Liang et al. 1994) using GCMs from the Intergovernmental Panel on Climate Change AR4 assessment to project future climates (Elsner et al., 2009). The VIC projections were prepared from an ensemble of GCM models that had the best match with observations in the historical period (see Littell et al. [2011] for details). Projections for the "2040s" cover an average from 2030 to 2059, and the "2080s" cover 2070 to 2099. Historical metrics were based on the period 1977-1997 (Wenger et al., 2010). The VIC data were computed on a 1/16th-degree (approximately 6 km) grid to produce daily flow data that were further analyzed for metrics important to aquatic ecology (Wenger et al. 2010, 2011b).

Snowpack

Effects of climate change on snowpack in watersheds of the Pacific Northwest can be broadly distinguished by mid-winter temperatures in each basin (Hamlet and Lettenmaier 2007). Rain-dominated basins are above freezing most of the time in winter, and snow accumulation is minimal (<10 percent of October through March precipitation). At a relatively coarse time scale, rain-dominated basins typically have one broad peak in streamflows in the winter that coincides with the regional winter peak in precipitation. However, at a finer time scale, rain-dominated basins may display multiple peaks in streamflow that coincide with individual storms or rain events. Mixed rain and snow (also called "transient" or "transitional") basins can collect substantial snowpack in winter (10 to 40 percent of October through March precipitation), but are typically only a few degrees below freezing on average in mid-winter. Mixed rain-and-snow basins typically have multiple seasonal streamflow peaks, with one primary peak in late autumn caused by rain, and another in late spring caused by snowmelt snowmelt. Snowmelt-dominated basins are relatively cold in winter and capture a larger percentage (>40 percent) of their October through March precipitation as snow. Snowmelt-dominated basins typically have relatively low flows through winter and a period of streamflow peaks in spring that coincides with seasonal snowmelt.

Increasing temperatures in the Pacific Northwest over the last 50 years have led to more precipitation falling as rain rather than snow, earlier snowmelt (Hamlet et al. 2007, Stewart et al. 2005), and reduced spring snowpack (Barnett et al. 2008, Hamlet et al. 2005, Mote 2003, Mote et al. 2005). Snowpack in the Pacific Northwest is expected to be sensitive to future temperature increases with changing climate. In response to warming, shifts from snowmelt-dominant to mixed rain-and-snow basins, and from mixed rain-and-snow to rain-dominant basins are projected by the 2040s in the Pacific Northwest (Tohver et al. 2014).

Kramer and Snook (unpublished data) developed a snowpack sensitivity map for the Pacific Northwest using data from the Snow Data Assimilation System (SNODAS) (NOHRSC 2004). SNODAS snow water equivalent (SWE) data from 2003-2012 were used to characterize the sensitivity of snowpack to climate variability (table 3.3). Luce et al. (2014a) also evaluated snow sensitivity to climate at Snowpack Telemetry (SNOTEL) sites in the Pacific Northwest, using a spatial analog method to make April 1 SWE projections under a future climate warming scenario of 3°C warmer than the last 20 years (expected by around 2050 for the RCP 8.5 scenario [fig. 3.3]). An analysis of snow cover data from strongly contrasting years gives some insight about potential sensitivity of late season snowpack to a changing climate (Kramer and Snook unpublished data) (table 3.3). Results of both studies suggest that there will likely be future declines in snowpack persistence and April 1 SWE throughout the Pacific Northwest, with the largest declines in mid-elevation and wetter locations.

In the Blue Mountains, large areas could lose all or significant portions of April 1 SWE under a 3°C temperature increase (expected by around 2050 for the RCP 8.5 scenario [fig. 3.3]) (fig. 3.4). Results indicate that snowpack sensitivity is relatively high in the Strawberry Mountains, Monument Rock Wilderness, Wenaha-Tucannon Wilderness, and at mid-elevations in the North Fork John Day, Eagle Cap Wilderness, and Hells Canyon Wilderness (fig. 3.4). Snowpack sensitivity is lower at high elevations in the Wallowa Mountains (Eagle Cap Wilderness), Greenhorn Mountains (North Fork John Day Wilderness), and Hells Canyon Wilderness Area. However, snowpack loss may still be significant (40-100 percent loss) in some of these areas (Luce et al. 2014a).

Similarly, the VIC model was used to project up to 100 percent loss of April 1 SWE in parts of the Blue Mountains by the 2080's (Hamlet et al. 2013). This study also projected that most of the watersheds in the Blue Mountains that were historically classified as mixed rain and snow will become rain dominant by the 2080's. These watersheds will likely receive more rain and less snow in the winter months.

Peak flows

Flooding regimes in the Pacific Northwest are sensitive to precipitation intensity, temperature effects on freezing elevation (which determines whether precipitation falls as rain or snow), and the effects of temperature and precipitation change on seasonal snow dynamics (Hamlet and Lettenmaier 2007, Tohver et al. 2014). Floods in the Pacific Northwest typically occur during the autumn and winter because of heavy rainfall (sometimes combined with melting snow) or in spring because of unusually heavy snowpack and rapid snowmelt (Hamlet and Lettenmaier 2007, Sumioka et al. 1998). Summer thunderstorms can also cause local flooding and mass wasting, particularly after wildfire (e.g., Cannon et al. 2010, Istanbulluoglu et al. 2004, Luce et al. 2012, Moody and Martin 2009).

Flooding can be exacerbated by rain-on-snow (ROS) events, which are contingent on the wind speed, air temperature, absolute humidity, intensity of precipitation, elevation of the freezing line, and existing snowpack when storms happen (Eiriksson et al. 2013, Harr 1986, Marks et al. 1998, McCabe et al. 2007). Warming affects future flood risk from ROS events differently depending on the importance of these events as a driver of flooding in different basins under the current climate. As temperatures warm, the ROS zone, an elevation band below which there is rarely snow and above which there is rarely rain, will likely shift upwards in elevation. This upward shift in the ROS zone will tend to strongly increase flooding in basins where the current ROS zone is low in the basin (with a large snow collection area above). In contrast, in basins in which the ROS zone is higher in the basin, the upward shift in the ROS zone may only modestly increase the fractional contributing basin area with ROS or potentially shrink the relative contribution of ROS.

In the latter half of the 20th century, increased temperatures led to earlier runoff timing in snowmelt-dominated and mixed rain-and-snow watersheds across the western United States (Cayan et al. 2001, Hamlet et al. 2007, Stewart et al. 2005). With future increases in temperature and potentially in amount of precipitation in the winter months, extreme hydrologic events (e.g., those currently rated as having 100-year recurrence intervals) may become more frequent (Hamlet et al. 2013).

An analysis for the Blue Mountains, using VIC model output from Wenger et al. (2010), projects that flood magnitude will increase in the Wallowa Mountains, Hells Canyon Wilderness Area, and northeastern portion of the Wallowa-Whitman National Forest by the 2080s, particularly in mid-elevation areas most vulnerable to ROS (fig. 3.5a). The frequency of midwinter flood events does not change over much of the area (fig. 3.5b,c), but the areas showing the greatest change in flood magnitude (fig. 3.5a) are also showing substantial changes in the frequency of the largest flows in each winter (fig. 3.5b,c), a measure of flood seasonality that is important to a number of fall-spawning fish species (Goode et al. 2013; Tonina et al. 2008; Wenger et al. 2011a,b).

Low flows

As a result of earlier snowmelt and peak streamflows over the last 50 years in the western United States, spring, early summer, and late summer flows have been decreasing, and fractions of annual flow occurring earlier in the water year have been increasing (Kormos et al. in review, Leppi et al. 2011, Luce and Holden 2009, Safeeq et al. 2013, Stewart et al. 2005). An analysis by Stewart et al. (2005) in eastern Oregon showed some of the largest trends toward decreasing fractional flows from March through June. In addition to decreased summer flows, Luce and Holden (2009) showed declines in some annual streamflow quantiles in the Pacific Northwest between 1948 and 2006; they found decreases in the 25th percentile flow (drought year flows) over the study period, meaning that the driest 25 percent of years have become drier across the Pacific Northwest.

Summer low flows are influenced not only by the timing of snowmelt, but also by landscape drainage efficiency, or the inherent geologically-mediated efficiency of landscapes in converting recharge (precipitation) into discharge (Safeeq et al. 2013, Tague and Grant 2009). The Blue Mountains, which have moderate groundwater contributions, experienced reduced summer flows of 21 to 28 percent between 1949 and 2010 (Safeeq et al. 2013). Safeeq et al. (2014) developed and applied an analytical framework for characterizing summer streamflow sensitivity to a change in the magnitude (mm mm^{-1}) and timing (mm day^{-1}) of recharge at broad spatial scales (assuming an initial recharge volume or 1 mm). This approach facilitates assessments of relative sensitivities in different locations in a watershed or among watersheds. Sensitivity, in this approach, has a very specific meaning: how much does summer streamflow (at some defined point during the summer, i.e., July 1 or August 1), change in response to a change in either the amount of water that recharges the aquifer during late winter and early spring, and the timing of that recharge. So magnitude sensitivity relates how much summer discharge will change (in mm) to a 1 mm change in amount of recharge, and timing sensitivity means how much will summer discharge (in mm) to a 1 day change in the timing of recharge. Both metrics make the simplifying assumption that all recharge happens on a particular day, which of course is not the case – recharge happens throughout the rain and snowmelt season. But this approach allows for expressing the intrinsic landscape response to a change in either magnitude or timing of recharge.

Snow-dominated regions with late snowmelt, such as the Wallowa Mountains, show relatively high sensitivity (fig. 3.6), especially early in summer (July), although they are less sensitive than the Cascade and Olympic Mountains. The rest of the Blue Mountains region shows moderate to low sensitivity to changes in the magnitude and timing of snowmelt (fig. 3.6), although sensitivity in the Wallowas is higher in early summer. The level and sensitivity and the spatial extent of highly sensitive areas was shown to diminish over time as summer progresses.

Projections of future low flows using the VIC hydrologic model (data from Wenger et al. 2010) also show relatively minor decreases in summer streamflow (<10 percent decrease) for 47 percent of perennial streams across the Blue Mountains region by 2080 (fig. 3.7). However, some portions of the region, such as the Wallowas, Greenhorn Mountains, and the Wenaha-Tucannon Wilderness show greater decreases (>30 percent in streamflow by 2080; fig. 3.7).

A direct comparison of the framework developed by Safeeq et al. (2014) and VIC projections for the 2040 time period at the Hydrologic Unit Code 10 watershed scale generally highlight the same portions of the Blue Mountains as being most sensitive to decreases in summer flows as the climate warms in future decades (fig.3.8). However, the exponential model

(Safeeq et al. 2014), which incorporates the role of groundwater, projects larger decreases in summer low flows across the Blue Mountains than the VIC model.

Water Quality

Historical trends in stream temperatures are variable among different studies. Isaak et al. (2010, 2012) found that temperatures at unregulated stream sites closely tracked air temperature trends at nearby weather stations across the Pacific Northwest from 1980 to 2009. Statistically significant stream temperature increases occurred during summer, autumn, and winter, with the highest rates of warming in the summer (reconstructed trend = 0.22 °C per decade). A statistically significant stream cooling trend occurred during the spring season in association with a regional trend towards cooler air temperatures (Abatzoglou et al. 2014, Isaak et al. 2012). Most of the variation in long-term stream temperature trends (80-90 percent) was explained by air temperature trends and a smaller proportion by discharge trends (10-20 percent). Arismendi et al. (2012) examined stream data from a larger number of sites and different periods of record and found variable trends in stream temperature, concluding that stream temperatures have increased at some minimally-altered sites in the Pacific Northwest (28-44 percent) and decreased at others (22-33 percent); no detectable trends were found at the remaining sites. Stream temperature trends were influenced by the length of record, period of record, and location relative to dams, with more warming trends becoming apparent where longer term records were available.

Luce et al. (2014b) analyzed summer stream temperature records from forested streams in the Pacific Northwest and found that cold streams were generally not as sensitive as warm streams to climatic conditions. Thus, temperature in low elevation, warmer streams (less shade, less cool groundwater inputs) will likely increase the most in the future. These results suggest that these warmer streams in the Blue Mountains are relatively sensitive to climate.

The NorWeST Regional Stream Temperature Database (http://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html) used extensive stream temperature observations and spatial statistical models to characterize stream temperatures throughout the Blue Mountains under recent historic conditions at a 1-km resolution (Isaak et al. 2015) (fig. 3.9). Future stream temperatures were then projected based on these historical conditions, assessments of past sensitivity to climate, and projections of future climatic conditions. Results project basin-wide average August stream temperatures in the Blue Mountains to increase by ~1°C by 2040 and by nearly 2°C by 2080 in direct response to climatic conditions (i.e., no consideration of secondary effects, such as increased fire). Warmer streams in the basin will likely warm to a greater degree than cooler ones (Luce et al. 2014b).

Decreasing summer water availability and warming temperatures across the western United States may contribute to forest mortality in some locations (Adams et al. 2009, Allen et al. 2010, Breshears et al. 2005, Meddens and Hicke 2014, van Mantgem et al. 2009) and increased wildfire area burned compared to the mid-20th century (Littell et al. 2009, Westerling et al. 2006). Increased area burned, particularly if fire in riparian areas results in decreased shade over streams, will contribute further to stream temperature increases (Dunham et al. 2007, Isaak et al. 2010). Increases in fire are also increasing basin-scale sediment yields in some basins (Goode et al. 2012).

Conclusions

The results and map products discussed in this chapter represent our current best understanding of the likely effects of climate change on key hydrologic processes. Nevertheless, these results should be applied with caution. Key uncertainties include the specific climate trajectories that the Blue Mountains will experience in the future, critical assumptions underlying all models used, and the myriad uncertainties and errors attached to the calibration of each of the models. Resource managers wishing to apply the results of this analysis in forest planning are encouraged to read the primary literature in which the strengths and limitations of different modeling and forecasting approaches are described.

In general, projections of future trends in streamflow and related processes are strongest in characterizing relative sensitivities of different parts of the landscape rather than absolute changes. In other words, the spatial pattern of trends is more robust than projections associated with any particular location. Similarly, more confidence applies to the interpretation of relative as opposed to absolute magnitudes of projected changes. Differences in results between modeling approaches, such as the low-flow analysis, should be interpreted as bracketing likely potential changes. Finally, the models used here contain uncertainties related to the quantification of soil, vegetation, and other characteristics used to generate hydrologic dynamics.

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Chapter 4: Climate Change, Water Resources, and Roads in the Blue Mountains

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Introduction

Water is a critical resource in dry forest and rangeland environments of western North America, largely determining the distribution of plant and animal species across a broad range of elevations and ecosystems. Water is also essential for human endeavors, directly affecting where and how human communities and local economies have developed. The Blue Mountains of northeast Oregon and southeast Washington are an important source of water for forest ecosystems and human uses. Surrounding communities rely on water from national forest lands in the Blue Mountains for drinking water, industrial uses, irrigation, livestock watering, and recreation, among other uses. Climate change affects water supply by changing the amount, timing and distribution of precipitation and runoff. These changes have the potential to impact water supply, roads and other infrastructure, and access to national forest lands in the Blue Mountains region. Reduced or less reliable water supply affects local economic activities, planning, and resource management. Damage to roads, bridges, and culverts creates safety hazards, affects aquatic resources, and incurs high repair costs. Reduced access to public lands reduces the ability of land managers to preserve, protect, and restore resources and to provide for public use of resources. Understanding vulnerabilities and the processes through which climate change affects hydrology will help U.S. Forest Service land managers identify adaptation strategies that maintain ecosystem function, a sustainable water supply, and a sustainable road system.

In this chapter, we (1) identify key sensitivities of water supply, roads, and infrastructure to changes in climate and hydrology, (2) review current and proposed management priorities and share management approaches that already consider climate or climate change, and (3) use the latest scientific information on climate change and effects on hydrologic regimes (see chapter 3) to identify adaptation strategies and tactics. During a workshop convened in La Grande, Oregon

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in April 2014, participants reviewed the latest science on the sensitivity of water resources and related water uses and infrastructure to climate change in the Blue Mountains (boxes 4.1, 4.2). Workshop participants worked collaboratively to identify adaptation options to reduce vulnerability to climate change and facilitate transition to new conditions. The results of this vulnerability assessment and adaptation planning process are described in the sections below.

Water Resources and Uses

In the predominantly dry climate of the Blue Mountains region, water availability is the most critical natural resource for human habitation and enterprises. Many streams and groundwater systems surrounding the Blue Mountains region originate from the Malheur, Umatilla, and Wallowa-Whitman National Forests, thus providing a valued ecosystem service to local communities and economies. There are approximately 6,800 water rights on national forest lands in the Blue Mountains; 43 percent provide water for domestic livestock, 32 percent support in-stream flows, 9 percent for wildlife, 5 percent for irrigation, and 3 percent for domestic uses (Gecy 2014). By volume, in-stream flows account for 75 percent of water rights, with irrigation and domestic uses each accounting for 1 percent by volume, and water for livestock accounting for 2 percent by volume (Gecy 2014). Six municipalities (Baker City, La Grande, Long Creek, Walla Walla, Pendleton, and Canyon City) rely directly on the national forests for municipal water supply. In addition, 20 smaller communities rely on surface or groundwater from the Blue Mountains forests for drinking water. There are 320 points of diversion (under a certificated water right) within the boundaries of national forests in the Blue Mountains that provide water for domestic use (Gecy 2014; e.g., see fig. 4.1).

Water is critical for livestock on the national forests and surrounding lands, and consumption for this purpose is broadly dispersed across different ecosystems. About 42 percent of national forest land in the Blue Mountains is considered suitable for sheep and cattle grazing, and grazing occurs on these lands in 455 of 552 subwatersheds within the Blue Mountains (Gecy 2014). Water for livestock is the largest permitted water use on national forest land by number of certificated water rights.

All basins in national forests of the Blue Mountains region are fully allocated in terms of water available for appropriation under state law in the dry summer season. In national forests, water is generally available for campgrounds and administrative sites and for other appropriated uses (e.g., livestock and wildlife), although in dry years availability may be limited at some sites, especially in late summer. Dams for storage facilities, stream diversions, and development of springs and ponds for livestock on the national forests affect hydrologic and ecologic function of groundwater-dependent ecosystems (see chapter 7). In drought years, downstream “junior” users in the water allocation system may not receive water for various purposes, primarily irrigation. It is uncertain if long-term climate change or short-term drought will alter permitted water use in the future, although significant changes in water use during the next decade or so are unlikely.

Climate Change Effects on Water Uses

Warming temperatures will lead to decreased snowpack and earlier snow melt, resulting in shifts in timing and magnitude of streamflow and decreased summer soil moisture (see chapter 3).

Across the Blue Mountains, the majority of precipitation occurs during the winter months, when consumptive demand is lowest. In summer, when demand is highest, rain is infrequent and streams are dependent on groundwater to maintain low or base flows. Because water supply in the Blue Mountains is limited, climate change may reduce water available to meet current demands in the summer months, especially during extreme drought years and after multiple consecutive drought years. Although current conflict over water use in the Blue Mountains is not a prominent issue, future water shortages may create social and political tension as different sectors (e.g., agriculture and municipal) compete for scarce water.

Regional water supplies depend on snowpack extent and duration, and late-season water availability (often characterized by April 1 snow water equivalent). Declining summer low flows caused by earlier snowmelt runoff could affect water availability during peak demand. Historical snowpack sensitivity (fig. 4.2) and projections of summer steamflow (fig. 4.3) across the Blue Mountains identify areas that may be particularly sensitive with respect to water supply. Lower elevation locations with mixed snow and rain will be the most vulnerable to reduced spring snowpack, but even the most persistent snowpacks at higher elevation are expected to decline by the 2080s (see chapter 3). Variable Infiltration Capacity hydrological model runs (for natural flows, not accounting for withdrawals, use, and storage) suggest that the Burnt, Powder, Upper Grande Ronde, Silver, Silvies, Upper John Day, Wallowa, and Willow subbasins are at highest risk for summer water shortage associated with low streamflow by 2080 (fig. 4.3). Decreases in summer low flows in these areas have the greatest potential to affect agricultural irrigation and municipal uses.

Water diversions and dams can also affect the resilience of watersheds to climate change. Although dams increase water storage during low flow, diversions also increase water extraction. Aging and inefficient diversion infrastructure can increase water loss. Engaging users within basins where water shortages can occur is critical for resolving addressing water distribution and climate change effects. Clarifying water demand, negotiating water allocations, ensuring environmental flows in the water rights process, adjudicating overallocated basins, and monitoring compliance can help reduce susceptibility to climate stresses.

Water quantity is an important attribute of the Watershed Condition Framework (WCF) classification system used by national forests to rate overall watershed condition (Potyondy and Geier 2011). Most subwatersheds across the Blue Mountains were rated as “functioning” or “functioning at risk” for this attribute, based on the magnitude of existing flow alterations from dams, diversions, and withdrawals relative to natural streamflows and groundwater storage. The Burnt, Powder, Upper Grande Ronde, and Wallowa subbasins have the highest number of subwatersheds rated as having “impaired function” for water quantity on national forest lands (fig. 4.4). Basins with the highest off-forest consumptive uses include Walla Walla, Umatilla, Burnt, Powder, Malheur, Silvies, and Silver Creek (Gecy 2014). Most of these areas are among those expected to experience the greatest changes in summer flows (fig. 4.3) and thus may be the most vulnerable from a water use perspective.

Besides projected changes in streamflows and the magnitude of existing water diversions, the presence or absence of back-up water systems is an important factor affecting the vulnerability of water supplies for human uses. Those systems with redundant supplies will generally be less vulnerable. Development of such systems, as well as increasing water conservation efforts, are key opportunities for adaptation.

Roads, Infrastructure, and Access

Roads, trails, bridges, and other infrastructure were developed in the Blue Mountains over more than a century to provide access for mineral prospectors, loggers, hunters, and recreationists. The national forests in the Blue Mountains were created to protect water supply, timber and range resources, and wildlife and to provide multiple uses and enjoyment by the public. Providing access to accomplish these objectives largely determined where these activities historically occurred. Today, reliable and strategic access is critical for people to recreate, extract resources, monitor and manage resources, and respond to emergencies. Access to public lands promotes use, stewardship, and appreciation of their value as a vital resource contributing to quality of life (Louter 2006).

The three national forests combined contain 37,350 km of roads (table 4.1, fig. 4.5). Of the existing roads, 850 km of roads are paved, 17,800 km are gravel, and the remaining are native surface roads. Road density is higher at low elevations and adjacent to mountain passes, such as near major highways (fig. 4.5). Roads and trails cross many streams and rivers because of the rugged topography. Most (96 percent) known road-water crossings are culverts installed decades ago. Some crossings are being replaced, but many have not been inventoried and conditions are unknown. In many landscapes, the older the road, the more likely it is near or adjacent to streams, greatly increasing risks for road damage and degraded aquatic resources.

Historically, the primary purpose for development of the road system in national forests was timber harvest. Reduced harvesting during the past 20 years has decreased the need for roads for timber purposes. However, local population growth and tourism have increased demand for access for a diversity of recreation activities. Hiking and camping are the most popular activities, but visitors are staying for shorter duration, often only day use; more than 60 percent of trips to national forests last 6 hours or less (USDA FS 2010). Short visits concentrate human impacts on areas that are easily accessible. Demand is increasing for trail use by mountain bikes and motorized vehicles and for routes designated for off-highway vehicles, as well as for winter recreation (USDA FS 2010).

Road Management and Maintenance

The condition of roads and trails differs widely across the Blue Mountains, as do the impact of roads to watersheds and aquatic ecosystems. Culverts were typically designed to withstand a 25-year flood. Road construction has declined since the 1990s, with few new roads being added to the system. Road maintenance is primarily the responsibility of the Forest Service, but county road maintenance crews maintain some roads. The Federal Highway Administration is also involved with the management, design, and funding of roads within the national forests.

Roads vary in their level of environmental impact. They tend to accelerate runoff rates and decrease late season flows, increase peak flows, and increase erosion rates and sediment delivery to the stream system. These impacts are generally greater from roads closer to rivers and streams; however roads in uplands also affect surface and shallow groundwater flows, and erosion processes (Trombulak and Frissell 2000).

Each national forest develops a road maintenance plan for the fiscal year, primarily based on priorities by operational maintenance level, then by category and priority. Maintenance of forest roads subject to Highway Safety Act standards receive priority for appropriated capital maintenance, road maintenance, or improvement funds over roads maintained for high clearance vehicles. Activities that are critical to health and safety receive priority in decisions about which roads to repair and maintain, but are balanced with demands for access and protection of aquatic habitat.

Given current and projected funding levels, national forest staff are examining tradeoffs between providing access and maintaining and operating a sustainable transportation system that is safe, affordable, and responsive to public needs, and which causes minimal environmental impact. Management actions being implemented to meet these sometimes competing objectives include reducing road maintenance levels, storm-proofing roads, upgrading drainage structures and stream crossings, reconstructing and upgrading roads, decommissioning roads, converting roads to alternative modes of transportation, and developing more comprehensive access and travel management plans.

Planning for transportation and access on national forests is included in Forest land management plans. The 2001 Road Management Rule (36 CFR 212, 261, and 295) requires national forests to use science-based analysis to identify a minimum road system that is ecologically and fiscally sustainable. The Malheur, Umatilla, and Wallow-Whitman National Forests are currently identifying a sustainable road network in accordance with the rule. The goals of transportation analysis are to assess the condition of existing roads, identify options for removing damaged or unnecessary roads, and maintaining and improving necessary roads without compromising environmental quality. Transportation analysis has four benefits: (1) increased ability to acquire funding for road improvement and decommissioning; (2) a framework to set annual maintenance costs; (3) improved ability to meet agreement terms with regulatory agencies; and (4) increased financial sustainability and flexibility. Consideration of climate change is not currently a formal part of the analysis.

Major road projects in national forests, such as reconstruction of roads and trails or decommissioning, must comply with the National Environmental Policy Act (NEPA) of 1969 (NEPA 1969), and require an environmental assessment and public involvement. Decommissioning roads is a process of restoring roads to a more natural state by reestablishing drainage patterns, stabilizing slopes, restoring vegetation, blocking road entrances, installing water bars, removing culverts, removing unstable fills, pulling back road shoulders, scattering slash on roadbeds, and completely eliminating roadbeds (36 CFR 212.5; Road System Management; 23 U.S.C. 101).

Spatial and terrain analysis tools developed to assess road risks, such as the Water and Erosion Predictive model (Flanagan and Nearing 1995), the Geomorphic Road Analysis and Inventory Package (GRAIP; Black et al. 2012, Cissel et al. 2012), and NetMap (Benda et al. 2007), are often used to identify hydrologic impacts and guide management on projects. For example, the Wall Creek watershed GRAIP analysis on the Umatilla National Forest identified 12 percent of the road system contributing 90 percent of the sediment, and focused treatment plans to the most critical sites (Nelson et al. 2010).

Climate Change Effects on Transportation Systems

Altered hydrologic regimes are expected as a result of climate change, especially in the latter half of the 21st century (see chapter 3). Specifically, climate and hydrology will influence the transportation system on the Blue Mountains national forests through reduced and earlier runoff of snowpack, resulting in a longer season of road use, higher peak flows and flood risk, and increased landslide risk on steep slopes associated with elevated soil moisture in winter (Strauch et al. 2014). Increased wildfire disturbance (see chapter 6), in combination with higher peak flows, may also lead to increased erosion and landslide frequency.

Changes in climate and hydrology can have both direct and indirect effects on infrastructure and access. Direct effects are those that physically alter the operation or integrity of transportation facilities. These include effects related to floods, snow, landslides, extreme temperatures, and wind. Indirect effects include secondary influences of climate change on access that can increase threats to public safety and change visitor use patterns. For hydrologic extremes such as flooding, the effect on access may be more related to weather events (e.g., the effects of a single storm) rather than climate trends, but the expansion of future extremes outside the historical range of frequency or intensity will likely have the greatest impacts (e.g., by exceeding current design standards for infrastructure).

Projected changes in soil moisture and precipitation form and intensity with climate change may locally accelerate mass wasting in the Blue Mountains. For example in the deeply dissected northern Blue Mountains, shallow rapid debris slides may become more frequent, impacting infrastructure and access. Climate projections indicate that the conditions that trigger landslides will increase because more precipitation will fall as rain rather than snow, and more winter precipitation will occur in intense storms (Salathé et al. 2014). These effects will likely differ with elevation, because higher elevation areas typically have steeper slopes and more precipitation during storms. Flooding can also be exacerbated by increased basin size during rain events, because since snow elevation is projected to move higher. Furthermore, reduced snowpack is expected to increase antecedent soil moisture in winter (Hamlet et al. 2013). Increasing trends in April 1 soil moisture have been observed in modeling studies as a result of warming, showing that soil moisture recharge is occurring earlier in spring and is now higher on April 1 than it was prior to 1947 (Hamlet et al. 2007).

Elevated soil moisture and rapid changes in soil moisture can affect the stability of a slope and are responsible for triggering more landslides than any other factor (Crozier 1986). Antecedent moisture, geology, soil conditions, land cover, and land use are also affect landslides (Kim et al. 1991, Strauch et al. 2014), and areas with projected increases in antecedent soil moisture (coupled with more intense winter storms) will have increased landslide risk. Although the Variable Infiltration Capacity model (VIC; see chapter 3) does not directly simulate slope stability failures or landslides, projections of December 1 total column soil moisture from VIC can be used as an indicator of landslide risk. Projections from VIC indicate that December 1 soil moisture will likely be higher as the climate warms, and thus there will be higher landslide risk in winter on unstable land types at higher elevations.

The vulnerability of roads to hydrologic change (see chapter 3) varies based on topography, geology, slope stability, design, location, and use. To assess vulnerability of the transportation system in the Blue Mountain national forests, we identified the traits of the transportation system most sensitive to projected climate changes (box 4.3). This vulnerability assessment of the transportation system can inform transportation management and long-range planning.

Roads and trails built decades ago have increased sensitivity because of age and declining condition. Many infrastructure components are at or near the end of their design lifespan. Culverts were typically designed to last 25 to 75 years, depending on structure and material. Culverts remaining in place beyond their design life are less resilient to high flows and bed load movement and have a higher likelihood of structural failure. As roads and trails age, their surface and subsurface structure deteriorates and less intense storms can cause more damage than a storm of high intensity would have when the infrastructure was new.

Advanced design of materials, alignment, drainage, and subgrade that are required standards today were generally not available or required when much of the travel network was developed in the Blue Mountains. Consequently, new or replaced infrastructure is likely to have increased resilience to climate change, especially if climate change is considered in the design. New culverts and bridges are often wider than the original structures to meet agency regulations and current design standards. In the past 15 years, many culverts across the Blue Mountains have been replaced to improve fish passage and stream function using open bottomed arch structures that are less constricted during high flows and accommodate aquatic organism passage at a range of flows. Natural channel design techniques that mimic the natural stream channel condition upstream and downstream of the crossing are being used at these crossings. In addition, culverts on non-fish bearing streams are being upgraded.

The location of roads and trails can increase vulnerability to climate change. Many roads and trails were built on steep slopes because of the rugged topography of the region, so cut slopes and side-cast material have created landslide hazards. Past timber harvesting and its associated road network in national forests have contributed to the sensitivity of existing infrastructure by increasing storm runoff and peak flows that can affect road crossing structures (Croke and Hairsine 2006, Schmidt et al. 2001, Swanston 1971). Many roads and trails were also constructed in valley bottoms near streams to take advantage of gentle grades, but proximity to streams increases sensitivity to flooding, channel migration, bank erosion, and shifts in alluvial fans and debris cones. Most road-stream crossings used culverts rather than bridges, and culverts are generally more sensitive to increased flood peaks and associated debris. Roads that are currently in the rain-on-snow zone, typically in mid-elevation basins, may be increasingly sensitive to warmer temperatures.

Management of roads and trails (planning, funding, maintenance, and response) affect the sensitivity of the transportation system, and the condition of one road or trail segment can affect the function of connected segments. Major highways within the Blue Mountains, built to higher design standards and maintained more frequently, will likely be less sensitive to climate change than their unpaved counterparts built to lower design standards in the national forests. Lack of funding can limit options for repairing infrastructure, which can affect the short- and long-term vulnerability of the transportation system. For example, replacing a damaged culvert with an “in kind” culvert that was undersized for the current streamflow conditions leads to continued sensitivity to both the current flow regime and projected higher flows.

Current and Near-Term Climate Change Effects

Assessing the vulnerability of the transportation network in the Blue Mountains to climate change (boxes 4.3, 4.4) requires evaluating projected changes in hydrologic processes (box 4.1). The integrity and operation of the transportation network in the Blue Mountains may be affected in several ways. Changes in climate have already altered hydrologic regimes in the Pacific Northwest, resulting in decreased snowpack, higher winter streamflow, earlier spring snowmelt, earlier peak spring streamflow, and lower streamflow in summer (Hamlet et al. 2007, 2010). Ongoing changes in climate and hydrologic response in the short term (in the next 10 years) are likely to be a mix of natural variability combined with ongoing trends related to climate change. High variability of short-term trends is an expected part of the response of the evolving climate system. Natural climatic variability, in the short term, may exacerbate, compensate for, or even temporarily reverse expected trends in some hydroclimatic variables. This is particularly true for strong El Niño years (high El Niño Southern Oscillation index) and during warm phases of the Pacific Decadal Oscillation (as well as years with high Pacific Decadal Oscillation index), which may provide a preview of future climatic conditions under climate change.

Higher streamflow in winter (October through March) and higher peak flows, in comparison to historical conditions, increase the risk of flooding and impacts to structures, roads, and trails. Many transportation professionals consider flooding and inundation to be the greatest threat to infrastructure and operations because of the damage that standing and flowing water cause to transportation structures (MacArthur et al. 2012, Walker et al. 2011). Floods also transport logs and sediment that block culverts or are deposited on bridge abutments. Isolated intense storms can overwhelm the vegetation and soil water holding capacity and concentrate high velocity flows into channels that erode soils and remove vegetation. During floods, roads and trails can become preferential paths for floodwaters, reducing operational function and potentially damaging infrastructure not designed to withstand inundation. If extreme peak flows become more common, they will have a major effect on roads and infrastructure.

In the short term, flooding of roads and trails will likely increase, threatening the structural stability of crossing structures and subgrade material. Roads near perennial and other major streams are especially vulnerable (fig. 4.6), and many of these roads are located in floodplains and are used for recreation access. Increases in high flows and winter soil moisture may also increase the amount of large woody debris delivered to streams, further increasing damage to culverts and bridges, and in some cases making roads impassable or requiring road and facility closures. Unpaved roads with limited drainage structures or minimal maintenance are likely to experience increased surface erosion, requiring additional repairs or grading.

Increasing incidence of more intense precipitation and higher soil moisture in early winter could increase the risk of landslides in some areas. Landslides also contribute to flooding by diverting water, blocking drainage, and filling channels with debris (Chatwin et al. 1994, Crozier 1986, Schuster and Highland 2003). Increased sedimentation from landslides also causes aggradation within stream, thus elevating flood risk. Culverts filled with landslide debris can cause flooding, damage, or complete destruction of roads and trails (Halofsky et al. 2011). Landslides that connect with waterways or converging drainages can transform into more destructive flows (Baum et al. 2007). Roads themselves also increase landslide risk (Swanson and Dyrness 1975, Swanston 1971), especially if they are built on steep slopes and through erosion-prone drainages. In the western United States, the development of roads increased the rate of debris avalanche erosion by 25 to 340 times the rate found in forested areas without roads (Swanston 1976), and Chatwin et al. (1994) and Montgomery (1994) found that the number of landslides is directly correlated with total kilometers of roads in an area. Consequently, areas with high road or trail density and projected increases in soil moisture that already experience frequent landslides may be most vulnerable to increased landslide risks.

Short-term exposures to changes in climate may affect safety and access in the Blue Mountains. Damaged or closed roads reduce agency capacity to respond to emergencies or provide detour routes during emergencies. Increased flood risk could make conditions more hazardous for river recreation and campers. More wildfires (see chapter 6) could reduce safe operation of some roads and require additional emergency response to protect recreationists and communities (Strauch et al. 2014). Furthermore, damaged and closed roads can reduce agency capacity to respond to wildfires.

Emerging and Intensifying Exposure in the Medium and Long Term

Many of the observed exposures to climate change in the short term are likely to increase in the medium (10-30 years) and long term (greater than 30 years) (box 4.4). In the medium term, natural climatic variability may continue to affect outcomes in any given decade, whereas in the long term, the cumulative effects of climate change may become a dominant factor, particularly for temperature-related effects. Conditions thought to be extreme today may be averages in the future, particularly for temperature-related changes (MacArthur et al. 2012).

Flooding in autumn and early winter is projected to continue to intensify in the medium and long term, particularly in mixed-rain-and-snow basins, but direct rain-and-snow events may diminish in importance as a cause of flooding (McCabe et al. 2007). At mid to high elevations, more precipitation falling as rain rather than snow will continue to increase winter streamflow. By the 2080s, peak flows are anticipated to increase in magnitude and frequency (fig. 4.8; see chapter 3). In the long term, higher and more frequent peak flows will likely continue to increase sediment and debris transport within waterways. These elevated peak flows could affect stream-crossing structures downstream as well as adjacent structures because of elevated stream channels. Even as crossing structures are replaced with wider and taller structures, shifting channel dynamics caused by changes in flow and sediment may affect lower-elevation segments adjacent to crossings, such as bridge approaches.

Projected increases in flooding in autumn and early winter will shift the timing of peak flows and affect the timing of maintenance and repair of roads and trails. More repairs may be necessary during the cool, wet, and dark time of year in response to damage from autumn flooding and landslides, challenging crews to complete necessary repairs before snowfall. If increased demand for repairs cannot be met, access may be restricted until conditions are more suitable for construction and repairs.

In the long term, declines in low streamflow in summer may require increased use of more expensive culverts and bridges designed to balance the management of peak flows with providing low flow channels in fish-bearing streams. Road design regulations for aquatic habitat will become more difficult to meet as warming temperatures hinder recovery of cold-water fish populations, although some streams may be buffered by inputs from snow melt or ground water in the medium term.

Over the long term, higher winter soil moisture may increase the risk of landslides in autumn and winter. Landslide risk may increase more in areas with tree mortality from fire and insect outbreaks, because tree mortality reduces soil root cohesion and decreases interception and evaporation, further increasing soil moisture (Martin 2006, Montgomery et al. 2000, Neary et al. 2005, Schmidt et al. 2001). Thus, soils will likely become more saturated and vulnerable to slippage on steep slopes during the wet season. Although floods and landslides will continue to occur near known hazard areas (e.g., because of high forest road density), they may also occur in new areas (e.g., those areas which are currently covered by deep snowpack in mid-winter) (MacArthur et al. 2012). Thus, more landslides at increasingly higher elevations (with sufficient soil) may be a long-term effect of climate change. Coinciding exposures in space and time may be particularly detrimental to access.

Climate change effects on access may create public safety concerns for national forests. A longer snow-free season may extend visitor use in early spring and late autumn at higher elevations (Rice et al. 2012). Lower snowpack may lead to fewer snow-related road closures for a longer portion of the year, allowing visitors to reach trails and campsites earlier in the season. However, warmer temperatures and earlier snowmelt may encourage use of trails and roads before they are cleared. Trailheads, which are located at lower elevations, may be snow-free earlier, but hazards associated with melting snow bridges, avalanche chutes, or frozen snowfields in shaded areas may persist at higher elevations along trails. Relatively rapid warming at the end of the 20th century coincided with greater variability in cool season precipitation and increased flooding (Hamlet and Lettenmaier 2007). If this pattern continues, early-season visitors may be exposed to more extreme weather than they have encountered historically, creating potential risks to visitors. In summer, whitewater rafters may encounter unfavorable conditions from lower streamflows in late summer (Mickelson 2009) and hazards associated with deposited sediment and woody debris from higher winter flows. Warmer winters may shift river recreation to times of year when risks of extreme weather and flooding are higher. These activities may also increase use of unpaved roads in the wet season, which can increase damage and associated maintenance costs.

Climate change may also benefit access and transportation operations in the Blue Mountains over the long term. Lower snow cover will reduce the need for and cost of snow removal, and earlier snow-free dates projected for the 2040s suggest that low- and mid-elevation areas will be accessible earlier. Earlier access to roads and trails will create opportunities for earlier seasonal maintenance and recreation. Temporary trail bridges installed across rivers may be installed earlier in spring as spring flows decline. A longer snow-free season and warmer temperatures may allow for a longer construction season at higher elevations. Less snow may increase access for summer recreation, but it may reduce opportunities for winter recreation particularly at low and moderate elevations (Joyce et al. 2001, Morris and Walls 2009). The highest elevations of the Blue Mountains may retain relatively more snow than other areas, which may create higher localized demand for winter recreation and river rafting in summer over the next several decades.

Adapting Management of Water Use and Roads in a Changing Climate

Through a workshop and subsequent dialogue, scientists and resource managers worked collaboratively to identify adaptation options that can reduce the adverse effects of climatic variability and change on water use and roads in the Blue Mountains. The workshop included an overview of adaptation principles (Peterson et al. 2011) and regional examples of agency efforts to adapt to climate change. Options for adapting hydrologic systems, transportation systems, and access management were identified, as well as potential barriers, opportunities, and information needs for implementing adaptation.

Adaptation Options for Water Use

Climate change adaptation options for water use on national forests must be considered within the broader context of multi-ownership watersheds, where most of the traditional consumptive uses occur off the forest but the forests are relied on for a majority of the supply. Many of the resource sensitivities addressed here already exist to some extent, but are expected to intensify as the climate warms. Adaptation options focusing on national forests were developed after consideration of the collective effects of several climate-related stressors: lower summer streamflow, higher winter peak streamflow, earlier peak streamflow, lower groundwater recharge, and higher demand and competition for water by municipalities and agriculture (table 4.2). The following adaptation strategies were developed to address these stressors: (1) restore function of watersheds; (2) connect floodplains; (3) support groundwater dependent ecosystems; (4) reduce drainage efficiency; (5) maximize valley storage; and (6) reduce fire hazard (table 4.2). The objective of most of these adaptation strategies is to retain water for a longer period of time at higher elevations and in riparian systems and groundwater of mountain landscapes. These strategies will likely help maintain water supplies to meet demands, especially during the summer, and reduce loss of water during times when withdrawals are low. This diversity of adaptation strategies requires an equally diverse portfolio of adaptation tactics that address different biophysical components of hydrologic systems and timing of uses, among other considerations (table 4.2).

The adaptation tactic of using a "climate change lens" when developing plans and projects, provides an overarching context for managing and conserving water supply (tables 4.1, 4.2). Including climate change in decision making generally reinforces practices that support sustainable resource management. Potential risk and uncertainty can be included in this process by considering a range of climate projections (based on different models and emission scenarios) (see chapter 3) to frame decisions about appropriate responses to climate change. In addition, user awareness of vulnerability to shortages, reducing demand through education and negotiation, and collaboration among users can support adaptation efforts.

Many adaptation tactics to protect water supply are current standard practices, or "best management practices (BMP)," for water quality protection. In 2012, the Forest Service implemented a national best management practices program to improve management of water quality consistently with the Federal Clean Water Act and state water quality programs (USDA FS 2012; http://www.fs.fed.us/biology/watershed/BMP.html). BMPs are specific practices or actions used to reduce or control impacts to water bodies from nonpoint sources of pollution, most commonly by reducing the loading of pollutants from such sources into storm water and waterways. BMPs are required on all activities with potential to affect water quality, including road management and water developments. A related tactic is improving roads and drainage systems to maintain high-quality water as long as possible within hydrologic systems in national forests. Although these tactics may be expensive, they have a significant impact on water retention and erosion control. Actions are typically needed at specific locations at key times even under normal conditions, and climate change will likely force more frequent maintenance and repair.

Several adaptation tactics related to biological components of mountain landscapes can reduce the effects of climate change on water resources. Reducing stand density and surface fuels in low-elevation coniferous forest reduces the likelihood of fires that severely impact soils, accelerate erosion and degrade water quality in streams. Vegetation treatments in high-snow areas may enhance snow retention and soil moisture, and may extend water yield into the summer, at the catchment scale, for a few years following treatment.

Similarly, restoration techniques that maintain or modify biophysical properties of hydrological systems to be within their pre-settlement historical range of variability can increase climate change resilience. Stream restoration techniques that improve floodplain hydrologic connectivity increase water storage capacity. Meadow and wetland restoration techniques that remove encroaching conifers can improve hydrologic function and water storage capacity. Adding wood to streams improves channel stability and complexity, slows water movement, improves aquatic habitat, and increases resilience to both low and high flows. Similarly, increasing beaver (*Castor canadensis* Kuhl) populations will create more ponds, swamps, and low-velocity channels that retain water throughout the year.

Lower soil moisture and low flows in late summer, combined with increasing demand for water, will likely reduce water availability for aquatic resources, recreation, and other uses. However, water conservation measures within national forests can potentially reduce water use. For example, resource managers can work with permitees to implement livestock management practices that use less water (e.g., install shut-off valves on stock water troughs). Over the long term, increasing water conservation and reducing user expectations of water availability (e.g., through education) are inexpensive and complementary adaptation tactics for maintaining adequate water supply.

At a broader level, it will be valuable to engage in and contribute to integrated assessments, such as the Oregon Integrated Water Resource Strategy, for water supply and availability and local effects of climate change. Vulnerability assessments for individual communities will provide better information on where and when water shortages may occur, leading to adaptation tactics customized for each location. Because discussions of water use and water rights are often contentious, it will be important to help foster open dialogue and full disclosure of data and regulatory requirements so that proactive, realistic and fair management options can be developed.

Adaptation Options for Roads and Infrastructure

Climate change adaptation options for roads and infrastructure were developed after consideration of the collective effects of several climate-related stressors: sensitivity of road design and maintenance to increasing flood risk, effects of higher peak streamflows on road damage at stream crossings, and safety hazards associated with an increase in extreme disturbance events (table 4.3, box 4.2). The following adaptation strategies were developed to address these stressors: (1) increase resilience of stream crossings, culverts, and bridges to higher streamflow; and (2) increase the resilience of the road system to higher streamflows and associated damage by stormproofing and reducing the road system.

The Forest Service travel analysis process (USDA FS 2005) and BMPs provide an overarching framework for identifying and maintaining a sustainable transportation system in national forests in the Blue Mountains, and climate change provides a new context for evaluating current practices (Raymond et al. 2014, Strauch et al. 2014). Incorporating climate change in the travel analysis process, which is already addressing some vulnerabilities by decommissioning and stormproofing roads, culverts, and bridges, will enhance resilience to higher streamflows. Improving and updating geospatial databases of roads, culverts, and bridges will provide a foundation for continuous evaluation and maintenance. If vulnerable watersheds, roads, and infrastructure can be identified, then proactive management (e.g., use of drains, gravel, and outsloping of roads to disperse surface water) can be implemented to reduce potential damage and high repair costs.

National forests in the Blue Mountains have a large backlog of culverts and road segments in need of repair, replacement, or upgrade, even under current hydrologic regimes. Limited funding and staff hinder current efforts to upgrade the system to current standards and policies, so the additional cost of upgrades to accommodate future hydrological regimes could be a barrier to adaptation. However, extreme floods that damage roads and culverts can be opportunities to replace existing structures with ones that are more resilient to higher peak flows. These replacements, called "betterments," can be difficult to fund under current Federal Highway Administration Emergency Relief for Federally Owned Roads (ERFO) program eligibility requirements when used to fix damage from extreme events, because the current policy is to replace in kind. In some cases, matching funds can be raised or betterments can be funded with sufficient justification and documentation of the environmental impacts. Justification for betterments based on the latest climate change science would facilitate this approach.

Increasing resilience to higher peak flows will not be possible for all road segments because of limited funding for maintenance. Adapting road management to climate change in the long term may require further reductions in the road system. Road segments that are candidates for decommissioning are typically those with low demand for access, high risks to aquatic habitat, a history of frequent failures, or combinations of the three. National forest road managers also consider use of roads for fire management (fire suppression, prescribed fire, and hazardous fuel treatments).

Engineers may consider emphasizing roads for decommissioning that are in basins with higher risk of increased flooding and peak flows, in floodplains of large rivers, or on adjacent low terraces. Information on locations in the transportation system that currently experience frequent flood damage (Strauch et al. 2014) can be combined with spatially explicit data on projected changes in flood risk and current infrastructure condition to provide indicators of where damage is most likely to continue and escalate with changes in climate (e.g., figs. 4.7 and 4.8). Optimization approaches (e.g., linear programming) can be used to compare the tradeoffs associated with competing objectives and constraints while minimizing the overall costs of the road system.

Reducing the road system in national forests presents both barriers and opportunities (table 4.3). Decommissioning roads or converting roads to trails is expensive and must be done properly to reduce adverse effects on water quality and aquatic habitat. Furthermore, reductions to the road system are often met with opposition from the public accustomed to using roads for recreational access, but public involvement in road decisions can also be an opportunity to increase awareness and develop "win-win" adaptation options. Thus, one adaptation tactic is to adjust visitation patterns and visitor expectations by actively involving the public in road decisions related to climate change. This has the added benefit of raising political support and possibly funding from external sources to help maintain access. Partnerships with recreation user groups will be increasingly important for raising public awareness of climate change threats to access and for identifying successful adaptation options.

Increased risk of flooding in some basins may require modification to current management of facilities and historic and cultural resources. In most cases, the high cost of relocating buildings and inability to move historic sites from floodplains will require that adaptation options focus on resistance through prevention of flood damage. Stabilizing banks reduces risk to infrastructure, and using bioengineering rather than rip-rap or other inflexible materials may have less environmental impact. In the long term, protecting infrastructure in place will be more difficult as flood risk continues to increase. Long-term adaptation strategies may require removing (or not rebuilding) infrastructure in the floodplain to allow river channels to migrate and accommodate the changing hydrologic regime. National forest land and resource management plans, which have relatively long planning horizons, are opportunities to implement these long-term adaptation tactics for management of facilities and infrastructure.

More frequent failures in the road and trail system may increase risks to public safety. Limited resources and staff make it difficult for national forests to quickly repair damage, yet the public often expects continuous access. In response to climate change, managers may consider implementing and enforcing more restrictions on access to areas where trails and roads are damaged and safe access is uncertain. Greater control of seasonal use, combined with better information about current conditions, especially during early spring and late autumn, before and after active maintenance, will ensure better public safety. Partnerships with recreation user groups may generate opportunities to convey this message to a larger audience, thus enhancing public awareness of hazards and the safety of recreation users.

Managers may consider adapting recreation management to changes in visitor use patterns in early spring and late autumn in response to reduced snowpack and warmer temperatures. An expanded visitor season would increase the cost of operating facilities (e.g., campgrounds), but revenue from user fees may also increase. Land management plans and transportation planning provide opportunities to address anticipated changes in the amount and timing of visitation. Limitations on staff because of funding or other constraints may also present obstacles to an expanded visitor season. Adaptive management can be used to monitor changes in the timing, location, and number of visitors, thus providing data on where management can be modified in response to altered visitor patterns.

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Chapter 5: Climate Change, Fish, and Aquatic Habitat in the Blue Mountains

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Introduction

National Forest System lands in the Blue Mountains region support a diversity of important native aquatic species that will be affected by climate change. As part of the Blue Mountains Adaptation Partnership, four of these species (spring Chinook salmon (*Oncorhynchus tshawytscha* Walbaum in Artedi), bull trout (*Salvelinus confluentus* Suckley), summer steelhead (*O. mykiss* Walbaum), and interior redband trout (*O. m. gibbsi* Suckley) were selected for climate vulnerability analysis because of their important role in land management planning (e.g., grazing, timber harvest, ESA consultations). These species occupy a broad range of aquatic habitats from small headwaters tributaries to large rivers, both within and downstream of the Malheur, Umatilla, and Wallowa-Whitman National Forests. Although habitats for the selected species overlap in places, each species uses a unique set of aquatic habitats in the Blue Mountains national forests and their associated subbasins, depending on their life stage, season of the year, and available habitat conditions. These species have a diverse array of life history strategies, including anadromy (steelhead and spring Chinook salmon), fluvial and adfluvial movements (bull trout), and residency (bull trout and redband trout).

Climate change affects the environments of these species in many ways (box 5.1). Warming air temperatures and changing precipitation patterns are resulting in warmer stream temperatures (Bartholow 2005, Isaak et al. 2010, Isaak et al. 2012b, Petersen and Kitchell 2001), altered stream hydrology (Hamlet and Lettenmaier 2007, Luce et al. 2013), and changes in the frequency, magnitude, and extent of climate-induced events such as floods, droughts, and wildfires (Holden et al. 2012, Littell et al. 2010, Luce and Holden 2009, Rieman and Isaak 2010). Fish populations have been adapting by shifting their phenology and migration dates (Crozier et al. 2008, Crozier et al. 2011, Keefer et al. 2008), using cold-water refugia during thermally stressful periods (Keefer et al. 2009, Torgersen et al. 1999, Torgersen et al. 2012), and shifting spatial distributions within river networks (Comte et al. 2013, Eby et al. 2014). These

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changes are adding additional stressors to many fish populations, but many populations are also likely to have sufficient resilience and habitat diversity to make the necessary adjustments.

In this chapter, we assess specific vulnerabilities of each of the four selected species to climate change in stream networks draining the Blue Mountains and Forest Service lands. Aiding this assessment are recently developed, high-resolution stream temperature and flow scenarios (Variable Infiltration Capacity model [VIC] ecological flow metrics - Wenger et al. 2010; NorWeST - Isaak et al. 2011; Hamlet et al. 2013) that translate outputs from Global Climate Models (GCMs) to reach-scale habitat factors relevant to aquatic biota. The concluding section of this chapter discusses management strategies that can be used to help species and aquatic ecosystems in the Blue Mountains better adapt to climate change. No attempt is made to provide an exhaustive review of the climate-aquatic-fisheries literature because good general reviews already exist for the Pacific Northwest (ISAB 2007, Mantua et al. 2009, Mantua et al. 2011, Mantua and Raymond 2014, Mote et al. 2003) and other geographic regions (Ficke et al. 2007, Furniss et al. 2010, Furniss et al. 2013, Isaak et al. 2012a, Luce et al. 2012, Poff et al. 2002, Rieman and Isaak 2010, Schindler et al. 2008).

Analysis Area

This chapter focuses on the 23 subbasins draining the Blue Mountains province, specifically the revision analysis area covered by the joint land management plan for the Malheur, Umatilla, and Wallowa-Whitman National Forests. The subbasins are distributed among four major hydrologic units or basins, referred to here as river basins (fig. 5.1): the Middle Columbia River Basin which drains the west face of the Blue Mountains; the Lower Snake River Basin, which drains the northern part of the Blue Mountains; the Middle Snake River Basin, which drains the southeast face of the Blue Mountains; and the Oregon Closed Basin, which drains the southern Blue Mountains but has no hydrologic connections to either the Columbia or Snake Rivers.

Approximately 20,000 km of streams occur in the Blue Mountains, of which almost 7,000 km are on national forest lands. The distribution of the four selected fish species varies in each subbasin depending on available habitat and unique life history characteristics. Generally speaking, redband trout are the only species in Closed Basin streams, redband trout and bull trout are the only species in the Middle Snake Basin (access to this basin by anadromous steelhead and spring Chinook salmon was blocked by construction of the Hells Canyon Dam complex), and all four species occur in the Lower Snake and Middle Columbia basins.

The majority of national forest lands in the analysis area is administered by the Malheur, Umatilla, and Wallowa-Whitman National Forests, but a small portion of the Ochoco National Forest is also present and administered in conjunction with Malheur National Forest lands. For convenience, references to the southern portion of the Malheur National Forest will be assumed to also include the small portion of Ochoco National Forest lands in the analysis area. The Middle Snake River Basin contains portions of the Wallowa-Whitman and Malheur National Forests. The Lower Snake River Basin contains portions of the Umatilla and Wallowa-Whitman National Forests, and the Middle Columbia River Basin encompasses portions of the three main Blue Mountains national forests.

Selected Aquatic Species

Spring Chinook Salmon

Spring Chinook salmon are a fall-spawning anadromous species. They can be found in medium-sized river reaches and tributaries within and downstream of national forest boundaries. Because they are fall spawners, they move upriver to spawning grounds in mid-late summer and can experience thermal barriers to upstream movement through migratory corridors in the analysis area caused by sudden increases in air and water temperatures at the onset of summer, known in past decades to occur more quickly than salmon can adjust physiologically. In some instances, sudden temperature increases have caused direct mortality of salmon adults.

Native spring Chinook salmon within the analysis area belong to two different Environmentally Significant Units (ESUs): the Middle Columbia River (MCR) spring Chinook salmon ESU in the Middle Columbia River Basin and the Snake River Basin (SRB) spring Chinook salmon ESU in the Lower Snake River Basin (Figure 5.1). Snake River Basin spring Chinook salmon populations spawn and rear in subbasins forming the Lower Snake River hydrologic Basin: Tucannon, Grande Ronde, Wallowa, and Imnaha subbasins. Native Middle Columbia River spring Chinook salmon spawn in subbasins forming the headwater forks of the John Day River: the North, Middle, Upper John Day and South Fork subbasins draining the west face of the Blue Mountains province in the Middle Columbia River Basin. Reintroduced hatchery stocks spawn in Walla Walla and Umatilla subbasins headwater habitats within the Middle Columbia River Basin in the northern Blue Mountains. For convenience, further discussions of native Middle Columbia River spring Chinook salmon will refer to these reintroduced populations as well, unless otherwise noted.

Bull Trout

Bull trout, like spring Chinook salmon, are fall spawners with eggs that overwinter in the gravels and fry that emerge from redds in late winter and early spring. Their habitat ranges from medium-sized, high-elevation tributaries to very small headwater streams. Migratory individuals are known to winter in larger rivers and tributaries but move upriver towards headwater resident tributaries as migratory corridors begin to warm in summer and as the fish prepare to spawn. Optimum habitats for bull trout provide year-round high quality cold water and high habitat complexity.

Bull trout in the Blue Mountains exist in a variety of life history forms, including freshwater migratory (fluvial and adfluvial) and headwater year-round resident. Migratory bull trout move between their natal streams and larger bodies of freshwater, such as lakes, reservoirs and mainstem rivers where they can grow much larger than the year-round residents that rear and mature in small colder headwaters.

Bull trout populations are often a mix of resident and migratory individuals, an adaptation to infrequent but major natural disturbances in their high-elevation habitats. When such disturbances cause a small resident population to be eliminated, migratory individuals that were elsewhere at the time of the event can then establish a new population in the vacant habitat, although such recolonization may not occur immediately. The benefits of such disturbances are that they often deliver pulses of large wood and streambed material that provide new spawning gravels and increase habitat complexity, providing for resting places and cover to shelter them from predators and reduce energy demands imposed by fast streamflow. A fresh assortment of large streambed substrate provides spaces in the streambed where juveniles can hide from predators.

Columbia River bull trout are found in all subbasins except for the Burnt River in the Middle Snake River Basin, the Oregon Closed Basins, and the South Fork Crooked River subbasin in the south end of the plan area (Ochoco National Forest lands; Middle Columbia River Basin). The migratory life history is still present in most of the populations inhabiting national forest lands in the Blue Mountains province, but some headwater resident populations in the analysis area have become isolated and are very small due to the effects of historic land use. Connectivity among such resident populations is no longer provided by migratory individuals, and these populations are considered to be at heightened risk of long-term extirpation.

Steelhead Trout/Redband Trout

Research suggests that steelhead and redband trout are alternate life history forms of the same species, and often constitute the same populations, where they co-occur (McMillan et al 2011, Mills et al. 2012). Both steelhead and redband trout spawn at lower elevations and tolerate warmer temperatures in their spawning and rearing habitats than do fall-spawning bull trout.

Steelhead trout are a large-bodied anadromous form of *O. mykiss*. They spawn in the spring in medium rivers to headwater tributaries and rear in cool medium and small rivers, tributary and headwater streams, and upstream portions of large rivers in the analysis area, within accessible portions of the Middle Columbia River and Snake River Basins. Steelhead in the analysis area belong to two spatially discrete Distinct Population Segments (DPS): the Middle Columbia River (MCR) steelhead DPS and the Snake River Basin steelhead DPS in the Lower Snake River Basin. Snake River Basin populations spawn and rear in subbasins within the Lower Snake River Basin: Tucannon, Asotin, Upper and Lower Grande Ronde, Wallowa and Imnaha subbasins. Middle Columbia River steelhead spawn in subbasins forming the headwaters of the John Day River: the North, Middle, Upper and South Fork subbasins, as well as in the Umatilla and Walla Walla subbasins, all within the Middle Columbia River Basin.

Redband trout are a much smaller bodied spring-spawning resident life form of *O. mykiss*. They represent a different range of needs both spatially and temporally than large-bodied spring Chinook salmon and steelhead. Redband trout inhabit small and medium rivers and tributary streams year round and spawn from late winter into May. They are found throughout the entire analysis area, including several subbasins where steelhead are absent, and are considered present wherever steelhead are present. They occur in subbasins of the Blue Mountains where no spring Chinook salmon, steelhead, or other native trout species are present.

The juvenile life stages of the two forms of *O. mykiss* are indistinguishable visually where they co-occur, but juvenile individuals eventually express one or the other of the two life histories, as they either develop physiologically into ocean-going steelhead or remain as freshwater resident redband trout. The likelihood that a juvenile will express one or the other life history is strongly influenced by environmental and physiological factors including water temperature, food supply, gender, growth rates, and body fat development that interact to determine which individuals out-migrate as steelhead smolts and which remain to mature in freshwater (Sloat and Reeves 2014). Where they co-occur, offspring of female steelhead may mature into resident redband trout, and offspring of female redband trout may ultimately out-migrate to the ocean and return to natal streams as adult steelhead (Carmichael et al. 2005).

Current Status and Trend

Current Population Conditions

Snake River Basin spring Chinook salmon are currently listed as threatened under the U.S. Endangered Species Act within the Blue Mountains analysis area. Middle Columbia River spring Chinook salmon are not listed at this time. Only the anadromous (steelhead) forms of *O. mykiss* are listed as threatened (two steelhead DPSs); whereas resident redband trout remain unlisted. Bull trout are also listed as threatened (two DPSs). Each listed DPS and ESU is considered a separate species under the Endangered Species Act, with recovery goals both for component populations of each listed DPS and ESU at the subbasin scale or comparable scales and for the DPS/ESU as a whole.

Status reviews for listed species populations occur approximately every five years after they have been listed. The most recent status review for listed anadromous species was based on the criterion of self-sustainability (Ford 2011). The majority of the biological review team members concluded that Snake River spring/summer Chinook salmon ESU and both Middle Columbia River and Snake River Basin steelhead DPS are still high risk and should remain listed as threatened. The recent five-year status and trend assessment summaries (NMFS 2011a,b) indicate that although most listed spring Chinook salmon and summer steelhead populations are still considered nonviable, most populations are generally increasing. The National Marine Fisheries Service describes and monitors population viability for anadromous fish populations and major population groups at different scales by subbasin, major tributary watershed, and/or groups of adjoining subbasins, depending on the population of interest.

For purposes of bull trout recovery planning, core populations have been designated as population units that correspond scale-wise to anadromous populations (Whitsell et al. 2004). These core populations have been generally defined at the subbasin scale, but may also be described based on groups of adjoining subbasins, depending on the population of interest. Core populations in the occupied subbasins are all part of the Columbia River bull trout DPS, which is listed as threatened. For the remainder of this analysis, core populations for Columbia River bull trout will be referred to simply as “populations,” and Columbia River bull trout will be referred to simply as “bull trout.” The most recent status and trend assessments for bull trout in the analysis area indicate that most populations are presently stable or increasing (USFWS 2008; http://www.fws.gov/pacific/bulltrout).

Interagency conservation planning efforts have stratified redband trout populations in the analysis area into spatially discrete conservation population groups based on several geographic clusters of subbasins (May et al. 2012) within each of the four major hydrologic basins draining the Blue Mountains. Most of these large-scale population groups ware targeted for conservation efforts.

Redband populations in the Middle Columbia River and Lower Snake River basins where resident populations co-occur with steelhead are generally considered by local biologists to be depressed. They parallel steelhead population conditions but to an unknown extent, because as resident individuals, redband trout do not experience the challenges that affect anadromous steelhead during the years spent outside their natal subbasins. Those challenges include adverse conditions in freshwater migratory corridors in the larger rivers, passage through and around mainstem hydropower dams, physiological transitions to and from saltwater environments in the Columbia River estuary, and additional hazards as they feed in the open ocean before returning to their natal streams. Interbreeding between redband and steelhead (Kostow 2003) may support population viability for both life histories where they co-occur, but the extent to which this occurs or the extent to which each life history supports viability of the other, is unknown.

Redband trout populations, which have long been naturally isolated from steelhead in the Oregon Closed Basins (*O. mykiss newberryi*), have also been separated from other redband populations in the Columbia and Snake River Basins (*O. m. gairdnerii*) for millennia, and are especially targeted for conservation. The redband populations in this group of subbasins are also collectively known as the Great Basin DPS. Although still considered viable (USFWS 2009), these populations are believed to be declining, primarily due to land management impacts, particularly downstream of national forest lands.

Current Fish Habitat Conditions

The majority of resident bull trout populations and spawning habitats are located in high-elevation habitats where management is limited. These areas are mostly allocated to wilderness, wild and scenic river corridors, municipal watersheds, and backcountry nonmotorized use where timber harvest, livestock grazing and roaded access are limited. Natural disturbances in high-elevation spawning areas generally operate at natural frequencies, magnitudes, and rates to which bull trout have adapted over centuries.

Habitat conditions in downstream migratory corridors, however, may affect fluvial bull trout adults as well as anadromous species. Current connectivity and habitat conditions in the Blue Mountains, especially in middle and lower elevations, are the result of past natural disturbances, particularly floods and wildfire, and past land management activities such as grazing, mining, timber harvest, irrigation diversion, and road construction. These disturbances have also occurred in private and state lands, including private inholdings that create checkerboard patterns of landownership within national forest boundaries.

This diversity of land uses interacting with natural disturbance regimes have contributed over time to degradation of stream channels and downcut floodplains, resulting in less water storage and release as base flows during the summer season. Water storage behind impassible dams, water withdrawals, and irrigation diversions for downstream use also contribute to reduced base flows and can result in high summer water temperatures that create thermal barriers to fish movement in lower river corridors. Dams and irrigation diversion structures reduce habitat and population connectivity wherever they occur, by partially or totally blocking fish passage, depending on the design and use of the individual structures. Fish habitats are naturally fragmented in the Oregon Closed Basins but are further fragmented by urban development and water withdrawals.

Many km of fish habitat for one or more species within national forest lands are blocked, or seasonally blocked, by culverts under roads. An inventory conducted in national forest lands in 2000 and 2001 revealed numerous passage concerns created by culverts, particularly in the (1) Upper and Lower Grande Ronde subbasins and the upper Imnaha subbasin in the Lower Snake Basin, (2) North and Middle Forks John Day subbasins in the Middle Columbia River Basin, (3) Upper Malheur subbasin in the Middle Snake River Basin, and (4) Silvies River subbasin in the Oregon Closed Basins. Most of these passage barriers can impede passage at base flow for juveniles of the various species and some impede adult passage for one or more species like bull trout or steelhead/rainbow in headwater streams. Passage barriers generally do not exist in the larger tributaries and rivers that provide spawning and rearing habitat for Chinook salmon.

Although many effects from past and current management will likely continue for the foreseeable future, restoration efforts have been ongoing for over 20 years in the Blue Mountains, achieved in part by reducing land management effects under direction in national forest land management plans, as amended by PACFISH and INFISH (USDA-FS 1995, USDA-FS and USDI-BLM 1995). Current Forest Service watershed restoration programs emphasize whole-watershed restoration and strategic investment in accelerated watershed and habitat restoration are focused on the Blue Mountains to provide better protections from further effects of land-use activities. Fish habitat trend analyses indicate improvements are occurring in the condition of aquatic and riparian habitats in national forest lands throughout the Blue Mountains (Archer et al. 2009, 2012).

Analysis of Projected Climate Change Effects

To assess the potential effects of climate change on stream environments in the Blue Mountains, we used national geospatial data products to delineate the stream network. Streamflow and temperature values from high-resolution stream climate models were then linked to reaches in that network to create a geospatial database that could be queried to summarize climate effects at different time periods.

Stream Network and Hydrology Model

To delineate a stream network for this assessment, geospatial data for the NHDPlus 1:100,000-scale national stream hydrography layer (Cooter et al. 2010) were downloaded from the Horizons Systems website (http://www.horizon-systems.com/NHDPlus/index.php) and clipped to the major watershed boundaries associated with the Blue Mountains. The network was filtered to exclude reaches with minimum summer flows less than 0.034 m^3s^{-1}, which approximates a low-flow wetted width of 1.5 m (based on an empirical relationship developed in Peterson et al. [2013]) in which fish occurrence is rare. For purposes of this assessment, the summer flow period was defined as beginning with the recession of the spring flood to September 30, and is considered to be a critical period for many fish populations because it coincides with maximum temperatures (Arismendi et al. 2013).

Summer flow values predicted by the VIC model (Hamlet et al. 2007, Wenger et al. 2010) were downloaded from the Western United States Flow Metrics website (http://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml) and linked to each reach in the hydrography layer through the COMID field. The VIC model is a distributed, physically-based model that balances water and energy fluxes at the land surface and takes into account soil moisture, infiltration, runoff, and base flow processes within vegetation classes (Liang et al. 1994). It has been widely used in the western United States to study past and potential future changes to water flow regimes (Hamlet et al. 2007, Hamlet et al. 2013), snowpacks (Hamlet et al. 2007), and droughts (Luo and Wood 2007).

Application of the minimum summer flow criteria reduced the original set of blue-lines in the NHDPlus hydrography layer for the Blue Mountains region to 20,123 stream km, of which 6,907 km were on U.S. Forest Service lands (fig. 5.2). In addition to summer flows, earlier validation work (Wenger et al. 2010) suggests the VIC model accurately predicts several other flow metrics relevant to fish: center of flow mass (date at which 50 percent of annual flow has occurred), winter 95 percent flow (number of days from December 1 to February 28 when flows are among highest 5 percent of year), and mean annual flow (Wenger et al. 2010).

Climate Scenarios

To assess stream responses to climate change, the VIC model was forced by an ensemble of 10 GCMs that best represented historical trends in air temperatures and precipitation for the northwestern United States during the 20th century (Mote and Salathé 2010, Hamlet et al. 2013). We considered changes associated with the A1B emission scenario (moderate emissions as defined by the Intergovernmental Panel on Climate Change) and summarized flow characteristics during a historical baseline period (1970-1999, hereafter the 1980s) and two future periods (2030-2059 hereafter 2040s; 2070-2099 hereafter 2080s). Within the Blue Mountain region, summer air temperatures were projected to increase 3.3 ºC by the 2040s and 5.4 ºC by the 2080s, with smaller increases during other seasons. Summer precipitation was projected to decrease 11 percent by the 2040s and 14 percent by the 2080s, but slight increases during other seasons meant total annual precipitation was projected to change less than 5 percent (Hamlet et al. 2013, Mote and Salathe 2010).

Most GCM projections are relatively consistent until the mid-21st century and diverge primarily in late century due to uncertainties about future greenhouse gas emissions (Cox and Stephensen 2007, Stocker et al. 2013). The climatic conditions associated with the A1B trajectory and historical period bracket that range of possibilities. Given uncertainties about the magnitude and timing of changes, it is reasonable to interpret future projections as a moderate change scenario (2040s) and an extreme change scenario (2080s) relative to the baseline period (1980s).

Stream Temperature Model and Scenarios

To complement the streamflow scenarios, geospatial data for August mean stream temperatures were downloaded for the same A1B trajectory and climate periods described above from the NorWeST website (http://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html). NorWeST scenarios are developed by applying spatial statistical models for data on stream networks to a crowd-sourced database contributed by more than 70 resource agencies (Isaak et al. 2011, Isaak et al. 2014). NorWeST scenarios account for differential sensitivity of streams to climate forcing through application of basin-specific parameters (Isaak et al. 2011, Luce et al. 2014b). NorWeST scenarios are available at a 1-km resolution and were modeled in the study area from more than 9,000 summers of measurement with thermographs at more than 3,000 unique stream sites monitored intermittently with digital sensors from 1993 to 2011. The density and spatial extent of the temperature dataset, combined with the predictive accuracy of the NorWeST model across those sites (r^2 = 0.94; RMSE = 0.90 °C) overcame the commonly cited weakness of coarse resolution in climate vulnerability assessments (Potter et al. 2013, Wiens and Bachelet 2010). The model also performed well over the wide range of climatic variation that occurred during the calibration period (interannual variation in August air temperatures of 5.0 °C and three-fold variation in August flows), which is notable because the warmest and driest years exceeded mean conditions projected to occur by the 2040s.

Climate Change Effects on Fish and Fish Habitat

Streamflows – General Patterns

The broad elevation range across the Blue Mountains translates to significant spatial heterogeneity in stream hydrology. Streams in low-elevation catchments have rain-dominated hydrographs with peak flows occurring much earlier in the year than high-elevation streams dominated by snowmelt runoff. Relative to the 1980s baseline period, runoff timing (center of flow mass) of all streams are projected to advance 9 to 23 days earlier in the year, with larger changes anticipated by the 2080s and at higher elevations (table 5.1, fig. 5.3). Similar patterns are projected with regards to larger changes late in the 21st century, and on national forests with respect to number of winter high flow days (fig. 5.4) and declines in average summer flows.

If summer flows decline by the projected 15 to 41 percent (table 5.1), the linear extent of the network supporting fish (flows greater than 0.034 m^3s^{-1}) could decrease by 5 to 12 percent, as the smallest headwater streams become more frequently intermittent (fig. 5.2). Summer flow reductions are predicted to be most prominent in the highest elevation watersheds like the Eagle Cap Wilderness where stream hydrologies are most dependent on winter snow accumulation. The direction of projected trends in stream hydrologic attributes is similar to that observed during the last 50 years of the 20th century across the Pacific Northwest (Isaak et al. 2012a, Luce et al. 2009, Safeeq et al. 2013, Sawaske and Freyberg 2014), but future changes and associated rates of change are expected to be larger.

Winter high flow frequency—

As air temperatures increase, the rain-on-snow (ROS) zone will move up in elevation, increasing stream flooding where the current ROS zone is lower in a subbasin. In contrast, subbasins with an already higher-elevation ROS zone will see only modest increased flooding risks (Hamlet and Lettenmaier 2007, Tohver et al. 2014). As ROS zones move higher over time, the zones themselves will shrink in size, reducing the potential contribution to peak winter runoff in some subbasins. The probability of ROS events occurring is also expected to decrease with warmer temperatures because of decreased snow occurrence and length of time that snow is on the ground (McCabe et al. 2007), especially in lower-elevation subbasins in the northern Blue Mountains.

Most high elevation areas under the baseline conditions (1980s) have 1 to 6 days (mean 4.6 days) of winter high flows, with the exception of the Wallowa Mountains in the Eagle Cap Wilderness which have less than a day (fig. 5.3). The highest elevations in the analysis area occur within the Wallowa Mountains at 3000 m. Projections indicate that risks from mid-winter peak flows triggered by ROS events increase moderately by 2040 (2.1 days) and accelerate in

2080 (4 days) for national forest lands under the A1B emission scenario (table 5.1). By 2080, all areas, with the exception of the highest terrain in the Wallowa Mountains, almost double the number of days (mean 8.6 days) with the highest 5 percent flows (fig. 5.4).

Increased frequency and severity of flood flows during winter can affect overwintering juvenile fish and eggs incubating in the streambed. Eggs of fall spawning spring Chinook salmon and bull trout may suffer higher levels of mortality when exposed to increased flood flows (Jager et al. 1997). Scouring of the streambed can dislodge eggs (Schuett-Hames et al. 2000), and elevated sediment transport caused by high flow can increase sediment deposition in redds, suffocating eggs (Peterson and Quinn 1996). Potential effects to fish from altered winter peak flows are likely to vary by species and strength of each population. Eggs from smaller fall spawning fish (e.g. bull trout) are likely to be at higher risk from winter channel scour events than larger fall spawning fish (e.g., spring Chinook salmon) because of shallower egg depths (Bjornn and Reiser 1991, Montgomery et al. 1999, Shellberg et al. 2010). Smaller fish also use smaller gravels that are more easily scoured than gravels used by larger spawning fish.

Despite similar incubation timing, bull trout had a higher risk of scour than spring Chinook salmon, which bury their eggs deeper (40 cm vs. 10-20 cm for bull trout) in the Middle Fork of the Salmon River (Goode et al. 2013). The risk of scour was also greater for resident bull trout (10 cm egg burial depth) than for larger, migratory forms (20 cm egg burial depth) for identical spawning locations and timing. Spring Chinook salmon had lower risk of channel scour associated with climate change because of their spawning preference for unconfined alluvial valleys that are somewhat buffered from scour caused by increased winter flows (Goode, et. al. 2013). Spring Chinook salmon redds also have lower risk because they are generally constructed from larger substrates that would be mobilized only during large flood events.

Winter floods may increase risks to fry that are vulnerable to displacement during the first month after emergence (Fausch et al. 2001, Nehring and Anderson 1993) or to juveniles with poor swimming ability in high-velocity water (Crisp and Hurley 1991, Heggenes and Traaen 1988). The retreat of snow level to higher elevations may lead to earlier fry emergence for some populations (Healey 2006). Individual populations have spawning times and egg development rates matched to the long-term environmental conditions of their spawning stream (Beacham and Murray 1990, Tallman 1986). Earlier emergence may expose the fry to increased mortality because of a lack of food or increased predation (Brannon 1987, Tallman and Healey 1994).

The above potential effects are most likely to occur in years with higher ROS risk. However, they will not occur every year or in every subbasin across the Blue Mountains. Risks of winter peak flow to fish habitat will vary by habitat and subbasin condition, valley confinement, and frequency and intensity of each ROS event. Smaller watersheds with higher road densities may concentrate flows into streams and magnify channel scour. Habitats that offer fewer refugia (e.g., infrequent pools, shallow pools, less woody debris) from high flows may result in higher fish mortality from winter floods.

Risks from winter peak flow will almost double in some Blue Mountains subbasins in the future, although risk will differ at different locations. Because salmonids have evolved within a highly dynamic landscape (Benda et al. 1992, Montgomery 2000), they may have sufficient phenotypic plasticity to buffer environmental changes, assuming that such changes are within the historical range of variability (Waples et al. 2008). However, it is unknown if phenotypic adjustment can keep pace with evolving disturbance frequency induced by contemporary climate change (Crozier et al. 2008, 2011).

Summer flows—

As described previously, spring and early summer flows have been decreasing as a result of earlier snowmelt and runoff over the last 50 years (Safeeq et al. 2013, Stewart et al. 2005). Streamflow magnitude in the Pacific Northwest also declined between 1948 and 2006, including decreased 25th percentile flow (Luce and Holden 2009), which means that the driest 25 percent of years have become drier across the majority of the Pacific Northwest. Overall, trends in the Blue Mountains suggest summer flows decreased 21-28 percent in the period from 1949 to 2010 (Safeeq et al. 2013). This trend is expected to continue because large portions of the Blue Mountains could lose all or significant portions of April 1 snow water equivalent (SWE) in future periods because of reduced snow accumulation and increased ROS events that reduce snowpack SWE prior to April 1. Snowpack sensitivity differs with elevation, and risk of loss is higher where there is snowpack currently persists in higher-elevation subbasins (see chapter 3).

Effects on fish and their habitats from changes in summer low flows will differ by the intensity and frequency of drought and early season runoff, as a function of geology, drainage elevation, and fish species across the Blue Mountains. Streams more dependent on snowmelt with minimal groundwater contribution will be affected more than streams sustained by groundwater. However, even these groundwater streams will have lower base flows in sustained droughts.

Fish populations most affected by this change will likely be in headwater areas inhabited by steelhead, redband trout, and bull trout. However, all fish populations will be stressed as streamflows decrease over the summer. Increased intensity of extreme low flows reduces the probability of survival in rearing juveniles (May and Lee 2004), with evidence that summer and early autumn low flows cause greater impacts (Crozier and Zabel 2006, Harvey et al. 2006). In some stream reaches, riffles will become shallower and perhaps intermittent (Sando and Blasch 2015). This may result in disconnected stream reaches, isolated pools, overcrowding of fish, increased competition for food and cover, and greater vulnerability to predators in remaining deep water habitat.

In years of extreme drought, native trout and salmon populations may retreat to shrinking coldwater refuges in drainages where unimpeded access allows fish to avoid warmer downstream conditions. Changes in low flow may reduce the likelihood of successful adult migration (Rand et al. 2006, Zeug et al. 2011) as adult fish return from the ocean and have difficulty negotiating waterfalls and other barriers.

Steelhead have high phenotypic plasticity and may shift the timing of a life stage transition to reduce probability of exposure to changes in stream temperature and flow. However, steelhead are limited in their ability to shift the timing of their life stages. Because changes in both temperature and flow initiate transitions among salmon life stages (Bjornn and Reiser 1991, Quinn and Adams 1996), the likelihoods of a temporal disparity between life stages and a decrease in probability of persistence increase when temperature and flow are beyond some critical threshold (Reed et al. 2010, Schlaepfer et al. 2002).

Stream Temperatures – General Patterns

Considerable thermal heterogeneity exists across Blue Mountain streams due to the complex topography and range of elevations within the area (table 5.2, fig. 5.5). August stream temperatures in the 1980s baseline period averaged 15.5 °C and ranged from 4.7 to 23.5 °C. Temperatures of streams flowing through higher-elevation national forest lands were cooler, averaging 12.4 °C. Summer temperatures are projected to increase across the Blue Mountains by an average of 0.9 °C in the 2040s and 1.9 °C in the 2080s. Larger than average increases are projected to occur in the warmest streams at low elevations and smaller than average increases are projected for the coldest streams. This differential warming occurs because cold streams are usually more buffered by local groundwater contributions than are warm streams (Luce et al. 2014a, Mayer 2012). Those projected temperature increases are smaller than the 2 to 4 °C increases projected for Pacific Northwest streams for the same A1B trajectory (Beechie et al. 2012, Mantua et al. 2010, Wu et al. 2012). But previous studies modeled short-term weekly maxima that often changed more rapidly than mean temperature conditions (Meehl and Tebaldi 2004).

In all cases, projections imply faster rates of stream temperature warming than has been observed in recent decades. Most streams in the Pacific Northwest have been warming 0.1 to 0.2 ºC per decade (Isaak et al. 2012a,b), which is slower than the rates of air temperature increases. Stream warming rates differ by season, with rates usually highest in summer and lower during winter and spring (for which a cooling trend has occurred in recent decades; Isaak et al. 2012a). Regional rates of stream warming have been dampened in recent years by a cool phase of the Pacific Decadal Oscillation (PDO; Mote et al. 2003; see http://cses.washington.edu/cig/pnwc/aboutpdo.shtml), a phenomenon that will not persist indefinitely as the PDO continues to cycle.

Stream temperature increases will almost certainly continue, but with some uncertainty about site-specific rates. Fish species in Blue Mountain streams will respond by adjusting their spatial distributions and phenologies to the evolving thermal environment. Adjustments in spatial distributions will be most pronounced near distributional boundaries that are currently mediated by temperature. Range contractions and habitat losses near warm downstream boundaries are often the focus of climate vulnerability assessments, but upstream boundaries controlled by cold temperatures may be equally relevant for some species. Colonization of new habitats further upstream as warming progresses could offset a portion of downstream habitat losses for some species and populations (Isaak et al. 2010). Populations may also adapt phenologically by using habitats at different times of the year to avoid stressful conditions.

Evidence exists that migration dates of some salmon species have been advancing in recent decades (Crozier et al. 2008, Keefer et al. 2008, Petersen and Kitchell 2001) and that these trends are related to warmer temperatures (Crozier et al. 2011). However, phenological adaptations involve tradeoffs elsewhere in the life cycle. In the case of salmon, earlier migrants have spent less time in the ocean and have smaller body sizes (Rand et al. 2006), and these fish arrive in spawning areas sooner where they may be more susceptible to thermal events and predation risks. Documentation of phenological shifts for fish species and life stages beyond adult migrations are limited, but the strong temperature dependencies of physiological and metabolic processes usually translate to earlier timing of life history events in most species (Parmesan and Yohe 2003, Root et al. 2004).

Spring Chinook salmon—

Populations of spring Chinook salmon in the Blue Mountains will be sensitive to future temperature increases, because many of their primary spawning and rearing streams occur at low elevation and are relatively warm for this species. Moreover, adults in these populations already exhibit heat-related stress symptoms, including clustering in cold water refugia during warm periods (Torgersen et al. 1999, 2012) and occasional thermally-induced mortality events in the lower river migration corridors in the John Day River system during upriver spawning migrations. The large body size of salmon and preference for spawning in unconfined valleys with gravel substrates (Isaak et al. 2007) may preclude their colonization of new habitats upstream from historical ones, so these fish are expected to lose habitat as warming continues.

To estimate potential habitat losses for spring Chinook salmon, a 19 ºC temperature threshold was applied to the stream temperature scenarios described previously (table 5.2; fig. 5.5) and these were clipped to match the upstream distribution of spring Chinook salmon in the Blue Mountains. This temperature was chosen because streams with higher temperatures are susceptible to invasion by non-native smallmouth bass (*Micropterus dolomieu* Lacepède) that predate heavily on juvenile Chinook salmon (Lawrence et al. 2014). Chinook salmon populations in streams warmer than 19 ºC also exhibit consistently higher pre-spawn mortality rates than cooler streams (Bowerman et al., unpublished data; Keefer et al. 2010). Based on that criterion, it was estimated that 1,921 km of salmon habitat existed during the 1980s baseline across the Blue Mountains region, 854 km of which were on national forest lands (table 5.3, fig. 5.6). In the future, 24 to 38 percent of streams across the analysis area are projected to exceed 19 ºC, but only 10 to 21 percent reductions are expected on national forest lands because streams are colder and farther from the temperature threshold. An earlier estimate of Chinook salmon habitat loss in a subset of the Blue Mountains, the John Day River basin, predicted larger reductions (30 to 75 percent decreases for similar time periods; Reusch et al. 2012), but also showed a similar pattern of larger habitat losses in warmer downstream areas.

Bull trout—

Populations of bull trout in the Blue Mountains will be sensitive to future temperature increases because this species requires streams with very cold temperatures (Dunham et al. 2003, Mesa et al. 2013, Selong et al. 2001). Although adult bull trout sometimes use main-stem rivers at times of year when temperatures may be relatively warm (Monnot et al. 2008, Howell et al. 2010), spawning and juvenile rearing during the first few years of life occur exclusively in the coldest streams (Isaak et al. 2010, Rieman and McIntyre 1995). As a result, existing populations of bull trout in the Blue Mountains are constrained and heavily fragmented among a few small headwater networks. The upstream extent of bull trout in those networks is further limited by stream slope and small flow volumes, so colonization of new upstream habitats in response to climate change is not possible.

A variety of temperature thresholds have been used in bull trout climate vulnerability assessments (Isaak et al. 2010, Rieman et al. 2007, Ruesch et al. 2012, Wenger et al. 2011a,b), but most focus on temperature criteria associated with spawning and rearing habitats because these are critical to bull trout population persistence. An 11.0 ºC temperature criterion was chosen for this assessment because cross-referencing of extensive fish survey databases (Rieman et al. 2007, Wenger et al. 2011a,b) with stream temperature measurements suggest more than 90 percent of juvenile bull trout (defined as fish less than 150 mm in length) occur in streams at least this cold (Isaak et al. 2010, 2014, 2015). Moreover, most juveniles found in reaches more than 11 ºC probably originated in colder, upstream areas and have subsequently moved downstream.

Based on that criterion, it was estimated that 1,953 km of bull trout habitat existed during the baseline 1980s conditions across the Blue Mountains, 94 percent of which occurred on national forest lands (table 5.4, fig. 5.7). Future projections suggest that 38 percent and 58 percent of those streams will exceed 11 ºC by mid- and late-21st century, respectively. Ruesch et al. (2012) made a comparable estimate of bull trout habitat loss over similar time periods in the John Day River system that predicted larger losses (66 to 100 percent decreases), but projected a qualitatively similar pattern of decline as temperatures increase in the future. None of the available estimates accounts for reductions in habitat volume that would be caused by ongoing decreases in summer flows. Neither do those estimates account for potential negative synergies between reductions in habitat size and increased environmental disturbances such as wildfires, debris flows, winter high flows, and drought (Dunham et al. 2007, Hamlet and Lettenmaier 2007, Littell et al. 2010, Luce and Holden 2007).

Steelhead trout/redband trout—

Populations of steelhead and redband trout will be affected by future temperature increases in the Blue Mountains, but probably less negatively than bull trout and spring Chinook salmon. Temperature and food availability play important roles in mediating the relative proportions of the two life history forms, but both have relatively warm thermal niches (Richter and Kolmes 1995, Rodnick et al. 2004, Zoellick 1999) and upstream distributions that are limited by cold temperatures in many streams (Isaak et al. 2015). The latter provides some flexibility as warming proceeds because the thermal suitability of upstream habitats will improve and may partially compensate for losses where streams become too warm. Several temperature categories were used to represent the breadth of the thermal niche for steelhead and redband trout, with temperatures less than 9 ºC considered unsuitably cold, 13 to 20 ºC optimal, and greater than 25 ºC unsuitably warm.

Summaries of future stream lengths in those categories indicate that upstream habitat gains could occur in many streams that were too cold for this species historically, and a small net increase in the length of thermally suitable habitat may occur (tables 5.5, 5.6; figs. 5.8, 5.9). That result contrasts with a loss of 10 to 43 percent of habitat in the John Day system projected by Ruesch et al. (2012), but that estimate was based on the assumption that unsuitably cold habitats were not upstream of current steelhead and redband trout in some streams. In a regional climate vulnerability analysis, Wade et al. (2013) indicated that steelhead migrating through the lower Columbia River and many large rivers and tributaries will be negatively affected by future temperature increases. However, the spatial resolution of the model was limited to the largest rivers, so their analysis could not assess effects throughout the full network of streams draining the Blue Mountains. Taken in combination, these studies suggest that although there may be some thermal effects on steelhead and redband trout (Ebersole et al. 2001, Torgersen et al. 2012), the species have some flexibility to adapt to temperature increases through different life histories, phenological adjustments, and distribution shifts.

Climate Cycles and Ocean Effects on Fisheries

The biology and legal status of anadromous steelhead and spring Chinook salmon in the Middle Columbia River and Snake River basins demonstrate that anadromous fish populations are heavily influenced by many factors outside and downstream of national forest-administered lands. For example, population dynamics and abundance of spring Chinook salmon and steelhead are strongly affected by conditions in the ocean environment (Mantua et al. 1997). The productivity of that environment for salmon growth and survival varies through time in response to sea surface temperatures and strength of coastal upwelling tied to regional climate cycles like the El Niño Southern Oscillation (ENSO; 5 to 7 year periods) and PDO (20 to 40 year periods). Although ocean productivity and climate cycles most strongly affect anadromous fishes, these cycles are also relevant to resident species like bull trout and redband trout because of inland effects on temperature, precipitation, and hydrologic regimes that alter the quality and quantity of freshwater habitat (Kiffney et al. 2002). In the Pacific Northwest, cool (wet) phases of ENSO and PDO are more beneficial to fish populations than are warm (dry) phases (Copeland and Meyer 2011, Mote et al. 2003).

Research summarized in the recent Intergovernmental Panel on Climate Change report (Stocker et al. 2013) provides little evidence to support concerns about climate change affecting the periodicity or magnitude of ENSO or PDO, either in the historical record, or in future climate projections. Therefore, we can expect climate cycles to periodically dampen or exacerbate the effects of climate change on fish populations and stream habitats. A cool PDO phase began in approximately 2000 and has largely persisted through 2014 (see http://www.jisao.washington.edu/pdo), but it is unknown how long this PDO phase will persist and help buffer fish populations against climate change.

Adapting Fish Management to Climate Change in the Blue Mountains

Management strategies and tactics for increasing resilience of fish populations to a warmer climate in streams of the western United States have been well documented in the scientific literature (e.g., Isaak et al. 2012a, ISAB 2007, Luce et al. 2013, Mantua and Raymond 2014, Rieman and Isaak 2010). This documentation and feedback from resource specialists contributed to a summary of climate change adaptation options for the Blue Mountains (table 5.7). Implementation of climate-smart management actions and watershed restoration objectives will benefit from a strategic hierarchical approach to ensure the most important work is occurring in the most important places (Hughes et al. 2014). Stream restoration, which is already well underway in the Blue Mountains and elsewhere, will in turn need to consider future biophysical conditions that will be affected by a warming climate (Beechie et al. 2012).

Responding to Shifts in Timing and Magnitude of Streamflow

Reduced snowpack as a function of increasing temperature has already been documented in the Pacific Northwest, including the Blue Mountains (see chapter 3 and previous discussion in this chapter). As this continues, hydrologic regimes will shift towards increasing dominance by rainfall and decreasing dominance by snow at all but the highest elevations, altering the timing and magnitude of streamflows. Specifically, winter peak flows will be higher, and extreme flows will be more frequent than they are now, causing considerable stress for some fish species.

Maintaining the overall integrity and functionality of stream systems will be critical for minimizing the effects of higher winter flows (table 5.7a). This can be accomplished by increasing soil water storage in floodplains and on hillslopes for instream base flows, thus reducing the “flashiness” of storm flows. Decreasing fragmentation of stream networks, currently somewhat disrupted by roads and water diversions, will ensure that aquatic organisms have options for accessing favorable habitats during extreme flows. Better information about streamflow regimes will ensure that climate change adaptation and stream restoration will be conducted effectively and efficiently (Luce et al. 2012, Wigington et al. 2013).

Many management tactics are available to alter the effects of increasing winter streamflows, including managing upland and riparian vegetation, managing roads to reduce accelerated runoff, reconnecting and increasing off-channel habitat in side channels and wetlands, and promoting American beaver (*Castor canadensis* Kuhl) populations and beaver-related overbank flow processes (Pollock et al. 2014). It is also desirable to identify critical downstream areas where leased water rights or short-term water swaps under low-flow conditions would benefit migrating spring Chinook populations subject to thermal stress, and where such actions would help sustain critical headwater areas for bull trout spawning and rearing. These adaptation tactics will be most efficient if they can be coordinated with existing stream management and restoration efforts conducted by the Forest Service and other landowners and stakeholders (Rieman et al. 2015).

Responding to the Effects of Increased Disturbance on Sediment and Debris Flows

Climate change may increase the frequency of extreme events, which in turn would affect streams and aquatic habitat. Increased frequency of wildfire and annual area burned are a near certainty in the Blue Mountains region (see chapter 6). Increased area burned, especially if it includes increased fire intensity will cause increased sediment removal from hillslope locations and higher episodic and chronic delivery of sediment to stream channels. Depending on the timing and magnitude of sediment delivery, some life-history stages of anadromous fish can be greatly affected. Large debris flows can be especially damaging to aquatic habitat.

Developing wildfire use plans that specifically address postfire effects on streams and aquatic habitat can help reduce disturbance-related sediment input from roads and timber harvest (table 5.7b). Identifying hillslope landslide hazard areas and susceptible roads prior to the occurrence of wildfire and as part of fire planning will be critical to mitigating the effects of erosion.

Restoring and revegetating burned areas, often a component of Burned Area Emergency Rehabilitation programs, can help to store sediment and maintain channel geomorphology following large wildfires. Effective implementation will require thorough pre-fire assessment of geomorphic hazards, especially areas prone to debris flows. Fortunately analytical and decision support tools are readily available to calculate debris flow runout distance and other parameters and to map the location of hazards. Coordination with resource specialists and programs in vegetation, fire, hydrology, geology, and soils will ensure accuracy and efficiency in protecting multiple resources. Prioritizing the tactics needed to protect multiple fish species and populations in the face of increasing disturbance will be a challenging but necessary task for guiding implementation of management actions.

Responding to Increased Stream Temperatures

Increasing stream temperatures, combined with decreasing summer flows, will one of the most obvious effects of climate change for several fish species in the Blue Mountains region, as detailed earlier in this chapter. Increased temperatures are expected to be widespread, especially at lower elevations where some species may already be at the limit of thermal tolerance during the summer. Considerable variation exists in thermal regimes as a function of water sources (Ebersole et al. 2003, 2015; Fullerton et al. 2015), topography, aspect, and other factors, influencing where and how management actions might be implemented to reduce deleterious effects.

Maintaining and restoring natural thermal conditions will be the most effective strategy for buffering future increases in stream temperatures (table 5.7c). It will be especially important to increase connectivity within stream networks, so aquatic organisms can have year-round access to cold water, especially during stressful summer periods. An effective, climate-smart approach to managing for cold-water refugia will be possible only if stream temperature regimes are well understood. Fortunately, the expanding NorWeST stream temperature network, including in the Blue Mountains, is helping to provide a scientific foundation for site-specific adaptation tactics.

Increasing resilience of fish populations to higher stream temperature focuses primarily on maintaining existing cold water refugia and improving the condition of streams that are vulnerable to increasing air temperature. Riparian function and hydrologic processes can be maintained by restoring riparian vegetation to ensure that exposure of stream channels to solar radiation is minimized. Increasing floodplain connectivity, diversity, and water storage will improve hyporheic and base flow conditions. Reducing damage by livestock grazing on riparian vegetation and stream banks can result in rapid improvements, but may also face opposition from those who access national forests for grazing. All of these adaptation tactics will be more effective if informed by stream temperature data collection and long-term monitoring

Responding to Changes in Headwater and Intermittent Streams

Headwater and other intermittent streams and water bodies (e.g., springs, ponds) are vital for local and seasonal aquatic habitat. Although they may be transient, these water sources influence stream temperature and downstream water quantity and quality (Ebersole et al. 2015). Intermittent streams will be especially vulnerable to the effects of increasing wildfire on sediment pulses and altered flood patterns and magnitudes.

As noted above, wildfire use plans will be needed to proactively reduce the effects of increasing fire disturbance and to identify hillslope landslide hazard areas and susceptible roads (table 5.7d). Restoring and revegetating burned areas can help to store sediment and maintain channel geomorphology following large wildfires, and should be informed by pre-fire assessment of geomorphic hazards, especially areas prone to debris flows. Coordination with resource specialists and programs in vegetation, fire, hydrology, geology, and soils will help ensure that multiple resources will be protected.

Conclusions

Adapting fisheries to the environmental trends associated with climate change will require a diverse portfolio comprised of many strategies and tactics as described above (table 5.7). Equally important is understanding a new concept of dynamic disequilibrium in which stream habitats will become more variable, undergo gradual shifts through time, and sometimes decline in quality. Some fish species and populations will retain enough flexibility to adapt and track their habitats (Eliason et al. 2011), but others may be overwhelmed by future changes. It may not be possible to preserve all populations of all fish species across the Blue Mountains. However, as better information continues to be developed, resource managers will be able to identify where resource commitments are best made to enhance the resilience of fish and fish habitat. As many species and populations adjust their phenologies and distributions to track climate change, Forest Service lands will play an increasingly important role in providing aquatic habitats.

Three factors will be especially important for improving and maintaining resilience of fish species and aquatic systems to climate change. First, it will be critical to restore and maintain natural thermal regimes to minimize increases in summer stream temperatures and effects on cold-water species. As noted above, many techniques can be used to promote stream shading and narrow unnaturally widened channels. Second, a strategic approach is critical for climate-smart fisheries management and restoration during the current period of declining budgets and increasing stresses. For example, it is important to identify high-priority culvert barriers to fish movements, especially in areas with the potential to provide high-quality refugia for cold-water species in the future. These types of tactics need to be considered at both the reach and watershed scales. Finally, long-term monitoring is the only means by which the effectiveness of climate-smart resource management can be determined, thus reducing uncertainties and informing the broader mission of sustainable fisheries management. More and higher quality data are needed for streamflow (more sites), stream temperature (annual data from sensors maintained over many years), and fish distributions. These data will improve status and trend descriptions, while also contributing to better models that more accurately predict responses to climatic change and land management.

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Chapter 6: Effects of Climatic Variability and Change on Upland Vegetation in the Blue Mountains

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Introduction

The Blue Mountains ecoregion (BME) extends from the Ochoco Mountains in central Oregon to Hells Canyon of the Snake River in extreme northeastern Oregon and adjacent Idaho, and then north to the deeply carved canyons and basalt rimrock of southeastern Washington (see fig. 1.1 in chapter 1). The BME consists of a series of mountain ranges occurring in a southwest to northeast orientation, allowing the BME to function ecologically and floristically as a transverse bridge between the Cascade Mountains province to the west, and the main portion of the middle Rocky Mountains province to the east.

Powell (2012) defines six vegetation zones within the BME, which range from low elevation grasslands to high elevation alpine areas, and Johnson (2004), Johnson and Clausnitzer (1995), Johnson and Simon (1987), and Johnson and Swanson (2005) describe upland plant associations. The lowest elevation plains zone contains grasslands and shrublands because moisture is too low to support forests except along waterways. The foothills zone is usually dominated by western juniper (*Juniperus occidentalis* Hook.), often with a mixture of curl-leaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.) and antelope bitterbrush (*Purshia tridentata* [Pursh] DC.) shrublands. In the northern Blue Mountains, the foothills zone generally supports tall shrublands (with, for example, western serviceberry [*Amelanchier alnifolia* [Nutt.] Nutt. ex M. Roem.], black hawthorn [*Crataegus douglasii* Lindl.], and western chokecherry [*Prunus virginiana* var. *demissa* [Nutt.] Torr.]), rather than western juniper, curl-leaf mountain-mahogany, and antelope bitterbrush. Located just above the western juniper woodlands is the lower montane zone, which contains dry conifer forests characterized by ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.). Although some of these forests are mixed conifer, many of the warm, dry forests are composed almost exclusively of ponderosa pine.

Warm, dry forests tend to be the most common forest zone in the Blue Mountains, and because they occur at lower and moderate elevations, they have a long history of human use both

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for commodity purposes (e.g., livestock grazing) and as an area where effective fire exclusion occurred and led to well-documented changes in species composition, forest structure, and stand density. The upper montane zone includes moist forests characterized by Douglas-fir, grand fir, and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.). The upper montane zone may be very narrow or nonexistent in the southern portion of the Blue Mountains. High elevations support a subalpine zone with Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), subalpine fir, and whitebark pine (*Pinus albicaulis* Engelm.) or an alpine zone near mountain summits where trees are absent. Subalpine and alpine environments are mostly confined to the high ridges of the Strawberry Mountains, the Elkhorn Crest, the Wallowa Mountains and the Seven Devils in eastern Oregon and western Idaho.

This chapter assesses potential climate change effects on upland vegetation in the BME using information from the literature and output from simulation models. The focus is on the BME in general, but emphasis is placed on the Malheur, Umatilla, and Wallowa-Whitman National Forests (see fig. 1.1 in chapter 1). First, we review information sources that were used as the basis of this assessment. Then we present the vegetation assessment, which is organized around an established vegetation classification system, followed by a brief conclusion.

Understanding Climate Change Effects

Climate change is expected to profoundly alter vegetation structure and composition, terrestrial ecosystem processes, and the delivery of important ecosystem services over the next century. Climate influences the spatial distribution of major vegetation biomes, the abundance of species and communities within biomes, biotic interactions, and the geographic ranges of individual species. Climate influences the rates at which terrestrial ecosystems process water, carbon, and nutrients and deliver ecosystem services like fresh water, food, and biomass. Climate also influences the disturbance processes that shape vegetation structure and composition, and altered disturbance regimes will likely be the most important catalyst for vegetation change (box 6.1). Climate-induced vegetation changes have important implications for wildlife habitat, biodiversity, hydrology, future disturbance regimes, ecosystem services, and the ability of ecosystems to absorb and sequester carbon from the atmosphere.

Several information sources are useful for assessing potential climate change impacts on vegetation and future forest composition and structure, including long-term paleoecological records; evidence from climate, carbon dioxide, and other experimental and observational studies; and model predictions for the future. We use these sources to conduct our assessment of potential effects of climate change on vegetation. When different lines of evidence are in conflict, we often rely on our ecological and local knowledge and derived logical inference to weigh different lines of evidence. For example, we emphasize the importance of interactions between climate change and disturbance, which many agree is likely to be the number one driver of change in the next few decades (box 6.1). However, these disturbances are not included in many of the models that we discuss. In addition, paeloecological evidence or empirical evidence form highly relevant studies is often given great weight than model output.

Paleoecological Records

Paleoecology uses data from some form of preserved plant material, or fossils and subfossils, to reconstruct vegetation of the past. More recent records (e.g., the Holocene or last 11,700 years) are commonly reconstructed by using plant materials such as pollen and phytoliths. For example, pollen grains that are washed or blown into lakes can accumulate in sediments and provide a record of past vegetation. Different types of pollen in lake sediments reflect the vegetation that was present around the lake and, therefore, the climate conditions favorable for that vegetation. Phytoliths are morphologically distinct silica bodies from plants that can exist in soil, sedimentary deposits, or in archaeological material. In addition, much historical vegetation information can be obtained from biological remains in nests and middens (waste accumulation) created by several species of birds and small mammals, and they may be preserved for thousands of years, providing a detailed fossil record of past environmental conditions (Betancourt et al. 1990, Rhode 2001).

Global paleoecological records indicate that during warm periods of the past, tree species tend to move poleward and upward in elevation. A species "leading edge" is fundamentally important under global change, as it is commonly accepted that range expansions depend mostly on populations at the colonization front. The leading edge is also seen as controlled by rare long-dispersal dispersal events followed by exponential population growth (Hampe and Petit, 2005). Unfortunately, existing environmental reconstructions of the Pacific Northwest are based mostly on pollen records obtained from lakes in forested areas west of the Cascade Range, whereas the dry interior of the region has received less attention because fewer of these sites exist. However, important insights can be derived from the existing literature (Blinnikov 2002, Hansen 1943, Mehringer 1997, Whitlock 1992, Whitlock and Bartlein 1997). In particular, as recent glaciation waned during the early Holocene period, a warmer and drier climate than today is inferred (Anderson and Davis 1988). Thus, inferred vegetation during the early- to mid-Holocene (9,000-6,000 yr BP) is used in this assessment as one line of evidence with respect to potential future changes. However, early- and mid-Holocene conditions may be drier than current projections from global climate models.

The rate of future climate change may be faster than any period in the Holocene record (500-1000 years) (Whitlock 1992). Hansen (1943) reports on pollen profiles from a peat deposit located at Mud Lake, one of the Anthony Lakes that lie near the crest of the Blue Mountains. The site is at 2,134 m elevation in a subalpine forest. The author reports on a cursory survey of the area surrounding the peat deposit: 50 percent lodgepole pine (*Pinus contorta* Douglas ex Loudon), 25 percent alpine fir (assumed to be subalpine fir), and 25 percent Engelmann spruce. Two peat cores were removed above a layer of impenetrable ash. Hansen states that they probably represent most, if not all, of the post-Pleistocene (Holocene), but no radiocarbon dates were obtained. The source of the ash layer is unknown. A rapid gain in ponderosa pine pollen pine roughly mid-profile is one of the more important climate indicator trends identified, signifying a warmer and drier climate as the influence of recent glaciation waned. This trend is consistent with early- to mid- Holocene warming. A western larch (*Larix occidentalis* Nutt.) maximum suggests that fire slowed this trend, but as fires decreased, ponderosa pine continued to increase, again suggesting that warmer and drier conditions prevailed. A general decline in whitebark pine from the bottom of the profile to the level of ponderosa pine maximum is substantiating evidence for this climatic trend. Ponderosa pine pollen then declines from its peak to proportions that are generally maintained to the surface, marking the shift to a cooler and wetter climate (compared to the earlier warm and dry period) and present day conditions. Today, ponderosa pine is not within the vicinity of the Anthony Lakes area (it is found at much lower elevations).

Mehringer (1997) reports on a pollen and macrofossil record from Lost Lake, which is located in the Umatilla National Forest on the edge of the Vinegar Hill-Indian Rock Scenic Area about 50 km southeast of Dale, Oregon and only 3 km north of the Malheur National Forest border. At 1,870 m elevation, the surrounding slopes of the lake are within a mixed conifer forests above the elevation of the ponderosa pine zone. Grand fir, Douglas-fir, western larch, and lodgepole pine are common. Engelmann spruce is restricted to cold area drainages and moist areas around the lake. A 3.8-m core located above the Mazama ash tephra layer was examined, thus the record spans the last 7,600 years. The record reveals that lodgepole pine, Douglas-fir and western larch have been the dominant conifers at Lost Lake for the past 7,600 years. However, a transition at about 4,000 yr BP is noted. Relatively more pine, spruce, and fir pollen, and less Douglas-fir and western larch pollen are noted at that point in the record. This change is attributed to an increase in effective moisture, coinciding with a transition from open woodland conditions to closed canopy forests. Although the author does not specifically discuss mid-Holocene warming, this change in vegetation conditions probably relates to the end of mid-Holocene warming and the development of present-day cooler and wetter conditions. This change brought an episode of intense wildfires starting about 3,600 yr BP and concluding around 1,860 yr BP; sagebrush pollen is also a more important component of the pollen profile 9,000-5,600 yr BP.

Whitlock and Bartlein (1997) present a pollen record from Carp Lake, located at 714 m in the eastern Cascades. Presently the site is dominated by ponderosa pine. The Carp Lake record indicates the development of a pine-oak woodland at 9,000 BP. Blinnikov et al. (2002) presents a phytolith record in loess from four different sites in the interior Columbia Basin, two of which were located in the Blue Mountains, using the Carp Lake record and the phylioth sites to infer vegetation in the Columbia Basin during the last 21,000 years (fig. 6.3).

Climate, Carbon Dioxide, and Other Studies

Dendroecological records (tree rings) from the past several hundred years also provide evidence about changes in tree growth with climatic variability. Experimental studies involving altered temperature, water availability, and carbon dioxide concentration provide information about how individual species may respond to climate change. Observational studies regarding response of species to current warming trends also provide clues to potential future responses of vegetation to climate. For example, van Mantgem et al. (2009) note that unmanaged old forests in the western United States showed that background (noncatastrophic) mortality rates have increased in recent decades. Increases occurred across elevations, tree sizes, dominant genera, and past fire histories, and the authors attributed elevated mortality to regional warming and consequent increases in water deficits.

Much of the evidence from climate, carbon dioxide, and other studies cited in this chapter is covered in greater detail in Peterson et al. (2014), which documents that climatic warming during the 20th century has led to a variety of plant responses, including altered phenology and geographic distributions of species (Crimmins et al. 2011, Kullman 2002, Parmesan 2006, Walther et al. 2005, Walther et al. 2012). Results from these types of studies are more appropriately discussed within the relevant vegetation type in the assessment section below.

Model Output

Many vegetation models have the ability to simulate the effects of climate change on vegetation, including altered productivity, distribution shifts, and disturbance regimes. These climate-aware vegetation models can be divided into three classes: species distribution (also called empirical, correlative, statistical, bioclimatic envelope, or niche), process-based (also called mechanistic), and landscape (Littell et al. 2011).

Species distribution models (SDMs) use historical correlations of climate and historical species distributions to develop relationships and then project future suitable habitat under climate change scenarios (Hamann and Wang 2006, Iverson et al. 2008, Kerns et al. 2009, McKenzie et al. 2003a). These models are typically species specific. Model output is not the actual species range or habitat, although it is often presented as such; it is simply the climate and or environmental conditions that are correlated with the species historical presence or abundance. Many types of statistical relationships and approaches are used to make projections about the future (i.e., generalized linear models, machine learning, maximum entropy).

Process models can be species specific, such as forest gap models (Bugmann 2001), or they can simulate groups of species with similar form and function in ecosystems (plant functional types), including dynamic global vegetation models (DGVMs) (Prentice et al. 2007). Although some dismiss this class of model because they do not include species-specific information (Morin and Thuiller 2009), relevant information can be inferred from these models, including regionally specific and biome-scale information from projections based on plant functional types. Moreover, information at multiple taxonomic levels (e.g., species vs. vegetation type) about how climate might affect vegetation change, and the difference between projections and their sources, are themselves useful tools with which to assess uncertainty (Littell et al. 2011). Process models are parameterized from theoretical or experimental, and observational information; they project the response of an individual or a population to environmental conditions by explicitly incorporating biological processes (Bachelet et al. 2001, Bugmann 2001, Campbell et al. 2009, Cramer et al. 2001, Morin and Thuiller 2009). Some operate by using average climate while others use transient climate data. Vegetation or other output by the models is an “emergent property” in relation to key model input information (e.g., climate, soils, carbon dioxide emissions). Some argue that process models may be more robust at projecting the future given their basis in known mechanisms and their theoretical capability to extrapolate into non-analog conditions (Coops and Waring 2011a, Thuiller 2007).

Landscape models expand either of the first two types to explicitly simulate landscape change and ecological processes such as succession and fire. The models are often spatial but may be distributional (Baker 1989, Cushman et al. 2007), such as nonspatial state-and-transition simulation models (Kerns et al. 2012). These models are well-suited to simulations of the effects of disturbance on vegetation, which can be a major limitation for some of the empirical and process models. They can use climate information as inputs, but climate is not always explicitly incorporated into landscape models (Littell et al. 2011).

Despite the specific strengths and weaknesses of different modeling approaches, there are caveats specific to all vegetation models. For example, most of the models do not deal with inertia in vegetation (the tendency of vegetation to remain unchanged through various mechanisms), although some landscape models have the capacity to deal with inertia if they are parameterized to include climate change (Halofsky et al. 2013). Most of the models also do not incorporate disturbance processes, although MC2, a DGVM, does address fire (box 6.1). The inclusion of disturbance and extreme events in most models is still early in their developmental process (Keane et al. 2004, Lenihan et al. 1998, Thonicke et al. 2001).

Most models do not consider biotic interactions (although some process models do consider competition) and phenotypic plasticity (box 6.2), and no models deal with dispersal or genetic adaptation. A new generation of models incorporating climatic variability with many of these important processes is clearly needed. In addition, many models are used without being methodically calibrated and without careful validation of model skill or sensitivity analysis, such that model uncertainty is not understood. For all models, if future conditions differ greatly from conditions used to build and calibrate a model, then accuracy for projecting ecological phenomena will be relatively low (Williams et al. 2007).

Multiple projections using different vegetation models can be used to assess a range of potential changes in climate habitat and implications for changes in species distribution under future climate scenarios. We assess vegetation model projections for the end of 21st century for the BME using different types of vegetation models. Our goal is to summarize and discuss agreement or disagreement in the models regarding general trends relating to potential changes in climate habitat for vegetation, and implications of the changes for future vegetation distribution. Although climate change projections simulated by more than 20 different global climate models (GCMs) have been available in recent years, most of the models we examined used climate data as inputs that were derived from only a small subset of the available GCM output. We tried to limit our assessment to common climate projections (GCM and greenhouse gas emission scenario combination) among the models, although this was not always possible. In addition, we assessed model outputs under only the “business as usual” emission scenarios (A2, A1FI, and RCP 8.5), which significantly depart from other scenarios after the mid-21st century (Nakicenovic et al. 2014, Nakicenovic and Swart 2000).

We selected species-specific model output from websites and peer-reviewed literature supporting the model output (table 6.1). We also used the MC2 model, a new version of MC1 rewritten for improved computational efficiency (Kim and Conklin, n.d.). MC2 simulates response of plant functional types to climate change at a monthly type step, including plant physiology, biogeography, water relations, and interactions with fire. We selected four GCM outputs published by Coupled Model Intercomparison Project Phase 5 (CMIP5) for the RCP8.5 greenhouse gas emissions scenario using impact-relevant sensitivities as a guide (Vano et al., n.d.). The four selected GCM outputs—CSIRO-MK360, HADGEM2-ES, MRI-CGCM3, and NORESM1-M—were selected based on their performance ranking and climate representation. We avoided poorly performing models based on Rupp et al. (2013), then selected models that represented the ensemble or multi-model mean for the hottest, coolest and wettest projections of all 30 available projections, respectively (fig. 6.4). The selected GCM outputs were downscaled to 30 arc-second spatial resolution using the delta method (Fowler et al. 2007) for model input. MC2 was calibrated for the historical period (1895-2009) using a hierarchical approach, using PRISM climate data (Daly et al. 2008) as input (Kim et al., n.d.). We used MODIS net primary productivity data (Zhao and Running 2010), an internally generated potential vegetation type map, and fire return interval data from LANDFIRE (Rollins 2009) as reference datasets for calibration. Future vegetation conditions were simulated for 2010-2100 using the four downscaled GCM data as input; output variables included vegetation type, carbon fluxes and stocks, and fire occurrence and effect.

Table 6.2 lists each species examined (from the species-specific models) and an overall summary of the model output. Projections indicate some loss to complete loss of habitat for most species. Specific model results are discussed in the vegetation assessment section below. The results in table 6.2 are typical for SDMs, showing large reductions in available climate habitat for many species. SDMs often produce projections with dramatic reductions in suitable habitat for a species simply because future novel climates do not correspond to modern conditions under which the species occurs. Climate change is expected to result in substantial areas that have novel climate with no modern analog ("novel climates," Williams and Jackson 2007, Williams et al. 2007). This may be especially true of the western United States where almost half the land area may have novel climatic conditions by the end of the century (Rehfeldt et al. 2006). Furthermore, as models of modern realized niches increase in complexity (more climatic predictors and interactions), the less likely they are to project that species will exist under future climate scenarios.

In some cases, a novel climate could be quite favorable for a species (e.g., Kerns et al. 2009). As a result, novel climates often create a bias in SDM projections toward reduced area for most species under future climate, without identifying which species would replace them. Thus, the projected complete loss of habitat of species may simply illustrate the widespread nature of novel future climate conditions in the BME. Some species have relatively broad ecological amplitudes (e.g., lodgepole pine, juniper; Miller et al. 2005, Miller and Wigand 1994, Pfister and Daubenmire 1975) and may be very competitive in a novel environment. In contrast, outputs from MC2 indicate that although some forest types may decrease in the future, increases in shrublands and grasslands might be expected (fig. 6.5). Agreement among the four simulation outputs is high, especially for the arid portions of the ecoregion (fig. 6.6). The northern portion of the Umatilla National Forest has considerable uncertainty in MC2 projections.

Effects of Climate Change on Vegetation

To structure the vegetation assessment and development of adaptation options, we used the potential natural vegetation (PNV) concept and an existing vegetation classification scheme. Potential natural vegetation is defined as the plant community that would establish under existing environmental conditions in the absence of disturbance and without interference by humans (Chiarucci et al. 2010, Cook 1996). PNV implies that over the course of time, similar types of plant communities will develop on similar sites, thus a potential plant association is an indicator of a particular biophysical conditions or setting of a site.

The PNV concept has been debated in the literature (e.g., Chiarucci et al. 2010, Jackson 2013), and climate change effects on vegetation will probably not be consistent with the PNV concept. Paleoecological evidence suggests that species responses to climate are individualistic and no-analog plant communities should be expected in response to future climate change. Nevertheless, similar biophysical settings that support broad potential vegetation types will still exist in the future and occupy similar habitat types, but it is likely these habitat types will be redistributed across the landscape. For example, it is likely that some form of upland shrublands, grasslands, and dry forest types will exist in the future, although the species that make up these groups could be quite different.

The PNV concept is still useful for discussing changes in future vegetation, and U.S. Forest Service management is largely based on publications, classification systems, and models that rely on PNV. We used Potential Vegetation Groups (PVGs; Powell et al. 2007), a mid-scale hierarchy system typically used for planning and strategic assessments at large spatial scales. These PVGs are similar to the potential plant functional types used for global biome vegetation modeling (e.g., Prentice et al. 1992), and they crosswalk reasonably well to MC2 output and potential vegetation functional types (table 6.3). Although these broad vegetation groups may exist in the future as representations of biophysical settings, the species that comprise them at fine spatial scales may differ greatly.

National forests in the Blue Mountains differ in their distribution of upland PVGs (table 6.4; figs. 6.7–6.9). Malheur National Forest uplands are dominated by the Dry Upland Forest PVG, and includes some Cold Upland Forest and moist forests. Umatilla National Forest uplands are dominated by Moist, Dry and Cold Upland PVGs, a large amount of moist forest, and a considerable amount of dry upland herbland. Wallowa-Whitman National Forest uplands are dominated by Dry, Cold and Moist Upland PVGs, with more cold upland forests than the other two Blue Mountains national forests.

To aid our assessment, we also used Potential Soil Drought Stress maps developed by the U.S. Forest Service Pacific Northwest Region (Ringo et al., n.d.) (box 6.3). The following information, organized by PVG, was evaluated by workshop attendees to create the final assessment worksheets.

Cold Upland Forest

Cold upland forests (UFs) occur at moderate or high elevations in the subalpine zone, characterized by cold, wet winters, and mild, relatively cool and dry summers. Deep, persistent winter snowpacks are common, although the depth and persistence of winter snowpacks has been declining since the 1950s in response to climate change (Furniss et al. 2010, Karl et al. 2009, Stewart et al. 2004). Cold UFs have relatively short growing seasons, low air and soil temperatures, and slow nutrient cycling rates.

Late-seral stands are typically dominated by subalpine fir, grand fir, Engelmann spruce, whitebark pine, and lodgepole pine, but whitebark pine, lodgepole pine, and western larch are often persistent, early-seral species. Whitebark pine will successfully dominate late-seral stands at timberline, on low-moisture soils and southern exposures. Cold UFs are adjoined by a treeless alpine zone at their upper edge (often separated by a narrow zone of dwarf or krummholz trees), and by moist upland forests at their lower edge.

The Cold UF PVG consists of three plant association groups (PAGs), two in the cold temperature regime (Cold Moist and Cold Dry) and one in the cool temperature regime (Cool Dry). The Cold Dry UF PAG is the most widespread member of the Cold UF PVG and includes the more xeric of the high-elevation forested communities. Cold Dry UF plant associations occur on all aspects and many different substrates, often on sites with moderate to high impact from wind scour and, at lower elevation, in cold air pockets and drainages.

Common undergrowth species include herbs and dwarf shrubs. Areas with physiographic and soil characteristics suitable for supporting forests with at least moderate canopy cover are frequently dominated by grouseberry (*Vaccinium scoparium* Leiberg ex Coville). Areas with steeper slopes or shallower soils support open-canopy stands and herb-dominated undergrowth, often featuring elk sedge (*Carex geyeri* Boott), Ross' sedge (*Carex rossii* Boott), and western needlegrass (*Achnatherum occidentale* ssp. *occidentale* [Thurb. ex S. Watson] Barkworth).

Cold UFs at the highest elevations often contain whitebark pine, and these open forest communities often have undergrowth similar to flora in subalpine and alpine meadows, including greenleaf fescue (*Festuca viridula* Vasey), prickly sandwort (*Eremogone aculeata* [S. Watson] Ikonn.), mountain heath (*Phyllodoce empetriformis* [Sm.] D. Don), and poke knotweed (*Aconogonon phytolaccifolium* var. *phytolaccifolium* [Meisn. ex Small] Small ex Rydb.).

Species of concern in this PVG include whitebark pine, limber pine (*Pinus flexilis* E. James), and mountain hemlock (*Tsuga mertensiana* [Bong] Carrière) (box 6.2). These forests are an important component of the landscape from a biodiversity perspective, and for snow retention and hydrologic function.

Potential future changes—

At broad scales, forests of western North America can be partitioned into energy-limited versus water-limited domains (Littell and Peterson 2005, Littell et al. 2008, McKenzie et al. 2003b, Milne et al. 2002). Energy-limiting factors are chiefly light (e.g., productive forests where competition reduces light to most individuals) and temperature (e.g., high-latitude or high-elevation forests). Tree growth in energy-limited forests appears to be responding positively to warming temperatures over the past 100 years (McKenzie et al. 2001). Productivity is projected to increase in subalpine and alpine zones across the Pacific Northwest (Latta et al. 2010).

High-elevation forests may increase productivity in response to moderate warming and elevated atmospheric carbon dioxide. Longer growing seasons, reduced snowpack and depth, and warmer summer temperatures (day and night) associated with future warming may promote increased tree growth within the treeline ecotone and an upward advance of treeline in some locations. However, a recent worldwide study concluded that there was evidence of treeline advance over the last century at only about half of the sites throughout the world with long-term measurements (Harsch et al. 2009). Upward migration of treeline must include the successful recruitment of tree seedlings, first within and then above the current treeline ecotone. This mechanism depends on microsite facilitation (Smith et al. 2003) and may be limited by unsuitable topographic and edaphic conditions of upslope areas, wind exposure and patterns of snow distribution (Holtmeier and Broll 2012, Macias-Fauria and Johnson 2013). Zald et al. (2012) note that the importance of snow, the mediation of snow by interacting and context dependent factors in complex mountain terrain, and the uncertainty of climate change impacts on snow, creates a challenge for understanding how these ecotones may respond to future climate conditions.

A high elevation site in the Blue Mountains that is presently at the transition between cold UF and moist UF and was forested (fig. 6.3), supported an *Agropyron*-dominated grassland/ponderosa pine parkland during early- to mid-Holocene warming. A rapid gain in ponderosa pine in the middle of this record indicates the influence of this species under drier and warmer conditions (Hansen 1943). Paleoecological evidence suggests cold UFs may be converted to herbaceous parklands with ponderosa pine, or the importance of ponderosa pine may increase under warmer and drier scenarios. Douglas-fir woodland types, which are found in the Okanogan valley of southern and central British Columbia and western Montana, may also be a potential analog for what could happen in some areas of the Blue Mountains that are currently dominated by cold forest species.

Species distribution models project that suitable climate available for most cold upland tree species will be moderately reduced to nonexistent in the Blue Mountains by the end of the 21st century (table 6.2). MC2 projects major loss of subalpine forest climate habitat as well (fig.

6.5). Based on model output, cold UFs may be vulnerable to climate change and high-elevation mountains (e.g., Wallowa Mountains, Seven Devils) may serve as refugia for subalpine species. Devine et al. (2012) consider subalpine fir, Engelmann spruce, and western white pine (*Pinus monticola* Douglas ex D. Don) to be highly susceptible to climate change (table 6.2), although lodgepole pine has a lower susceptibility score. Although western white pine also has a high susceptibility score (Devine et al. 2012), its generalist life history (Rehfeldt et al. 1984) may confer phenotypic plasticity, allowing it to better adjust to changing environmental conditions (box 6.2). Another study, which suggests that subalpine species were among the most vulnerable to recent climatic variation, assigned lodgepole pine and western hemlock as the first and second most vulnerable species, respectively, in the Blue Mountains (Waring et al. 2011).

Disturbance—

Historically, large-scale disturbances have been infrequent in the subalpine and alpine zones but can still play an important role in shaping vegetation distribution and abundance. Smaller scale disturbances, such as windthrow, are common in the Cold UF (fig. 6.13). Deep snowpacks allow only a short fire season, fuels are often wet, and spatial discontinuities inhibit fire spread (Agee 1993). However, forest vegetation can be greatly altered by rare wildfire events, and tree establishment following stand-replacing wildfires can require decades to centuries for trees in subalpine forests that do not include lodgepole pine (Agee and Smith 1984, Little et al. 1994). In the future, wildfire events in subalpine systems may be more common as the summer dry period grows longer. Recovery of subalpine forests following wildfire requires nearby seed sources, an extended period of favorable climate, and favorable biotic and abiotic microsite conditions (Bansal et al. 2011, Stueve et al. 2009, Zald et al. 2012). Seral whitebark pine communities could benefit from increased fire occurrence but depend on Clark's nutcracker (*Nucifraga columbiana* (A. Wilson, 1811)) for dispersal (Arno and Hoff 1990, Barringer et al. 2012, Keane et al. 2012). However, even with warmer temperatures, most cold UFs would continue to support high-severity and mixed-severity fire regimes.

A warmer climate could increase the potential for insect and disease outbreaks (box 6.2). Insects in cold UFs include non-native balsam woolly adelgid (*Adelges piceae* (Ratzeburg)) in subalpine fir, spruce beetle (*Dendroctonus rufipennis* [Kirby, 1837]) in Engelmann spruce, mountain pine beetle in lodgepole pine, and larch dwarf mistletoe (*Arceuthobium campylopodum* Engelm.) in western larch. Whitebark pine populations are currently declining due to introduced white pine blister rust (*Cronartium ribicola* A. Dietr.) and mountain pine beetles. The status of limber pine, another species susceptible to white pine blister rust, is unknown. The balsam woolly adelgid is increasing in the Blue Mountains and may be a major stressor of subalpine fir in future decades.

Synthesis—

Based on multiple lines of evidence from modeling and autecological assessment, climate change is likely to produce significant changes in cold UFs over time, including altered growth, altered phenology, and establishment and persistence of trees in current meadow communities. However, results from experimental and observational studies are not as clear and even suggest potential contrary responses. Cold UFs may be converted to high-elevation herbaceous parklands or woodlands with ponderosa pine or Douglas-fir under warmer and drier scenarios. Remnant populations may persist in the highest of elevations within the Blue Mountains (e.g., Wallowa Mountains). The cool dry PAG may be more tolerant of future warming compared to the cold dry and cold moist PAGs. However, increased wildfire may constrain tree reestablishment in these slow-growing systems, particularly for sites without serotinous lodgepole pine as a common, pre-fire component. Increased insect and disease activity with climate change may also increase stress and mortality in these forests.

Cold Upland Shrub

Cold upland shrublands (US; fig. 6.14) occur at moderate to high elevations in climate conditions similar to cold upland forests. This PVG is not common in the Blue Mountains (table 6.4). The principal species characterizing these shrublands form associations that range from xeric to mesic, although this system may be associated with exposed sites, rocky substrates, cold air drainages, and dry conditions that limit tree growth. This PVG includes cold very moist, cold moist, cool dry, and cool moist US PAGs. Typical species include Sitka alder (*Alnus viridis* ssp. *sinuata* [Regel] Á. Löve & D. Löve), mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* [Rydb.] Beetle), and shrubby cinquefoil (*Dasiphora fruticosa* [L.] Rydb.). However the habitat conditions for these species are quite different. Cold moist shrublands are common in subalpine and alpine systems in the Wallowa Mountains at about 2,200 m elevation. Cold very moist shrublands consist of Sitka alder in snow slides in the northern portion of the Blue Mountains. The cooler US PAGs are more common in the montane zone.

Potential future changes—
Species distribution model results for sagebrush indicate complete loss of habitat for this species in the future, although only one model was available for assessment. MC2 model results project a major loss of subalpine forest climate habitat, and an analogous loss of cold US habitat might be inferred from this result (fig. 6.5). In contrast, the Biome-BGC model projects increased net productivity in eastern Oregon shrublands after 2030, including areas currently occupied by sagebrush (Reeves et al. 2014). It should be noted that paleoecological studies generally suggest an increase in sagebrush during warmer conditions in the early to mid Holocene (fig. 6.3), providing an important context for interpretation of modeling studies.

Disturbance—
Large-scale disturbances are infrequent in subalpine and alpine systems, but can still play an important role in shaping vegetation distribution and abundance. Deep snowpacks produce a short fire season, fuels are often wet, and spatial discontinuities can inhibit fire spread (Agee 1993). In the future, wildfire events in subalpine systems may be more common, and shrub species may be able to more rapidly regenerate compared to subalpine tree species. Sitka alder and shrubby cinquefoil can survive low to moderate intensity fires and resprout vigorously. However, mountain big sagebrush is readily killed by fire and requires at least 15 years or more to recover after fire (Bunting et al. 1987).

Synthesis—
Climate change is likely to produce changes in cold US over time, including changes in plant growth and phenology. Although limited model data suggest that sagebrush may be highly vulnerable to climate change, paleoecological evidence does not support this inference. Warming at higher elevations and a longer growing season may increase productivity in cold US.

Increased future fire activity in subalpine environments may not constrain shrub establishment as much as tree establishment in these slow-growing areas, although recovery times may still be long.

Cold Upland Herb

Subalpine and alpine meadows are found at high elevations where temperatures are too cold or snow covers the ground too long for trees to grow. This PVG is not common in the Blue Mountains (table 6.4), but includes cold moist, cold dry, cool moist, and cool dry upland herb (UH) PAGs. Typical plant communities consist of grasslands with greenleaf fescue and Idaho fescue (*Festuca idahoensis* Elmer), elk sedge and Hood’s sedge (*Carex hoodii* Boott). Many of these high elevation grasslands have been degraded from livestock and elk grazing (Irwin et al. 1994 and Johnson 2003). Partial to complete loss of vegetation cover resulted in significant erosion of deep, fine textured, loess soils and decreased productivity, and greenleaf fescue grasslands in the Wallowa Mountains are still recovering (Johnson 2003, Reid et al. 1991). Some sites appeared to have transitioned to stable forb communities with poke knotweed, Nuttall's linanthus (*Leptosiphon nuttallii* ssp. *nuttallii* [A. Gray] J.M. Porter & L.A. Johnson) and alpine golden buckwheat (*Eriogonum flavum* Nutt.). This PVG includes many rare and endemic meadow species (fig. 6.15).

Potential future changes—

Snowpack melting and subsequent soil heating have been shown to influence flowering, growth phenology, and vegetation community patterns in alpine meadows (Dunne et al. 2003, Inouye 2008). Spatial variability in snowpack persistence influences flowering and growth phenology within species and, to the extent that spatial patterns of snowmelt are consistent among years, also influences species composition and community types in alpine meadows (Canaday and Fonda 1974, Evans and Fonda 1990).

Subalpine conifers have been documented as infilling alpine tundra and meadows in the Pacific Northwest, a trend related to periods of warmer climate (Rochefort and Peterson 1996, Woodward et al. 1995). Snow-dominated meadows on Mount Rainier (Washington) (Rochefort and Peterson 1996) and in the Olympic Mountains (Woodward et al. 1995) have been infilled by subalpine fir and mountain hemlock during the 20th century in association with warm phases of the Pacific Decadal Oscillation. Tree establishment has also been documented in the Wallowa and Elkhorn Mountains (Skovlin et al. 2001) and in some locations may have been aided by the heavy grazing pressures in the early 1900s. Because of the shallow, rocky soils that characterize this PVG, it is unlikely that tree encroachment will be a major problem in the future. Meadow plant species are able to colonize new habitat that was previously covered by ice or bare ground under more favorable climatic conditions, but the process of soil formation is slow. It is more likely that subalpine trees will establish in cold US areas, many of which are characterized by deeper soils.

The effects of climate change on alpine or subalpine meadow species have not been modeled. However, model results for subalpine tree species and MC2 results generally show loss of subalpine and alpine habitat, which may suggest loss of habitat for other subalpine and alpine life forms.

Disturbance—
As noted above, large-scale disturbances are infrequent in subalpine and alpine systems. In the future, wildfire events in subalpine systems may be more common, and herbaceous species may be able to more rapidly regenerate compared to shrubs and trees. Many species in cold UH communities are fire survivors because they sprout or sucker from roots, root crowns, corms, or rhizomes.

Synthesis—
There is considerable uncertainty about the future spatial extent of subalpine and alpine meadows. Continued warming in future decades could cause the geographic range of upland grass and forbs to contract, expand, or remain the same. Trends will most likely depend on the rates at which meadow species colonize exposed soil following disturbance.

Moist Upland Forests

Moist UFs occur at moderate elevations in the montane vegetation zone, or at low elevations in the subalpine zone (fig. 6.16). They are adjoined by cold forests at their upper edge and by dry forests at their lower edge. They are characterized by slightly longer growing seasons compared to the cold UF, and have cooler temperatures and higher precipitation than dry UF.

Late-seral stands are generally dominated by subalpine fir, grand fir, or Douglas-fir, and lodgepole pine or western larch often occur as early-seral species (except where lodgepole pine dominates). Douglas-fir and western white pine are mid-seral species (except in three potential vegetation types where Douglas-fir is the dominant (climax) species). Species of concern in this PVG include Alaska cedar (*Callitropsis nootkatensis* [D. Don] D.P. Little) (box 6.4) and quaking aspen (*Populus tremuloides* Michx.) (box 6.5).

For the Blue Mountains, the Moist Upland Forest PVG consists of five plant association groups (PAGs)—three in the cool temperature regime (Cool Wet UF, Cool Very Moist UF, Cool Moist UF), and two in the warm temperature regime (Warm Very Moist UF, Warm Moist UF). The Cool Moist UF is the most common component of this PVG. Cool moist forest understories are dominated by forbs, several mid-height shrubs, and a few tall shrubs on the warm end of the PAG. These forests tend to occupy the most productive forested environments in the Blue Mountains because moisture is less limiting, and their temperate nature is demonstrated by high species diversity and closed forest structure. High species diversity pertains to both forest overstory and understory composition.

Moist-forest understories are dominated by forbs, some mid-height shrubs, and a few tall shrubs in warmer locations. Moist-site plants such as bride’s bonnet (*Clintonia uniflora* [Menzies ex Schult. & Schult. f.] Kunth), twinflower (*Linnaea borealis* L.), false bugbane (*Trautvetteria caroliniensis* [Walter] Vail), western swordfern (*Polystichum munitum* [Kaulf.] C. Presl), and British Columbia wildginger (*Asarum caudatum* Lindl.) occur here, but most mesic environments in the Moist Upland Forest PVG are dominated by thinleaf huckleberry (*Vaccinium membranaceum* Douglas ex Torr.). Moist forests at the warm end of the temperature spectrum (Warm Very Moist and Warm Moist PAGs) include mid or tall shrubs such as Rocky Mountain maple (*Acer glabrum* Torr.), mallow ninebark (*Physocarpus malvaceus* [Greene] Kuntze), and oceanspray (*Holodiscus discolor* [Pursh] Maxim.).

Potential future changes—
Many (but not all) productive moist upland forests at higher elevation are more energy limited than water limited. Light is a limiting factor in productive forests where competition reduces light to most individuals. Tree growth in energy-limited forests appears to be responding positively to warming temperatures over the past 100 years (McKenzie et al. 2001). In the Blue Mountains, lower elevation moist forests may transition to being primarily water limited, particularly areas without much ash or loess which would enhance water holding capacity.

Moderate warming may lead to a positive response and increased productivity. However, more extreme warming and increased drought stress, particularly at lower elevations and in the southern portion of the Blue Mountains (e.g., Malheur National Forest) will likely cause decreased tree growth and forest productivity. However, suitable climate habitat currently occupied by cold upland forests may offset these losses.

Paleoecological evidence demonstrates that during early- to mid-Holocene warming, areas that are currently between cold upland forest and moist upland forest were forested (fig. 6.3), supporting *Agropyron*-dominated grassland/ponderosa pine parkland. Areas currently dominated by grand fir and Douglas-fir were dominated by ponderosa pine.

Species distribution models (table 6.1) project that the suitable climate available for most tree species characteristic of this PVG will be reduced to nonexistent by the end of the 21st century, although some models project minor loss of climate habitat for Douglas-fir. Losses were more extreme for the Malheur National Forest (where this PVG is currently limited in distribution) and southern portions of the Umatilla and Wallowa-Whitman National Forests. In contrast, MC2 projects either very small (3 scenarios) or moderate (1 scenario) increases in this forest type (fig. 6.5), particularly in the northern Blue Mountains. The latter scenario has a large projected increase in annual precipitation, including some increased precipitation in the summer, coupled with only moderate warming. None of the MC2 scenarios project a decrease in this forest type.

As discussed earlier, SDMs tend to produce large reductions in species climate habitat with future novel conditions, so the model results for these species are not surprising. MC2 is not constrained as much by future novel conditions. Moreover, most GCM scenarios project increased precipitation for the Pacific Northwest (fig. 6.4). Higher precipitation, coupled with warming, and the availability of climate habitat currently occupied by colder forest species, confirms the logic of MC2 model output. One recent study suggests that precipitation may decrease, because slower westerlies associated with jet-stream changes will result in less orographic lifting (enhancement) of precipitation for mountainous regions (Luce et al. 2013). MC2 does not account for decreasing orographic effects and tends to underestimate the effects of decreased snowpack expected in the future. Even with these shortcomings, the MC2 projections are more robust, and it is unlikely that any future climate would result in a large loss of moist forest habitat.

Disturbance—
Moist upland forests generally support mixed-severity fire regimes, although low-severity and high-severity regimes also occur (Stine et al. 2014). High levels of coarse woody debris, litter, and live biomass can produce occasional large, high-severity wildfires when fire weather and dry fuel conditions coincide. Increased frequency and duration of summer drought would allow wildfires to burn wetter and cooler sites, where high fuel loads become more available when fuel

moisture is low. However, the primary effect of severe fire in these forests is to reduce total standing biomass rather than change forest composition.

Warming temperatures could increase the potential for insect and disease outbreaks (box 6.1). Insect and disease agents in moist upland forest include western spruce budworm (*Choristoneura occidentalis* Freeman), Douglas-fir tussock moth (*Orgyia pseudotsugata* [McDunnough, 1921]), Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins, 1905), fir engraver (*Scolytus ventralis* LeConte, 1868), spruce beetle, mountain pine beetle in lodgepole pine, Douglas-fir dwarf mistletoe (*Arceuthobium douglasii* Engelm.), larch dwarf mistletoe, and root pathogens (particularly *Armillaria*, *Annosus* root rot (*Heterobasidion annosum* [Fr.] Bref.), and laminated root rot (*Phellinus sulphurascens* [Pilat]).

Many moist upland forests (e.g., those not burned in recent decades) have high stand densities, small trees, and few large fire-tolerant trees, and are dominated by shade tolerant and fire-intolerant tree species (Stine et al. 2014). Because of the increase in surface and canopy fuel loads, there is a greater risk of large and severe wildfires, particularly when fire weather conditions are severe. Increased stand densities also increase competition for water and nutrients, which leads to higher susceptibility to insect and disease outbreaks. These conditions threaten the long-term survival of large trees and sustainability of these forests in general.

Synthesis—

Paleoecological and some model evidence suggest that climate change will cause moderate to extreme loss of moist upland forests and characteristic species. However, MC2 model results suggest the opposite. Future warming with increased precipitation may lead to increased importance of this PVG across the landscape. This outcome is somewhat supported by recent trends in response to warming in energy limited forests. Unlike cold upland forests, these forests may be able to adapt to future climate change by expanding into new available habitats. Warm and very warm moist forest PAGs may be able to better adapt to warming compared to cool PAGs. However, increased summer drought stress may make these forests more vulnerable to other stressors, particularly at lower elevations and on southern sites in the Blue Mountains. Wildfire activity and insect and disease outbreaks will most likely increase with future warming, and may reduce the distribution of this PVG.

Dry Upland Forest

Dry UFs generally occur at low to moderate elevations in the montane vegetation zone (fig. 6.17). Climate varies with elevation, but common features include warm, dry summers, with warm to hot daytime temperatures and cool nighttime temperatures, and cold, wet winters. Much of the annual precipitation falls as snow in winter or during spring rainstorms. Late-seral stands are dominated by ponderosa pine, grand fir, or Douglas-fir, and ponderosa pine or Douglas-fir also function as early- or mid-seral species depending on plant association. Western juniper is expanding rapidly into this PVG as a result of fire exclusion and climate change, moving upward from a foothills woodland zone located below the montane zone (Knapp and Soulé 1998, Miller et al. 2005). Dry UFs are adjoined by moist upland forests at their upper edge, and by the woodlands and shrublands of the foothills vegetation zone at their lower edge.

The Dry UF PVG consists of three PAGs—one from the warm temperature regime (Warm Dry UF), and two from the hot temperature regime (Hot Moist UF, Hot Dry UF).

Warm, dry forests are the most common forest zone in the Blue Mountains (table 6.4), and they have a long history of human use for commodity purposes (e.g., livestock grazing and timber production; fig. 6.18). Effective fire exclusion has led to significant changes in species composition, forest structure, and stand density. Dry UF sites were historically dominated by ponderosa pine which is well adapted to survive in low-severity fire regimes.

Common dry UF understory species include graminoids and mid-height shrubs. Elk sedge and pinegrass (*Calamagrostis rubescens* Buckle) are ubiquitous, and white spirea (*Spiraea betulifolia* Pall.), snowberry (*Symphoricarpos* spp.), mallow ninebark, antelope bitterbrush and curl-leaf mountain-mahogany are common shrubs. On hot and dry sites, mountain big sagebrush, bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve), and western juniper are important species.

Potential future changes—

Dry UFs in the Blue Mountains are water-limited, and productivity is projected to decline in a warmer climate (Latta et al. 2010). Water stress during the warm season is the primary factor limiting tree growth at low elevations in the Pacific Northwest (Brubaker 1980), and negative water balances constrain photosynthesis (Hicke et al. 2002), although this may be partially offset if carbon dioxide fertilization significantly increases water-use efficiency in trees (Neilson et al. 2005). For example, Littell (2006) found that most montane Douglas-fir forests in the northwestern United States appear to be water-limited, and water limitation will increase in warmer climate (assuming that precipitation does not increase in the summer).

Kusnierczyk and Ettl (2002) found that ponderosa pine growth was positively correlated with precipitation in the fall and winter prior to the growing season, but was not significantly correlated with temperatures, suggesting that ponderosa pine growth is more sensitive to changes in water balance than to temperature. Ponderosa pine may be able to adapt to increased summer drought stress by allocating more biomass to sapwood conducting area, reducing the ratio of leaf area to sapwood area and presumably reducing risks of hydraulic failure (Callaway et al. 1994). This is an example of phenotypic plasticity and not genetic adaptation, suggesting that this could be a future response to changing climate (Maherali et al. 2002).

Increased drought stress will likely result in decreased tree growth and forest productivity in dry UFs in the Blue Mountains, particularly at the current dry forest-steppe ecotone. Areas with increased tree density due to recent fire exclusion may be particularly vulnerable to future climate change because of increased drought stress, although suitable climate habitat currently occupied by moist UFs may offset these losses.

Ponderosa pine was generally abundant during periods of major climatic change in the past (Whitlock 1992). Hansen (1943) documents an increase in ponderosa pine pollen during the past (inferred early- to mid- Holocene) at a site currently dominated by subalpine species. Phytolith data from sites in the Blue Mountains indicate that the ponderosa pine forests were located at a higher elevation than today and during the early- to mid-Holocene (fig. 6.3). Paleoecological evidence suggests that dry UFs may persist, but will be located farther north and at higher elevations.

Species distribution models suggest that climate habitat for ponderosa pine and Douglas-fir will decrease greatly, particularly in the southern Blue Mountains (table 6.2). Other model results suggest that Douglas-fir distribution would decrease only slightly or even increase. Vulnerability scores suggest that Douglas-fir might be more vulnerable to climate change than ponderosa pine (table 6.2). Output from MC2 suggests that temperate needleleaf forest and dry temperate needleleaf forest (both crosswalk to the dry upland PVG) will decrease in relative abundance on the landscape. In summary, model output suggests that dry upland forests will decrease somewhat in abundance across the landscape, particularly in the southwestern portions of the Blue Mountains. However, there is considerable model disagreement about the magnitude of this effect.

Disturbance—

For dry, frequent-fire forests, present-day stand structure, species composition, fuel accumulation and associated risk of severe fires, and insect and disease outbreaks are generally regarded as historically uncharacteristic and undesirable. In the Blue Mountains, grand fir has recently encroached in many ponderosa pine stands, and ponderosa pine has moved down in elevation into adjacent woodlands. These conditions are commonly attributed to decades of fire exclusion and suppression, timber harvesting, historical periods of overgrazing, and shifts in climate (Hessburg et al. 2005, Tiedemann et al. 2000, Wright and Agee 2004). The role and importance of fire as a disturbance process (Agee 1993, Fulé et al. 1997, Hessburg and Agee 2003) and disruption of fire regimes coinciding with Euro-American settlement and associated fire exclusion (Covington and Moore 1994, Hessburg et al. 2005, Swetnam et al. 1999) have been extensively documented. Management goals for these forests, such as reducing the risk of severe wildfires and sustaining and promoting biodiversity, have prompted the use of thinning and prescribed fire to reduce fuels, lower stand densities and alter stand composition. Prescribed fire simulates the frequent low-intensity surface fires considered characteristic of the historic environment of dry fire-prone forests throughout the interior West prior to non-indigenous settlement (Agee 1993, Cooper 1960, Covington and Moore 1994, Weaver 1943). However, prescribed fires are typically conducted mostly in the spring and autumn, whereas historical wildfires occurred mostly during summer and early autumn. The ecological implications of the current seasonal timing of prescribed burns have are rarely explored (but see Kerns et al. 2006).

Dry forest sites were historically dominated by ponderosa pine because it is well-adapted to survive in a low-severity fire regime (Agee 1996, Hall 1976, 1980). Heyerdahl et al. (2001) noted that within the same plant association, fires were more than twice as frequent at plots in the southern watersheds of the Blue Mountain compared to northern watersheds. Plots to the south experienced 15 and 13 fires on average (1687-1900), whereas plots to the north experienced only six, leading to mean fire return intervals of 14-35 years. Heyerdahl et al. (2001) also noted that fire frequency varied more based on location (south versus north) rather than forest type. It is likely that at least some mixed- and high-severity fires also occurred within the matrix of primarily low-severity fire prior to Euro-American settlement (Hessburg et al. 2007).

Longer summer droughts will potentially lengthen the fire season for most dry coniferous forest types, and may increase the risk of large wildfires. Because of the current accumulation of live and dead fuels, large and severe wildfires may become the norm for these forest types. At lower elevations, these fires may cause conversion to shrublands or grasslands, a trend that is supported by MC2 output, particularly for hotter and drier scenarios.

A warmer climate could increase the potential for insect and disease outbreaks (box 6.1). Insect and disease agents include western spruce budworm, Douglas-fir tussock moth (frequently where Douglas-fir and grand fir invaded stands historically dominated by ponderosa pine), Douglas-fir dwarf mistletoe, western dwarf mistletoe (*Arceuthobium campylopodum* Engelm.) and bark beetles (*Dendroctonus* spp.). Pine white (*Neophasia menapia* [C. Felder and R. Felder, 1859]), or pine butterfly, is also important in ponderosa pine. A recent pine white outbreak in the Blue Mountains caused significant defoliation (Kerns and Westlind 2013) (fig. 6.19), although mortality from this event is not expected to be high.

Synthesis—

Some dry UFs may undergo undesirable changes in the face of future climate change. These forests have already experienced a long history of human land use. Many dry UFs are already experiencing severe and uncharacteristic wildfire, and equally uncharacteristic insect and disease outbreaks, which will most likely increase in the future. It is likely that the hottest and driest sites will shift to woodland or steppe vegetation. Species characteristic of hot dry PAGs may be better adapted to future conditions and these species may become more common. Some model output suggests that Douglas-fir and ponderosa pine may decrease in the future, although paleoecological evidence conflicts somewhat with this conclusion, suggesting that ponderosa pine was able to adapt to warmer climate by migrating north or up in elevation. However, the extent to which these species can adapt under current and future stressors is unclear. The overall vulnerability assessment score for ponderosa pine is quite low, whereas Douglas-fir has a somewhat higher score (table 6.2). Given the strong paleoecological evidence regarding the persistence of ponderosa pine, coupled with its potential low vulnerability and availability of habitat currently occupied by moist forests, it is likely that this forest type will persist and remain an important component of the landscape, although shifts in the distribution of dry UFs and changes in relative abundance of different PAGs might be expected (or the formation of novel plant associations).

Dry and Moist Upland Woodlands

Upland woodlands (UW) in the Blue Mountains occupy the transition zone between shrublands at lower elevation and dry upland forests at higher elevation (fig. 6.20). These woodlands occupy the driest of the tree-dominated vegetation zones in the Blue Mountains. Summers are hot and very dry, while winters are cold and relatively wet. Annual precipitation in western juniper savannas and woodlands ranges from 13 to 75 cm, but most sites fall within the range of 25-50 cm yr^{-1} (Gedney et al. 1999). Much of this precipitation falls during the winter as rain or snow.

There are two woodland PVGs in the Blue Mountains—Moist UW (hot moist UW PAG) and Dry UW (hot dry UW PAG). These two PVGs are discussed together for this assessment, although the Dry UW PVG is the least common PVG. Western juniper is dominant in all of the PAGs. The Moist UW PVG is characterized by understory shrubs such as mountain big sagebrush, curl-leaf mountain-mahogany, antelope bitterbrush or the grasses Idaho fescue and bluebunch wheatgrass. The Dry UW PVG is characterized by understory species such as bluebunch wheatgrass, low sagebrush (*Artemisia arbuscula* Nutt.), and scabland sagebrush (*Artemisia rigida* [Nutt.] A. Gray).

Western juniper has expanded its range in the interior Pacific Northwest during the past 130 years, invading and creating savannas and woodlands in semi-arid ecosystems that were formerly shrub-steppe and grassland communities (Miller et al. 2000). More than 90 percent of the 3.2 million ha of current juniper savannas and woodlands developed in the past 100 years (Miller et al. 2000). The area of juniper forest and woodland is estimated to have increased fivefold between 1936 and 1988 (Gedney et al. 1999). Much of this expansion is attributed to heavy livestock grazing and reduced fire frequencies (Miller et al. 2000), but woodland expansion may have started between 1850 and 1870 in some areas during wet, mild climatic conditions (Miller et al. 2005). Western junipers tolerate very dry conditions and can live for up to 1,000 years in the absence of disturbance (Miller et al. 2000).

Potential future changes—
Minimal experimental research has been conducted on the response of junipers to elevated carbon dioxide or warming temperatures. Western junipers might be expected to benefit from increasing atmospheric carbon dioxide if it reduces stomatal conductance and delays depletion of deep soil water, although it is unclear if improved water-use efficiency would significantly increase growth, or simply reduce drought stress. Increased growth of western juniper in recent decades suggests that elevated carbon dioxide may be increasing growth through increased water-use efficiency (Knapp et al. 2001), but such evidence is not conclusive.

Tree-ring analyses suggest that western juniper growth is driven primarily by soil moisture availability and drought (Knapp et al. 2004, Knutson and Pyke 2008). Growth was positively correlated with winter and spring precipitation (October to June) and negatively correlated with spring and summer temperatures (Knutson and Pyke 2008). Growth sensitivity to drought was greatest at lower elevations and on steep, rocky sites. Cold winter temperatures also exert influence on vegetation communities and are almost as important as drought for limiting photosynthesis in juniper woodlands (Runyon et al. 1994). Extreme cold temperatures can also function as a disturbance agent (Knapp and Soule 2005).

Juniper woodlands may increase in abundance in future scenarios associated with increased winter and spring precipitation. However, higher spring and summer temperatures may negatively impact juniper woodlands, particularly at lower elevations and for hot dry UW PAGs. Areas with increased juniper density from recent land-use history may be particularly vulnerable to future climate change. However, suitable climate habitat currently occupied by dry UFs may offset these losses.

Recent research suggests that increased precipitation intensity (the same amount of precipitation falling in fewer storms) can result in moisture reaching deeper portions of the soil profile, and that juniper and other woody plants are able to access the deeper moisture better than grasses and herbs. Woody plant encroachment observed over the last century could continue into the future should precipitation intensity increase (Kulmatiski and Beard 2013).

As temperatures warmed during the early Holocene, western juniper began migrating north into its present range in the Pacific Northwest. Since the arrival of western juniper in central and eastern Oregon (circa 6,600-4,800 BP), northeastern California, and southeastern Idaho, its abundance and distribution have fluctuated (Miller et al. 2005). Dry climatic periods can cause regional declines of juniper, whereas wetter periods (wet summers, mild winters) can cause expansion. Paleoecological evidence suggests that juniper woodlands may increase in abundance in future scenarios associated with increased winter temperatures (virtually all scenarios) and increased spring and summer precipitation. The Moist UW PVG may become more abundant with this type of scenario. A warm, dry scenario may result in decreased western juniper abundance, although species characteristic of the Dry UW PVG may be better adapted to these conditions. Data from the Blue Mountains indicate that the shrub-steppe boundary was higher in elevation in the past during the warmer and drier Holocene (fig. 6.3), and a similar shift in the forest-woodland boundary might be expected.

Model output—

Species distribution models indicate that climate habitat for western juniper, curl-leaf mountain-mahogany, and big sagebrush will be almost nonexistent (table 6.2). However, MC2 output shows that temperate needleleaf woodlands will most likely increase in abundance, except for one scenario that represents the least amount of change (fig. 6.5). Assuming process models are more robust than species distribution models for projecting vegetation responses to climate change, we would expect an increase in these PVGs across the landscape, an inference that is consistent with the temperature and moisture tolerances of major species associated with woodland PVGs.

Disturbance—

Historical fire regimes are not well described for juniper savannas and woodlands in Oregon (Agee 1993, Young and Evans 1981). Young junipers have thin bark and are readily killed by fires. Junipers that avoid fires early in their lifespan can subsequently escape injury and death from fire by having thicker bark and suppressing understory herbaceous fine fuels through competition for water (Agee 1993). As a result, fire-scarred junipers are limited to microsites with limited fine fuel production. Fire scars in scattered or adjacent ponderosa pine forests might suggest a mixed-severity fire regime, with mean fire return intervals of 15 years to more than a century and occasional large fires (Agee 1993, Miller et al. 2005, Miller and Rose 1999). Romme et al. (2009) note that in many places in the west, fire return intervals in juniper woodlands were very long (generally measured in centuries). In this predominantly fuel-limited biome, climate change effects on fire frequency and severity will likely depend on changes in soil water availability and its effect on understory plant productivity (fuel generation).

Many of these woodlands were also homesteads at the turn of century, have been subject to livestock grazing for over a century, and are still used for grazing. These systems are also highly changed because of exotic annual grass invasion, particularly after tree harvest and natural disturbances. In general, sites supporting moist and cooler upland woodlands and sites dominated by mountain big sagebrush are more resistant to invasion by juniper than sites dominated by other species or subspecies of sagebrush (e.g., Wyoming big sagebrush [*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young]) found in warmer and drier conditions (Miller et al. 2013).

Synthesis—

Higher spring and summer temperatures may negatively affect some very hot and dry juniper woodlands at lower elevations. However, it is unlikely that these PVGs will be greatly reduced across the landscape (although locations of juniper woodlands could shift), given the (1) adaptability of juniper to drought, (2) potential availability of habitat currently occupied by dry upland forests, and (3) continued fire suppression policies. In the future, years of wet and mild climatic conditions, particularly above-average spring and summer precipitation, will most likely facilitate the continued expansion of juniper. Increased fire frequency and severity associated with future warming could reverse this trend and lead to conversion of some of these woodlands to persistent grasslands. This disturbance-mediated effect may help reduce ongoing conversion of shrublands and grasslands to juniper woodlands, although grassland dominance may be accompanied by increasing dominance of non-native annual grasses.

Moist and Dry Upland Shrublands

Moist upland shrublands (warm moist, hot moist, very hot moist) and dry upland shrublands (hot dry and warm dry) are not common in the Blue Mountains and are discussed together for this assessment. These PVGs tend to occupy the transition zone between woodlands at the upper elevation and grasslands at the lower elevation, but can also be found in forest openings or near ridge tops at higher elevation (e.g., snow openings, snowberry shrublands). Numerous community types differ according to the dominant grasses and shrubs, and whose distribution largely reflects underlying gradients in annual mean precipitation and soil properties (Franklin and Dyrness 1973). Characteristic species for the moist US PVG include mountain big sagebrush, antelope bitterbrush, snowberries, bitter cherry (*Prunus emarginata* [Douglas] Eaton), curl-leaf mountain-mahogany, and cool season (C3) bunchgrasses (e.g., Idaho fescue, bluebunch wheatgrass, Sandberg bluegrass [*Poa secunda* ssp. *secunda* J. Presl]). The dry US PVG is characterized by low, scabland, and threetip sagebrush (*Artemisia tripartita* Rydb.). The climate is classified as arid to semi-arid with low precipitation, hot dry summers, and relatively cold winters, and these shrublands have often been described as cold desert or high desert (Franklin and Dyrness 1973). Some of these sites, particularly those in the dry US PVG, have high value for traditional and tribal use. Camas (*Camassia* spp.), bitterroot (*Lewisia rediviva* Pursh), cous biscuitroot (*Lomatium cous* [S. Watson] J.M. Coult. & Rose), yampah (*Perideridia* spp.), and other Native American first food plants are often associated with scabland environments assigned to the dry US PVG.

Potential future changes—

Soil moisture and winter temperature are the major limiting factors influencing vegetation composition and productivity in Pacific Northwest steppe communities; precipitation, temperature, soil texture, and soil depth are the primary abiotic determinants of soil moisture (Bates et al. 2006, Comstock and Ehleringer 1992, Schlaepfer et al. 2012). Winter snow and rain are important for recharging water storage in deep soil layers (Schlaepfer et al. 2012, Schwinning et al. 2003), and most precipitation in the Blue Mountains occurs during the winter and spring. However, high temperatures and low summer precipitation combine to produce extended periods of soil moisture deficits each summer. Although summers are warm and dry, winters can be quite cold throughout much of the steppe region in the Pacific Northwest (Comstock and Ehleringer 1992).

Shrubs in this PVG are generally well-adapted to both cold winter temperatures and summer drought, particularly the dry US PVG. Sagebrush is tolerant of summer drought and is unresponsive to shifts in the seasonality of precipitation in regard to cover and density (Bates et al. 2006). Some shrubs are able to tolerate drought and remain photosynthetically-active during periods of water and heat stress (Depuit and Caldwell 1975) or avoid severe drought stress by developing deep root systems that allow them to access deep soil water reserves throughout the summer (Franklin and Dyrness 1973). Schlaepfer et al. (2014) recently synthesized knowledge about natural big sagebrush regeneration. They note that increases in temperature may not have a large direct influence on regeneration due to the broad temperature optimum for regeneration. However, indirect effects could include selection for populations with less stringent seed dormancy. Drier conditions may direct negative effects on germination and seedling survival and could also lead to lighter seeds, which lowers germination success further.

Moist US may be particularly vulnerable to future climate shifts, particularly at the lower ectone. Upland shublands at low elevations may decrease in productivity in response to prolonged summer drought. Increased winter precipitation may allow soil recharge, but it is unclear if the capacity of these systems is adequate to avoid high stress and mortality in the summer. Paleoecological evidence suggests an increase in sagebrush with warming. Data from the Blue Mountains indicates that the shrub-steppe boundary was higher in elevation in the past during the warmer and drier early- to mid-Holocene (fig. 6.3).

Species distribution models project nearly complete loss of habitat for big sagebrush, curl-leaf mountain-mahogany, and antelope bitterbrush (table 6.2). Models by others suggest that sagebrush distribution is may decrease, with strong decreases in the southern part of the range and increases were in the northern parts and at higher elevations (Schlaepfer et al. 2012). Simulation results for big sagebrush from a process model (under 2070–2099 CMIP5 climate scenarios) support expectations of increased probability of regeneration at the leading edge of the current big sagebrush range and decreased probability at the trailing edge compared to current levels. MC2 projects that although temperate (C3) shrublands will decline, xeromorphic (C4) shrublands (shrublands with an understory C4 grass component) will increase markedly. Currently C4 grasses occur only in Hells Canyon National Recreation Area, typically within grassland communities. It is unclear if C4 grasses will actually increase in this region without adequate summer precipitation, but it is likely that more drought tolerant species like sagebrush will increase in dominance within this PVG at the expense of species adapted to moister and cooler conditions. Thus, model output suggests that some shrubland species may decline, but that shrublands overall may increase markedly across the landscape.

Disturbance—

In productive mountain big sagebrush plant associations, such as those characterized by Idaho fescue, mean fire return intervals ranged from 10 to 25 years, with large fires every 38 years (Miller et al. 2005). However, fire was much less frequent in the more arid plant associations such as Wyoming big sagebrush/Thurber’s needlegrass (*Achnatherum thurberianum* [Piper] Barkworth) (50–70 years) and low sagebrush/Sandberg bluegrass (Miller and Rose 1999, Young and Evans 1981), where fire-free periods of 90 (Young and Evans 1981) and 138 years (Miller and Rose 1999) were reported in northern California and south-central Oregon, and fire-free periods probably exceeded 150 years for some sites. Baker (2006) argues that historical fire rotations were 70–200 years in mountain big sagebrush and longer in other types. Long-term charcoal records suggest that fire regimes in these vegetation types are climate- and fuel-driven; sagebrush densities and fire frequencies increased during wet periods (decades to centuries) and declined during dry periods (Mensing et al. 2006).

Multiple shrub species are characteristic of these PVGs. Big sagebrush, antelope bitterbrush and curl-leaf mountain-mahogany are fire sensitive and can be temporarily eliminated from a site by burning. Recovery of shrub canopy cover to predisturbance levels can require 10–50 years or more, with recruitment of new shrubs from soil seed banks being an important factor controlling recovery time (Ziegenhagen and Miller 2009). Short fire return intervals can cause significant changes in species and productivity if shrub communities have not fully recovered between disturbances (Davies et al. 2012).

Shrublands, particularly dry USs, are also prone to invasion by non-native annual grasses (box 6.6). In some areas, introductions of invasive plant species such as cheatgrass and medusahead (*Taeniatherum caput-medusae* [L.] Nevski) have significantly altered fire regimes by producing sufficient fine fuels to carry wildfires (D'Antonio and Vitousek 1992). Other important disturbances and stressors in these systems include juniper expansion, livestock grazing, and land conversion due to agriculture or development.

Synthesis—
Paleoecological data suggest sagebrush generally increased during warm periods in the past. However, in the early- to mid-Holocene (warmer, drier) the shrub-steppe boundary was located higher in elevation, and results regarding shifts in sagebrush distribution and regeneration support this expectation. Less is known about other shrub species. However, species distribution models indicate highly reduced available climate habitat for many shrubs, but output from MC2 projects that xeromorphic shrublands will increase significantly. It is likely that increased warming would result in increased coverage of ecosystems better adapted to arid conditions ecosystems, such as shrublands. However, as wildfires and warmer conditions increase, there is a risk of conversion to non-native annual grasslands in these systems, particularly for the drier PAGs.

Moist and Dry Upland Herbland

Moist UH (warm very moist, warm moist, hot moist, very hot moist) and Dry UH (hot dry and warm dry) are discussed together for this assessment. Dry UH is a more common PVG than Moist UH. Grasslands encompass numerous grass-dominated vegetation community types that differ according to the dominant grasses, and whose distribution largely reflects underlying gradients in annual precipitation and soil properties (Franklin and Dyrness 1973). The climate is classified as arid to semi-arid with low precipitation, hot dry summers, and relatively cold winters. Most precipitation occurs as rain and snow in winter and spring. Moist upland grasslands are characterized largely by Idaho fescue and bluebunch wheatgrass (fig. 6.21). The absence of sagebrush suggests an improved moisture condition (Franklin and Dyrness 1973). Dry upland herblands are dominated by bluebunch wheatgrass and Sandberg's bluegrass.

Potential future changes—
Soil moisture and winter temperatures are the major environmental limiting factors influencing grassland composition, with precipitation, temperature, soil texture, and soil depth being the primary abiotic determinants of soil moisture (Bates et al. 2006, Comstock and Ehleringer 1992, Schlaepfer et al. 2012). Grassland vegetation is generally well-adapted to both cold winter temperatures and summer drought. Grasses can avoid summer drought stress by concentrating growth in the spring and early summer, when soil water is still available and cooler temperatures promote high water-use efficiency (Comstock and Ehleringer 1992). Several studies also show that grasslands may be resistant to climate change effects (Dukes et al. 2005, Grime et al. 2008). Short-term changes in the interannual precipitation regime may not result in large changes in semi-arid vegetation communities (Jankju 2008).

In a warmer climate, grasslands at lower elevation may shift in dominance towards more drought-tolerant species (e.g., less Idaho fescue) (Blinnikov et al. 2002) (fig. 6.5). There is essentially no model output available for individual grassland species, although Reeves et al. (2014) project that net primary productivity will increase in eastern Oregon grasslands in a warmer climate. MC2 projects that C3 grasslands will decline significantly, but that warm season (C4) grasslands will expand. However, the expansion of C4 grasses in the model is based on changes in temperature alone, and it is unlikely that C4 grasses will increase markedly in the future unless summer precipitation increases as well. Most GCMs project increased aridity in the summer. A potential shift from C3 to C4 grassland species is uncertain, although a shift in species composition to drought tolerant species might be expected.

Disturbance—

Grasslands have been greatly affected by human activities, including livestock grazing, introduction of non-native species, and agriculture (Humphrey 1943, Tisdale 1961). Grazing first became a major factor in Pacific Northwest steppe communities with the introduction of cattle grazing in 1834 and sheep grazing in 1860. Settlers also introduced numerous non-native grasses, including cheatgrass, which was well-adapted to climate within parts of the steppe region, and heavy grazing allowed non-native grasses to invade native communities where they became highly persistent (Mack 1981). In addition to grazing, many grasslands have been cultivated for dryland agricultural crops like winter wheat or irrigated to produce summer fruits, vegetables, and grains. Homesteading on what is now public land was common in many tributaries of the Snake River and in Hells Canyon.

Wildfire occurrence is generally limited by lack of ignitions during the fire season or by lack of continuous fuels in particular on scabland communities on shallow soils. Cold season bunchgrass communities often have continuous fuels to carry fire. The extensive grasslands in Hells Canyon National Recreation Area have experienced numerous large grassfires, and much of the area has burned at least once since 1980. Many of the dominant bunchgrasses recover well from fires by resprouting from belowground organs and can achieve pre-fire abundance within five years. Fire return intervals of less than five years have been documented only in isolated instances. However, in some areas, introductions of invasive plant species such as cheatgrass, medusahead, and North Africa grass (*Ventenata dubia* [Leers] Coss.) have altered fire regimes by producing sufficient fine fuels to carry wildfires at higher frequencies than tolerated by perennial native grasses (D'Antonio and Vitousek 1992) (box 6.4).

Synthesis—

Grasslands may shift in dominance towards more drought-tolerant species, particularly at lower elevations and on more arid sites. Non-native annual grasses may also increase in importance. In general, it is likely that with increased warming and fire occurrence, grasslands will become a bigger component of the landscape, particularly where shrublands and woodlands are no longer able to support woody species

Conclusions

Climate change is expected to alter vegetation structure and composition, terrestrial ecosystem processes, and the delivery of ecosystem services in the Blue Mountains. Climate influences the spatial distribution of major vegetation biomes, abundance of species and communities within biomes, biotic interactions, and geographic ranges of individual species. Climate also influences disturbance processes that shape vegetation structure and composition, which are often the catalysts for vegetation change. However, there is considerable uncertainty in what the actual effects on vegetation due to climate change could be. Waring et al. (2011) report that the Pacific

Northwest has seen a significant decrease in the competitiveness of over 50 percent of the evergreen species in six ecoregions, with the highest concentration towards the northern and southern limits of their analysis area. However, these authors assigned the Blue Mountains a relatively low vulnerability score.

In a warmer, drier climate (especially in summer), the following may occur in the Blue Mountains by the end of the 21st century, although we note that there is considerable uncertainty about the future:

- The importance of pine and sagebrush species may increase.
- The forest-steppe ecotone may move north of its present position and/or up in elevation.
- Ponderosa pine may be found at higher elevations.
- Subalpine and alpine systems are potentially vulnerable, and subalpine tree species may be replaced by high-elevation grasslands, pine, or Douglas-fir.
- Juniper woodlands, which have been increasing in recent decades, may be reduced if longer and drier summers lead to more wildfire.
- Grasslands and shrublands at lower elevations may increase across the landscape but shift in dominance towards more drought-tolerant species.
- Non-native species, including annual grasses, may increase in abundance and extent.

In general, species with life histories tolerant of frequent disturbance and highly altered environments will be more dominant because they can establish and persist in rapidly changing environments.

Tree growth in energy-limited portions of the landscape (high elevations, north aspects) may increase as the climate warms and snowpack decreases, whereas tree growth in water-limited portions of the landscape (low elevations, south aspects) will probably decrease. Some species may respond positively to higher concentrations of ambient carbon dioxide as a result of increased water-use efficiency, although this "fertilization" effect may diminish as other factors become limiting.

Ecological disturbance (e.g. fire, insect and disease outbreaks), which is expected to increase in a warmer climate, will be extremely important in affecting species distribution, tree age, and forest structure, facilitating transitions to new combinations of species and vegetation patterns. Mountain pine beetle may be particularly important in lodgepole pine and ponderosa pine forests, and western spruce budworm and Douglas-fir tussock moth may also increase periodically. Annual area burned by wildfire is expected to increase substantially, and fire seasons will likely lengthen. In dry forest types where fire has not occurred for several decades, crown fires may result in high tree mortality. In addition, the interaction of multiple disturbances and stressors will create or exacerbate stress complexes. For example, an extended warm and dry period may increase bark beetle activity which would increase short-term fine fuels.

Considerable uncertainty exists about how climate change will affect species distribution, forest productivity, and ecological disturbance in the Blue Mountains. Simulation models provide science-based projections of how a warmer climate could modify the growth environment of species and broad patterns of ecological disturbance, supplemented by studies of the paleoecology of the region. However, because the future climate may differ considerably from what has been observed in the past, it is difficult to project vegetative response accurately for specific locations and time periods.

Adapting Vegetation Management to Climate Change in the Blue Mountains

Despite the uncertainty associated with climate change, there are many proactive steps that land managers can take that are likely to increase ecosystem resilience to climate change (e.g., Halofsky et al. 2011, Raymond et al. 2015). Based on the vulnerability assessment information presented in this chapter, and on documented adaptation principles (e.g., Millar et al. 2007, Peterson et al. 2011, Swanston and Janowiak 2012), workshop participants identified strategies, or general approaches, for adapting vegetation management to climate change (table 6.8). Participants also identified more specific on-the-ground tactics, or actions, associated with each adaptation strategy and considered the implementation of those tactics, specifically the time frame for implementation, opportunities and barriers to implementation, and information needs (table 6.8). Adaptation strategies and tactics were focused on addressing potential increases in fire and invasive species (table 6.8a,b,c), insect outbreaks (table 6.8d), increases in temperatures and droughts (table 6.8e,f), and effects of increasing temperatures on alpine and subalpine plant communities (table 6.8g,h). These adaptation strategies and tactics are summarized below.

While the strategies and tactics described here do not include all potentially appropriate vegetation-related actions to take under a changing climate, they do include those that workshop participants thought were most important. However, not all of these strategies and tactics are appropriate in all places and in all situations, and they should be evaluated by managers on a case by case basis. Many of these ideas will require additional thought and analysis before they can be implemented.

Responding to Increased Fire and Invasive Species Establishment

Increased temperatures with climate change will likely lead to increased wildfire area burned (Littell et al. 2010, McKenzie et al. 2004, Westerling et al. 2006). With increasing fire in forested ecosystems, managing vegetation to reduce fire severity and decrease fire patch size could help to protect fire refugia and maintain old trees (Peterson et al. 2011). For example, incorporating openings in silvicultural prescriptions decreases forest density and fuel continuity, which may reduce wildfire severity and protect old trees (Churchill et al. 2013, Stine et al. 2014) (table 6.8a). Management practices that help fire to play a more natural role in ecosystems, such as density management, prescribed fire, and wildland fire use, may also increase ecosystem resilience to wildfire under a changing climate (Peterson et al. 2011, Stephens et al. 2010, Stine et al. 2014) (table 6.8a).

In shrubland and grassland systems, increased area burned will likely lead to increased mortality of shrub species and native grasses, and increased abundance of non-native species, annual grasses in particular (Creutzburg et al. 2014). Adaptation strategies and tactics to address these sensitivities include increasing the resilience of native ecosystems through grazing management (i.e., avoid grazing practices that promote invasive species establishment), active restoration of less resilient sites (e.g., plant natives on sites dominated by invasive species), and management of soil resources to maintain stability and productivity (e.g., establish native vegetation to stabilize eroded areas).

Responding to Increased Insect Outbreaks

Native insect species have long played a role in Blue Mountain ecosystem dynamics (Oliver et al. 1994), and it will be important to recognize this role and accept that there will be insect-caused tree mortality under changing climate (table 6.8d). However, there are some management actions that may increase ecosystem resilience to native insect outbreaks, such as mountain pine beetle outbreaks. For example, restoring historical fire regimes in dry forests, and increasing diversity of forest structure and age and size classes may help to minimize the impacts of insect outbreaks (Churchill et al. 2013). Increasing tree species diversity may also help to increase resilience to insect outbreaks (Dymond et al. 2014), particularly in low-diversity stands (e.g., stands where ponderosa pine and western larch were removed and grand fir dominates).

Responding to Increasing Temperatures and Droughts

Increasing temperatures will likely decrease productivity in water-limited forests and inhibit regeneration of some species (Littell et al. 2010). Protecting trees that exhibit adaptation to water stress (e.g., trees with low leaf area to sapwood ratios) and collecting seed from these individuals for future regeneration could help to increase resilience to water stress (table 6.8e). To ensure success of future revegetation efforts, seed use plans should reflect seed needs based on projected climate change and disturbance trends. Managers may want to ensure that seed orchards contain tree species and genotypes that are well-adapted to drought and disturbance, and for some species, resistant to disease (e.g., white pine blister rust). Managers may also want to push the limits of seed zone boundaries and include seed from lower elevations in plantings.

With changing climate, it will be important to promote forest productivity and ecosystem function (table 6.8e). In some cases, managers may want to protect certain species that will be susceptible to increased drought stress, such as western larch in moist mixed conifer forests (table 6.8f). Silvicultural practices that maintain densities to maximize tree growth and vigor and that protect soil productivity will likely help to maintain ecosystem function.

Responding to Effects of Increasing Temperatures on Alpine and Subalpine Plant Communities

Higher temperatures are likely to increase water stress for some species in cold upland and subalpine plant communities. Rare and disjunct populations (e.g., of Alaska cedar, limber pine, whitebark pine, and mountain hemlock) may require protection to ensure their continued survival under changing climate (table 6.8g). Planting in appropriate locations could help to prevent loss of these populations. For whitebark pine, planting of genotypes that are resistant to white pine blister rust will be critical. For alpine plant species, such as those in the Wallowa Mountains, monitoring will be necessary to improve understanding of how climate variability and change will affect them (table 6.8h).

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Chapter 7: Climate Change and Special Habitats in the Blue Mountains: Riparian Areas, Wetlands, and Groundwater-Dependent Ecosystems

Kathleen A. Dwire and Sabine Mellmann-Brown[7]

Introduction

In the Blue Mountains, climate change is likely to have significant, long-term implications for freshwater resources, including riparian areas, wetlands (box 7.1), and groundwater-dependent ecosystems (box 7.2). Climate change is expected to cause a transition from snow to rain, resulting in diminished snowpack and shifts in streamflow to earlier in the season (Leibowitz et al. 2014, Luce et al. 2012; see chapter 3). Additional effects include changes in extreme high and low flow events, alteration of groundwater recharge rates, changes in the fate and transport of nutrients, sediments, and contaminants, and temporal and spatial shifts in critical ecosystem processes and functions (Johnson et al. 2012, Raymondi et al. 2013). Another consequence of climate change is higher frequency and severity of droughts (Seager et al. 2007), which will influence distribution of plant species, and likely increase susceptibility to insects attacks, as well as increase the frequency and severity of wildfires (see chapter 6).

In this chapter, we synthesize existing information and describe the potential effects of climate change on riparian areas, wetlands, and groundwater-dependent ecosystems of the Malheur, Umatilla, and Wallowa-Whitman National Forests. We begin by defining riparian areas, wetlands, and groundwater dependent ecosystems, highlighting the considerable overlap among these ecosystems, as well as the numerous definitions for them. We briefly describe the range of plant communities that occur in these special habitats, partly to highlight the existing diversity of wetland/riparian vegetation, but also as a basis for discussing the potential influences of climate change. Much of this chapter is devoted to summarizing existing information on the current condition of special habitats in the Blue Mountains, with focus on wetland/ riparian plant communities. Although we describe potential changes for different riparian/wetland vegetation groups, we also emphasize that there is considerable uncertainty about the rates and direction of change, which depend on the physical watershed and stream channel conditions, past and present land use, and the reliability of climate-change predictions for a given area.

Definitions

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Riparian Areas

Riparian areas have been ecologically defined as "three-dimensional zones of direct physical and biotic interactions between terrestrial and aquatic ecosystems, with boundaries extending outward to the limits of flooding and upward into the canopy of streamside vegetation" (Gregory et al. 1991). The first dimension of riparian areas is the longitudinal continuum from headwaters to the mouths of streams and rivers and ultimately the oceans (Vannote et al. 1980). The second is the vertical dimension that extends upward into the vegetation canopy and downward into the subsurface and includes hyporheic and belowground interactions for the length of the stream-riparian corridor (Stanford and Ward 1988, 1993). The third dimension is lateral, extending to the limits of flooding on either side of the stream or river (Stanford and Ward 1993). The dynamic spatial and temporal extent of each of these three dimensions depends on the watershed hydrologic regime, location within the stream network of the watershed (elevation, connectivity), and watershed physical characteristics and geomorphic processes, which in turn influence floodplain water availability and the distribution of different riparian communities. These physical characteristics and processes largely regulate the structure and function of riparian ecosystems (Gregory et al. 1991, Naiman and Décamps 1997, Naiman et al. 2005).

In the Blue Mountains, riparian ecosystems occur along low-gradient, U-and-trough shaped glacial valleys in alpine, high elevation sites; along steep-gradient, low-order headwater streams; along montane channels flowing through segments of varying valley width; and along low-gradient, alluvial rivers in the wider reaches of the Grande Ronde and the John Day Rivers and their tributaries (Crowe and Clausnitzer 1997, Johnson 2004, Wells 2006). The diversity of stream sizes, landforms, valley widths and gradients, and hydrologic regimes determine the types of biotic communities that occur along streams in a given region; each of these communities could have distinct responses to changing climate.

To assist in managing riparian areas, numerous administrative definitions and various terms have been developed (USDA FS 2012c). In the Blue Mountains, riparian areas, wetlands, and intermittent streams are included within Riparian Habitat Conservation Areas (RHCAs), which specify minimum buffers from each side of the channel or stream edge: intermittent streams (15 m), wetlands and non-fish-bearing perennial streams (46 m), and fish-bearing streams (91 m). Active management within these buffers must comply with a number of Riparian Management Objectives designed to improve habitat conditions for fish species that have been federally-listed as threatened or endangered under the Endangered Species Act (USDA FS 1995). Along many stream segments, the dimensions of the riparian buffers differ from the ecologically-defined riparian area described above.

Wetlands

Numerous definitions for wetlands have been developed for a range of administrative, academic, and regulatory delineation purposes (National Research Council 1995). For all Federal regulatory activities, wetlands are ecosystems "that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions" (Federal Interagency Committee for Wetland Delineation 1989). Wetlands can be extremely diverse, exhibiting a wide range of vegetation, soil, and hydrologic characteristics (Cowardin et al. 1979, National Research Council 1995). However, all definitions emphasize hydrologic variables, particularly duration, seasonality, and depth of inundation and soil saturation, that result in distinctive hydric soils and wetland vegetation.

For the Blue Mountains, the Oregon Wetlands Geodatabase provides a summary map of wetlands (figs. 7.1, 7.2, 7.3), color-coded by wetland type, as classified by the U.S. Fish and Wildlife Service (http://oregonexplorer.info/wetlands/DataCollections/GeospatialData_Wetlands) (Cowardin et al. 1979). The maps were compiled from existing National Wetlands Inventory data and many additional sources, including local surveys and academic studies. Three broad categories of wetlands occur in the Blue Mountains: palustrine, lacustrine, and riverine (Cowardin et al. 1979). Palustrine wetlands are freshwater wetlands that include marshes, wet meadows, and forested wetlands, and may be dominated by trees, shrubs, or emergent vegetation. Some palustrine wetlands may be associated with streams, particularly in headwaters, whereas many are isolated, occurring in basins, depressions, or wet meadows. Lacustrine wetlands border lake shores.

Riverine wetlands are associated with streams and rivers, and occur along stream channels. In this database, most riparian areas are treated as riverine wetlands (figs. 7.1, 7.2, 7.3), demonstrating the overlap in definitions of riparian areas and wetlands. This designation may result in an overestimate of wetland area, because some riparian areas may not qualify as jurisdictional wetlands (Federal Interagency Committee for Wetland Delineation 1989), but it does provide a basis for management, because all wetland and riparian areas in national forests in the Blue Mountains are managed as RHCAs (USDA FS 2012c). The mapped wetlands (shown in figs. 7.1, 7.2, 7.3) illustrate the extent and diversity of these resources in the three national forests of the Blue Mountains.

Although the Oregon Wetlands Geodatabase is an excellent resource for national forests in Oregon, it covers only wetlands that occur within the state's boundaries. The portion of the Umatilla National Forest in Washington is therefore excluded, as well as the small portion of the Wallowa-Whitman National Forest along the Snake River in Idaho. Wetland databases are not available for Washington and Idaho.

Groundwater-Dependent Ecosystems

Groundwater is broadly defined as "all water below the ground surface, including water in the saturated and unsaturated zones" (USDA FS 2012c). Groundwater-dependent ecosystems (GDEs) are "communities of plants, animals and other organisms whose extent and life processes are dependent on access to or discharge of groundwater" (USDA FS 2012a,b), which can greatly contribute to local and regional biodiversity (Murray et al. 2006). GDEs occur at aquifer discharge locations, such as springs, rheic, lentic or alluvial systems (Aldous et al. 2015), which are also referred to as surface/terrestrial GDEs (Bertrand et al. 2012, Goldscheider et al. 2006). Many wetlands, lakes, streams, and rivers receive inflow from groundwater, which can contribute substantially to maintenance of water levels, as well as water temperature and chemistry required by native biota (Lawrence et al. 2014, Winter 2007).

Along stream segments referred to as “gaining reaches”, groundwater enters the stream from the banks or the channel bed, and the volume of downstream streamflow is subsequently increased (Winter 2007, Winter et al. 1996). Groundwater can contribute substantially to late-summer streamflow (Gannett 1984) and is the source for cool-water upwellings that serve as refugia for coldwater aquatic species (Lawrence et al. 2014, Torgersen et al. 1999, 2012). Springbrooks, defined as “runout channels from springs, which may become a stream at some distance from the spring source” (USDA 2012a), may also contribute to the mediation of stream temperature. Groundwater is important to stream and river ecosystems in the John Day River basin (Gannett 1984) and most watersheds in northeastern Oregon (Brown et al. 2009).

In the Blue Mountains, GDEs include springs, springbrooks, certain high elevation lakes, fens, streams and rivers (Brown et al. 2009, 2010) and riparian wetlands along gaining river reaches, all of which may provide habitat for rare flora and fauna. Fens are wetlands supported primarily by groundwater with a minimum depth (usually 30-40 cm) of accumulated peat (Chadde et al. 1998, USDA FS 2012a,c). Springs are entirely supported by groundwater.

Five types of GDEs have been sampled in the Blue Mountains: helocrene, hillslope, hypocrene, mound, and rheocrene (USDA FS 2012a,b; modified from Springer and Stevens 2009). Helocrene springs emerge diffusely from low-gradient wetlands, often discharging from indistinct or multiple sources. Hillslope GDEs are springs or fens located on hillslopes, usually on 20-to 60-degree slopes, often with indistinct or multiple sources of groundwater. Springs associated with mounds actually emerge near the top of elevated surfaces, i.e. mounds composed of peat or mineralized carbonate, and may be located within fens or wetland complexes near subsurface faults. Rheocrene springs emerge directly into stream channels, and are also referred to as springbrooks or spring runs. Other types of GDEs may occur in the Blue Mountains, but have not yet been described or inventoried.

Dependence of Special Habitats on Different Water Sources

In contrast to surrounding upland ecosystems, the occurrence and characteristics of riparian areas, wetlands, and GDEs depend on the availability of abundant water. The fundamental hydrologic processes that influence these special habitats are: (1) the amount, timing, and type of precipitation (rain or snow); (2) streamflow variables described by magnitude, frequency, timing, duration, and rate of change (Nilsson and Svedmark 2002) and other characteristics of surface water runoff; (3) groundwater recharge; (4) groundwater discharge; and (5) evapotranspiration (Lins 1997).

Because precipitation is the ultimate source of water and directly influences streamflow characteristics and groundwater dynamics, it is expected that climate-induced changes in precipitation will affect riparian areas, wetlands, and GDEs. The availability of water to riparian areas, wetlands or GDEs is also influenced by physical watershed characteristics that affect infiltration and surface and hillslope runoff, including lithology, soil depth and topography (Jencso et al. 2009). However, determining how climate-induced changes in hydrologic sources and processes will affect special habitats is complex and has not been directly studied in watersheds of the Blue Mountains. Here we draw on research that has been conducted in other locations in the western United States with similar plant species or communities, and infer potential climate-induced changes in riparian and wetland vegetation in northeast Oregon and southeast Washington.

Riparian ecosystems depend on the presence of flowing water, although streamflow may not be perennial along all stream segments and can vary considerably with season, physical features of the watershed, and water source. The volume of streamflow largely regulates the transport and deposition of sediment, influencing the creation and erosion of stream banks, floodplains, point bars, and meandering, braided, and abandoned channels. Depending on the physical characteristics of a given stream segment, the volume of streamflow can also drive the seasonal changes in water table elevation of the adjacent riparian area (Jencso et al. 2011). These hydrologic and fluvial processes and resulting geomorphic surfaces are essential for the establishment, development, and persistence of riparian vegetation, and strongly influence the local distribution of different plant species and communities (Naiman et al. 2005). Based on long-term daily flow data (from U.S. Geological Survey stream gauging stations), different streams in the Blue Mountains have been characterized as supported by perennial runoff, snow plus rain, and super-stable groundwater (Poff 1996).

As noted above, streamflow volume along gaining reaches increases with the inflow of groundwater to the channel. Stream water can also drain from the channel bed and banks to the groundwater system, resulting in a loss of downstream surface flow volume (Winter et al. 1996); these stream segments are referred to as “losing reaches.” The extent and location of hyporheic and groundwater exchange along a channel segment is influenced by valley bottom features, including width, gradient, substrate size, and depth to bedrock, and can determine whether a reach is gaining or losing (Winter et al. 1996). Gaining and losing stream reaches result in different aquatic communities in the channels and different riparian plant communities on the floodplains. The extent to which specific reaches are gaining or losing may change in response to climate-induced changes in precipitation, streamflow characteristics, and groundwater discharge.

Wetlands can be supported by surface water, groundwater, and precipitation, or frequently by combinations of these sources that differ seasonally (Goslee et al. 1997, Winter 2001). Fens are primarily supported by springs or local aquifers and can maintain fairly stable water table elevations despite changes in timing and amounts of precipitation (Winter 1999). Other wetlands with different or multiple water sources will likely respond differently to climate-induced changes and variability (Winter 1999).

In wetlands and riparian ecosystems worldwide, hydrologic variables are consistently the strongest predictors of plant species distributions (Cooper and Merritt 2012, Franz and Bazzazz 1977, Lessen et al. 1999, Merritt and Cooper 2000, Shipley et al. 1991). Ordination and other analyses repeatedly show that riparian and wetland species and vegetation communities are distributed along gradients (usually elevational or microtopographic) relating to streamflow duration (Auble et al. 1994, 1998, 2005; Franz and Bazzazz 1977; Friedman et al. 2006); growing-season streamflow volume (Stromberg 1993); depth, duration, or timing of flooding (Richter and Richter 2000, Toner and Keddy 1997); inundation duration (Auble et al. 1994, Franz and Bazzazz 1977, Friedman et al 2006); and water table elevation or depth to groundwater (Busch and Smith 1995, Castelli et al. 2000, Cooper et al. 1999, Dwire et al. 2006, Rains et al. 2004, Scott et al. 1999). In wetlands, variables related to water table elevation and hydroperiod are the primary determinants of plant species distributions (Goslee et al. 1997, Magee and Kentula 2005, Weiher and Keddy 1995).

Current understanding of the water sources used by riparian/wetland plants is limited to a few indicator, keystone, and either highly valued or highly invasive species (mostly woody) (Cooper and Merritt 2012). However, it has been shown that riparian/wetland plant species use water from multiple sources (surface water, soil water, groundwater), depending on life stage and season (Busch and Smith 1995, Cooper et al. 1999, Goslee et al. 1997). In assessing the vulnerability of riparian/wetland species to climate-induced changes in streamflow or groundwater, the availability of water at all life stages must be considered, from plant recruitment and establishment, to reproducing adults, to persistence at later life stages (Cooper and Merritt 2012).

Lack of scientific information makes it difficult to directly infer climate change effects on riparian vegetation or to describe physical mechanisms regulating water availability to special habitats in the Blue Mountains. However, based on research from other locations, we assume that climate-induced changes in precipitation and streamflow will exert influences on the distribution of riparian vegetation via changes in local hydrologic regimes. Summer base flows are predicted to decrease (Cayan et al. 2001, Luce and Holden 2009). If riparian water table elevation can be assumed to be in equilibrium with water levels in the stream, reduced base flows could result in lower riparian water table elevations and subsequent drying of some streamside areas, particularly in wider valley bottoms. Increasing air temperature will result in increased evapotranspiration across the landscape, could reduce the hydrologic connectivity between uplands and riparian areas (Jencso et al. 2009, 2011), and subsequently contribute to the drying of some streamside areas. Dominant wetland/riparian plant communities will respond to climate-induced changes in hydrologic variables differently due to differences in their species composition (Merritt et al. 2010, Weltzin et al. 2000).

Current Resource Conditions

The Blue Mountains have a rich diversity of riparian and wetland plant associations and community types at mid-montane elevations (Crowe and Clausnitzer 1997) and at higher elevations and within deep canyons (box 7.3) (Johnson 2004, Wells 2006). Several quaking aspen (*Populus tremuloides* Michx.) communities and associations, which occur in upland locations as well as wetlands and riparian areas, have also been classified for the Blue Mountains (Swanson et al. 2010). Riparian and wetland aspen communities are highly valued throughout the Blue Mountains and are included here as special habitats.

Past land use and management activities have affected riparian and aquatic resources, but in different ways and to different extents, depending on valley setting, location within the watershed, and land use (Dwire et al 1999; Kauffman et al. 2004; Magee et al. 2008; McAllister 2008; McIntosh et al. 1994a,b; Parks et al. 2005; Ringold et al. 2008). Streams, wetlands, and associated plant communities are possibly the most heavily impacted ecosystems in the Blue Mountains. In many cases, the effects of past land use and management activities may be considerably greater than the anticipated gradual influence of climate change. However, for these altered ecosystems, climate-induced changes will exert additional stress, possibly resulting in further degradation. In this section, we briefly describe the current condition of different riparian and wetland vegetation types and how they have been affected by past land use and management activities.

Riparian Areas

We utilize existing vegetation classifications to highlight the diversity and complexity of riparian areas in the Blue Mountains, and as a basis for discussing how different vegetation types might respond to climate-induced changes. We present information for distinct riparian/wetland potential vegetation types (PVTs), plant association groups (PAGs), and potential vegetation groups (PVGs) that have been described for the Blue Mountains (Crowe and Clausnitzer 1997, Powell et al. 2007, Swanson et al. 2010, Wells 2006). These groupings are hierarchical, aggregated from fine-scale units to mid-scale units, and are explained in detail in Powell et al. (2007). Potential vegetation types (PVTs) are fine-scale hierarchical classification units that include plant associations and plant community types (Powell et al. 2007). PVTs are aggregated into mid-scale plant association groups (PAGs) representing similar ecological environments as characterized by temperature and moisture regimes (Powell et al. 2007). PAGs are then aggregated into potential vegetation groups (PVGs) with similar environmental regimes and dominant plant species; each PVG typically includes PAGs representing a predominant temperature or moisture influence (Powell 2000, Powell et al. 2007).

Potential vegetation describes the plant species composition occurring under existing climatic conditions and in the absence of disturbance (Powell et al. 2007), implies some knowledge of successional pathways, and is most useful for well-studied upland vegetation. However, riparian environments can be subject to frequent and unpredictable disturbances with a range of possible, but largely unstudied, successional trajectories. Plant associations and community types are interspersed along stream-riparian corridors as a mosaic, sometimes co-occurring over short stream lengths, responding to valley bottom width, geomorphic surfaces, and local differences in hydrologic variables (Naiman et al. 2005). Although successional pathways cannot be reliably determined for these riparian classifications, the plant community/association descriptions provide detailed floristic information and a basis for assessing both current conditions and future changes in species composition in response to management, natural disturbance, and climate-induced changes. Below, we discuss broad riparian vegetation types and note the number of PAGs and PVTs and other groupings that have been classified for each.

Conifer-dominated riparian areas—
Many kilometers of streams in the Blue Mountains are bordered by conifer-dominated riparian communities. Conifer-dominated riparian areas are valued for maintenance of riparian

microclimates, wildlife habitat, and a source of large wood for streams (table 7.1). Powell et al. (2007) describe a "cold riparian forest" PVG that includes 3 PAGs and 25 PVTs for conifer-dominated riparian areas. Depending on the PAG, dominant conifer species are subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.), Engelmann spruce (*Picea engelmannii* Parry ex. Engelm.), or lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. ex S. Watson). The "warm riparian forest" potential vegetation group includes 2 PAGs with 7 PVTs dominated by either Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) or grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), and one PVT by western white pine (*Pinus monticola* Douglas ex D. Don). These conifer-dominated riparian vegetation types typically occur at high to moderate elevations, mostly along first- and second-order streams, and mostly in moderately to highly confined valley bottoms. Although several other PVTs have high cover of ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), grand fir or Douglas-fir, they occur at lower elevations and are not consistently surrounded by conifer-dominated uplands (Powell et al. 2007).

Conifer-dominated riparian vegetation types have been affected by past forest harvest, mining, grazing, road building, fire exclusion, and to a lesser extent, invasive species (Wickman 1992) (table 7.2). Natural disturbances include wildfire, infestations by forest insects and fungal pathogens, landslides, and debris flows (Luce et al. 2012). In some locations at lower elevations, the ponderosa pine-common snowberry (*Symphoricarpos albus* [L.] S.F. Blake) community may be increasing in streamside areas previously dominated by deciduous woody species in response to channel incision and decreasing riparian soil moisture (table 7.2).

Riparian and wetland aspen plant communities—

Stands of quaking aspen are an uncommon, valued habitat type throughout the Blue Mountains region, and their sustainability has been a focus for management in uplands, riparian areas and wetlands (Swanson et al. 2010). Classification of wetland and riparian aspen communities for the Blue Mountains region showed the largest number of aspen community types (CT) were associated with herbaceous species in meadows (8 CTs), followed by associations with common snowberry (4 CTs) and other tall shrubs (3 CTs) in riparian areas (Swanson et al. 2010). One aspen community type was associated with Engelmann spruce, and another with tall shrub wetland types on slopes, likely occurring at points of groundwater emergence (Swanson et al. 2010).

Aspen CTs have been affected by fire suppression and herbivory by livestock and native ungulates (Baker et al. 1997, Bartos and Campbell 1998, Shinneman et al. 2013). They are currently threatened by herbivory and conifer encroachment, especially those occurring in meadows (Swanson et al. 2010) (table 7.2). Many stands are declining, without signs of regeneration, and are susceptible to a variety of insects and fungal pathogens (Swanson et al. 2010). Most aspen stands are less than 1 ha in size.

Cottonwood-dominated riparian areas—

Black cottonwood (*Populus trichocarpa* T. & G. ex Hook.) is a keystone riparian species occurring along a variety of valley types in the Blue Mountains, ranging from high gradient, V-shaped valleys to moderately confined or open, low-gradient valleys (Crowe and Clausnitzer

1997). Powell et al. (2007) classified a "warm riparian forest" PVG that includes three PAGs dominated by black cottonwood.

Cottonwood-dominated riparian areas were among the earliest settled in the mid-1800s. Settlers quickly recognized the economic potential for raising livestock, especially along the wider valley bottoms at mid to low elevations with abundant forage and water resources (Dwire et al. 1999). The widespread decline of cottonwood and willows (*Salix* spp.) has been widely attributed to land management practices associated with livestock production (McIntosh et al. 1994a,b; Beschta and Ripple 2005). Many floodplains formerly dominated by woody riparian species, including portions of the John Day River and its tributaries, were converted to cattle pastures and hay fields by modifying or relocating portions of the stream channels, removing woody species, and planting with introduced grasses (Dwire et al.1999). Other factors, such as use of cottonwood as a wood source, removal of streamside woody plants as "phreatophyte control," and hydrologic modification of streams and rivers for agricultural production and irrigation have contributed to the decrease in the distribution and abundance of deciduous riparian species, including cottonwoods and aspen, willows, and alders (*Alnus* spp.).

Several cottonwood species have been shown to depend on flood frequency and duration for recruitment and establishment (Mahoney and Rood 1998; Scott et al. 1996, 1997). Although recruitment of black cottonwood has not been studied relative to streamflow characteristics, germination and seedling survival depend on continuously moist conditions (http://plants.usda.gov/plantguide/pdf/cs_pobat.pdf), which are provided in part by high flows during spring runoff. It is possible that flow alteration of streams and rivers in the Blue Mountains has reduced the recruitment of new cottonwoods, thus contributing to the decline of existing stands. Limited recruitment has also been attributed to grazing of young cottonwood plants by livestock (Beschta and Ripple 2005) (table 7.2).

Willow-dominated riparian areas—

Willow-dominated riparian areas are found across elevation ranges, but are most extensive at mid to lower elevations. Willows provide numerous valued ecological functions, including shade and organic matter for streams, increased bank stability and sediment retention, and wildlife habitat for many resident and migratory vertebrate species, such as Neotropical migratory birds (Kauffman et al. 2001, Kelsey and West 1998) (table 7.1). Willow-dominated riparian areas also maintain water quality through trapping sediment and pollutants from upslope and upstream areas, thus reducing the volume or concentrations delivered to streams (Johnson and Buffler 2008).

Potential vegetation analysis for willow-dominated riparian areas of the Blue Mountains region resulted in the classification of a "cold riparian shrub" PVG that includes 4 PAGs and 10 PVTs (Powell et al. 2007). The cold riparian shrub PVG occurs at higher elevations or along channels with frequent cold air drainage at lower elevations (Crowe and Clausnitzer 1997). The dominant willow species include Booth's willow (*Salix boothii* Dorn), undergreen willow (*S. commutata* Bebb.) and Drummond's willow (*S. drummondiana* Barratt). The "warm riparian shrub" PVG includes 3 PAGs with 8 PVTs dominated by willow species that generally occur in moderately confined or open valley bottoms, including unconfined and glaciated valleys with low slopes (less than 3 percent) in montane and subalpine settings (Powell et al. 2007). These warm riparian shrub PVTs are also referred to as the "alluvial bar" willow group, because they frequently occur on coarse-textured sands, gravel and cobble bars. They are generally dominated by sandbar willow (*S. exigua* Nutt.), dusk willow (*S. melanopsis* Nutt.), and Pacific willow (*S. lasiandra* Benth.) (Powell et al. 2007).

In many streams throughout North America, the historic removal of American beaver (*Castor canadensis* Kuhl) has influenced the geomorphic and hydrologic characteristics of stream channels (Wohl 2001) as well as the distribution of woody riparian species, especially willows (Naiman et al. 1988). Dam building by beaver modifies local hydrology, thus expanding wetland area and contributing to retention of sediment and organic matter (Butler and Malanson 1995; Meentemeyer and Butler1999; Westbrook et al. 2006, 2011). Willows are a preferred source of food and dam-building material for beaver and readily establish along the edges of beaver ponds and beaver-influenced stream reaches. In the Blue Mountains, the removal of beaver likely contributed to the reduction of willow-dominated riparian areas and abundance of aspen (Kay 1994, McAllister 2008, Swanson et al. 2010). Beavers and functioning beaver dams are still infrequent in the Blue Mountains (Swanson et al. 2010).

Willow-dominated riparian areas have been heavily impacted by livestock use, including direct effects of grazing and browsing. Livestock grazing reduces cover and stem density of adult plants (Brookshire et al. 2002), and in many areas, has eliminated seedlings and saplings, thus reducing establishment of new plants. Elk (*Cervus elaphus* L.) utilize willows throughout the year (Singer et al. 1994, Zeigenfuss et al. 2002), and in floodplains with combined herbivory pressure from both livestock and native ungulates, willows can be highly impacted. Flow alteration has also affected willow-dominated riparian areas; downstream of diversions, species composition tends to consist of more drought-tolerant species (Caskey et al. 2014).

Other woody-dominated riparian areas (deciduous shrubs and trees)—
Geographic location, complex geology, and highly variable channel forms create a rich floristic diversity of woody riparian species in the Blue Mountains. Powell et al. (2007) describe a "warm riparian forest" with 7 red alder (*Alnus rubra* Bong.)-dominated and 2 white (*Alnus rhombifolia* Nutt.) alder-dominated PVTs. In drier riparian areas, classified as "low soil moisture riparian shrub," 16 additional PVTs are described, dominated by 13 different shrub species that occur across a range of valley bottom types, including steep canyons. In a "warm riparian shrub" PVG, they describe the following PVTs, dominated by different riparian woody species:

- mountain alder (*Alnus viridis* subsp. *crispa* [Chaix] DC.) (16 PVTs),
- Sitka alder (*A. v.* subsp. *sinuata* [Chaix] DC.) (3 PVTs),
- water birch (*Betula occidentalis* Hook.) (3 PVTs),
- red-osier dogwood (*Cornus sericea* L.) (3 PVTs),
- currant (*Ribes* spp.) (3 PVTs),
- twinberry (*Lonicera involucrata* [Richardson] Banks ex Spreng.) (1 PVT),
- shrubby cinquefoil (*Dasiophora fruticosa* [L.] Rydb.) (1 PVT),
- alderleaf buckthorn (*Rhamnus alnifolia* L'Hér.) (1 PVT).

In some locations, certain woody riparian plant associations are likely the result of land use, particularly hydrologic modification that has caused the conversion of willow-dominated areas to communities dominated by more dry-tolerant shrub species, such as shrubby cinquefoil, currant, and common snowberry. Woody-dominated riparian areas, including those with cottonwood, willow, and aspen, have also been impacted by livestock grazing, herbivory pressure from native ungulates, and conversion to pastures and other agricultural uses (table 7.2).

Herbaceous-dominated riparian areas—
Several herbaceous-dominated riparian and wetland plant associations have been identified in the Blue Mountains, reflecting both environmental conditions and past land use. Herbaceous-dominated riparian and wetland communities occur over a wide elevation range from alpine to lower montane environments. Crowe and Clausnitzer (1997) identified 11 herbaceous plant associations and 17 plant community types located in meadows, most of which were dominated by different sedge (*Carex*, *Eleocharis*, and other genera) species. In addition, they described seven herbaceous plant associations and six plant community types that occurred along shaded streams or springs (GDEs). Herbaceous-dominated riparian areas occur most commonly in moderately-confined to wide valley bottoms, usually along low-gradient stream segments.

At mid elevations, herbaceous-dominated meadows have been impacted by heavy elk grazing. At nearly all elevations, meadows have also been impacted by livestock use (Kauffman et al. 2004), with lasting impacts in many areas (Skovlin and Thomas 1995). At lower elevations, changes in species composition, density, and cover have resulted from either the complete or partial conversion of natural meadows to pasture along some floodplains. As with willow-dominated communities, hydrologic modifications, including water diversions and construction of ditches, and stream modifications (e.g., relocation or alteration of natural channels) have influenced channel characteristics, seasonal water supply and water table elevations (McIntosh et al. 1994a,b). In riparian meadows, the distribution of herbaceous species is largely determined by seasonal water table elevation (Castelli et al. 2000, Dwire et al. 2006, Loheide and Gorelick 2007), which can be influenced by patterns of streamflow runoff. In many meadows, a combination of hydrologic alteration and livestock grazing has resulted in drier conditions and increased dominance by non-native grasses and grazing-tolerant native species (Johnson et al. 1994) (table 7.2).

Subalpine and alpine riparian areas and wetlands—
Wells (2006) described 13 wetland alpine/subalpine plant associations:

- Three are dominated by willow species and generally occur in low gradient, U-shaped glacial cirques, U-and-trough-shaped glacial valleys, and higher gradient glaciated valleys.
- Two low shrub associations are identified:
    i) "alpine laurel (*Kalmia microphylla* [Hook.] A. Heller)/black alpine sedge (*Carex nigricans* C.A. Mey) plant association" that occurs in low gradient, U-and-trough-shaped valleys, and
    ii) "pink mountainheath (*Phyllodoce empetriformis* [Sm.] D. Don) mounds plant association" that occurs in the upper terminus of glacial valleys.

- Four wet graminoid associations are described, all of which occur most frequently in U-shaped, low gradient valleys. They are dominated by the following sedge species:
    - i) Water sedge (*Carex aquatilis* Wahlenb.),
    - ii) Northwest Territory sedge (*C. utriculata* Boott),
    - iii) blister sedge (*C. vesicaria* L.),
    - iv) few-flower spikerush (*Eleocharis quinqueflora* [Hartmann] O. Schwarz). The "few-flower spikerush plant association" is found in fens (GDEs) near springs at high elevations (2067 to 2348 m) in the Eagle Cap and Elkhorn Mountains.
- Three moist graminoid associations are described, dominated by the following sedge species:
    - i) Holm's Rocky Mountain sedge (*C. scopulorum* T. Holm),
    - ii) woodrush sedge (*C. luzulina* Olney),
    - iii) black alpine sedge (*C. nigricans* C.A. Mey)
- A fourth moist graminoid plant association is a sedge-forb mix, most commonly associated with headwater springs in the Strawberry Mountains. Referred to as the "northern singlespike sedge (*C. scirpoidea* Michx.)/brook saxifrage (*Micranthes odontoloma* [Piper] A. Heller)-spring plant association," it is considered an indicator for GDEs (Wells 2006).

Although alpine wetlands and meadows have been affected by past livestock grazing and ungulate browsing, they are typically in better condition than their low-elevation counterparts.

PACFISH INFISH Biological Opinion Effectiveness Monitoring—
The PACFISH INFISH Biological Opinion (PIBO) Effectiveness Monitoring was developed as a response to declining populations of steelhead trout and bull trout in the upper Columbia River Basin (http://fsweb.r4.fs.fed.us/unit/nr/pibo/index.shtml). Its main objective is to monitor biological and physical components of aquatic and riparian habitats (Meredith et al. 2011). As part of the Columbia River Basin project, 191 monitoring sites were established in randomly located watersheds across the Blue Mountains. Sites have been designated as 'reference' or 'managed' and are sampled on a five-year rotation. Reference sites (18 of the 191) are located mostly in wilderness areas at somewhat higher elevations and with more annual precipitation. No reference sites are available for the Malheur National Forest, which complicates comparisons between reference and managed sites in the Blue Mountains. However, PIBO monitoring data are the primary source of quantitative information on the condition of riparian areas occurring along 'response reaches'.

PIBO monitoring provides a regional evaluation of the condition of riparian vegetation for both reference and managed sites (Archer et al. 2012a, Meredith et al. 2011). At each site, plant species cover is sampled along the densely-vegetated streamside zone, or "greenline" (Winward 2000), and along cross-sectional transects established perpendicular to the channel or valley bottom. For each site, "wetland ratings" are calculated based on relative abundance of wetland indicator species (Coles-Ritchie et al. 2007). For Blue Mountain PIBO sites, the Wilcoxon rank sum test was used to compare managed vs. reference sites, and the Wilcoxon signed rank test was used for comparisons between measurement cycles.

Data from 2007 to 2011 showed lower total cover ($p = 0.04$) and woody cover (including conifers; $p = 0.01$) along the greenline for managed sites compared to reference (fig.7.4). Non-native species cover, however, was significantly higher at managed sites relative to reference sites ($p < 0.001$, fig.7.4). A comparison of data from 2003 to 2006 to the later sampling cycle (2007-2011) showed no detectable change in greenline total cover ($p = 0.83$), cross-section wetland rating ($p = 0.30$), and native species richness ($p = 0.79$). Greenline woody cover appears to have increased slightly at both managed and reference sites ($p < 0.001$ and $p = 0.03$, respectively) while non-native cover has decreased (managed sites $p = 0.03$, reference sites $p = 0.002$). There is evidence that wetland ratings along the greenline have decreased on managed sites ($p < 0.001$). Definitive trends in vegetation and habitat quality will likely take more than two five-year sampling cycles to detect.

Invasive weed species occurred in 109 of 178 managed sites (61 percent), compared to 8 of 18 reference sites (44 percent). The five most commonly encountered invasive plant species for the Blue Mountain PIBO monitoring sites were Canada thistle (*Cirsium arvense* [L.] Scop.), reed canarygrass (*Phalaris arundinacea* L.), Oxeye daisy (*Leucanthemum vulgare* Lam.), tall buttercup (*Ranunculus acris* L.), and bull thistle (*Cirsium vulgare* [Savi] Ten.), similar to findings for the entire Columbia River Basin (Archer et al. 2012b). Archer et al. (2012b) concluded that invasive plant species are widespread across the interior Columbia River Basin, consistent with results reported by others (Magee et al. 2008, Ringold et al. 2008). The continued spread of invasive species could contribute to future degradation of riparian plant communities.

Wetlands

The number of wetlands in national forests of the Blue Mountains, as derived from the Oregon Wetlands Geodatabase, is shown in table 7.3 (wetlands for the portion of the Umatilla National Forest in Washington are not shown). Depending on the forest, 42 to 51 percent of the mapped wetlands are classified as “riverine” or riparian wetlands associated with streams, indicating the overlap in definitions of riparian areas and wetlands (table 7.3). Riverine wetlands account for the largest area among all wetland types on the Wallowa-Whitman National Forest (table 7.3). Other important wetland types in the Blue Mountains are: palustrine wetlands (freshwater wetlands including marshes and forested wetlands) and lacustrine wetlands (bordering lake shores). In Malheur National Forest, palustrine wetlands account for the largest area among all wetland types (table 7.4). The Oregon Wetlands Geodatabase also identified “potential fens” if a wetland, usually palustrine, occurred near a spring (tables 7.3, 7.4; figs. 7.1, 7.3). All fens are GDEs, defined and discussed in more detail below. In the National Wetlands Inventory database, fens are frequently classified as a type of palustrine wetland, again indicating overlap in definitions for riparian areas, wetlands, and GDEs. In the Cowardin et al. (1979) system, fens typically fall into the Palustrine Emergent Class (PEM) with a saturated water regime. However, because characterization based on remotely sensed information is sometimes inaccurate, fens may remain undetected or be classified as other wetland types (Aldous et al. 2015, Werstak et al. 2012).

Comparing riparian and wetland conditions—

The current condition of riparian and wetland ecosystems differs considerably depending on location within the watershed, valley configuration, and past and current land use. The riparian and wetland communities at low to mid elevations have been the most altered by land use, including grazing, development of water infrastructure (dams, diversions), road building along floodplains, and conversion of floodplains to agricultural uses (Crowe and Clausnitzer 1997; McIntosh et al. 1994a,b) (table 7.2). Riparian and wetland communities that occur in wide, accessible valley bottoms have been more heavily impacted than higher elevation, narrow, conifer-dominated riparian corridors. Effects of climate change on precipitation and streamflow, combined with agricultural and municipal demands for water, will continue to affect river segments and riparian areas in lower portions of watersheds (Theobald et al. 2010). Wetlands and riparian areas that have been impacted by land use are more vulnerable to natural disturbances like flooding or wildfire (Dwire and Kauffman 2003). Less degraded wetlands and riparian areas may be more resilient to climate-related stressors (Luce et al. 2012).

Groundwater-Dependent Ecosystems

Steep elevation gradients, varied bedrock, and glacial landforms in the Blue Mountains influence the distribution, characteristics, and water chemistry of GDEs. Although the U.S. Forest Service recognizes that groundwater is a key component of the water resources on National Forest Systems lands (USDA FS 2007), existing information on the condition and distribution of GDEs in national forests of the Blue Mountains is limited. Here, we again rely on data compiled by The Nature Conservancy (Brown et al. 2010), the National Hydrology Dataset (http://nhd.usgs.gov), and the Oregon Wetlands Geodatabase (http://oregonexplorer.info/wetlands/DataCollections/GeospatialData_Wetlands) to assess the current condition of GDEs in the Blue Mountains.

Springs—

The number of currently mapped springs for Malheur, Umatilla, and Wallowa-Whitman, National Forests is shown in table 7.3. The percentage of named springs, which implies a known perennial water source, ranges from 9 percent of mapped springs in Umatilla National Forest (Oregon portion) to 14 percent in both Malheur and Wallowa-Whitman National Forests. Most springs are unnamed, and many may not be perennial, especially during drier years. The number of springs is presented here to document the currently known occurrence of spring GDEs in the Blue Mountains. Although many more springs likely exist, such as rheocrene springs discharging directly to streams, they are not yet mapped.

Springs play a key role as groundwater discharge zones that deliver cool water to warming streams and support late-season streamflows in summer, and may deliver relatively warm water during winter months (Lawrence et al. 2014, Winter 2007). Using criteria developed by The Nature Conservancy, most streams and rivers in the Blue Mountains are at least partially groundwater dependent. Santhi et al. (2008) estimated that 59 percent of annual streamflow in the semiarid mountains of eastern Oregon is attributable to groundwater discharge.

Locations of groundwater discharge to streams have been identified using remote sensing (Torgersen et al. 1999) and field techniques (Torgersen et al. 2012), but have not been systematically mapped (see chapter 4). The focus on stream temperature in relation to salmonid habitat has increased awareness of the ecological relevance and importance of groundwater discharge to streams and rivers.

Fens—

The Oregon Wetlands Geodatabase identified "potential fens" if a wetland, usually palustrine, occurred near a spring. To determine if these wetlands are indeed fens, each would require a field visit to determine that the wetland is supported (at least in part) by groundwater and that a minimum depth of peat (30-40 cm) has accumulated within the wetland (Chadde et al. 1998; USDA FS 2012a,c). Fens occupy less than 1 percent of the Blue Mountains landscape (table 7.4), but they contribute substantially to regional biodiversity of plants and animals (Blevins and Aldous 2011). In an otherwise arid region, perennially saturated fens are critical habitat for invertebrate and amphibian species. Although not explicitly differentiated as fen vegetation, several herbaceous-dominated plant associations frequently occur in fens. These are underlain by organic soils, and dominated by different sedge species, including Northwest Territory sedge, Cusick's sedge (*Carex cusickii* Mack. ex Piper & Beattie), Holm's Rocky Mountain sedge, and woodrush sedge (Crowe and Clausnitzer 1997).

Current condition of groundwater-dependent ecosystems—

Since 2008, 133 GDEs, mostly springs, have been inventoried and documented Blue Mountains national forests using draft and final versions of the Groundwater-Dependent Ecosystems Level I and Level II inventory methods (USDA FS 2012a,b). The Level I guide (USDA FS 2012a) describes basic methods for assessment of GDEs within a given area (e.g. national forest, ranger district, or specific project area) and is intended to qualitatively document the location, size, and basic characteristics of each GDE site. It also presents a "management indicator tool," described in more detail below. The Level II guide presents more detailed inventory (USDA FS 2012c) in addition to protocols for more comprehensive characterization of the vegetation, hydrology, geology, and soils at a given site.

In Malheur and Wallowa-Whitman National Forests, these inventories targeted strategically selected sites, because of concerns about disturbance and management, proposals for water development, and the high value of the resource. GDE inventories in Umatilla National Forest targeted portions of grazing allotments and watersheds with specific management concerns. Most inventories in all three national forests used the Level 1 inventory protocol (USDA FS 2012b). As of 2014, eight Level II inventories have been conducted, including collection of quantitative vegetation data suitable for monitoring.

In Umatilla National Forest, Level I inventories were conducted at 102 GDEs. Nearly 72 percent of these GDEs were identified as rheocrene springs that discharge directly into stream channels (table 7.5). Helocrene springs were the second most common GDE type and typically support a larger, low gradient wetland (0.1 to greater than 1 ha). Water diversions that withdrew emerging water away from the spring habitat and/or adjacent stream were observed at 46 of the inventoried springs (table 7.5). The amount of water diverted away from the GDE was estimated at 20 sites, averaging 93 percent of the available water at the time of sampling. Information on diverted water and other variables was not recorded at two of the 102 GDEs ("missing data"; table 7.5).

In the GDE Level I protocol (USDA FS 2012a), a series of 25 management indicator statements assist in identifying potential concerns and needs for management action based on observations recorded during field inventories. Information for the following three management indicators is presented for GDEs in Umatilla National Forest (table 7.6). Assessments for each of these are described in more detail in USDA FS (2012a):

- Aquifer functionality—There is no evidence to suggest that the aquifer supplying groundwater to the site is being affected by groundwater withdrawal or loss of recharge.
- Soil integrity—Soils are intact and functional; for example, saturation is sufficient to maintain hydric soils, if present, and erosion or deposition is not excessive.
- Vegetation composition—The site includes anticipated cover of plant species associated with the site environment, and upland species are not replacing hydric species.

Over 56 percent of the GDEs had evidence that aquifer functionality was compromised in some way, usually through groundwater extraction (table 7.6). Soil alterations in the form of ground disturbance, soil compaction, or soil pedestaling affected 24 percent of inventoried sites (soil integrity; table 7.6). Upland species cover was higher than expected in nearly 18 percent of the GDEs, suggesting that hydric species may have been replaced as a result of altered local hydrology.

Trails created by animals or people were noted in 44 percent of the sites, grazing/browsing by livestock was noted in 36 percent of sites, and grazing/browsing by wildlife was observed in 16 percent of sites. Thirty-one percent of the sites were disturbed through animal trampling. Disturbances were severe enough to question the long-term functioning of the most severely impacted GDEs. In summary, the inventoried GDEs in Umatilla National Forest showed significant resource impacts through water diversion, soil disturbance, and livestock impacts on vegetation composition (table 7.6). GDE inventories for Malheur and Wallowa-Whitman National Forests documented similar trends (data not shown).

In an assessment of GDEs in Oregon, Brown et al. (2009, 2010) examined existing data to determine distribution of GDEs and associated threats, focusing on water quantity and quality. They used the National Hydrography Dataset to identify locations of GDEs at the scale of Hydrologic Unit Code 6 (HUC6), and focused their assessment on watersheds containing two or more types of GDEs (e.g., wetlands and rivers), which they termed "GDE clusters." To evaluate threats to water quantity supporting GDEs, they examined the extent of water extraction or pumping through permitted wells used primarily for irrigation, industrial, and municipal uses, and unregulated (exempt) wells, which are used for livestock and domestic purposes. Where possible, they incorporated pending groundwater pumping permits and projections of residential growth to assess future threats from groundwater extraction. They found that GDE clusters in the Grande Ronde Valley, which is largely surrounded by Wallowa-Whitman National Forest, are threatened by diminished groundwater quantity (Brown et al. 2009).

To evaluate threats to groundwater quality supporting GDEs, Brown et al. (2009, 2010) examined contamination by nitrogen and phosphorus (high levels of fertilizer use), pesticides, and other toxic chemicals, using the compiled information to locate where GDEs (and GDE clusters) may be threatened by contaminated groundwater. Based on location and type of surrounding industrial and agricultural land use, they estimated that 22-40 percent of the GDE clusters in the John Day, Malheur/Owyhee, and Northeast Oregon HUCs were potentially at risk by groundwater contamination from pesticides or nutrients. Although most of the threatened GDE clusters they identified were located in agricultural valleys, not in national forests, contaminated groundwater may influence water quality in other portions of these basins, depending on physical features of the aquifers.

Potential Climate Change Effects

Riparian Areas and Wetlands

Changing climate in the Pacific Northwest is projected to alter streamflow in rivers and streams in a number of ecologically significant ways (see chapters 3 and 5). Warmer temperatures will influence changes in precipitation, evapotranspiration and snow accumulation, timing, and rate of melt (chapter 3). Earlier spring snowmelt will affect the timing and magnitude of peak flows, leading to higher peak flows in winter (Mote et al. 2005). Summer low flows are projected to decrease throughout the West (Cayan et al. 2001, Luce and Holden 2009).

In this section, we describe the potential effects of climate change in special habitats in the Blue Mountains, based on research that has examined responses of riparian vegetation to hydrologic alteration, primarily dams and diversions, as described above. However, there is considerable uncertainty in our projections, because empirical data are lacking on specific mechanisms through which climate change will influence riparian and wetland plant communities in the Blue Mountains. Climate change is likely to affect diverse riparian/wetland plant communities differently, depending on elevation, location within the watershed, land use, and species composition. Shifts in riparian vegetation and reduction in riparian area will probably occur in response to changes in streamflow characteristics, direct effects of higher temperatures, and seasonal and spatial distributions of soil moisture independent of streamflow (atmospheric and non-alluvial groundwater) (table 7.7, box 7.1).

Reduced riparian extent could result in direct losses in quantity and quality of ecosystem services provided by riparian vegetation, such as wildlife habitat, recreational value, shade over streams, and buffer capacity for maintenance of stream water quality. Less quantifiable are the loss of aesthetic values associated with reduced riparian cover and changes in streamside species composition, vegetation and age-class structure. Reduced width of riparian areas associated with projected changes in streamflow characteristics (see chapter 3), increased severity and frequency of drought, and higher agricultural and municipal demands for water could result in lower buffer capacity between aquatic and upland habitats.

Conifer-dominated riparian areas—
In headwater portions of forested watersheds, riparian areas are frequently dominated by the same species as surrounding uplands, although stands may differ in age, stem density, and

relative proportions of different species or size classes (Dwire et al. 2015). With progression of climate change, conifer-dominated riparian forests are increasingly subject to disturbances occurring in upland forests, including more numerous and severe fires and more frequent incidence of insect infestations.

In the Blue Mountains, Olson (2000) compared the fire history of upland and riparian forests, dominated by ponderosa pine, Douglas-fir, and grand fir. In these dry forest types, characterized by a low-severity fire regime, fires in riparian areas were slightly less frequent than uplands of the same forest types; however, differences were not significant. Williamson (1999) studied fuel characteristics and potential for crown fire initiation (torching) in paired upland-riparian stands of ponderosa pine, Douglas-fir, grand fir, and subalpine fir in the Blue Mountains. The potential for torching was high in both upland and riparian forests of all forest types, suggesting that high-severity fire could extend downslope into the valley bottoms. With projected changes in fire intensity and severity, fuel conditions in riparian areas may not be sufficiently different from uplands to stop or reduce the intensity of large wildfires during hot, dry weather (Luce et al. 2012).

Recent warmer climate has been associated with frequent and extensive insect outbreaks, as well as outbreaks in places where historical insect activity was low or unknown (Logan and Powell 2009; see chapter 6). Warming temperatures are projected to promote insect outbreaks in forested areas by increasing water stress in host trees while conferring physiological advantages to insects (Bale et al. 2002). Riparian trees, which grow in moist soils and cool microclimatic conditions, may be more resistant to insect infestation. However, climatic-induced increases in air temperature and changes in precipitation may result in drier streamside conditions, leading to stress in riparian trees. In addition, high insect densities may overwhelm local resistance to attack, making host trees vulnerable despite location along a stream channel. Wildfire, an important forest disturbance that is directly influenced by climate change (Peterson et al. 2014, Westerling et al. 2006), can reduce the resistance of surviving trees to insect attack. In addition, insect-caused canopy mortality alters the amount, composition, and arrangement of fuels (Jenkins et al. 2008, 2012). As fire- and insect-caused mortality transform the structure of dry forests, effects on associated riparian forests may also become more prominent.

In a vulnerability assessment of forest trees in the Pacific Northwest, tree species were ranked by risk factors: distribution, reproductive capacity, habitat affinity, adaptive genetic variation, and threats from insects and disease (Devine et al. 2012). Each risk factor incorporated several variables evaluating the vulnerability of each species to climate change. Authors found that subalpine fir and Engelmann spruce, dominant conifer species in many high-elevation, forested riparian areas in the Blue Mountains, were rated as highly vulnerable to climate change (Devine et al. 2012). Although some riparian conifer species could decline in cover, others may increase. For example, at lower elevations, ponderosa pine, grand fir, and Douglas-fir could increase in cover and density along drier floodplains.

Riparian and wetland aspen plant communities—

Quaking aspen is one of the few broadleaf deciduous trees in northeastern Oregon, providing vegetative diversity in the Blue Mountains region (Swanson et al. 2010). Its relative rarity, high value for wildlife, and colorful autumn foliage contribute to its aesthetic value. Over the last 25 years, many aspen stands in the Blue Mountains have been declining in number, area, and stem density (Swanson et al. 2010); similar dieback has been observed in other locations in western North America (Worrall et al. 2013). The reasons for broad-scale decline remain uncertain, but may be related to low soil moisture in severely affected stands (Worrall et al. 2013; see chapter 6). In the Blue Mountains, aspen communities will likely continue to decrease in extent if climate-related changes reduce water availability, thus affecting streamflow characteristics, available groundwater, and drought conditions (see chapter 6).

Cottonwood-dominated riparian areas—

Black cottonwood is a fairly short-lived tree (Braatne et al. 1996) that likely depends on seasonal flooding for recruitment and stand replacement (Lytle and Merritt 2004, Mahoney and Rood 1998, Merigliano 2005, Shafroth et al. 1998), and on baseflow for stand maintenance (Lite and Stromberg 2005). Relationships between streamflow and aspects of cottonwood ecology have been described for different species, geomorphic settings, and regions in western North America, mostly in response to dams and other flow alterations (Auble et al. 1994, Braatne et al. 1996, Merritt and Cooper 2000). Research results have also indicated that numerous cottonwood populations are in serious decline, and that non-native woody riparian species, notably tamarisk (*Tamarix* spp.), are expanding in distribution and displacing native cottonwoods throughout the western United States, particularly along rivers with altered flow regimes (Friedman et al. 2005, Merritt and Poff 2010).

Although tamarisk is not currently an issue in national forests of the Blue Mountains, habitat suitability modeling suggests that riparian habitat for tamarisk will increase throughout the Pacific Northwest over the next century (Kerns et al. 2009), which could have negative consequences for native cottonwoods. Decreases in distribution and declines in condition of cottonwood stands are likely for cottonwood-dominated riparian areas in the Blue Mountains in response to climate-related changes in availability of stream and groundwater, and increased frequency and severity of droughts. Many cottonwood stands are already compromised by limited recruitment, livestock grazing (Beschta and Ripple 2005), and floodplain conversion and development. Additional stress from climate-related changes in the hydrologic regime could have negative effects on the distribution and abundance of cottonwood.

Willow-dominated riparian areas—

Throughout the western United States, willow-dominated riparian areas occur in broad valley bottoms, including unconfined and glaciated valleys with low slopes in montane and subalpine landscapes (Patten 1998, Rocchio 2006). Floods, streamflow, shallow subsurface drainage, and American beaver activities all contribute to maintenance of high water tables and willow dominance (Demmer and Beschta 2008, Gage and Cooper 2004). The relative importance of streamflow and hillslope discharge for maintenance of willow ecosystems depends on elevation, geology, season, and other factors (Westbrook et al. 2006, 2011; Wolf et al. 2007).

Climate-induced changes in precipitation could affect both streamflow characteristics and groundwater discharge and may result in the spatial contraction of willow communities and in local loss of species near limits of their distribution. Similar to most cottonwood species, many willow species are thought to be reproductive specialists, requiring open substrates with certain particle size distributions that are able to maintain soil moisture levels for germination and establishment (Karrenberg et al. 2002). Hydrologic modification of streams has altered flood frequency and duration along many riparian corridors, which has likely influenced the local recruitment and persistence of willows and contributed to the decline of some willow species and communities, particularly those in the “cold riparian shrub” PVG (table 7.2; see also Stromberg et al. 2010). Willow communities in the “warm riparian shrub” PVG tend to be dominated by clonal willow species that frequently establish and spread via vegetative propagules. However, clonal willow species depend on flow characteristics for creation and re-working of sand and gravel bars, where they frequently establish. In addition, drying of willow-dominated plant communities, in combination with higher air temperature and projected decreases in stream baseflow could reduce soil and foliar moisture and limit their ability to serve as fuel breaks during wildfires.

Other shrub-dominated riparian areas—
Depending on the species, some shrub-dominated riparian areas could increase in areal extent and displace more moisture-dependent vegetation, including willow communities and sedge-dominated meadow communities. Conifers could encroach into shrub-dominated riparian areas, particularly at lower elevations.

Herbaceous-dominated riparian areas—
Wetland herbaceous species are highly sensitive and responsive to water table elevation, which could become more variable and less predictable with changes in streamflow characteristics and increased frequency and severity of drought. In some locations, wet meadows could contract in area, and vegetation could shift from sedge-dominated communities to more drought tolerant native and non-native grasses and possibly shrubs. Changes in species composition and cover of riparian vegetation could have cascading effects on water quality by reducing infiltration of runoff, and on stream channel morphology by weakening bank stability.

Summary—
As noted above, climate change could influence riparian and wetland vegetation in the Blue Mountains in various ways (box 7.1), depending on species composition and physical setting, specifically valley bottom width and geometry and location within the stream network and watershed. Some riparian plant communities and associations could contract in area, many could change in species composition over time, and others could increase in cover. The following trends are expected:

- **Conifer-dominated riparian areas** will become more susceptible to drought, wildfire, and insect infestations. Shifts in latitudinal and altitudinal distribution of dominant conifers will likely track trends in uplands (see chapter 6). Conifer-dominated communities will increase in cover, particularly at lower elevations, encroaching on shrub-dominated riparian areas and herbaceous-dominated meadows.
- **Riparian and wetland aspen** plant communities will likely continue to decrease in extent and decline in vigor due to drought and decreased water availability. Some

populations (e.g., those associated with springs) may be lost because of altered local hydrology.

- **Cottonwood-dominated riparian areas** will decrease in extent. Reductions in late summer base flows will likely compromise the persistence of existing stands. Changes in timing and magnitude of spring runoff could influence the recruitment and establishment of new individuals, thus affecting the replacement of existing stands.
- **Willow-dominated riparian areas** will decrease in extent as riparian width contracts in response to changes in frequency and magnitude of flooding, and lower water table late in the growing season as a result of lower baseflows. Changes in timing and magnitude of spring runoff could influence recruitment and establishment of new individuals, thus affecting replacement of existing stands. Species composition of willow communities will likely shift, favoring the most drought tolerant willows and other shrub species.
- **Other woody-dominated riparian areas** will increase in extent in some riparian areas, displacing more mesic willow species and communities, and favoring more drought-tolerant species. In communities dominated by more drought tolerant species, encroachment of conifers could increase, possibly replacing some shrub species over time.
- **Herbaceous-dominated riparian areas** will decrease in extent as riparian width contracts in response to decreased water availability due to lower baseflows, and changes in the magnitude, duration, and extent of flooding. Some sedge species will be replaced by more drought tolerant (and grazing tolerant) native and non-native grass species, and invasive species will likely increase in cover.

Groundwater-Dependent Ecosystems

Climatic variables affect hydrological processes, and in the Pacific Northwest, increased warming will influence the amount, timing, and distribution of runoff, as well as groundwater recharge and discharge (Elsner et al. 2010, Waibel et al. 2013). In the Blue Mountains, air temperatures are projected to become warmer during all seasons, with the largest increases occurring in summer (chapter 3), which will increase evapotranspiration in all ecosystems, including the special habitats discussed in this chapter. Snowpack is the main source of groundwater recharge in mountainous terrain (Winograd et al. 1998).

Higher minimum temperatures can reduce the longevity of snowpack, and decrease the length of time aquifer recharge can occur, potentially leading to faster runoff and less groundwater recharge. Groundwater recharge has been examined in only a few locations (Tague and Grant 2009), and little is known about groundwater recharge processes in many watersheds, including those that may be shifting from snow-dominated to more rain-dominated hydrologic regimes (Safeeq et al. 2013, 2014; see chapter 3). Snowmelt is generally considered a more efficient recharge agent than rainfall, so snow-to-rain shifts could potentially drive declines in groundwater recharge in snow-dominated areas (Earman and Dettinger 2011). Depending on elevation and the hydrogeologic setting, however, slowly infiltrating precipitation that includes both rain and snow may recharge some groundwater aquifers as effectively as rapid, seasonal snowmelt runoff. Although rain-on-snow zones are expected to shift upwards in elevation (see chapter 3), the influence of these shifts on groundwater recharge is unknown.

In the Blue Mountains, annual precipitation is projected to remain within the natural range of variability (see chapter 3). However, summers will be drier, the onset of snowmelt will be earlier (Luce et al. 2012), the rate of snowmelt will be more rapid, and the snow water equivalent (SWE) of snowpack will decrease (Folland et al. 2001; see chapter 3), all of which will influence snowpack volume. The biggest declines in snowpack persistence and April 1 SWE are projected to occur in mid elevations (see chapter 3). Although effects will differ considerably depending on local physical features and land use, these changes will likely affect groundwater recharge rates and, in turn, influence groundwater levels and the amount of groundwater available to support springs, groundwater-dependent wetlands, stream baseflows, and soil moisture (Ludwig and Moench 2009).

When assessing potential climate-induced changes to groundwater resources, recharge, and GDEs, it is critical to consider the hydrogeologic setting. Geologic units respond differently to changes in precipitation because of differences in hydraulic conductivity, transmissivity, primary vs. secondary porosity, and fracture patterns. In a study that combined examination of aerial photography (over 50-80 years) and climate analysis, Drexler et al. (2013) showed that five fens in the Sierra Nevada (California) decreased 10 to 16 percent in area. This decrease in GDE area occurred over decades with documented increases in annual mean minimum air temperature and decreases in SWE and snowpack longevity. However, two fens in the southern Cascade Range, underlain by different geology than the Sierra Nevada, did not change in area, suggesting that the hydrogeologic setting plays an important role in mediating changing climate variables on GDEs.

In the Blue Mountains, several different hydrogeologic categories can be delineated, including igneous/metamorphic, basalt, sedimentary, and older volcanic units (Gonthier 1985). Igneous and metamorphic rocks that exhibit low permeability and porosity, low volume groundwater discharges to GDEs, and are recharged only during large infrequent precipitation or snowmelt events, may not be very vulnerable to changes in temperature and precipitation regimes. However, aquifers in sedimentary or basalt formations, which generally have high permeability and porosity, larger volume discharges to GDEs, and are recharged more frequently, may be more sensitive to altered climate.

Small, unconfined aquifers, especially surficial and shallow aquifers, are more likely to have renewable groundwater on shorter time scales and may respond rapidly to changes in climate (Healy and Cook 2002, Lee et al. 2006, Sophocleous 2002, Winter 1999). Larger, deeper, and confined aquifers are more likely to have non-renewable groundwater, may be less sensitive to the direct effects of climatic variability and change, and are projected to have a slower response (Wada et al. 2012). Hydrogeologic units in the Blue Mountains exhibit both confined and unconfined conditions. The deeper basalt units and older volcanic aquifers tend to be more confined (Gonthier 1985).

Groundwater storage can act as a moderator of surface water response to precipitation (Maxwell and Kollet 2008), and changes to groundwater levels can alter the interaction between groundwater and surface water (Hanson et al. 2012). Climate-induced changes in connectivity between groundwater and surface water could directly affect stream baseflows and associated wetlands and other GDEs (Candela et al. 2012, Earman and Dettinger 2011, Kløve et al. 2012, Tujchneider et al. 2012). Simulation modeling shows that short flow-path groundwater systems, including many that provide baseflow to headwater streams, could change substantially in the timing of discharge in response to changes in seasonality of recharge (Waibel et al. 2013). By contrast, regional-scale aquifer systems with flow paths on the order of tens of kilometers, are much less affected by shifts in seasonality of recharge (Waibel et al. 2013). These effects may be highly variable, and largely depend on local hydrogeology. In wetlands, changes in groundwater levels can lead to reduced groundwater inflow, leading to lower water table levels and altered wetland water balances. For local and intermediate scale systems, the spatial extent of some GDEs will likely contract in response to decreasing surface water and groundwater and increasing temperatures. Changes in groundwater and surface water will also vary depending on location within the watershed and stream network, as well as future land use.

Effects of changing climate on the ecology of GDEs will depend on changes in groundwater levels and recharge rates, as influenced by the size and position of groundwater aquifers (Aldous et al. 2015). GDEs supported by small, local groundwater systems tend to exhibit more variation in temperature and nutrient concentrations than regional systems (Bertrand et al. 2012). It is likely that larger systems will be more resilient to climate change. Freshwater springs are dependent on continuous discharge of groundwater and form ecotones between subsurface-surface water and aquatic-terrestrial environments, which contribute to local and regional aquatic biodiversity (Ward and Tockner 2001). Springs and springbrooks are physically stable environments that support locally unique biological communities (Barquin and Death 2006). However, climate-induced changes in recharge rates may be reflected in decreased summertime flows with possible drying, as well as increased winter flow and associated flooding that could have negative impacts on biological communities (Green et al. 2011).

Taylor and Stefan (2009) estimated that groundwater temperatures would rise by up to 4º C in a temperate region under a doubling of carbon dioxide. Because many biogeochemical processes are temperature dependent, climate-induced changes in groundwater temperature may negatively affect the quality of groundwater and, in turn, influence aquatic communities (Figura et al. 2011). However, because the thermal regime of groundwater systems is less dependent on air temperature patterns than surface waters, the effects of rising air temperatures are likely to be less pronounced in springs and other GDEs.

For fens, peat accumulating processes will be influenced by increasing temperatures and local and regional changes in hydrologic regime. Reduced groundwater levels tend to promote soil aeration and organic matter oxidation. Generation and maintenance of peat soils over time depend on stable hydrological conditions. In recent studies of peatlands exposed to groundwater lowering, responses such as soil cracking, peat subsidence, and secondary changes in water flow and storage patterns have been observed (Kværner and Snilsberg 2011). Wetland plant species can respond to even slight changes in water table elevation (Magee and Kentula 2005, Shipley et al. 1991, Vitt et al. 1984), and shifts in composition of both vascular and bryophyte species could occur in fens with lowered water tables.

Land-use changes can alter watershed conditions and generate responses in biological communities and ecological processes, and in some cases, may override hydrologic modifications caused by large-scale climate shifts. As noted above for wetland and riparian ecosystems, effects of land use and management activities may have more immediate and detectable impacts on GDEs and the species they support than changing climate. For example, a recent study on spring-channel water diversion indicated that substantial decreases in physical and aquatic habitat occur with relatively small (10 to 20 percent) discharge reductions (Morrison et al. 2013). In the Umatilla National Forest, approximately 45 percent of inventoried springs undergo water withdrawals from the spring habitat (table 7.5). However, some spring organisms appear to be resilient to human-induced disturbances. Ilmonen et al. (2012) showed that invertebrate communities in springs affected by logging approximately 30 years prior to sampling did not differ appreciably from those in unaffected reference springs.

Management Context

Current Management Objectives and Desired Outcomes

Riparian areas and wetlands are protected under the Clean Water Act, which regulates the development and modification of floodplains; minimizes the destruction, loss, and degradation of wetlands; and enhances the natural and beneficial value of wetlands. Current management objectives for riparian areas in eastern Oregon are mainly informed by the aquatic strategies PACFISH and INFISH (USDA FS 1995, USDA FS and USDI BLM 1995) that were developed and adopted by the U.S. Forest Service and Bureau of Land Management. These strategies were considered short-term interim direction to protect native fish populations and their aquatic habitat. They will be revised by desired riparian conditions and management objectives with the adoption of new land management plans for all three national forests in the Blue Mountains.

Riparian goals in PACFISH and INFISH address water quality, stream channel integrity, in-stream flow, natural timing and variability of water-table elevation, diversity and productivity of riparian plant communities, and other riparian and aquatic habitat qualities necessary to support populations of inland native and anadromous fish. Riparian vegetation is to be maintained or restored to provide instream and riparian large wood, thermal regulation (including stream shading), and protection of floodplain surfaces and banks against uncharacteristic erosion. PACFISH and INFISH interim direction establishes riparian management objectives (RMOs) for all watersheds that include inland native or anadromous fish. These RMOs describe habitat conditions as a range of features that need to be met or exceeded. The key feature is pool frequency that varies by channel width. Supporting features include maximum water temperature, instream large wood, width:depth ratios, and measures of bank stability and bank angle. In the absence of site-specific watershed analysis, the RMOs provide a benchmark for all management actions and apply to all Riparian Habitat Conservation Areas (RHCAs), including streams with and without fish, wetlands, and intermittent streams.

The U.S. Forest Service groundwater management program has made progress in increasing awareness of the importance and vulnerability of groundwater resources, and providing guidance on identification, assessment, and analysis of GDEs (USDA FS 2012a,c). National forests in the Blue Mountains are still in the early stages of identifying and understanding the extent of groundwater resources, as well as potential threats. However, resource managers are increasingly considering GDEs in watershed assessments and project-level planning.

Management Practices

The establishment of RHCAs has altered management priorities to provide primary emphasis on riparian-dependent resources. Management activities in RHCAs are subject to specific standards and guidelines that limit timber harvest (including fuelwood cutting). As a consequence, fuel management, timber sales, and forest restoration projects commonly exclude RHCAs from any treatment. This management approach may be creating uncharacteristic fuel conditions within some riparian corridors (Messier et al. 2012, Meyer et al. 2012). Avoiding active management within RHCAs that have been altered by fire suppression and streamflow regulation (dams, roads, culverts, diversions) could further influence disturbance regimes and contribute to more uniform, late-seral forest structure, or to increased fire hazard. This management approach could also affect post-disturbance conditions, resulting in decreased riparian plant diversity over time.

In upland forest watersheds where fuel treatments are implemented, it is recommended that adjacent riparian areas also be considered for treatments (Meyer et al. 2012) to avoid concentration of fuels in streamside areas. Although treating fuels in riparian areas can potentially affect desired functions and ecosystem services (Dwire et al. 2010) (table 7.1), research has indicated that effects of prescribed fire are largely short-term (Arkle and Pilliod 2010, Béche et al. 2005). However, the effects of other fuel reduction treatments (e.g., mechanical thinning or various treatment combinations) on stream and riparian attributes have not been evaluated.

RHCA standards and guidelines require adjustment or elimination of grazing practices that are inconsistent with attainment of RMOs, but appear to have a smaller effect on management practices compared to timber management. The requirements have resulted in increased fencing of sensitive riparian and wetland resources within grazing allotments. Other management actions include active movement of cattle out of riparian zones, and placement of cut conifers to discourage access to treated aspen stands and streamside meadows. To monitor the effects of grazing practices and ensure that they do not prevent the attainment of RMOs, national forests have implemented riparian monitoring protocols such as Multiple Indicator Monitoring (Burton et al. 2011) in addition to PACFISH/INFISH effectiveness monitoring.

Adapting Special Habitats to Climate Change in the Blue Mountains

Management strategies and tactics for increasing resilience of vegetation in the Pacific Northwest to a warmer climate are well documented (e.g., Gaines et al. 2012; Halofsky et al. 2011; Littell et al. 2012, 2014; Raymond et al. 2013, 2014; see chapter 6). Although adaptation options for habitats associated with special hydrologic conditions are a small part of this knowledge base, these habitats have a disproportionately large effect on biological diversity in the region. Adaptation options for water resources (Dalton et al. 2013, Halofsky et al. 2011, Strauch et al. 2014; see chapter 4) are often synonymous with or related to adaptation options relevant for special habitats (e.g., maintaining and restoring in-stream flows). These sources of information, combined with feedback from resource specialists, contributed to a summary of climate change adaptation options for special habitats in the Blue Mountains (tables 7.7 a, b, and c). Implementation of climate-smart management actions and restoration objectives will benefit from a strategic approach to ensure that the most important work is occurring in the most important places (Hughes et al. 2014). Because special habitat conservation is at the interface of vegetation and stream restoration, opportunities exist for coordination of restoration programs and on-the-ground actions.

Riparian Areas and Wetlands

The productivity of wetland and riparian ecosystems could decrease in the future as a result of increased evapotranspiration and reduced snowpack, causing lower water supply during the growing season and more variable streamflow. Maintaining appropriate densities of native species, propagating drought tolerant native species, and controlling or eliminating non-native species are strategies that may increase riparian resilience to a warmer climate (table 7.7a,b). It would also be beneficial to plant species that have a broad range of moisture tolerances, such as Lewis' mock orange (*Philadelphus lewisii* Pursh) and choke cherry (*Prunus virginiana* L.), which are resistant to variable water availability during the growing season. Finally, removing infrastructure (e.g., campsites, utility corridors, spring houses, and spring boxes) from riparian areas and wetlands will reduce soil compaction and other physical damage, thus allowing natural physical processes to occur and improving hydrologic function. Opposition by the public to facilities removal is likely, so relocation (rather than removal) of some facilities to areas with less environmental impact can be considered.

Improving soil health and bank stability to reduce erosion and enhance native vegetation is an adaptation strategy that would improve riparian conditions (Kauffman et al. 2004). The most important measure is to reduce degradation of riparian areas by livestock through fencing and rest-rotation grazing. Livestock grazing has caused considerable damage to riparian systems over many decades, and efforts to repair and reduce this damage will improve resilience, although opposition from range permittees is likely if changes are instituted. Along stream segments with highly valued deciduous riparian vegetative cover, fencing to exclude native ungulates could also be considered. Elk and deer are frequently able to enter enclosures or riparian areas that have been fenced to exclude cows.

Riparian areas and wetlands are important components of alpine and subalpine ecosystems. In these systems, an important adaptation strategy is to reduce existing stresses, such as conifer encroachment, livestock grazing, and ungulate browsing (table 7.7b). Specific adaptation tactics include controlling livestock grazing, and removing non-native species where feasible, especially following wildfire. Collaboration with range permittees, fire and fuels managers, and coordination with ongoing restoration activities, will enhance the effectiveness of adaptation actions.

Groundwater-Dependent Ecosystems

Reduced snowpack could decrease water supply, potentially reducing productivity in all types of GDEs. The primary strategy for increasing resilience in GDEs is to manage for their functionality in the spatial context of the broader forest landscape (table 7.7c), because the structure and function of GDEs are largely influenced by surrounding vegetation and hydrology. An adaptation strategy is to maintain GDEs by maintaining water supply and improving soil quality and stability. This can be accomplished through three different tactics. First, decommissioning roads and reducing road connectivity is likely to increase interception of precipitation and local retention of water. Second, trampling of GDEs by domestic livestock and native ungulates can be better managed with fencing. Third, water can be maintained at developed spring sites through improved engineering, including use of float valves, diversion valves, and pumps. These tactics require significant costs, and there may be opposition to road removal and grazing restrictions by the public and range permittees.

As ecosystems, GDEs are understudied, primarily because subterranean systems are difficult to access. The scientific community is in early stages of research and management of these ecosystems and faces important knowledge gaps. The current lack of knowledge on groundwater has limited the consideration of groundwater resources in integrated forest planning, inventory, monitoring, and permitting. A framework for managing groundwater resources in national forests is needed, one that includes a more consistent approach for evaluating and monitoring the effects of management actions on groundwater. With respect to conservation of GDEs, guidance on how groundwater resources are considered in agency activities is needed, and will require evaluation of potential effects of groundwater withdrawals on national forest resources. Guidance should also provide a strategy through which groundwater and vegetation can be jointly managed, thus facilitating management of riparian areas, wetlands, and GDEs in a warmer climate.

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Chapter 8: Conclusions

David L. Peterson, Robyn Darbyshire, and Becky Gravenmier[8]

The Blue Mountains Adaptation Partnership (BMAP) made significant progress on the climate change response of national forests, synthesized the best available scientific information to assess climate change vulnerability, developed adaptation options, and catalyzed a collaboration among national forests and other stakeholders seeking to address climate change in the Blue Mountains region. This vulnerability assessment and corresponding adaptation options enabled the national forests to address several components of the U.S. Forest Service (USFS) climate change response strategy as outlined in the National Roadmap for Responding to Climate Change (USDA FS 2010) and the Climate Change Performance Scorecard (USDA FS 2012) (see chapter 1). The BMAP process allowed the Malheur, Umatilla, and Wallow-Whitman National Forests to respond with "yes" to scorecard questions in the organizational capacity, engagement, and adaptation dimensions.

To maintain focus on key near-term issues, the BMAP assessment emphasizes four resource areas—water, fish, upland vegetation, and special habitats—regarded as the most important resources for local ecosystems and communities. Water is critical because downstream users rely on snowpack and water that comes from national forests in the Blue Mountains. Cold-water fish species, which depend on that water, are the primary species listed under the Endangered Species Act and are valued by Native Americans, and for recreation and tourism. Upland vegetation provides a multitude of ecosystem services in the form of forest products for local economies, wildlife habitat, shade for streams, soil protection, livestock forage, and scenery. Finally, special habitats (e.g., wetlands, groundwater-dependent ecosystems) benefit a high diversity of terrestrial and aquatic species even though these habitats occupy a relatively small proportion of the landscape. In the future, it may be possible to expand the assessment to include other systems and issues such as wildlife, recreation, and socioeconomic effects.

The BMAP built on previous science-management partnerships by creating an inclusive forum for resource managers and stakeholders to address issues related to climate change vulnerability and adaptation. This partnership was conducted for national forests only, so additional work is needed to achieve an "all lands" approach to climate change adaptation in the Blue Mountains region. The scope of this vulnerability assessment was to cover a range of natural resources that are critical for human communities and ecosystem services in the Blue Mountains. By exploring resources in detail, participants identified species and ecosystems that are sensitive to climate change. More detailed quantitative and spatially explicit vulnerability

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assessments would improve the scientific basis for detecting the effects of climate change and developing site-specific management responses and plans. It would also allow resource managers to prioritize locations for implementation. Finally, integrating carbon assessments would allow managers to develop linkages between adaptation and mitigation actions.

Relevance to Forest Service Climate Change Response Strategies

In this section, we summarize the relevance of the BMAP process to the climate change strategy of the USFS and to the forest plan revision process for national forests. Information presented in this report is also relevant for other land management agencies and stakeholders in the Blue Mountains. This process can be replicated and implemented by any organization, and many of the adaptation options are applicable beyond the Blue Mountains. Like previous adaptation efforts (e.g., Halofsky et al. 2011), a science-management partnership was critical to the success of the BMAP. For others interested in emulating this approach, we encourage them to pursue this partnership as the foundation for increasing climate change awareness, assessing vulnerability, and developing adaptation plans.

Organizational capacity to address climate change, as outlined in the USFS Climate Change Performance Scorecard requires building institutional capacity in management units through training and education for employees. Training and education were built into the BMAP process through workshops and webinars that provided information about the effects of climate change on water resources, fisheries, and vegetation. The workshops introduced climate tools and processes for assessing vulnerability and planning for adaptation.

The BMAP science-management partnership and process were as important as the products that were developed, because partnerships are the cornerstone for successful agency responses to climate change. We built a partnership that included three national forests, the Pacific Northwest Regional office, the USFS Pacific Northwest Research Station, and two universities (Oregon State University, University of Washington), and is relevant for ongoing plan revision and restoration conducted by the national forests in collaboration with several stakeholders.

Elements 4 and 5 of the USFS Climate Change Performance Scorecard require units to engage with scientists and scientific organizations to respond to climate change (element 4) and work with partners at various scales across all boundaries (element 5). The BMAP process therefore contributed to the USFS in achieving unit-level compliance in their agency-specific climate responses.

Elements 6 and 7 of the USFS Climate Change Performance Scorecard require units to assess the expected effects of climate change and which resources as a result will be most vulnerable, and identify management strategies to improve the adaptive capacity of the national forest lands. The BMAP vulnerability assessment used the best available science to identify sensitivity and vulnerability of multiple resources in the Blue Mountains. Adaptation options developed for each resource area can be incorporated into resource-specific programs and plans. The identification of key vulnerabilities and adaptation strategies can also inform the forest plan revision process.

The science-management dialogue identified management practices that are useful for increasing resilience and reducing stressors and threats. Although implementing all adaptation options developed in the BMAP process may not be feasible, resource managers can still draw from the menu of options as needed. Some adaptation strategies and tactics can be implemented on the ground now, whereas others may require changes in policies and practices or can be implemented when forest land management plans are revised or as threats become more apparent.

Implementation

Implementing adaptation is the most challenging next step for the BMAP. This can occur gradually over time, often motivated by extreme weather and large disturbances, and facilitated by changes in policies, programs, and land management plan revisions. It will be especially important for ongoing restoration programs to incorporate climate change adaptation to ensure effectiveness. Landowners, management agencies, and Native American tribes will need to work together for implementation to be effective.

In several cases, similar adaptation options were identified for more than one resource sector, suggesting a need to integrate adaptation planning across multiple disciplines. Adaptation options that yield benefits to more than one resource are likely to have the greatest benefit (Halofsky et al. 2011, Peterson et al. 2011). However, some adaptation options involve tradeoffs and uncertainties that need further exploration. Assembling an interdisciplinary team to tackle this issue will help to assess risks and develop risk management options.

Integration of the information in this assessment in everyday work through “climate-smart thinking” is critical, and can be reflected in resource management and planning, as well as for management priorities such as safety. Flooding, wildfire, and insect outbreaks may all be exacerbated by climate change, thus increasing hazards faced by federal employees and the public. Resource management can help minimize these hazards by reducing fuels, modifying forest species composition, and restoring hydrologic function. These activities are commonplace, demonstrating that much current resource management is already climate smart. This assessment can improve current management practice by helping to prioritize and accelerate implementation of specific options and locations for adaptation.

Putting adaptation on the ground will often be limited by insufficient human resources, insufficient funding, and conflicting priorities. However, the magnitude and likelihood for some changes to occur in the near future (especially water resources and fisheries) are high, as are the consequences for ecosystems and human values, and some adaptation options may be precluded if they are not implemented soon. This creates an imperative for timely action for the integration of climate change as a component of resource management and agency operations.

The climate change vulnerability assessment and adaptation approach developed by the BMAP can be used by the USFS and other organizations in many ways. From the perspective of federal land management, this information can be integrated within the following aspects of agency operations:

- *Landscape management assessments/planning*: Provide information on departure from historic range of variability and on desired conditions (e.g., land management plans, watershed assessments).
- *Resource management strategies:* Incorporate information into conservation strategies, fire management plans, infrastructure planning, and State Wildlife Action Plans.
- *Project National Environmental Policy Act (NEPA) analysis:* Provide best available science for documentation of resource conditions, vulnerabilities of resources to climate change and the development of alternatives.
- *Monitoring plans:* Identify knowledge gaps that can be addressed by monitoring in broad-scale strategies, plan-level programs, and project-level data collection.

Agencies can use climate change vulnerability information and adaptation strategies and tactics within:

- *National forest land management plan revision process:* Provides a foundation for understanding key resource vulnerabilities causec by climate change for the assessment phase of forest plan revision. Information from vulnerability assessments can be applied in assessments required under the 2012 Planning Rule, describe potential climatic conditions and effects on key resources, and identify and prioritize resource vulnerabilities to climate change in the future. Climate change vulnerabilities and adaptation strategies can inform forest plan components such as desired conditions, objectives, standards, and guidelines.
- *Resource management strategies:* Incorporate information into forest resiliency and restoration plans, conservation strategies, fire management plans, infrastructure planning, and State Wildlife Action Plans
- *Project design/implementation:* Provide mitigation and design tactics at specific locations.
- *Monitoring evaluations:* Provide periodic evaluation of monitoring questions.

We are optimistic that climate-change awareness, climate-smart management and planning, and implementation of adaptation in the Blue Mountains region will continue to evolve. We anticipate that by the end of the decade:

- Climate change will become an integral component of business operations.
- The effects of climate change will be continually assessed on natural and human systems, with a stronger focus on ecosystem services.
- Monitoring activities will include indicators to detect the effects of climate change on species and ecosystems.
- Agency planning processes will provide opportunities to manage across boundaries.
- Restoration activities will be implemented in the context of the influence of a changing climate.
- Management of carbon will be included in adaptation planning.
- Institutional capacity to manage for climate change will increase within federal agencies and local stakeholders.

- Resource managers will implement climate-informed practices in long-term planning and management.

This assessment provides a foundation for addressing key climate change vulnerabilities by implementing adaptation options that help reduce the negative effects of climate change and transition resources to a warmer climate. Maintaining the science-management partnership that was developed as part of the BMAP process will enhance the scientific capacity of the national forests. The effectiveness of the partnership will be improved by collaborating with organizations focused on restoration and other issues, thus increasing the capacity of national forests to address specific issues and landscapes. We hope that the assessment and an enduring partnership will help implement climate change in resource management and planning throughout the Blue Mountains region.

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Tables

Table 3.1—Summary of changes in temperature projections for the Representative Concentration Pathways (RCPs) 4.5 and 8.5 annually and by season for the Pacific Northwest for the change from historical (1950-1999) to mid-21st century (2041-2070). Values are for the maximum model projection, multi-model mean, and minimum model projection

	Annual		**Winter DJF**		**Spring MAM**		**Summer JJA**		**Autumn SON**	
RCP	4.5	8.5	4.5	8.5	4.5	8.5	4.5	8.5	4.5	8.5
	--- *˚C* ---									
Maximum	3.7	4.7	4.0	5.1	4.1	4.6	4.1	5.2	3.2	4.6
Mean	2.4	3.2	2.5	3.2	2.4	3.0	2.6	3.6	2.2	3.1
Minimum	1.1	1.7	0.9	1.3	0.5	1.0	1.3	1.9	0.8	1.6

Table 3.2—Summary of changes in precipitation projections for Representative Concentration Pathways (RCP) 4.5 and 8.5 emission scenarios annually and by season for the Pacific Northwest for the change from historical (1950-1999) to mid-21st century (2041-2070). Values are for the maximum model projection, multi-model mean, and minimum projection

	Annual		**Winter DJF**		**Spring MAM**		**Summer JJA**		**Autumn SON**	
RCP	4.5	8.5	4.5	8.5	4.5	8.5	4.5	8.5	4.5	8.5
	Percent change									
Maximum	10.1	13.4	16.3	19.8	18.8	26.6	18.0	12.4	13.1	12.3
Mean	2.8	3.2	5.4	7.2	4.3	6.5	-5.6	-7.5	3.2	1.5
Minimum	-4.3	-4.7	-5.6	-10.6	-6.8	-10.6	-33.6	-27.8	-8.5	-11.0

Table 3.3—Snowpack sensitivity definitions (Kramer and Snook, unpublished data) used in fig. 3.4

Sensitivity class	**Definition**
Persistent—least sensitive	Timing of peak snowmelt differed by >30 days between the warmest, driest year and coldest, wettest year in >30 percent of the subwatershed.
Persistent—more sensitive	Timing of peak snowmelt in the warmest, driest year (2003, El Niño year) occurred >30 days earlier than the coldest, wettest year (2011, La Niña year) in >50 percent of the subwatershed.
Ephemeral snow	April 1 snow water equivalent was <3.8 cm during dry years (no snow) and >3.8 cm during wet years (snow cover) in >80 percent of the subwatershed.

Table 4.1—Kilometers of road by maintenance level on national forests in the Blue Mountains

Operational maintenance levels (ML)			National forests	
Code	Description	Malheur	Umatilla	Wallowa-Whitman
			Kilometers	
ML 1	Basic custodial care (closed)	6,059	3,543	7,216
ML 2	High clearance cars/trucks	8,814	3,131	6,719
ML 3	Suitable for passenger cars	587	545	435
ML 4	Passenger car (moderate comfort)	0	114	29
ML 5	Passenger car (high comfort)	0	132	210
Total	All roads	15,460	7,465	14,609

Table 4.2—Adaptation options that address climate change effects on water use in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Lower summer flows; higher winter peak flows; earlier peak flows; lower groundwater recharge; higher demand and competition for water by municipalities and agriculture.				
Adaptation strategy: Restore function of watersheds; connect floodplains; support groundwater dependent ecosystems; reduce drainage efficiency; maximize valley storage; reduce fire hazard.				
Add wood to streams and increase beaver populations		Collaboration with other agencies	Concerns about effects of beavers on private property	Identification of priority areas
Use a “climate change lens” during project analysis		Use the Climate Project Screening Tool for analysis of projects		
Improve livestock management to reduce water use (e.g., shut-off valve on stock ponds)	Some < 10 years, some > 30 years	Collaboration with range managers Evaluation of new projects	Minimal technology available for operations	Improved technology to reduce water use
Reduce surface fuels and stand densities in low-elevation forest	< 10 years, ongoing	Collaboration with fire managers	Opposition to active management, prescribed burning	
Restore meadows	Ongoing		Interactions with roads Elk herbivory	Inventory and classification of meadows

Adaptation strategy: Address demands for water (including water rights); improve water conservation.

Conduct integrated assessment of water and local effects of climate change	Ongoing, opportunistic	Case by case	Concerns about property rights High existing workload	Updated water resource use assessments at priority sites
Implement vegetation treatments in high water retention areas (e.g., snow retention)	Ongoing	Enabled by national forest land management plan	Lack of funding for long-term collection of gage data	Centralized source of data and metadata on water resources
Improve efficiency of drainage and ditches	10-30 years			
Encourage communication and full disclosure of information	Ongoing	Existing management plans Communication between line officers and local communities		Identification of key messages
Conduct vulnerability assessments by community	10-30 years	Source water assessments in some cases		
Treat roads where needed to retain water and maintain high water quality	< 10 years	Collaboration with road engineers	Lack of support for treatments	Development of effective road treatments

Table 4.3—Adaptation options that address climate change effects on roads and infrastructure in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Road design and maintenance are sensitive to increasing flood risk; higher peak flows lead to increased road damage at stream crossings (because of insufficient culvert capacity, more culvert blockage, and low bridges); safety compromised by more extreme events (e.g., landslides and debris flows).				
Adaptation strategy: Increase resilience of stream crossings, culverts, and bridges to higher peak flows.				
Replace culverts with higher capacity culverts or other appropriate drainage (e.g., fords or dips) in high-risk locations	< 10 years, opportunistic	Structure failures, especially at fish crossings	Funding Current backlog of deferred maintenance and upgrades Historic use patterns	Database of projects and upgrades Safety priorities Campground analysis
Complete geospatial database of culverts and bridges	< 10 years, opportunistic	Recording of geospatial locations as projects are completed	Funding Incomplete culvert survey information	Database of projects and upgrades

Adaptation strategy: Facilitate response to higher peak flows by reducing the road system and thus flooding of roads and stream crossings; disconnect roads from streams.

Continue to decommission roads with high risk and low access	Opportunistic, > 30 years	Partnerships and collaboration with other agencies that are also disconnecting roads from waterways	High cost of decommissioning roads	Implementation of Geomorphic Road Analysis and Inventory Package for decision making
Convert use to other transportation modes (e.g., from vehicle to bicycle or foot)	Opportunistic, > 30 years	Transportation planning, travel management plans, land and resource management plans	High cost of decommissioning roads	
Invoke Travel Analysis Process to prioritize road management	< 10-30 years	Travel Analysis Process and Minimum Roads Analysis are underway		
Use drains, gravel, and outsloping of roads to disperse surface water	Ongoing, 10-30 years	National Forest land management plan	High cost of road management Priorities unclear in some cases	Implementation of Geomorphic Road Analysis and Inventory Package for decision making

Table 5.1—Summary of streamflow statistics relevant to fish populations in the Blue Mountains climate change analysis area, based on changes associated with the A1B emissions trajectory

Flow metric	Climate period	All lands		Forest Service lands	
		Day of year[a]	Days advance	Day of year	Days advance
Center of flow mass	1980s	181	-	189	-
	2040s	172	- 9	176	-13
	2080s	165	-16	166	-23
		Number of days	**Days increase**	**Number of days**	**Days increase**
Winter 95% flow	1980s	5.7	-	4.6	-
	2040s	7.3	1.6	6.7	2.1
	2080s	8.7	3.0	8.6	4.0
		Stream kilometers	**Percent change**	**Stream kilometers**	**Percent change**
Stream length[b]	1980s	20,123	-	6,907	-
	2040s	19,130	-4.9	6,362	-7.9
	2080s	18,681	-7.2	6,070	-12.1
		Cubic meters per second	**Percent change**	**Cubic meters per second**	**Percent change**
Mean summer flow[c]	1980s	0.98	-	0.68	-
	2040s	0.83	-15.3	0.48	-28.9
	2080s	0.77	-21.7	0.40	-41.4
Mean annual flow	1980s	1.65	-	1.09	-
	2040s	1.75	5.8	1.17	7.0
	2080s	1.82	10.1	1.23	12.5

[a]Refers to day of water year starting October 1.
[b]Stream reaches in network with mean summer flows greater than 0.039 $m^3 s^{-1}$.
[c] Average flow across all reaches in the network.

Table 5.2—Summary of August mean stream temperatures in the Blue Mountains climate analysis area during the baseline period and two future periods associated with the A1B emissions trajectory

	< 8 ºC	8–11 ºC	11–14 ºC	14–17 ºC	17–20 ºC	> 20 ºC
	-------------------------------- *Stream kilometers*					------------------------------------
All lands						
1980s	448	1,672	5,150	8,846	5,000	1,496
2040s	131	1,122	3,082	7,640	7,679	2,959
2080s	46	770	2,240	6,351	8,457	4,752
Forest Service lands						
1980s (1970-1999)	441	1,457	3,237	1,525	263	124
2040s (2030-2059)	131	1,054	2,353	2,716	588	203
2080s (2070-2099)	46	739	1,867	3,187	923	284

Table 5.3—Changes in stream length (kilometers; percent change in parentheses) across the Blue Mountains analysis area with temperatures suitable for spring Chinook salmon spawning and rearing during three climate periods associated with the A1B emissions trajectory.

	All lands	**Forest Service lands**
	------------------ *kilometers < 19.0 ºC*[a] -----------------	
1980s (1970-1999)	1,921	854
2040s (2030-2059)	1,453 (-24.4%)	767 (-10.2%)
2080s (2070-2099)	1,200 (-37.5%)	673 (-21.2%)

[a]A critical temperature threshold of 19 ºC was chosen because reaches warmer than this are susceptible to invasion by nonnative smallmouth bass that predate on juvenile salmon (Lawrence et al. 2014), and because pre-spawn mortality rates of adult salmon are much higher above this threshold (Bowerman et al., unpublished manuscript; Keefer et al. 2010).

Table 5.4—Changes in stream length (kilometers; percent change in parentheses) across the Blue Mountains analysis area with temperatures suitable for bull trout spawning and rearing during three climate periods associated with the A1B emissions trajectory.

	All lands	Forest Service lands
	------------------ *kilometers < 11.0 °C*[a] ---------------	
1980s (1970-1999)	1,953	1,827
2040s (2030-2059)	1,174 (-40%)	1,129 (-38%)
2080s (2070-2099)	786 (-60%)	771 (-58%)

[a]Stream reaches with mean August temperatures >11.0 ºC are warmer than optimal for bull trout spawning and rearing, so future warming will decrease habitat suitability in these areas. More than 90 percent of bull trout spawning and early juvenile rearing occurs in reaches with August temperatures <11.0 ºC, so these areas are critical to population persistence (Isaak et al. 2010; Isaak et al. 2014; Isaak et al. 2015; Rieman et al. 2007).

Table 5.5—Changes in stream length (kilometers; percent change in parentheses) with suitable temperatures for steelhead and redband trout downstream of Hells Canyon within the Blue Mountains analysis area during three climate periods associated with the A1B emissions trajectory

	Kilometers < 9 ºC[a]	**Kilometers 9–13 ºC**	**Kilometers 13–20 ºC**	**Kilometers 20–25 ºC**[b]
All lands				
1980s (1970-1999)	685	2,684	10,630	358
2040s (2030-2059)	246 (-64%)	1,724 (-36%)	11,182 (5%)	1,208 (337%)
2080s (2070-2099)	105 (-85%)	1,317 (-51%)	10,423 (-2%)	2,512 (702%)
Forest Service lands				
1980s (1970-1999)	681	2,183	2,185	8
2040s (2030-2059)	246 (-64%)	1,555 (-29%)	3,181 (46%)	75 (937%)
2080s (2070-2099)	105 (-85%)	1,262 (-42%)	3,527 (61%)	163 (2,037%)

[a]Stream reaches with mean August temperatures < 9 ºC are too cold for redband trout, so future warming will increase habitat suitability in these areas.
[b]Stream reaches with mean August temperatures > 20 ºC are warmer than optimal for steelhead/redband trout so future warming will decrease habitat suitability in these areas. Temperatures > 25 ºC are considered unsuitable (Cassinelli and Moffitt 2009, Rodnick et al. 2004, Sloat and Reeves 2014).

Table 5.6—Changes in stream length (kilometers; percent change in parentheses) with suitable temperatures for redband trout upstream of Hells Canyon within the Blue Mountains analysis area during three climate periods associated with the A1B emissions trajectory

	Kilomeers < 9 ºC[a]	Kilometers 9–20 ºC	Kilometers 20–25 ºC[b]	Kilometers > 25 ºC
All lands				
1980s (1970-1999)	168	3,914	71	0
2040s (2030-2059)	73 (-57%)	3,516 (-10%)	565 (796%)	0
2080s (2070-2099)	34 (-80%)	3,179 (-19%)	941 (1,325%)	0
Forest Service lands				
1980s (1970-1999)	158	976	0	0
2040s (2030-2059)	70 (-56%)	1,062 (9%)	2	0
2080s (2070-2099)	34 (-78%)	1,094 (12%)	6	0

[a]Stream reaches with mean August temperatures <9 ºC are too cold for redband trout, so future warming will increase habitat suitability in these areas.
[b]Stream reaches with mean August temperatures >20 ºC are warmer than optimal for redband trout, so future warming will decrease habitat suitability in these areas. Temperatures >25 ºC are considered unsuitable (Cassinelli and Moffitt 2009, Rodnick et al. 2004, Sloat and Reeves 2014).

Table 5.7a—Adaptation responses for fisheries and aquatic environments to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Shift in hydrologic regime involving changes in timing and magnitude of flows. Anticipated changes include lower summer flows and higher, more frequent winter peak flows.				
Adaptation strategy: Maintain or restore natural flow regime to buffer against future changes.				
Use watershed analysis, watershed condition framework, etc. to develop integrated, interdisciplinary tactics associated with vegetation and hydrology	< 10 years	Collaborate with nongovernmental organizations, other federal and state agencies, and private landowners	Competing priorities Lack of funding	Compare Mid-Columbia River Steelhead Recovery Plan and Oregon Department of Fish and Wildlife climate change analysis and other products Identify priority areas
Protect groundwater and springs	< 10 years	Explore opportunities with range permittees, nongovernmental organizations, and others	Lack of information on locations of groundwater and springs Lack of funds to address sites affected by livestock Competing priorities	Identify locations of groundwater dependent systems and spring influences

Restore riparian areas and beaver populations to maintain summer base flows and raise water table	< 10 years to > 30 years	Collaborate with states, range permittees, and private landowners regarding beaver re-entry Use range allotment management, riparian shrub planting and protection, riparian aspen restoration and management, road infrastructure planning, and valley form analysis to assess potential sites for beaver colonies and channel migration	Effects on infrastructure (culverts and diversions) Adequate food supply at for growing beaver colonies and dispersing individuals Public and private landowner acceptance of beaver colonies Effects of rising water levels on streamside roads and camping	Inventory riparian vegetation Monitor beaver populations Identify potential watersheds or stream reaches Inventory riparian vegetation and stream morphology status and trends
Address water loss at water diversions and ditches	< 10 years	Coordinate with the water district managers, water master, and water users	Competing priorities. Lack of funding Water law does not address ditch water loss and inefficiencies	Identify diversions and ditches with water losses or other resource issues
Reconnect and increase off-channel habitat and refugia in side channels and channels fed by wetlands	< 10 years to > 30 years	Collaborate with partners	Competing priorities Lack of funding	Identify potential watersheds or stream reaches
Revegetate, use fencing to exclude livestock,	< 10 years	Collaborate with partners on fencing or acquire/lease water rights Plant in and protect riparian areas Restore aspen	Competing priorities Lack of funding Difficulty of maintaining livestock and ungulate exclosures	Investigate baseline vs. desired conditions in riparian corridors Conduct effectiveness monitoring

Acquire water rights, use low-flow channel design	< 10 years	Collaborate with partners in acquiring and leasing water rights	Competing priorities Lack of funding Opposition from permittees	Investigate baseline vs. desired conditions in riparian corridors Identify water rights holders willing to collaborate
Disconnect roads from streams to reduce drainage efficiency	< 10 years to > 30 years	Identify areas with high drainage efficiency, using GIS and Netscape and Geomorphic Road Analysis and Inventory tools	Controversial from a political perspective Lack of travel plan Competing priorities Lack of funding Time and cost for full analyses	Acquire GIS data to identify areas with high drainage efficiency Identify problematic road segments contributing high sediment
Adaptation strategy: Decrease fragmentation of stream network so aquatic organisms can access similar habitats.				
Identify stream crossings that impede fish movements and prioritize culvert replacements	< 10 years to > 30 years	Collaborate with partners	Competing priorities. Lack of funding	Update culvert inventory
Use stream simulation design (e.g., bottomless arches, bridges), adjusting designs to provide low-flow thalweg	< 10 years to > 30 years	Collaborate with partners to enhance skills and funding Use GIS analyses to inform low-flow and high-flow stream design at specific crossings	Insufficient staff time for design, contracting, and effectiveness monitoring Insufficient NEPA funding and staff time for project planning	Incorporate climate change information to ensure culvert priorities consider long-term cost-effectiveness and ecological effectiveness

Rebuild stream bottoms by increasing floodplain connectivity, riparian vegetation, and water tables; decrease road connectivity	< 10 years to > 30 years	Restore beaver habitat and beaver colonies Relocate streamside roads away from streams as possible Manage roads, vegetation, and fuels to ensure large wood recruitment for in-channel potential	Hazard tree management, roadside salvage, firewood cutting, road proximity to stream channels and beaver colonies, developed recreation site locations Mining claim and private land access obligations Competition with livestock and large ungulates for limited food supply	Assess beaver habitat potential Compare natural fire regime restoration with potential for fish habitat restoration Options for alternate access routes for mining claims and private land access
Maintain minimum streamflows (buy and lease water rights, install modern flow structures, monitor water use)	< 10 years to > 30 years	Collaborate with partners	Competing priorities Lack of funding and personnel Few opportunities to acquire senior water rights	Identify areas of concern and opportunities for return on investment
Design channels at stream crossings to provide a deep thalweg for fish passage during low-flow periods	< 10 years to > 30 years		Competing priorities Lack of funding	Identify and prioritize channels and crossings of high concern
Design stream crossings to accommodate higher peak flows	< 10 years to > 30 years		High costs Competing priorities	Update stream crossing inventory

Adaptation strategy: Develop better information about streamflow regimes.				
Increase flow data collection and monitoring to better describe patterns in flow and improve hydrologic models	< 10 years to > 30 years	Collaborate with USGS, watershed councils, etc.	Competing priorities Lack of funding and personnel	Identify and prioritize areas to monitor

Table 5.7b—Adaptation responses for fisheries and aquatic environments to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Steams and other water bodies (e.g., springs) will be affected by more wildfires and disturbances associated with sediment pulses and debris flow torrents.				
Adaptation strategy: Develop wildfire use plans that address sediment inputs and road failures; reduce sediment input from roads and management activities.				
Restore and revegetate burned areas to store sediment and maintain channel geomorphology	< 10 years	Collaborate with fire management program	Lack of funding if outside of Burned Area Emergency Rehabilitation Competing priorities	
Develop a geospatial layer of debris flow potential for pre-fire planning	< 10 years	Collaborate with soils and geology programs	Lack of funding and personnel Competing priorities	
Use Montgomery-Buffington classification and other tools to calculate runout distance and woody debris source areas	< 10 years		Lack of funding Competing priorities	

Adaptation strategy: Identify hillslope landslide hazard areas and at-risk roads prior to wildfires and as part of fire planning.				
Link stream inventory with topographic, geomorphic, and vegetation layers to assess existing hazard and risk	< 10 years	Collaborate with soils/geology	Lack of funding Competing priorities	Identify hillslope landslide hazard areas and at-risk roads
Develop a process to prioritize tactics needed to protect multiple fish species and populations	< 10 years	Collaborate with fisheries at multiple levels	Competing forest priorities Lack of funding	Need updated fish distribution layer

Table 5.7c—Adaptation responses for fisheries and aquatic environments to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Stream temperatures will increase from rising air temperatures and declining summer flows which will affect many life stages of aquatic organisms.				
Adaptation strategy: Maintain or restore natural thermal conditions to buffer against future climate changes.				
Maintain or restore riparian vegetation to ensure channels are not exposed to increased solar radiation	< 10 years to > 30 years	In project analysis, consider departure from desired conditions/natural range of variability	Competing priorities Lack of funding	Conduct ecologically-based investigations into appropriate vegetation for restoration
Manage livestock grazing to restore ecological function of riparian vegetation and maintain stream bank conditions	< 10 years to > 30 years	Work with range permittees, watershed councils, and other partners to fund or install fencing	Reluctant permittees do not want to do more fence maintenance Competing priorities Lack of funding	Identify streams within range allotments, then prioritize streams based on fish population status
Work with vegetation managers to identify interdisciplinary tactics for restoring riparian function and hydrologic processes	< 10 years to > 30 years	Vegetation management is a high priority for resource management	Project planning and objectives are largely limited to timber products	Identifying how to pay for additional riparian treatments
Increase floodplain connectivity, diversity, and water storage to improve hyporheic and base flow conditions	< 10 years to 30 years	Can identify treatment areas during project planning	High cost and/or social barriers to restoration	Identify and prioritize areas for treatment

Adaptation strategy: Increase connectivity within stream networks so aquatic organisms can access cold water refugia when needed.				
Same as above in table 4.7c	Same as above in table 4.7c	Same as above in table 4.7c	Same as above in table 4.7c	Same as above in table 4.7c
Adaptation strategy: Develop better information about stream temperature regimes.				
Increase temperature data collection and monitoring to better describe patterns in thermal regimes and improve hydrologic models	< 10 years to 30 years	Collaborate with federal and other partners	Competing priorities Lack of funding	Identify and prioritize areas for data collection

Table 5.7d—Adaptation responses for fisheries and aquatic environments to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Headwater and other intermittent streams and water bodies (e.g., springs) will be vulnerable to increasing wildfire; sediment pulses and altered flood patterns and magnitudes will increase with increasing fire frequency.				
Adaptation strategy: Develop wildfire use plans that address sediment delivery.				
Restore and revegetate burned areas to store sediment and maintain channel geomorphology	< 10 years	Burned Area Emergency Rehabilitation (BAER) and fire restoration planning for road decommissioning Fire-killed timber management can provide large wood for slope stabilization per natural fire regime characteristics Other fire restoration needs can be included in fire salvage NEPA	Public/internal desire to provide useable wood to local mills and support local employment and economy Tight timelines for NEPA to conduct fire salvage Lack of funding to conduct non-BAER fire restoration opportunities identified in fire salvage NEPA documents	Identify trees likely to live beyond 10 years postfire Assess how balance ecological values and community economic needs
Develop a geospatial layer of debris flow potential for pre-fire planning	< 10 years	Hydrologists and soil scientists can provide data and criteria to geospatial analyst	Competing priorities May not be a big concern, depending on current availability of data	Identify geology/topography/soil maps/locations of known slide areas and their characteristics

Use Montgomery-Buffington classification and other tools to calculate runout distance and woody debris source areas	< 10 years	Runout distance matters only if debris flow potential is an actual concern	Minimal concerns	Identify geology/topography/soil maps/locations of known slide areas and other areas of concern
Adaptation strategy: Identify hillslope landslide hazard areas and at-risk roads prior to as part of fire planning.				
Link stream inventory with topographic, geomorphic, and vegetation layers to assess existing hazard and risk	< 10 years	Hydrologists and soil scientists can provide data and criteria to geospatial analyst	Minimal concerns	Identify geology/topography/soil maps/locations of known slide areas and other areas of concern
Develop a process to prioritize tactics needed to protect multiple fish species and populations	< 10 years	Convene interdisciplinary fisheries, hydrology, soils, and roads discussions as part of development of watershed action plans, watershed restoration action plans, and water quality restoration plans	Staff time to integrate concerns and information	

Table 6.1—Summary of species-specific model output and scenarios examined. CGCM and Hadley were the common global climate models (GCMs) among the output available

Model name	Scenario / GCM	End of century	Citation	Link
3-PG (hybrid)	A2 CGCM2	2080	Coops and Waring (2011b)	http://www.pnwspecieschange.info/index.html
ForeCASTS	A1(A1FI) HADCM3 no elevation	2100	Hargrove and Hoffman (2005)	http://www.geobabble.org/~hnw/global/treeranges3/climate_change/atlas.html
Plant hardiness	HADGEM and Composite-AR5	2170 - 2100	McKenney et al. (2011)	http://planthardiness.gc.ca/ph_futurehabitat.pl?lang=en
Plant species and climate profile projections	A2/CGCM3 and A2/Hadley	2090	Rehfeldt et al. (2006)	http://forest.moscowfsl.wsu.edu/climate/species/index.php

Table 6.2—Summary of model projections and vulnerability assessment scores for some common upland species in the Blue Mountains. Loss or gain refers to climate habitat, not species range. Species are ordered from the highest vulnerability score to the lowest. Model output is summarized from all four models shown in table 6.1 unless otherwise noted

Potential vegetation or species	Common name	Species model output summary	Vulnerability assessment score[a]
Callitropsis nootkatensis	Alaska cedar	Complete loss[c]	Elevated[b]
Pinus flexilis	Limber pine	Major to complete loss[c]	Elevated[b]
Tsuga mertensiana	Mountain hemlock	Major to complete loss	Elevated[b]
Pinus albicaulis	Whitebark pine	Major to complete loss	74
Abies lasiocarpa	Subalpine fir	Moderate to complete loss, potential refugia in Wallowa Mountains	70
Picea engelmannii	Engelmann spruce	Moderate to complete loss, potential refugia in Wallowa Mountains[c]	61
Pinus monticola	Western white pine	Some loss to major; other models show shifts and minor loss	57
Populus tremuloides	Quaking aspen	Major to complete loss; one model showed minor loss	57
Abies grandis	Grand fir	Some loss to almost complete loss	51
Abies concolor	White fir	Major to complete loss[c]	51
Pinus contorta	Lodgepole pine	Moderate to complete loss	43
Pseudotsuga menziesii	Douglas-fir	Minor to major loss	42
Juniperus occidentalis	Western juniper	Major to complete loss[c]	30
Larix occidentalis	Western larch	Minor loss, shifts to complete loss	32
Pinus ponderosa	Ponderosa pine	Minor to major loss; one model shows minor shift	22
Artemisia tridentata	Big sagebrush	Complete loss[d]	NA
Cercocarpus ledifolius	Curl-leaf mountain-mahogany	Major loss[e]	NA
Purshia tridentata	Antelope bitterbrush	Complete loss[d]	NA

[a] Based on Table 15 in Devine et al. (2012).
[b] These species were not offically ranked in Devine et al. (2012). However, they are considered to have an elevated vulnerability to climate change because they are rare in the Blue Mountains.
[c] Based on three models.
[d] Based on one model.
[e] Based on two models.

Table 6.3—Summary of the upland potential vegetation groups (PVGs) used in this chapter, and the crosswalk between these PVGs and MC2 potential vegetation functional types

PVG	Typical species	MC2 potential vegetation functional type
Cold Upland Forest	Subalpine fir, Engelmann spruce, lodgepole pine, whitebark pine	Subalpine forest
Cold Upland Shrub	Mountain big sagebrush, Sitka alder, shrubby cinquefoil	NA
Cold Upland Herb	Greenleaf fescue, Idaho fescue, forbs, sedges	NA
Dry Upland Forest	Ponderosa pine, grand fir, or Douglas-fir	Temperate and dry temperate needleleaf forest
Dry Upland Herb	bluebunch wheatgrass and Sandberg bluegrass	Temperate (C3) grassland
Dry Upland Woodland	Western juniper, mountain big sagebrush, curl-leaf mountain-mahogany	Temperate needleleaf woodland
Dry Upland Shrub	Low, scabland, and threetip sagebrush	Temperate (C3) shrubland
Moist Upland Forest	Subalpine fir, grand fir, Douglas-fir, lodgepole pine, western larch	Moist temperate needleleaf forest
Moist Upland Woodland	Western juniper, bluebunch wheatgrass, low and scabland sagebrush	Temperate needleleaf woodland
Moist Upland Shrub	Mountain big sagebrush, antelope bitterbrush, snowberries, bitter cherry	Temperate (C3) shrubland
Moist Upland Herb	Idaho fescue, bluebunch wheatgrass	Temperate (C3) grassland

Table 6.4—The distribution of upland potential vegetation groups (PVGs) among the three Blue Mountains Forests. Data are based on maps and boundaries shown in figs. 6.12 – 6.14. Percentages do not add up to 100 because other PVGs (e.g., riparian areas) are not included.

Potential vegetation group	Malheur National Forest	Umatilla National Forest	Wallowa-Whitman National Forest
	------------------------ *percent*	------------------------------	
Cold Upland Forest	11	12	18
Cold Upland Shrub	<1	1	1
Cold Upland Herb	<1	1	4
Dry Upland Forest	70	29	32
Dry Upland Woodland	<1	<1	<1
Dry Upland Shrub	6	1	2
Dry Upland Herb	3	11	13
Moist Upland Forest	5	38	10
Moist Upland Woodland	3	2	<1
Moist Upland Shrub	<1	2	1
Moist Upland Herb	<1	3	2

Table 6.5—Potential soil drought stress index (PSDI) categories derived from an overlay of departure from potential transpiration (AET/PET) and available water holding capacity data. Additional details are provided in Box 6.3. PSDI categories are an estimate of potential stress and range from 1 to 5, where 1 = low, 2 = medium low, 3 = medium, 4 = medium high, and 5 = high

	Departure from potential evapotranspiration				
Available water holding capacity (0-150 cm)	**Low (0.8-1.0)**	**Medium low (0.60-0.80)**	**Medium (0.4-0.6)**	**Medium high (0.2-0.4)**	**High (≤ 0.20)**
High	1	1	2	2	3
Medium high	1	2	3	3	3
Medium	1	2	3	3	4
Medium low	1	2	3	4	5
Low	2	3	4	5	5

Table 6.6—Potential soil drought stress index for the three national forests in the Blue Mountains. Note that not all percentages within a Forest add up to 100 due to rounding and/or because some areas contain no data. Categories are based on mapped data in figs. 6.12–6.14.

Potential soil drought stress index	**Umatilla National Forest**		**Wallowa-Whitman National Forest**		**Malheur National Forest**	
	Spring	**Summer**	**Spring**	**Summer**	**Spring**	**Summer**
	--- *percent* ---------------------------------------					
1 – Low	0	0	0	0	0	0
2 – Low to moderate	5	2	2	1	0	0
3 – Moderate	52	29	46	22	28	11
4 – Moderate to high	42	30	34	26	30	28
5 – High	1	39	9	43	41	60

Table 6.7—Potential vegetation groups (PVGs) and their relative proportion of area mapped in each potential soil drought stress category for each for each national forest in spring (April, May, June) and summer (July, August, September). Values less than 0.5 percent are listed as 0. Categories are based on mapped data shown in figs. 6.12–6.14. Note that for Wallowa-Whitman National Forest, not all mapped PVGs had associated PSDS data. "Unmapped" PSDS data were removed to calculate the relative percentages.

Potential vegetation group	**Potential soil drought stress: spring**					**Potential soil drought stress: summer**				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Malheur National Forest	--------------- *relative percent* ---------------					--------------- *relative percent* ---------------				
Cold Upland Forest	0	0	70	21	10	0	0	40	39	21
Cold Upland Shrub	0	0	31	27	42	0	0	5	23	72
Cold Upland Herb	0	0	28	30	42	0	0	6	16	77
Dry Upland Forest	0	0	23	34	44	0	0	6	30	64
Dry Upland Woodland	0	0	9	51	40	0	0	1	8	92
Dry Upland Shrub	0	0	1	19	79	0	0	1	3	97
Dry Upland Herb	0	0	10	28	62	0	0	2	9	89
Moist Upland Forest	0	0	77	19	4	0	0	43	38	19
Moist Upland Woodland	0	0	4	31	65	0	0	0	5	95
Moist Upland Shrub	0	0	5	19	75	0	0	1	8	91
Moist Upland Herb	0	1	15	69	14	0	0	7	9	83
Umatilla National										

Forest										
Cold Upland Forest	0	1	78	21	0	0	1	46	36	18
Cold Upland Shrub	0	1	29	70	0	0	3	23	40	34
Cold Upland Herb	0	2	21	77	0	0	1	16	31	52
Dry Upland Forest	0	2	55	42	1	0	1	24	33	42
Dry Upland Woodland	0	0	4	96	0	0	0	3	1	96
Dry Upland Shrub	0	1	14	83	2	0	0	3	9	88
Dry Upland Herb	0	1	13	85	1	0	0	3	15	82
Moist Upland Forest	0	12	60	28	0	0	4	41	32	23
Moist Upland Woodland	0	0	14	73	14	0	0	5	6	89
Moist Upland Shrub	0	1	24	74	1	0	0	9	32	59
Moist Upland Herb	0	1	15	82	2	0	0	2	12	86
Wallowa-Whitman National Forest										
Cold Upland Forest	0	0	57	30	12	0	0	32	28	39
Cold Upland Shrub	0	0	7	31	61	0	0	2	13	85
Cold Upland Herb	0	0	32	51	17	0	0	2	16	81
Dry Upland Forest	0	2	58	34	5	0	1	27	33	38
Dry Upland Woodland	0	0	20	35	45	0	0	6	18	76

Dry Upland Shrub	0	0	14	54	32	0	0	2	10	88
Dry Upland Herb	0	0	17	68	15	0	0	2	12	85
Moist Upland Forest	0	4	73	21	2	0	2	39	41	19
Moist Upland Woodland	0	0	8	40	52	0	0	3	10	86
Moist Upland Shrub	0	1	21	51	27	0	0	8	21	72
Moist Upland Herb	0	0	22	63	16	0	0	2	14	83

Table 6.8a—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Larger and more severe fires in forest ecosystems				
Adaptation strategy: Manage forest vegetation to reduce severity and patch size; protect refugia (e.g., old trees)				
Map fire refugia	< 10 years, opportunistic	Coordination with project planning	Lack of knowledge on current location and conditions of fire refugia	Compilation of locations of fire refugia with a consistent protocol
Use gaps and other methods in silvicultural prescriptions to reduce fuel continuity	< 10 years, opportunistic	Coordination with project planning		
Identify processes and conditions that create fire refugia	< 10 years, opportunistic	Coordination with project planning	Lack of knowledge on current location and conditions of fire refugia	Compilation of locations of fire refugia with a consistent protocol

Adaptation strategy: Manage forest landscapes to encourage fire to play a natural role				
Implement fuel breaks at strategic locations	< 10 years	Coordination with roadside treatments Blue Mountains restoration strategy	Limited capacity for National Environmental Policy Act analysis	Maps of lightning and human-caused ignitions
Identify areas where wildland fire use may help to meet management objectives	< 10 years	Use and interpretation of the Wildland Fire Decision Support System	Risk aversion by fire managers	Incorporation of additional information in the Wildland Fire Decision Support System, especially on resources that benefit from fire
Implement strategic density management through forest thinning	< 10 years			
Incorporate climate change in Wildland Fire Decision Support System	< 10 years to > 30 years			Public education on the benefits of fuel reduction and prescribed fire

Table 6.8b—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher wildfire frequency will cause increased mortality of shrub species and increased dominance of invasive species.				
Adaptation strategy: Increase resilience of native sagebrush-grass ecosystems				
Promote the occurrence and growth of early-season native species	< 10 years	Collaboration with range permitees		Effects of a warmer climate on plant phenology
Reduce grazing in July and August to encourage perennial growth	10 to 30 years	Collaboration with range permitees	Opposition by some range permitees; economic issues	
Revise grazing policies, and review and evaluate grazing allotment plans	> 30 years	Possible in some political environments	Entrenched policies for grazing; economic issues; effects on lifestyle	

Adaptation strategy: Maintain vigorous growth of native shrub, perennial grass, and other perennial species, while minimizing the spread of invasive species				
Remove encroaching conifers	< 10 years	Collaboration with range permitees; coordination with fuel reduction projects; coordination with sagebrush restoration projects	Funding issues; potential challenges with invasive species	Synthesis of information on effects of conifer removal; prioritization of treatment areas
Plant seed of native species	< 10 years	Post-fire seeding; coordination with fuel reduction projects; coordination with sagebrush restoration projects	Low success of seeding	Improved acquisition of native seed
Monitor successional patterns of vegetative communities	< 10 years	New LIDAR imaging	Cost of long-term monitoring; difficulty of on-the-ground location accuracy	Framework for timely and efficient monitoring
Adaptation strategy: Maintain reproducing populations of curl-leaf mountain-mahogany, so it can expand as needed				
Strategically protect mountain mahogany populations to allow for natural reseeding after fire	< 10 years to 30 years	After wildfire; during planning for grazing allotment management	Lack of resources; opposition by grazing permitees	Map of existing curl-leaf mountain-mahogany populations

Table 6.8c—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures and increased fire frequency and intensity will reduce the dominance of native grasses and increase dominance of non-native annual species.				
Adaptation strategy: Increase the resilience of native perennial grasses and other non-tree vegetation				
Apply prescribed burning in the spring	Ongoing	Integration in existing fuels reduction programs	Potential increase in invasive species	Information on range condition and high priority areas for treatment
Adaptation strategy: Manage grazing by livestock and ungulates to reduce impacts on perennial grasses				
Focus grazing on non-native species in spring; do not graze natives in summer	< 10 years to 30 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Find locations where late-season grazing has minimal impacts	< 10 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Adaptation strategy: Manage fire to avoid increase in non-native annual species				
Apply prescribed burning in the spring	< 10 years	Integration in existing fuels reduction programs	Potential increase in invasive species; small period of time for implementation	Identification of wildland-urban interface; seasonal fire forecasts
Adaptation strategy: Manage soil conditions to avoid increased runoff after wildfire				

Maximize native vegetative ground cover	< 10 years		Variability of local site conditions	Soil mapping; erosion modeling
Adaptation strategy: Determine potential resilience of different locations, and actively restore less resilient sites				
Increase resilience of native species where intact or productive communities exist	< 10 years	Monitoring of existing vegetative communities		Monitoring of areas that need protection
Decrease resilience of existing non-native species with appropriate management practices	< 10 years to > 30 years	Spring grazing		Improved communication with public
Identify and promote early-successional natives that may be able to compete with non-natives	10 years to > 30 years	Coordination with other projects that have identified suitable early-successional native species and reduction in non-natives		Improved local knowledge and testing of native cultivars

Table 6.8c—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures and increased fire frequency and intensity will reduce the dominance of native grasses and increase dominance of non-native annual species.				
Adaptation strategy: Increase the resilience of native perennial grasses and other non-tree vegetation				
Apply prescribed burning in the spring	Ongoing	Integration in existing fuels reduction programs	Potential increase in invasive species	Information on range condition and high priority areas for treatment
Adaptation strategy: Manage grazing by livestock and ungulates to reduce impacts on perennial grasses				
Focus grazing on non-native species in spring; do not graze natives in summer	< 10 years to 30 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Find locations where late-season grazing has minimal impacts	< 10 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Adaptation strategy: Manage fire to avoid increase in non-native annual species				
Apply prescribed burning in the spring	< 10 years	Integration in existing fuels reduction programs	Potential increase in invasive species; small period of time for implementation	Identification of wildland-urban interface; seasonal fire forecasts

Adaptation strategy: Manage soil conditions to avoid increased runoff after wildfire				
Maximize native vegetative ground cover	< 10 years		Variability of local site conditions	Soil mapping; erosion modeling
Adaptation strategy: Determine potential resilience of different locations, and actively restore less resilient sites				
Increase resilience of native species where intact or productive communities exist	< 10 years	Monitoring of existing vegetative communities		Monitoring of areas that need protection
Decrease resilience of existing non-native species with appropriate management practices	< 10 years to > 30 years	Spring grazing		Improved communication with public
Identify and promote early-successional natives that may be able to compete with non-natives	10 years to > 30 years	Coordination with other projects that have identified suitable early-successional native species and reduction in non-natives		Improved local knowledge and testing of native cultivars

Table 6.8d—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Uncharacteristic size and severity of ecological disturbances in forest ecosystems: insects				
Adaptation strategy: Recognize natural role of insect disturbances, and identify areas at high risk				
Allow for natural mortality within the historical range of variability for specific insects	< 10 years, depending on insect cycle	National forest land management plan revision	Public opposition	Identify scale at which mortality is acceptable relative to current objectives
In dry forest, restore low-severity fire to lower stand density and increase resilience to bark beetle outbreaks	10 to > 30 years	Coordination with project planning	Inadequate resources to conduct restoration at large spatial scales	
Adaptation strategy: In forest types where the risk of insect outbreaks is high, promote diversity of forest age and size classes				
Diversify large contiguous areas of single age and size classes		Coordination with project planning	Inadequate resources; public opposition; internal opposition	Historical/desired conditions for landscape pattern and patch size

Table 6.8e—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: A warmer climate with more droughts will reduce growth in most forests and make regeneration more difficult for some species.				
Adaptation strategy: Protect genotypic and phenotypic diversity				
Protect trees that exhibit adaptation to water stress (e.g., trees with low leaf area:sapwood); collect seed for future regeneration	<10 years	Coordination with project planning	Identification of trees with low leaf:sapwood area by marking crews; Identification of successful adaptation by trees to stress	Better understanding of the phenotypic adaptive capacity of different tree species
Maintain variability in species and in tree architecture	10 to 30 years	Coordination with project planning		
Adaptation strategy: Maintain and enhance forest productivity regardless of tree species; focus on functional ecosystems and processes				
Manage tree densities to maintain tree vigor and growth potential	< 10 to >30 years	Coordination with project planning		
Prepare for species migration by managing for multiple species across large landscapes	10 to 30 years	Coordination with project planning		
Maintain soil productivity through appropriate silvicultural practices	>30 years	Coordination with project planning		

Adaptation strategy: Use judicious managed relocation of genotypes where appropriate				
Push boundaries of seed zones and plant genotypes from warmer locations; used a variety of genotypes rather than just one	10 to 30 years, opportunistic	Coordination with forest regeneration projects		
Adaptation strategy: Use tree improvement programs to ensure availability of drought tolerant tree species and genotypes				
Develop seed orchards that contain a broader range of tree species and genotypes than in the past	10 to > 30 years	Good foundation in existing seed orchards	Time required to develop new capacity	Data on tree genetics across broad landscapes to guide development of nursery stock and regeneration strategies

Table 6.8f—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures and increasing drought will stress some species in moist mixed conifer forests, especially western larch.				
Adaptation strategy: Maintain vigorous existing western larch and encourage its regeneration				
Create gaps in forests to reduce competition and increase larch vigor	<10 years, opportunistic	Tree mortality of other species, which creates gaps for larch regeneration (after wildfire and insect outbreaks)	Public opposition to large forest openings	Data on current distribution of larch; studies on gap sizes needed for regeneration
Regenerate larch with appropriate site preparation (e.g., prescribed burning, followed by planting)	<10 years, opportunistic	Following timber harvest, wildfire, and insect outbreaks		

Table 6.8g—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures may increase stress for some species in cold upland and subalpine forests.				
Adaptation strategy: Protect rare and disjunct tree species (Alaska cedar, limber pine, mountain hemlock, and whitebark pine)				
Plant and encourage regeneration of rare and disjunct species in appropriate locations	Opportunistic	Following wildfire		
Plant whitebark pine genotypes that are resistant to white pine blister rust	< 10 years, opportunistic		Limited availability of rust-resistant nursery stock	

Table 6.8h—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures may increase stress for some alpine plant species (including rare plants), especially in the Wallowa Mountains.				
Adaptation strategy: Improve our understanding of the effects of climatic variability and change on alpine plant species.				
Install GLORIA plots to monitor species distribution and abundance	< 10 years			
Collaborate with other federal agencies to monitor alpine species	< 10 years to 30 years			

Table 7.1—Functions of riparian areas and key relationships to ecological services (modified from Dwire et al. 2010, Naiman et al. 2005, NRC 2002)

Riparian functions	Indicators of riparian functions	On-site or off-site effects of functions	Goods and services provided	Relevant riparian plant communities
Hydrology and sediment dynamics				
Short-term storage of surface water	Connectivity of floodplain and stream channel	Attenuates downstream flood peaks	Reduces damage from floodwaters	Shrub (especially willow) and herbaceous dominated; all communities to some extent
Maintenance of high water table	Presence of flood tolerant, hydrophytic, and mesic plant species	Maintains distinct vegetation, particularly in arid climates	Contributes to regional biodiversity through provision of habitat	Shrub (especially willow) and herbaceous dominated; all communities to some extent
Retention and transport of sediments; decreased stream bank erosion	Riffle-pool sequences, point bars, floodplain terraces, and bank stability	Contributes to fluvial processes	Creates predictable yet dynamic channel and floodplain features	Shrub (especially willow) and herbaceous dominated; all communities to some extent
Biogeochemistry and nutrient cycling				
Riparian vegetation provides source of organic carbon (allochthonous inputs to streams; organic matter inputs to soils)	Healthy mosaic of riparian vegetation	Maintains aquatic and terrestrial food webs	Supports terrestrial and aquatic biodiversity	Highest quality inputs are from deciduous shrubs and herbaceous dominated; all communities to some extent

Transformation and retention of nutrients and pollutants	Water quality and biotic indicators	Intercepts nutrients and toxicants from runoff; water quality	Improves and maintains water quality	Retention and interception most likely in shrub (especially deciduous) and herbaceous dominated communities; all communities to some extent
Sequestration of carbon in riparian soils	Occurrence, extent, and distribution of organic-rich soils	Contributes to nutrient retention and carbon sequestration	Potentially ameliorates climate change; provides source of dissolved carbon to streams via subsurface flow	Carbon sequestration in soils highest in herbaceous dominated meadows; all communities to some extent
Distinctive terrestrial and aquatic habitat				
Contributes to overall biodiversity and biocomplexity	High species richness of plants and animals	Provides reservoirs for genetic diversity	Supports regional biodiversity	All communities, but especially those with multiple canopy strata, {e.g. cottonwood, aspen and willow dominated)
Maintenance of streamside microclimate	Presence of shade-producing canopy; healthy populations of native terrestrial and aquatic biota	Provides shade and thermal insulation to stream; provides migratory corridors for terrestrial and aquatic species	Maintains habitat for sensitive species (e.g., amphibians, cold-water fishes)	All communities, but especially those with multiple canopy strata, (e.g. conifer, aspen, cottonwood and willow dominated)

Contribution to aquatic habitat; provision of large wood	Aquatic habitat complexity (pool-riffle sequences, debris dams); maintenance of aquatic biota	Maintains aquatic biota	Maintains fisheries, recreation	Conifer, aspen and cottonwood dominated
Provision of structural diversity	Availability of nesting/rearing habitat; presence of appropriate indicator wildlife species (e.g. neotropical migrants)	Maintains biodiversity; provides migratory corridors for terrestrial and aquatic species	Provides recreation opportunities (e.g., birding, wildlife enjoyment, hunting)	All woody communities, but especially those with multiple canopy strata, e.g. conifer, aspen, cottonwood and shrub dominated

Table 7.2—Stressors in riparian and wetland ecosystems, all of which are likely to be exacerbated by climate change (modified from Theobald et al. 2010). Although nearly all riparian and wetland plant associations are affected by each stressor, the most affected associations are noted in the last column

Stressor	Direct and indirect causes	Potential effects	Riparian and wetland plant communities and associations most affected by climate change
Changes in flow regime and dewatering	Surface water: dams, diversions, land-use changes, climate change Groundwater: pumping, land-use change, climate change	Water stress in vegetation Shifts in plant species composition Homogenization of riparian area and simplification of biota Isolation of floodplain from stream Altered stream-riparian organic matter exchange and trophic dynamics Altered floodplain biogeochemistry Altered channel structure Decreased lateral extent of riparian area	Cottonwood, aspen, willow, and herbaceous-dominated communities located along low-gradient, wide valley bottoms
Channelization	Bank hardening Levee construction Structural changes in channel-deepening Berm development	Isolation of floodplain from stream Altered fluvial processes Altered hydraulics (aquatic habitat and	Cottonwood, aspen, willow, and herbaceous-dominated communities located along low-gradient, wide valley bottoms

	Meander cutoff	channel forms) Altered floodplain biogeochemistry	
Conversion of floodplains to other uses	Removal of woody riparian vegetation	Elimination of local populations of cottonwood, aspen, willow, and herbaceous communities Reduced extent of riparian area, thus reducing ecosystem services (maintenance of water quality, wildlife habitat, recreation)	Cottonwood, aspen, willow, and herbaceous-dominated communities located along low-gradient, wide valley bottoms
Invasive species	Altered physical and ecological processes that facilitate establishment and spread (e.g., herbivory, changes in flow regime)	Displacement of native species Formation of monoculture Altered site characteristics (e.g., biogeochemistry, soil properties, water balance) Shifts in community composition Altered habitat structure	Nearly all riparian and wetland communities, especially those that occur in drier environments Potential increase in tamarisk in Hells Canyon
Changes in sediment delivery to channel	Off-road vehicle use Roads (drainage, gravel application) Livestock and herbivore	Shifts in channel and floodplain form (through increased or decreased delivery to channel)	Nearly all riparian and wetland communities, although direct causes and severity will differ

	trampling Altered vegetative cover in watershed and along channel Direct mechanical effects on channel, dams, and diversions	Altered channel processes (e.g., incision and aggradation)	
Herbivory	Grazing by cattle and wild ungulates	Bank trampling and compaction Altered cover and composition of vegetation Stream capture Nutrient inputs	Aspen, cottonwood, willow and herbaceous communities are the most heavily impacted, but most riparian and wetland communities affected to some extent
Wildfire and fuels, fire suppression	Fuel buildup from invasive species and fire exclusion Reduced flooding Slower decomposition of organic material	Increased frequency and intensity of fires Loss of fire intolerant taxa Altered structure of riparian vegetation and habitat quality and distribution, with subsequent shifts in composition	Conifer-dominated riparian plant associations with dominant tree species similar to adjacent uplands
Insects and disease	Fire exclusion and past harvest activities have resulted in susceptible stand structure	Altered fuel loads and distribution associated with increased canopy mortality	Conifer-dominated riparian plant associations with dominant tree species similar to adjacent uplands

Table 7.3—Number of springs (named and unnamed) and wetlands for the Malheur, Umatilla, and Wallowa-Whitman National Forests. The number of springs was derived from the National Hydrography Database. The number of wetlands was derived from the Oregon Wetlands Geodatabase and excludes national forest land in Washington (Umatilla) and Idaho (Hells Canyon NRA, Wallowa-Whitman). This database identified "potential fens" if a wetland, usually palustrine, occurred near a spring, so overlap exists between the number of palustrine wetlands and number of potential fens

	Springs			Wetlands				
National Forest	Named	Unnamed	Total	Palustrine	Lacustrine	Riverine	Total	Potential fens
Malheur	389	2,462	2,851	4,405	8	4,648	9,061	1,132
Umatilla	268	381	649	2,472	5	1,780	4,257	568
Wallowa-Whitman	273	1,635	1,908	5,419	77	4,886	10,382	1,037
Total	930	4,478	5,408	12,296	90	7,314	23,700	2,737

Table 7.4—Area of different wetland types and percentage of forest area for the Malheur, Umatilla, and Wallowa-Whitman National Forests. Wetland area was derived from the Oregon Wetlands Geodatabase and excludes national forest land in Washington (Umatilla) and Idaho (Wallowa-Whitman, Hells Canyon National Recreation Area). Potential fens are classified primarily as palustrine wetlands and are included in the area calculated for palustrine wetland area.

National Forest	Area	Wetland type: Palustrine		Lacustrine		Riverine		Potential fens	
	Hectares	*Hectares*	*Percent*	*Hectares*	*Percent*	*Hectares*	*Percent*	*Hectares*	*Percent*
Malhuer	696,895	4,552	0.7	62	<0.001	1,963	0.3	967	0.15
Umatilla	442,428	2,091	0.5	104	<0.001	1,669	0.4	556	0.001
Wallowa-Whitman	914,115	3,897	0.4	1,447	0.01	4,458	0.5	619	0.06
Total		10,540		1,613		8,090		2,142	

Table 7.5—Number of different types of groundwater-dependent ecosystems (GDEs) affected by water diversion for 102 GDEs in Umatilla National Forest. Permanent diversion includes some types of infrastructure that withdrew emerging water away from the spring habitat. See text for definitions of GDE types

	Helocrene	Hillslope	Hypocrene	Mound	Rheocrene	Unclassified	Total
No permanent diversion	8	2	1	1	39	3	54
Permanent diversion	5	2	0	1	33	5	46
Missing data	0	1	0	0	1	0	2
Total	13	5	1	2	73	8	102

Table 7.6—Selected management indicators (sensu USDA FS 2012c) for 62 groundwater-dependent ecosystem sites in Umatilla National Forest. See text for explanation of management indicators

Management indicator	No concern	Issue of concern	Not applicable
Aquifer functionality	20	35	7
Soil integrity	42	15	5
Vegetation composition	47	11	4

Table 7.7a—Adaptation responses to climate change for riparian and wetland systems in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Shifts in hydrologic regime include changes in timing and magnitude of flows, lower summer flows and higher, more frequent winter peak flows. Reduced snowpack will decrease water supply during growing season and lead to more variable streamflow, thus reducing productivity in riparian ecosystems.				
Adaptation strategy: Maintain appropriate densities of native species, and propagate more drought tolerant native species.				
Plant species that have a broader range of moisture tolerance (e.g., mock orange, choke cherry)	10-30 years, > 30 years	Collaborate with Idaho Power, range permittees, and private landowners	High cost Limited availability of local plant material	Identify segments along stream-riparian corridors with need for shade that meet growth requirements of desirable woody species
Eradicate and control invasive species where possible (especially after fire)	< 10 years, 10-30 years	Collaborate with range permittees, private landowners, counties and Idaho Power	High cost	Identify riparian areas (stream segments) with high cover of most noxious non-native species; prioritize treatment areas.
Remove infrastructure where appropriate (e.g., campsites, utility corridors, springhouses, and spring boxes)	< 10 years	Various management plans	High cost Public opposition; constraints of existing water rights for modifying spring developments.	Identify infrastructure that can be removed; prioritize decommissioning and removal.
Adaptation strategy: Maintain or restore natural flow regime to buffer against future changes.				
Develop integrated, interdisciplinary tactics to maintain or restore natural flows; purchase and obtain in-stream	< 10 years	Collaborate with nongovernmental organizations, other federal and state agencies, and private landowners	Competing priorities Lack of funding	Using watershed analysis, watershed condition framework, or other approaches to identify priority watersheds and stream

flow rights where possible				segments
Restore riparian areas and beaver populations to maintain summer base flows and raise water table	< 10 years to > 30 years	Collaborate with states, range permittees, and private landowners regarding beaver re-entry Use range allotment management, riparian shrub planting and protection, riparian aspen restoration and management, road infrastructure planning, and valley form analysis to assess potential sites for beaver colonies and channel migration	Effects on infrastructure (culverts and diversions) Adequate food supply at for growing beaver colonies and dispersing individuals Public and private landowner acceptance of beaver colonies Effects of rising water levels on streamside roads and camping	Improve GIS layer for distribution of riparian community types relative to valley bottom classification Identify watersheds and stream reaches with highest potential for successful establishment Monitor beaver populations Inventory riparian vegetation and stream morphology status and trends
Address water loss at points of water diversion and along ditches.	< 10 years	Coordinate with the water district managers, water master, and water users	Competing priorities Lack of funding Water law does not address ditch water loss, inefficiencies, and other impacts on forest resources	Identify diversions/ and ditches with water losses or other impacts on forest resources
Anticipate new proposals for development of water infrastructure (additional diversions or reservoir expansions).	< 10 years	Develop approach for protection of streamflow and maintenance of water levels in wetlands	Competing priorities Constraints of water law	Identify priority watersheds and stream reaches; define flows required for sediment transport, and maintenance of riparian and wetland vegetation and aquatic habitat.
Reconnect and increase off-channel habitat and refugia in side channels and channels supported by sideslope wetlands.	< 10 years to > 30 years	Collaborate with partners	Competing priorities Lack of funding	Identify potential watersheds and stream reaches

Revegetate, fence to exclude livestock, acquire water rights	< 10 years	Collaborate with partners to fence or acquire or lease water rights Revegetate and protect riparian areas Restore aspen	Competing priorities Lack of funding Difficulty of maintaining livestock and ungulate exclusions	Investigate baseline vs. desired conditions in riparian corridors Conduct effectiveness monitoring Identify water rights holders willing to collaborate
Adaptation strategy: Improve soil health (including bank stability) and increase resilience of native vegetative communities to maintain natural water storage.				
Reduce degradation by livestock; fence riparian areas, and use rest rotation.	Ongoing	Adjust range allotment management where possible; collaborate with range permittees and private land owners	Public opposition	Identify stream-riparian reaches for fencing, and/or changes in grazing management.
Manage fuel loads through various prescriptions (prescribed fire, mechanical treatments).	10-30 years	Forest plans for fire and fuels management	High cost Logistical challenges of prescribed fire use in riparian areas	Quantify riparian fuel loads (relative to uplands) and effectiveness of various fuel reduction treatments

Table 7.7b—Adaptation responses to climate change for high-elevation riparian systems in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Reduced snowpack will decrease water supply during growing season and lead to more variable streamflow, thus reducing productivity in riparian systems in alpine and subalpine ecosystems.				
Adaptation strategy: Reduce stresses such as conifer encroachment, livestock grazing, and ungulate browsing				
Consider riparian fuel reduction strategies in forested subalpine areas, including small-scale fuel breaks	< 10 years	Coordination with other fuels management and restoration projects	Logistical constraints	Assessment of riparian fuel characteristics Identify strategic locations for fuel breaks
Reduce degradation by livestock	Ongoing	Adjust range allotment management where possible; collaborate with range permittees and private land owners	Public opposition	Identify stream-riparian reaches for fencing, and/or changes in grazing management.

Table 7.7c—Adaptation responses of groundwater-dependent ecosystems to climate change in the Blue Mountains

Adaptation tactic	Time frame	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Reduced snowpack will decrease water supply during growing season, thus reducing productivity in groundwater dependent systems, including springs and wetlands.				
Adaptation strategy: Manage for resilience of springs and wetlands by considering the broader forest landscape including uplands.				
Consider impacts and potential benefits of vegetation management treatments (prescribed fire, mechanical thinning)	< 10 years	Coordination with other fuels management and restoration projects	Lack of scientific and site-specific information on spring ecology	Inventory and characterization of springs, other GDEs and surrounding forest conditions
Protect groundwater recharge areas	< 10 years	Explore opportunities with range permittees, nongovernmental organizations, and others	Limited information on locations of springs, recharge areas, and groundwater input to stream channels Lack of funds to address sites affected by livestock Competing priorities	Locate and characterize groundwater dependent ecosystems and spring influences
Adaptation strategy: Manage water to maintain springs and wetlands; improve soil quality and stability.				
Decommission roads and reduce road connectivity to encourage interception and retention of water	< 10 years	Incorporate into Forest Plans as watershed improvement strategy	High cost Public opposition Competing priorities	Identify and prioritize project areas

Reduce ungulate trampling with fencing	< 10 years	Work with range permittees, watershed councils, and other partners to fund or install fencing	High cost Need for maintenance	Identify and prioritize springs and wetlands for fencing Monitor effectiveness of fencing projects
Maintain water on site through water conservation techniques such as float valves, diversion valves, and hose pumps	< 10 years	Incorporate into Forest Plans as watershed improvement strategy; specify in allotment management plans	Range permittees do not like these practices	Identify permittees and water users who are willing to collaborate
Encourage spring development project designs that will ensure environmental flows for native species and habitat.	< 10 years	Proposed, new, or redeveloped livestock watering projects		Conduct environmental flow/level analysis
• Collect no more water than is sufficient to meet the intended purpose of the spring development.	< 10 years	Proposed, new, or redeveloped livestock watering projects		Conduct environmental flow/level analysis
• Include implementation and effectiveness monitoring to evaluate success of the project in meeting design objectives and avoiding or minimizing unacceptable impacts to spring ecology.	< 10 years	Incorporate into Forest Plans as watershed improvement strategy		
• Use suitable measures to maintain desired downstream temperatures, dissolved oxygen levels, and aquatic habitats when water is released from a	< 10 years	Proposed, new, or redeveloped livestock watering projects		

pond trough or impoundment.				
• Use float valves or other flow control devices to provide flow only when a demand is present.	< 10 years	Incorporate into Forest Plans as watershed improvement strategy		
• Use a flow splitting devise to retain as much flow in the spring and associated habitat as possible.	< 10 years	Proposed, new, or redeveloped livestock watering projects		
• Consider discontinuing use of a water resource in critical habitats.	< 10 years	Allotment management plan revision		Ecological assessment
Relocate water troughs away from springs and riparian areas to limit trampling.	< 10 years	Proposed, new, or redeveloped livestock watering projects		
Change the duration, season, or intensity of grazing if the current grazing strategy inhibits natural recovery at a given site.	< 10 years	Allotment management plan revision		Determine the duration, season, and intensity of grazing with the least impact for a given site.

Boxes

Box 3.1—Summary of climate change effects on hydrology in the Blue Mountains

Broad-scale climate change effect
Warming temperatures will lead to decreased snowpack accumulation, earlier melt out, resulting in shifts in timing and magnitude of streamflow and decreased summer soil moisture. Both peak and low flows may be affected.

Habitat, ecosystem function, or species
Changes in streamflow timing and magnitude could potentially affect all aquatic species and riparian vegetation through either increased or decreased peak flows and/or decreased summer streamflows. Changes in soil moisture could potentially affect most terrestrial vegetation.

Current condition, existing stressors
Vegetation water stress varies annually under current climatic regime but may become more pronounced in the future; topographic influences on precipitation (orientation and orographic effect); legacy land use impacts.

Sensitivity to climatic variability and change
Because of the fundamental role of water in all ecosystems, changes in availability, timing, and volume of water will have ramifications for most terrestrial and aquatic ecosystems.

Expected effects of climate change
The most pronounced changes in snow/streamflow in the Blue Mountains are likely to occur in headwater basins of the Wallowa Mountains, notably the higher elevation radial drainages out of the Eagle Cap Wilderness, with other large changes occurring in the more northerly sections of the Umatilla and Wallowa-Whitman National Forests along the Oregon-Washington border.

Adaptive capacity
Variable. Key challenges posed by changes in hydrology include increased water stress to vegetation with consequences for fire, mortality, growth, etc. Hydrological changes to streamflow are likely to impact some fish species more than others (see Chapter 5).

Geographic locations most vulnerable
Mid-elevations; areas where snow is not persistent (e.g., Northern Blue Mountains, margins of Wallowa, Elkhorns, Greenhorn and Strawberry Mountains)

Risk Assessment

Potential magnitude of climate change effects

- For those regions determined to be sensitive, as identified in the hydrological assessment products
    - Moderate magnitude by 2040
    - High magnitude by 2080

Likelihood of climate change effects

- For those regions determined to be sensitive, as identified in the hydrological assessment products
    - Moderate likelihood by 2040
    - High likelihood by 2080

Box 4.1—Summary of climate change effects on water resource use in the Blue Mountains

Broad-scale climate change effect
Decreasing snowpack and declining summer flows alter timing and availability of water supply.

Resource entity affected
Drinking water supply for municipal and public uses both downstream from and on the national forests, other forest uses including livestock, wildlife, recreation, firefighting, road maintenance, and in-stream fishery flows. Change in availability of water supply to meet human uses increases the risk of scarcity and not satisfying consumptive and in-stream needs.

Current condition, existing stressors
All basins are fully allocated in terms of water available for appropriation under state law. On national forests, water is generally available for campgrounds and administrative sites and for other appropriated uses (e.g., livestock and wildlife), although in dry years availability may be limited at some sites, especially in late summer. In drought years, downstream "junior" users may not receive water for various purposes, primarily irrigation. Six municipalities rely directly on the national forests for drinking water supply (Baker City, La Grande, Walla Walla, Pendleton, Long Creek and Canyon City). Ecological effects include the following: on-forest dams for storage facilities and stream diversions affect stream channel function; and development of springs and ponds for livestock impact many "groundwater dependent ecosystems". Changes in water supply are expected to influence water use by vegetation, exacerbate low soil moisture, and influence fire frequency with various secondary effects on water supply and quality.

Sensitivity to climatic variability and change
Regional water supplies are highly dependent on snowpack extent and duration; April 1 snow water equivalent is the traditional indicator of late season water availability. Declining summer low flows will affect water availability during this period of peak demand (e.g., for irrigation and power supply).

Expected effects of climate change
Decreased snowpack extent and duration are expected to affect the timing and availability of water supply, particularly in late summer when demand is high for both consumptive and in-stream uses. Decreased summer low flows will limit water availability during critical times and for multiple uses.

Adaptive capacity
Within-forest impacts will likely be less than downstream, although some facilities may see reduced water supply in late summer. Annual operating plans could be adjusted to limit potential for effects on streams and springs by permitted livestock and wildlife during dry years. Off-forest effects may be greater, although some municipalities have back-up wells, water supply for

drinking water and other uses likely to be affected. There is a need for coordination at the county and state level for conservation planning. There may be an increase in proposals for groundwater and surface storage development.

Risk assessment

Potential magnitude of climate change effects

- For those watersheds determined to be sensitive
  - Moderate magnitude by 2040
  - High magnitude by 2080

Likelihood of climate change effects

- For those watersheds determined to be sensitive
  - Moderate likelihood by 2040
  - High likelihood by 2080

Box 4.2—Summary of climate change effects on roads and infrastructure in the Blue Mountains

Broad-scale climate change effect
Increase in magnitude of winter/spring peak streamflows.

Resource entity affected
Infrastructure and roads near perennial streams, which are valued for public access.

Current condition, existing stressors
Many miles of roads are located close to streams on the national forests, and these roads have high value for public access and resource management. A large backlog of deferred maintenance exists because of decreasing budget and maintenance capacity. Many roads are in vulnerable locations subject to high flows.

Sensitivity to climatic variability and change
Roads in near-stream environments are periodically exposed to high flows. Increased magnitude of peak flows increases susceptibility to effects ranging from minor erosion to complete loss of the road prism, resulting in effects on public safety, access for resource management, water quality, and aquatic habitat.

Expected effects of climate change
Projections for increased magnitude of peak flows indicate that more miles of road will be exposed to higher flow events and greater impacts.

Adaptive capacity
Knowing the extent and location of potentially vulnerable road segments will help with prioritizing scarce funding, targeting "storm damage risk reduction" treatments, and communicating potential hazard and risk to the public.

Risk Assessment

Potential magnitude of climate change effects

- For those watersheds determined to be sensitive
  - Moderate magnitude by 2040
  - High magnitude by 2080

Likelihood of climate change effects

- For those watersheds determined to be sensitive
  - Moderate likelihood by 2040
  - High likelihood by 2080

Box 4.3—Sensitivities of the transportation system in national forests in the Blue Mountains

- Aging and deteriorating infrastructure increases sensitivity to climate impacts, and existing infrastructure are not necessarily designed for future conditions (e.g., culverts are not designed for larger peak flows).

- Roads and trails built on steep topography are more sensitive to landslides and washouts.

- A substantial portion of the transportation system is at high elevation, which increases exposure to weather extremes and increases the costs of repairs and maintenance.

- Roads built across or adjacent to waterways are sensitive to high streamflows, stream migration, and sediment movement.

- Funding constraints, insufficient funds, or both limit the ability of agencies to repair damaged infrastructure or take preemptive actions to create a more robust system.

- Design standards or operational objectives that are unsustainable in a new climate regime may increase the frequency of infrastructure failure in the future.

Box 4.4—Exposure of access to climate change in the Blue Mountains

Current and short-term exposures (less than 10 years)

- Roads and trails are damaged by floods and inundation because of mismatches between existing designs and current flow regimes.
- Landslides, debris torrents, and sediment and debris movement block access routes and damage infrastructure.
- Traffic is affected by temporary closures to clean and repair damaged roads and trails.
- Frequent repairs and maintenance from damages and disruption incur higher costs and resource demands.

Medium-term exposures (intensifying or emerging in approximately 10 to 30 years)

- Flood and landslide damage will likely increase in late autumn and early winter, especially in mixed-rain-and-snow watersheds.
- Current drainage capacities may become overwhelmed by additional water and debris.
- Increases in surface material erosion are expected.
- Backlogged repairs and maintenance needs will grow with increasing damages.
- Demand for travel accommodations, such as easily accessible roads and trails, is projected to increase, which could increase travel management costs.
- Increased road damage will challenge emergency response units, making emergency planning more difficult.

Long-term exposures (emerging in 30 to 100 years)

- Fall and winter storms are expected to intensify, greatly increasing flood risk and infrastructure damage and creating a greater need for cool-season repairs.
- Higher streamflows will expand channel migration, potentially beyond recent footprints, causing more bank erosion, debris flows, and wood and sediment transport into streams.
- Changes in hydrologic response may affect visitation patterns by shifting the seasonality of use.
- Shifts in the seasonality of visitation may cause additional challenges to visitor safety, such as increased use in areas and during seasons prone to floods and avalanches.
- Managers will be challenged to provide adequate flexibility to respond to uncertainty in impacts to access.

Box 5.1—Summary of climate change effects on fisheries in the Blue Mountains

Broad-scale climate change effects

- Water temperatures will increase.
- Snowpack will decrease, causing a shift of peak flows from summer to spring, and a decrease in summer flows.
- Peak flows will be flashier, likely resulting in channel scouring and increased sedimentation.
- Wildfires will increase, creating the potential for increased erosion.
- Ocean productivity may change, affecting anadromous fish species.

Species affected

- Fish species affected will differ by location. This assessment focuses on bull trout, spring Chinook salmon, and redband trout/steelhead.

Current condition and existing stressors

- Altered riparian areas (from grazing, roads, recreation, etc.) – lead to elevated water temperatures
- Reduced resiliency of stream habitat affected by removal of woody debris, roads, grazing, recreation, etc.
- Reduced summer flows – related primarily to water withdrawals
- Invasive species – predation on spring Chinook salmon and steelhead
- Influence of hatchery fish – spring Chinook salmon and steelhead
- Overfishing for spring Chinook salmon

Sensitivity to climatic variability and change

- Bull trout – Bull trout require very cold water temperatures for spawning and juvenile rearing, so their populations are already greatly constrained by the limited availability of cold habitats in the Blue Mountains. Moreover, these natal habitats are confined at their upstream extent by small stream size and channel slope, so bull trout populations are the fish world's equivalent to terrestrial species that are trapped on "mountain-top islands." Increasing temperatures will eliminate cold habitats at the downstream extent of these populations. Decreasing summer flows will make some streams too small to support fish at the upper extent, and the habitats that remain will be subject to increased environmental fluctuation associated with wildfires, debris flows and increased winter flooding.
- Spring Chinook salmon – Adults require cool water for holding in fresh water prior to spawning. Juveniles could also be affected by warmer temperatures. A major impact may be changes in the size of adults and fecundity of females resulting from ocean acidification and warming. Spring Chinook salmon adults migrate upstream into low elevation mainstem rivers and streams to spawn during the warmest months of the year. During especially warm summers, adult salmon sometimes experience

direct mortality from thermal stress prior to spawning. These mortality events are exacerbated by decreases in summer flows, which confine the fish to smaller areas. Anticipated temperature increases will increase the frequency and severity of thermal stresses on spring Chinook salmon.

- Redband trout/steelhead – Most steelhead populations in the Pacific Northwest have reduced populations because they already have experienced changes in stream temperature and flow patterns as a result of land and water use practices. Both the resident and anadromous forms are important to maintain many overall populations. Both life forms have a lot of phenotypic plasticity to withstand and adapt to change, but climate change will likely increase competition between these life forms and will certainly affect populations that use lower elevations that are more susceptible to change. The loss or reduction in the larger anadromous form will mean fewer eggs which may affect the overall population in each Major Population Group. This species has a large adaptive capacity with a range of life histories and wide environmental tolerances, which will reduce sensitivity to the potential effects of climate change. Altered ocean conditions may have an effect on the expression of the steelhead life history.

Expected effects of climate change

- Bull trout – Changes in water temperature will be an important determinant of persistence. Long-term climate patterns suggest both an expected decrease in the total amount of cold water stream habitat and fragmentation of some colder areas into disconnected “patches” of suitable habitat. Bull trout populations will likely increase retreat into these shrinking summer cold water refuges to avoid warming conditions. These restricted tributary populations may become more vulnerable to local extirpation. Many remaining patches will be subjected to more frequent winter peak flows that will scour the streambed and destroy redds and/or kill newly emerged fry. Conceivably, the combined effects of shrinking patch size and increasing frequency or magnitude of stream channel disturbance could chip away at the low resiliency of these populations, leaving them in a poorer condition to withstand the next series of disturbances, and accelerating the rate of local extirpations beyond that driven by temperature alone. Wildfire effects are less clear, although minimal empirical data suggest that wildfire will have minimal effects on bull trout.
- Spring Chinook salmon – Elevated temperatures can reduce the energy available for reproduction and cause mortality, particularly where there are sudden increases in water temperatures. Effects from higher winter floods will probably be minimal unless there is a very large increase. These fish are large, use large substrates for spawning, and spawn in areas where the energy of flood water can be dissipated, so the potential for increased scouring is likely to minimal.
- Redband trout/steelhead – This species is a spring spawner and has a relatively warm thermal niche; it can move into some cold upstream areas as refugia. Because it spawns in the spring, eggs are not as susceptible to increased winter flooding as fall

spawners. Shifts in the timing of peak flows will likely result in changes in outmigration timing, changes in survival, changes in distribution, and changes in the availability of spawning and rearing habitats. Increases in the temperature of cooler waters (i.e., those <9°C currently) could improve habitat for redband trout. Further temperature increases (12-20 °C) could lead to greater expression of the steelhead life history. It is less clear what the trade-off between redband trout and steelhead life histories will be at water temperatures > 20 °C. Altered ocean conditions could reduce the expression of the steelhead life history, particularly in males, and reduce the size of returning adults.

Adaptive capacity

- Bull trout – This species has very limited capacity to adapt given their limited current habitats and restrictive ecological tolerances.
- Spring Chinook salmon – The adaptive capacity is probably presently compromised because of low population numbers. T. Beechie, NOAA Fisheries Seattle, has suggested the spring Chinook salmon (stream-type life history) in Puget Sound may become fall Chinook salmon (ocean-type life history) in response to changes of climate change (warmer water and lower summer flows). This could be possible in the Columbia Basin but is not as likely as in Puget Sound. These fish may be able to adjust to altered ocean conditions by reducing the time spent in the ocean, resulting in smaller, younger fish returning to streams.
- Redband trout/steelhead – This species has high adaptive capacity for adjusting to potential effects of climate change.

Vulnerable geographic locations

- Some spring Chinook salmon stocks (e.g. Catherine Creek stock and Upper Grande Ronde stock) are already in poor shape.
- Spring Chinook salmon in Catherine Creek may lose one outmigration period due to shift in hydrograph and therefore lose species life history diversity.
- Fish of all species will be vulnerable to decreased summer flows in the future. Flow reductions will make some headwater streams too small to support fish. In the remainder of the network, summer flow declines will also reduce the overall amount of "living space" for fish.
- The nests of fall spawning fish like bull trout or spring Chinook salmon could be vulnerable to higher winter flows causing channel scour and egg mortality in channels with confined valley bottoms. This vulnerability would occur just upstream at higher elevations of where winter high flows have occurred because warming will make things creep upstream.
- Places where fish distribution and abundance are currently limited by warm temperatures could experience fish declines with small amounts of additional warming. Bull trout distributions are limited at downstream sites in many streams and will have to shift upstream. Spring Chinook salmon adults have experienced heat stress, die-offs in a few streams, and are forced to cluster near cold-water refuges in

other streams. Additional warming will exacerbate these issues.

- Fish populations that are isolated in small headwater streams could be susceptible to local extirpations caused by more environmental variation (wildfire, debris flows, droughts, floods) as climate change progresses.

Risk assessment

Potential magnitude of climate change effects

- Bull trout – High magnitude of temperature effects by 2050 given that both species are already strongly and negatively affected by warm temperatures.
- Spring Chinook salmon – High magnitude of temperature effects by 2050 given that both species are already strongly and negatively affected by warm temperatures. The effects of altered ocean conditions may pose the greatest risk to these fish. In fresh water, elevated summer temperatures will be the greatest challenge and it is likely to increase with time. The latter can be potentially offset with restoration of riparian areas in streams throughout the stream network.
- Redband trout/steelhead – This species could be less affected by altered ocean conditions because of the ability to express life histories that are resident in freshwater. However, the steelhead component of a population may decline. Steelhead exhibit high phenotypic plasticity and may shift the timing of a life stage transition to reduce the probability of exposure to changes in stream temperature and flow, although there is a limit to how much steelhead can shift the timing of life stages.
- *Low summer flows* – Low effect on spring Chinook by 2050; moderate by 2100. Moderate effect on bull trout by 2050; high by 2100.
- *Winter flood frequency* – Low effect on spring Chinook by 2050 and 2100. Moderate effect on bull trout by 2050; high by 2100.
- *Wildfires/debris flows* – Low effect on spring Chinook by 2050 and 2100. Moderate effect on bull trout by 2050; high by 2100.
- *Invasive species* – Moderate effect on spring Chinook from smallmouth bass by 2050, high effect by 2100. Low effect on bull trout from brook trout by 2050; moderate by 2100.

Likelihood of climate change effects

- *Temperature effects* – High likelihood by 2050.
- *Low summer flows* – Moderate likelihood by 2050, but hydrology models need to improve for this parameter. Also need more flow data from small headwater streams to calibrate hydrology models.
- *Winter flood frequency increase* – High likelihood by 2050 because it is largely controlled by temperature.
- *Wildfires/debris flows* – High likelihood by 2050.
- *Invasive species* – High likelihood by 2050 and 2100 because overlap with native species is strongly controlled by temperatures.

Box 6.1—Ecological disturbance and climate change

Ecological disturbances include fires, floods, windstorms, and insect outbreaks, as well as human-caused disturbances such as forest clearing and the introduction of non-native species. Disturbances are part of the ecological history of most forest ecosystems, influencing vegetation age and structure, plant species composition, productivity, carbon storage, water yield, nutrient retention, and wildlife habitat. Natural disturbances, which are influenced by climate, weather, pre-disturbance vegetation conditions, and location, can have profound and immediate effects on ecosystems across large spatial and temporal scales.

Climate (and weather at a shorter time scale) influences the timing, frequency, and magnitude of disturbances in any particular location. For example, a one-year drought may not have significant direct effects on a forest, but it can reduce tree resistance to insect attack and can desiccate living and dead vegetation sufficiently to increase fire hazard. *As climate continues to warm during the 21st century, the most rapidly visible and significant short-term effects will be caused by altered disturbance, often occurring with increased frequency and severity.* Increased disturbance will be facilitated by more frequent extreme droughts, amplifying conditions that favor wildfire, insect outbreaks, and invasive species (Adams et al. 2010, Allen et al. 2009, Anderegg et al. 2012). The type and magnitude of disturbances will differ regionally and will pose significant challenges for resource managers to mitigate damage to resource values and transition systems to new disturbance regimes.

A warmer climate will cause an increase in the frequency and extent of wildfire in most dry forest and shrubland ecosystems (e.g., Westerling et al. 2006, 2011). By around 2050, annual area burned in most of the western United States is projected to be at least 2-3 times higher than it is today (Littell et al. 2010, McKenzie et al. 2004, Littell (n.d.) cited in Ojima et al. 2014). The Blue Mountains ecoprovince is also expected to experience increased area burned by the mid 21st century (fig. 6.1). Recent research shows that the occurrence of large fires in the western United States has increased since around 1980 (Dennison et al. 2014). Many dry forests that have not burned for several decades have high fuel accumulations, and initial fires may cause uncharacteristic tree mortality compared to low levels of mature tree mortality associated with historical surface-fire regime. If these areas recover as forested ecosystems, recurrent fires (if allowed to burn and are not suppressed) may more closely resemble the frequency characteristic of pre-settlement, low-severity fire regimes. However, *in the driest portions of the Blue Mountains, it is possible that these areas will not recover to forested conditions, and uncharacteristic fires combined with climatic warming could initiate a transition to shrub- or herb-dominated ecosystems.*

Critical thresholds in ecosystem structure and function may be exceeded in a warmer climate. Warmer temperatures may increase the potential for insect and disease outbreaks, particularly as a transient response in colder temperate zones where insect and pathogen vigor has been limited by suboptimal temperatures (Bentz et al. 2010). Higher warm-season temperatures should also increase growth rates for temperate insect herbivores, although the rate of increase will vary by species (Bale et al. 2002). For some species, faster growth rates and reduced development time could enhance juvenile survivorship by reducing predation rates during the larval and nymphal feeding stages (Bernays 1997, cited by Bale et al. 2002).

Increased growth rates could reduce generation times for some species, which could significantly increase population growth rates (Bale et al. 2002, Mitton and Ferrenberg 2012). Increasing population success increases the potential for insect outbreaks to develop, although outbreaks can be limited by host and predator constraints as well (Bale et al. 2002, Boone et al. 2011). Increasing temperatures may also facilitate migration of insects and diseases toward higher elevations and latitudes (Bale et al. 2002, Bentz et al. 2010). Similarly, species ranges could contract at lower elevations and latitudes if warm-season temperatures exceed tolerance levels during the juvenile (or other) growth stages (Bale et al. 2002). Depending on the future scenario, some research indicates that western spruce budworm defoliation in Oregon would decrease under climate change scenarios that do not have an increase in both temperature and precipitation (Williams and Liebhold 1995). Warmer temperatures could also increase host stress levels and the ability to defend against insect attacks (Boone et al. 2011).

The *recent proliferation of mountain pine beetles* (*Dendroctonus ponderosae* Hopkins, 1902) in lodgepole pine forests in western North America is a good example of how a warmer climate can propagate widespread disturbance (Bentz et al. 2009, Kurz et al. 2008, Raffia et al. 2008). *Mountain pine beetles have affected 36 million ha of forests in the western United States and British Columbia over the last two decades*, largely as a consequence of increasing temperatures in mostly older, low vigor forests. Both wildfire and insect outbreaks can rapidly “clear the slate” in landscapes, initiating a regeneration phase in which species will compete, allowing for a potential change in dominant species. A rapid change in dominant vegetation suggests that a threshold has been crossed, at least temporarily, and that new conditions established in a warming climate may be difficult to reverse. Higher atmospheric carbon dioxide levels could reduce insect impacts by increasing carbon availability for defenses in plants and reducing substrate quality in host plants (Stiling and Cornelissen 2007). However, the inhibitory effects of increasing carbon dioxide concentrations may be offset by the stimulatory effects of warmer temperatures on insect activity (Zvereva and Kozlov 2006).

Interacting disturbances and other stressors, termed stress complexes, are a normal component of forest ecosystems, affecting species composition, structure, and function. Altering one particular factor can potentially magnify the effects of other stressors, leading to a rapid and possibly long-lasting change in forest ecosystems. The effects of disturbance across large geographic areas are especially pronounced where forest regeneration is slow or delayed, leading to a potential change in dominant vegetation. A warmer climate is expected to alter and often exacerbate the effects of stress complexes (McKenzie et al. 2009). In an effort to examine the intersection of multiple stressors that affect Oregon and Washington, Kerns et al. (n.d.) examined spatial data regarding wildfire potential, insect and disease risk, projected urban and exurban development, and a warmer climate. They mapped where these stressors occur in concentrated places on the landscape and where they occur in combination (Kline et al. 2013) (fig. 6.2). *For the Blue Mountains, insect and disease risk is the most spatially extensive stressor*. It overlaps with wildfire potential and early climate departure in a few areas.

Forest resource managers are tasked with assessing the risk of disturbances and opportunities for mitigation. Most federal lands have fire management plans that describe current fuel conditions, potential fire occurrence, likely effects of wildfire, fire suppression strategies, and often postfire activities designed to reduce secondary damage such as erosion.

Similarly, plans assessing the management of insects and non-native plants evaluate the risk of their occurrence as the basis for developing appropriate responses. In some cases, the effectiveness of pre-disturbance and post-disturbance actions may be limited. For example, many non-native species are so well established that it is not feasible to remove them from a particular location; therefore, prevention or rapid response is usually the best approach. Following large, intense wildfires, it may be impractical and expensive to install erosion control across a mountainous landscape with minimal access. Active management on certain public lands—wilderness, riparian habitat conservation areas, designated old growth, lands with endangered species—may be severely restricted due to regulations or lack of social license, thus limiting mitigation of known risks.

Disturbances, which have always been a dominant influence on the dynamics of forest ecosystems in western North America, will be even more important if climate change leads to increased frequency and magnitude of extreme weather. In most cases, it will be advisable to develop strategies that allow us to live with increasing disturbance and stress in forests, especially as the climate continues to warm. Active management and planning in anticipation of these changing conditions can reduce risk and the severity of short-term and long-term hazards.

Box 6.2—Model limitations: phenotypic plasticity and biotic interactions

Models that predict vegetation responses to climate change, particularly species distribution models, are generally unable to account for the effects of several important factors such as *phenotypic plasticity* (Nicotra et al. 2010, Valladares et al. 2014) or the outcomes of *biotic interactions* (Araújo and Peterson 2012). However, it is well-established that biotic interactions can profoundly influence how plants respond to changing environmental conditions (Tylianakis et al. 2008), and that competition will play moderate species distribution patterns and responses to climate change (Brooker 2006). Moreover, long-lived organisms, such as trees can substantially modify their ability to tolerate stress or acquire resources as a consequence of plastic responses to external environmental conditions experienced in their lifetimes. This often results in significant differences in phenotypes among individuals, resulting in individualistic responses to environmental change (Nicotra et al. 2010, Richter et al. 2012). Therefore, assessing the effects of climate change on vegetation requires understanding (1) how climate affects species distributions of species across landscapes, (2) how these relationships are modified by biotic interactions and expressions of phenotypic plasticity (Araújo and Guisan 2006), and (3) how management actions influence resilience and adaptive capacity of forest ecosystems (Choi 2007).

Phenotypic plasticity refers to the range of functional traits that can be expressed by a particular genotype or individual plant in response to the environment. Traits that can vary in response to the environment include plant physiological processes (e.g., respiration rates, growth phenology), morphology (e.g., height, allocation to roots), and reproduction (e.g., method, timing) (Peterson et al. 2014). However, costly tradeoffs make plasticity less common than one might expect, given the apparent benefits to individuals and populations (Peterson et al. 2014). If plasticity cannot allow plants to adjust to changing environmental conditions, genetic adaptation or migration may be required to maintain species viability.

Phenotypic plasticity allows plants to adjust to seasonal changes in climate and longer-term climatic variability, and helps avoid extreme vegetation responses to typical (historical) climatic variability. The ability to persist as a juvenile (advance regeneration) facilitates rapid response to increased resource availability following disturbance (or a favorable period of climate), whereas the ability to persist as an adult provides a continuing opportunity to reproduce during favorable periods of climate (Peterson et al. 2014). High phenotypic plasticity may be associated with ecological generalists and benefit plants as the climate changes, because high plasticity allows a single genotype to occupy different environments (Matesanz et al. 2010).

Phenotypic plasticity can help plants become established and persist under low resource availability caused by climatic variability, biotic interactions, and management actions. Unlike most short-lived species, trees can modify their stress tolerance or acquire resources in response to environmental conditions, resulting in individualistic responses to environmental change. For example, the effects of long-term adaptations to water stress (changes in carbon allocation and hydraulic architecture) may influence the ability to cope with environmental variability. Altering water relations by shifting biomass allocation from leaves to woody tissue is an adaptive response of trees exposed to increased warming and drying (Cregg 1994, Zwiazek and Blake 1989, Parmesan 2006).

Competition and stand dynamics affect plant communities and vegetation dynamics (Connell and Slatyer 1977, Grime 1979, Tilman 1982). Assessing the relationship between competition and climate sensitivity for mature trees requires information on the proximate effects of neighbors on resource availability and understanding how expressions of phenotypic plasticity and long-term adaptations to competitive stress may influence the ability to cope with environmental variability (Barnard et al. 2011, Woods 2008). At a local scale, stand density and structure, which can be modified through silvicultural treatments, can affect environmental conditions and create sharp gradients in factors that regulate tree growth (light, water, temperature) (Aussenac 2000, Zhu et al. 2000). As a result, trees growing with high competition or in sub-dominant canopy positions are consistently exposed to different environmental conditions that affect morphological and physiological characteristics, including a lower ratio of leaf-area to sapwood-area (McDowell et al. 2006, Renninger et al. 2007), higher shoot-to-root ratio (Newton and Cole 1991), and reduced rooting depths (McMinn 1963). Trees experiencing high competition have reduced allocation to roots, which reduces water available to support transpiring leaves and necessitates large amounts of sapwood for water storage. However, these adaptations could also buffer the negative effects of stressful years and decrease drought sensitivity.

Differences in climate sensitivity between dominant and sub-dominant trees vary significantly among species depending on life history traits. In one study, dominant ponderosa pines were nearly twice as sensitive to high temperatures as ponderosa pines growing in subdominant canopy positions, but sensitivity of Douglas-fir did not vary across canopy classes (Carnwath et al. 2012). The contrasting effects of canopy position on temperature responses of these two species likely reflect fundamental differences in their physiology, morphology and hydraulic conductivity.

Biotic interactions can significantly influence species distributions and community composition at multiple scales (Araújo and Luoto 2007). For example, rising carbon dioxide concentrations and changing climate could alter the outcomes of biotic interactions if competing individuals or species respond differently to changes in the environment. If plants respond differently, the results could lead to increased or reduced relative rates of growth and reproduction. Changing climate and carbon dioxide concentrations could also alter the intensity of biotic interactions by increasing or reducing overall resource availability. A better understanding of the effects of climate on biotic interactions will assist resource managers with implementation of treatments that can affect those interactions (silviculture, fuel reduction, etc.).

Facilitation, or positive interactions in community structure, is not as well understood as competition (Brooker et al. 2008, Callaway 1995, Callaway and Walker 1997). Plants often rely on beneficial biotic interactions for initial establishment (e.g., nurse plants or logs, mycorrhizal infection), pollination, seed dispersal, and protection from herbivores. They also respond to or defend against detrimental biotic interactions such as competition for limiting resources, herbivory, and seed predation. Positive interactions can occur in all types of conditions, but appear to be most important in stressful environments, like alpine, semi-arid, and arid ecosystems (Callaway 1995, Callaway et al. 2002).

Large trees and shrubs can facilitate establishment and persistence of understory plants (including tree seedlings) by modifying the microclimate (e.g., shading) and availability of soil nutrients and water (McPherson 1997, Scholes and Archer 1997). For example, it has been demonstrated that sagebrush (*Artemisia* spp.) can increase water availability near the soil surface by transporting water from deeper layers at night via "hydraulic lift" (Caldwell et al. 1998, Caldwell and Richards 1989, Richards and Caldwell 1987), but this may also benefit invasive species such as cheatgrass (*Bromus tectorum* L.) (Griffith 2010). Subalpine conifers can reduce snowpack duration and increase growing season length, thereby facilitating establishment of herbaceous plants and tree seedlings (Brooke et al. 1970), or can increase local soil moisture by altering wind flow patterns and enhancing local snow deposition (Holtmeier and Broll 2005).

Box 6.3—Soil drought in forests of the Blue Mountains

One of the most important mechanisms by which climate change can influence vegetation composition and dynamics is through its effects on soil water availability, water uptake by plants, and evapotranspiration (Peterson et al. 2014). Soil water availability depends on precipitation inputs (form, amount, intensity, and seasonality), surface and subsurface water movement, evaporative demand, and vegetation structure and composition. Evapotranspiration is driven primarily by temperature and humidity, but can be constrained by reductions in stomatal conductance or reduced water uptake owing to low soil water availability. Peterson et al. (2014) reviewed the likely effects of projected changes in annual and seasonal mean temperature and precipitation on important components of the hydrologic cycle, including the fraction of winter precipitation received as snowfall, snowpack water storage, snowmelt rates, evapotranspiration, and soil water availability.

By examining soil indices related to drought throughout the Blue Mountains, land managers can potentially identify areas with drought-prone soils, although the relationship between low soil moisture in soils and vegetation vulnerability may not be straightforward. For example, soil parent materials that differ widely in texture may have little effect on mortality of trees and shrubs (Koepke et al. 2010). In addition, some experimental studies have shown that conifers preconditioned by exposure to water stress can actually have higher survival rates and improved water relations during subsequent drought events (Cregg 1994, Zwiazek and Blake 1989). Therefore, the existence of drought-prone soils may not necessarily imply vulnerability.

We examined the Potential Soil Drought Stress (PSDS) index (under development by U.S. Forest Service Pacific Northwest Region) to assess soil drought stress in the Blue Mountains in spring (April, May, June) and summer (July, August, September). The PSDS index ranged from low (1) to high (5) and was developed using available water storage, soil, and modeled actual and potential evapotranspiration data. Methods are summarized below, based on Ringo et al. (n.d.).

Available water holding capacity (AWHC) is the amount of plant-available water that can be held in each inch of soil if that soil is at field capacity (i.e., after free drainage of water following a storm has ceased). AWHC was calculated from Natural Resources Conservation Service soil surveys in the Soil Survey Geographic Database (SSURGO) at a scale of 1:24,000 and U.S. Forest Service Soil Resource Inventories (SRIs) at a scale of 1:63,560 where SSURGO was not available. Available water storage (AWS) for the entire soil profile was calculated by multiplying the AWHC for each inch of soil by the depth of the soil to 59 in or to a root restricting layer or bedrock, whichever was shallower. All calculations were for the dominant soil in the soil map unit. Data were classified into five AWHC categories from low to high.

Potential evapotranspiration (PET) is an estimate of the evaporation and transpiration that would occur if an adequate supply of moisture were available. *Actual evapotranspiration* (AET) measures the actual loss of moisture from soil and plant surfaces, and so the degree to which AET falls below PET may be interpreted an indicator of moisture limitation. Modeled actual and potential evapotranspiration datasets were derived from the Numerical Terradynamic Simulation

Group at the University of Montana (http://www.ntsg.umt.edu/project/mod16). In their MODIS Global Evapotranspiration Project, they used remotely-sensed land cover, leaf area index, and albedo data together with daily meteorological data for air temperature, air pressure, humidity, and shortwave radiation to model AET and PET monthly averages for the years 2000-2012. Departure from PET was calculated and classified into one of five categories that ranged from low (AET/PET = 0.80 – 1) to high (AET/PET = 0.20 or less) (table 6.5). Inaccuracies of this approach include (1) atmospheric interference from aerosols and cloud cover and reflectance from snow creates inaccuracies in the reflectance data and derivative products used to create the PET dataset, and (2) there are some years in which sensors were malfunctioning and data were estimated rather than measured.

A map of PSDS index was created by overlaying AET/PET and AWHC data (table 6.4). If there is little departure from PET (AET/PET is low) then vegetation is transpiring at full potential and there is enough moisture to allow this level of transpiration, then climatic factors, not soil water storage, limit available moisture for plant use. However, the degree to which AWHC or climatic factors are governing plant transpiration is not actually known.

As the departure from PET increases (e.g., AET/PET is much less than 1), then plant transpiration is below full potential. If AET/PET is extremely low (< 0.20 or close to 0), then it follows that all forms of available moisture for plants have largely been exhausted. However, the AET/PET spatial data describe average conditions across each grid cell and the AWHC data are finer scale. It is possible that even if AET/PET is classified as low within a grid as an average, areas that have soils with high AWHC within that grid cell may not be as dry as soils that have a lower AWHC. This means that although AET/PET may be mapped as extremely low, soils with higher AWHC may still have the ability to sustain growth. PSDS index allows the combination of two scales of data (coarser scale AET/PET data and finer scale AWHC data) to provide a potentially more refined understanding of plant drought stress.

PSDS maps are shown for each national forest in the Blue Mountains in figs. 6.10-12. The PSDS index has not yet been peer reviewed and relies heavily on the accuracy of the SSURGO data at a very fine scale. Additional field testing will help validate many assumptions and test data accuracy. Reliance on the fine-scale data could lead to an inappropriate spatial focus and assumptions about accuracy ("my favorite pixel"). Although the SSURGO soil data are available at relatively fine spatial scales, soil can is heterogeneous at even finer scales (as small as a few feet). Examination of the two spatial layers separately may be more appropriate. The PSDS index may simply provide an indication of where on the landscape vegetation may be vulnerable to drought stress, particularly across similar vegetation types or species.

Malheur National Forest

- Overview—Patterns in the Malheur National Forest are distinctive from other areas in the Blue Mountains, owing to the southerly location, lack of maritime influence, and lower elevation terrain (fig. 6.10). Soils are typified by fewer volcanic soils and more clay-dominated soils. A substantial portion of the landscape (41 percent) is classified as high PSDS (table 6.6). Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups.

- Spring—Seventy-one percent of the forest is classified into the two highest PSDS categories combined. Only 11 percent of the forest remains in the moderate PSDS category. All PVGs have substantial portions in the moderate and moderate to high PSDS categories (table 6.7). Nonforested PVGs have more area in the moderate to high categories compared to forested PVGs. The Dry UW, US, UH and Moist UW and US groups all have 90 percent or more area in moderate to high or high PSDS categories combined. However, the Dry UF group actually has a higher proportion in the two highest PSDS categories (78 percent) than some of the other nonforested PVGs. The Dry US group has the highest spatial extent in moderate to high and high PSDS (98 percent).
- Summer—Sixty percent of the forest is mapped as high potential drought stress and 28 percent as moderate to high. Therefore, almost 90 percent of the forest could be under extreme drought conditions in the summer. All PVGs have more area classified in moderate to high and high PSDS categories. Nonforested PVGs have high proportions of mapped drought-prone soils compared to forested PVGs. Ninety-seven percent of the Dry US PVG and 95 percent of Moist UW are in high categories. Values are lower for the forested PVGs, but all forested PVGs have substantial portions in the high category. Ninety-four percent of the Dry UF PVG is in the moderate to high and high combined categories.

Umatilla National Forest

- Overview—About half of the Umatilla National Forest is classified as having moderate PSDS (table 6.6, fig. 6.11); the relative percent PSDS index classified within each upland PVG is shown in table 6.7. Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups.
- Spring—Although 94 percent of the Forest is in the moderate and moderate to high categories combined, only 1 percent of the forest is mapped with high soil drought stress during this time period. Only 5 percent of the Forest has low to moderate drought stress. All PVGs have substantial portions in moderate and moderate to high PSDS categories. All nonforested PVGs have more area classified in the moderate to high categories (70 percent or more) compared to forested PVGs. The Dry UW PVG has the highest spatial extent in the high PSDS (14 percent). The spatial extent of category 4 and 5 PSDS for the three forested PVGs is much lower than the nonforested PVGs. As might be expected, the most drought-prone soils for the forested PVGs is for the Dry UF group, with more than 40 percent in moderate to high. However, 28 percent of the Moist UF and 21percent of the Cold UF are in moderate to high.
- Summer—About 69 percent of the forest is mapped as having moderate to high and high drought stress. All PVGs have more area classified in the moderate to high, and high PSDS categories. Nonforested PVGs have high proportions of drought-prone soils compared to forested PVGs. Ninety-six percent of the Dry UF PVG is in high PSDS, and even 23 percent of the Moist UW and 18 percent of the Cold UF are high. However, almost twice as much of Dry UF group is in the high category (42 percent), and about 75 percent of this PVG is in the moderate to high and high categories combined.

Wallowa-Whitman National Forest.

- Overview—Similar to the Umatilla National Forest, about half of the Wallowa-Whitman National Forest is classified in moderate PSDS (table 6.6, fig. 6.12), and 80 percent is in moderate and moderate to high categories combined.
- Spring—About 9 percent of the forest is in the high PSDS category, and only 2 percent is in the low to moderate category. All PVGs have substantial portions in moderate and moderate to high PSDS (table 6.7). Unlike the Umatilla National Forest, substantial portions of the PVGs are in high PSDS status. This forest has large areas of glaciated terrain with scoured landscapes and shallow rocky soils and fewer deep ash caps as compared to the Umatilla. All of the nonforested PVGs have more area classified in moderate to high categories compared to forested PVGs. The Moist UW PVG has the highest spatial extent in high PSDS (> 50 percent). Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups. The spatial extent of categories 4 and 5 PSDS for the three forested PVGs is lower than for nonforested PVGs. The Cold UF PVG has the most drought-prone soils (30 percent) in moderate to high, most likely a function of shallow rocky, soils and glaciated terrain in these areas.
- Summer—About 69 percent of the forest is mapped as moderate to high and high potential drought stress (categories 4 and 5 combined). As with the Umatilla National Forest, all PVGs have more area in the moderate to high and high PSDS categories. Similar to spring patterns described above, nonforested PVGs have high proportions of drought-prone soils compared to forested PVGs. Eighty-eight percent of the Dry US PVG and 86 percent of the Moist UW have high PSDS. Values are lower for the forested PVGs, but both the Cold and Dry UF PVGs substantial portions in the high category.

Box 6.4—Cold and moist upland conifer species of concern: whitebark pine, limber pine, mountain hemlock, Alaska cedar

Limber pine, Alaska cedar, and mountain hemlock in the Blue Mountains are represented by populations that are separated from the main parts of the species' distributions. A single population of Alaska cedar occurs in eastern Oregon, the Cedar Grove Botanical Area in the Malheur National Forest (Frenkel 1974). This population is likely a relict of a time period when cooler and wetter conditions prevailed and the species occurred across a larger area (Devine et al. 2012). The distribution of limber pine is discontinuous throughout its range and two relict populations occur in the Eagle Cap Wilderness. Infrequent mountain hemlock stands occur in the northern half of the Wallowa Mountains. These rare, disjunct populations represent important genetic resources that may be at risk to climate change and other stressors.

Whitebark pine and limber pine are also threatened by *white pine blister rust* (Smith et al. 2013, Tomback and Achuff 2010), a non-native fungus that forms cankers of necrotic tissue that girdle tree stems. Alternate hosts for the fungus are currant (*Ribes*) or the herbs paintbrush *Castilleja*) and lousewort (*Pedicularis*) (Geils et al. 2010). Indirect pathogen-related effects could occur if climate increasingly favors blister rust. Higher variability in weather conditions may create conditions conducive to infection, although drier summers could inhibit the formation and spread of rust spores and fruiting body development.

High elevation species can also be rapidly altered by rare wildfire events, because recovery from stand replacing wildfires is slow. This is especially true for species with limited distributions. Mountain hemlock in the Wallowa Mountains is favored by a lack of fire, because a stand-replacing fire here would likely favor lodgepole pine (Devine et al. 2012). The Alaska cedar grove in the Malheur National Forest burned in 2006, and most of the mature trees were killed. In 2014, wildfire burned through some of the limber pine habitat in the northern Wallowa Mountains, although the effects on the trees is unknown.

Fire effects on whitebark pine are difficult to generalize. Mixed-severity and stand-replacing fires are beneficial to seral whitebark pine communities, because the pine is better adapted to recolonize burned sites compared to more shade tolerant subalpine fir that may outcompete pine in the absence of fire (Arno and Hoff 1990, Keane et al. 2012). With declining populations, loss of cone-bearing trees with potential resistance to blister rust will limit future management options. In addition, probability of dispersal by Clark's nutcracker declines with diminishing cone production (Barringer et al. 2012).

During the past two decades, warmer temperatures have allowed mountain pine beetles to shift upward and persist in higher-elevation forests (Logan et al. 2010). Mountain pine beetle is the primary cause of whitebark pine mortality at Crater Lake National Park, presently at a rate of 1 percent annually (Murray 2010), and may have been facilitated by an increasing late-summer dry season (Daly et al. 2009). Longer, high-elevation growing seasons could enhance the growth of subalpine tree species (e.g., Bunn et al. 2005, Peterson and Peterson 2001). However, because whitebark pine grows so slowly, it may be at a competitive disadvantage compared to other species.

Box 6.5—Quaking aspen in upland forests

Because quaking aspen stands in the Blue Mountains are typically less than 1 ha in size and fragmented across the landscape, they are vulnerable to a warmer climate and other stressors (table 6.2). Aspen is an early seral component of forest communities, and stands have been declining in number, area, and stocking density. Current stressors include competition with shade tolerant conifers, herbivory by livestock and native ungulates, and fire exclusion. Aspen can occur in cold, moist, and dry upland forest PVGs, although the only aspen potential vegetation type is in the Cool Very Moist PAG in Moist Upland Forest (Swanson et al. 2010).

Sudden aspen death (SAD) was noted in the Rocky Mountains over a decade ago and has affected large areas of aspen in southwestern Colorado (Worrall et al. 2010). SAD is characterized by rapid, synchronous branch dieback, crown thinning, and mortality of stems, without the involvement of primary pathogens and insects. Some affected stands may fail to produce suckers in response to crown loss and mortality. Worrall et al. (2008) proposed a disease model for SAD as follows: (1) predisposing factors include low elevations, south/southwestern aspects, physiological maturity, and low stand density, (2) inciting factors are acute drought with high temperatures during the growing season, and (3) contributing factors are insects and pathogens that tend to invade and kill stressed trees.

A warmer climate and drier growing season could increase susceptibility to SAD in the Blue Mountains, because aspen requires mesic soil moisture conditions, and moisture stress is an underlying factor for SAD (Worrall et al. 2010). Aspen stands in the dry upland PVG are already near their soil moisture limit for survival, and increased loss of aspen in the Malheur National Forest might be expected at lower elevations.

Increased frequency of stand replacing fires in the future may favor aspen regeneration, because aspen respond to stand-replacing fire with abundant vegetative propagation (suckering) and rapid growth. Increased low-severity fire may benefit aspen if competition with conifers is reduced and soil moisture status is improved, although once competing conifers like ponderosa pine, Douglas-fir, and grand fir are relatively large, their fire tolerance exceeds that of aspen (Swanson et al. 2010). In addition, severe fires and reburns may kill shallow root systems and eliminate small clones, particularly in stressed stands.

Box 6.6—Non-native annual grasses

Non-native annual grasses such as cheatgrass, medusahead, and North Africa grass alter fire regimes and are some of the most important ecosystem-altering species on the planet (Brooks et al. 2004). Cheatgrass is widely distributed in western North America (USDA and NRCS 2013) and is abundant and dominant in steppe communities (Mack 1981). Following disturbance, this species is capable of invading low-elevation forests (Keeley et al. 2003, Keeley and McGinnis 2007, Kerns et al. 2006) and creating surface fuel continuity between arid lowlands and forested uplands (fig. 6.22). Highly competitive traits enhance its ability to exploit soil resources after fire and to increase its status in the community (Melgoza and Nowak 1991, Melgoza et al. 1990), and dominance may be persistent (Zouhar 2003). After establishment, cheatgrass tends to increase the probability of subsequent wildfire occurrence (D'Antonio and Vitousek 1992) because the fine, continuous, and highly combustible fuels dry early in the season, increasing the length of the fire season in some ecosystems. Aside from the potential threat to biodiversity from this type of change, conversion of forests and woodlands to grasslands has important implications for carbon cycling and feedbacks between climate and the biosphere, because forests sequester large amounts of carbon (Bonan 2008).

A species distribution model that assumed lower (summer) precipitation in the future projected expansion of cheatgrass in a warmer climate (Bradley et al. 2009). However, when higher precipitation is included in the model, the area of reduced habitat for cheatgrass was reduced as much as 70 percent. Increases productivity in response to elevated carbon dioxide has been documented, and elevated atmospheric carbon dioxide may have already contributed to cheatgrass productivity and fuel loading and associated effects on fire (Ziska et al. 2005). Because of the fire and invasive grass cycle, changes in future fire regimes are important climate considerations for non-native annual grasses. More area burned, more frequent large wildfires, larger extent of high-severity fire, longer wildfire durations, and longer wildfire seasons are expected in the future (Lutz et al. 2009, Miller et al. 2008, Nydick and Sydoriak 2014, Westerling et al. 2006), thus increasing the invasion risk of non-native annual grass species.

The likelihood of forest, woodlands and shrublands being invaded by non-native annual grasses in a warmer climate will increase because of more disturbance, effects of warming on species distributions, enhanced competitiveness of non-native plants from elevated carbon dioxide, and increased stress to native species (Breshears et al. 2005, Dukes and Mooney 1999, Pauchard et al. 2009, Ziska and Dukes 2011) (fig. 6.8). Warming alone may increase the risk of non-natives, because many invasive species have range limits set by cold temperatures, which has tended to limit their establishment in forests, particularly the higher elevation and continental western forests. Of particular concern in the Blue Mountains is the recent increase in North Africa grass and its apparent ability to outcompete and replace areas dominated by cheatgrass. The effectiveness of management actions to control invasive plants is decreasing in some areas as a result of reduced herbicide efficacy (Archambault et al. 2001, Ziska and Teasdale 2000), and biocontrol methods may not be as effective in a warmer climate (Hellmann et al. 2008).

Box 7.1—Climate change effects on riparian areas and wetlands in the Blue Mountains

Broad-scale climate change effects

- Higher air temperature
- Higher frequency and severity of droughts
- Decreased snowpack
- Increased rain:snow ratio
- Altered streamflow

Habitat and species

- Cottonwood-dominated riparian communities
- Wetland and riparian aspen communities
- Willow-dominated riparian communities
- Herbaceous-dominated riparian and wetland communities

Current condition, existing stressors

Cottonwood, aspen, and willow

Condition:

- Decreased area due to conversion and development of floodplains
- Degradation of stands due to altered flow regimes (dams, diversions) and changes in hydrology due to floodplain land use

Stressors:

- Structural simplification of channels (e.g., levee construction), roads, livestock and native ungulate browsing
- Intentional clearing/ removal of native riparian woody species, e.g. to increase herbaceous forage or pasture area for livestock grazing

Herbaceous

Condition:

- High cover and frequency of non-native pasture grasses
- High cover of grazing-tolerant native species

Stressors:

- Heavy herbivory from livestock and native ungulates
- Increasing cover of invasive and/or noxious plant species
- Planting or seeding of non-native pasture grasses

Sensitivity to climatic variability and change

Cottonwood, willow, and aspen

- Decreased establishment of willow and cottonwood
- Displacement of wetland/riparian plant species with upland species
- Decreased riparian cover

- Decreased plant growth and increased mortality

Herbaceous

- Further decreases in native species cover and richness
- Shift in community composition
- Increased success of non-natives
- Loss of sensitive species

Expected effects of climate change

Cottonwood, willow, and aspen

- Increased high flows in winter
- Decreased low flows in summer
- Increased demand for water (additional diversions, reservoir expansions)
- Increased browsing pressure

Herbaceous

- Decreased low flows in summer
- Increased demand for water
- Increased demand for forage and grazing

Adaptive capacity

Cottonwood, willow, and aspen

- Cottonwood and willow populations have evolved within the range of regional streamflow variability. They are highly dependent on natural flow characteristics for seed germination, seedling establishment, and stand persistence. They have limited adaptive capacity where flow regimes have been altered.
- Aspen and most willow species have high vegetative regenerative capacity following disturbance (fires, floods), which contributes to adaptive capacity. However, the ability to persist depends on site conditions, particularly soil moisture and depth to water table.

Herbaceous

- Native, herbaceous wetland species have high soil moisture requirements. When water table elevations decline and soil moisture conditions become more limiting, these species are no longer competitive against more drought-tolerant species, and have limited adaptive capacity.
- Common, dominant native species (e.g. *Carex aquatilis*) can occur over a fairly wide range of soil moisture conditions, as well as grazing pressure, and have some adaptive capacity. Less is known about the adaptive capacity of many native wetland sedge and forb species, but most occur within narrow ranges of soil saturation/soil moisture conditions. Where these conditions are not met, sensitive and uncommon herbaceous species will not persist.

Vulnerable geographic locations

Cottonwood, willow, and aspen

- Cottonwood gallery forests along low-gradient river segments are extremely vulnerable, particularly in floodplains with flow diversions or groundwater pumping. More isolated stands along higher-gradient stream segments are less vulnerable, but still require components of the natural flow regime for long-term persistence.
- Most willow stands at lower elevations along low-gradient stream segments have been impacted by floodplain land use and are highly vulnerable. At higher elevations and along smaller streams, willow stands may be less vulnerable. However, willow stands may be comprised of 2-8 species and the requirements for establishment, growth, and persistence are largely unknown for some species. Willow stands may persist, but some species may not survive locally.
- Riparian aspen stands along low-gradient river segments are extremely vulnerable, particularly in floodplains with flow diversions or groundwater pumping. Some isolated aspen stands in uplands environments may be largely dependent on groundwater, so vulnerability depends on underlying lithology.

Herbaceous

- The most vulnerable herbaceous communities are those that have already been extensively impacted by grazing and other land and water uses.
- Alpine and subalpine herbaceous communities are highly vulnerable due to decreases in amount and persistence of snow.
- Herbaceous communities at mid elevations will experience shifts in community composition, with increased cover of non-natives and loss of uncommon or sensitive species, but will likely persist.

Risk assessment

Potential magnitude of climate change effects

- Cottonwood-dominated riparian communities: high magnitude of effects along major rivers, given that cottonwood forests are currently impacted and declining in many locations.
- Willow-dominated riparian communities: moderate-high magnitude of effects for some species and communities. Highest risks are for communities located along stream segments already impacted by grazing and flow alteration.
- Wetland and riparian aspen communities: high magnitude of effects, because many aspen populations are known to be declining.
- Herbaceous-dominated riparian and wetland communities: moderate magnitude of effects for communities; high magnitude of effects for rare and sensitive species that will not be competitive in drier environments.

Likelihood of climate change effects

- Cottonwood-dominated riparian communities: high likelihood of effects for cottonwood communities located along larger floodplains.
- Wetland and riparian aspen communities: high likelihood of effects given current (declining) condition.
- Willow-dominated riparian communities: high likelihood of effects given predictions of changes in streamflow, increases in air temperature and higher frequency and severity of droughts, increased human demands for water.
- Herbaceous-dominated riparian and wetland communities: moderate likelihood of effects given predictions of changes in streamflow and higher frequency and severity of droughts, increased human demands for water and other resource use.

Box 7.2—Summary of climate change effects on groundwater-dependent ecosystems (GDEs) in the Blue Mountains

Broad-scale climate change effects

- Higher air temperature
- Higher frequency and severity of droughts
- Higher groundwater temperature
- Decreased snowpack, especially at lower elevations
- Possible changes in groundwater recharge quantity and levels

Habitat and systems

- Springs and associated wetlands and fens, hyporheic zones, groundwater contribution to streamflow (baseflow)

Current condition, existing stressors

Current condition:

- Numerous springs developed for watering livestock
- GDEs used by livestock, and native ungulates (source of water and forage)

Stressors:

- Continued water development
- Grazing, browsing and trampling by livestock and native ungulates

Sensitivity to climatic variability and change

- GDEs (springs, wetlands) and stream baseflows are supported by groundwater recharge from rain and annual snowpack, especially in more permeable lithologies
- GDEs may contract in size or dry out in summer
- Increased air and water temperatures and drought will stress moisture-dependent flora and fauna
- Small aquifer systems are generally more vulnerable than larger systems
- Groundwater resources may be less sensitive to climate change than surface water, depending on local and regional geology, and surrounding land and water use

Expected effects of climate change

- Reduced groundwater discharge to GDEs
- Reduced areas of saturated soil
- Perennial springs may become ephemeral
- Ephemeral springs may disappear, except during high-snow years
- For springs discharging to streams, local cooling influence on stream temperature

may be reduced
- Increased stress from effects of grazing
- Shifts in aquatic flora and fauna communities
- Higher groundwater temperatures
- “Gaining” reaches of streams may contribute less or become “losing” reaches

Adaptive capacity
- Because GDEs and the biota they support depend on continued availability and volume of groundwater, they have limited adaptive capacity.
- Current information about the role of groundwater on water budgets at different scales is very limited for wildland watersheds. Although ongoing research may reveal adaptive capacity in some locations, current information suggests that groundwater resources are declining.

Vulnerable geographic locations
- Vulnerability largely depends on elevation and underlying lithology, which influence the storage and movement of groundwater.
- GDEs located at higher elevations are likely the most vulnerable, given predicted changes in snowpack volume and persistence. As snowpacks decrease, less water will infiltrate into subsurface aquifers, and the amount of groundwater discharge will decrease. High elevation springs and other GDEs may be the first to become ephemeral, dry out, and eventually disappear.
- GDEs located at mid-elevations may be the least vulnerable, depending on underlying geology and water demands. GDEs may persist in lithologies that support large aquifer systems.
- GDEs located at lower elevations, including many rheocrene springs or springbrooks, are extremely vulnerable to increasing water demands, pressure for increased diversion or water development, and other watershed-scale land use effects.

Risk assessment
Potential magnitude of climate change effects
- High magnitude of effects for GDEs, especially those located at higher elevations or occurring where underlying geology only supports small shallow aquifer systems.

Likelihood of climate change effects
- Moderate likelihood of some GDEs disappearing by 2050, but groundwater research and modeling are needed to identify most vulnerable aquifers and GDEs.

Box 7.3—Deep river canyons and climate change: the lower Snake River and its tributaries

The lower Snake River runs along the Idaho and Oregon state line and forms the deepest river gorge in North America, commonly referred to as Hells Canyon. From Hells Canyon Dam, one of three dams in the Hells Canyon Project, the river winds its way for 114 km to the northern boundary of Hell Canyon National Recreation Area (HCNRA), managed by Wallowa-Whitman National Forest. With river elevations of 512 m at the dam to 263 m at the northern boundary of HCNRA, this river canyon provides the warmest and driest environments in the Blue Mountains national forests.

The vegetation of the canyon is characterized by extensive bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve) and Idaho fescue (*Festuca idahoensis* Elmer) grasslands, with stringers of conifer forests at cooler aspects and higher elevation. Riparian communities are often confined to narrow strips along river corridors and moisture gradients are steep. The floodplains, rocky bars and terraces at elevations below 700 m support a number of unique riparian plant communities characterized by black cottonwood, white alder, netleaf hackberry (*Celtis reticula* [Torr.] L.D. Benson), and Barton's raspberry (*Rubus bartonianus* M. Peck), a narrow endemic shrub species of Hells Canyon and surrounding canyonlands (Wells 2006).

With the settlement of the canyon in the early 19th century came the introduction of many non-native plant species, including tree of heaven (*Ailanthus altissima* [Mill.] Swingle), false indigobush (*Amorpha fruticosa* L.) and Himalayan blackberry (*Rubus armeniacus* Focke). The spread of Himalayan blackberry is of particular concern for the endemic Barton's raspberry, because Himalayan blackberry is able to occupy the same habitats but is a better competitor (Ferriel and Ferriel 2010). Native *Rubus* species are restricted by drought conditions during summer, whereas Himalayan blackberry can store more water and achieves high growth and reproductive rates (Caplan and Yeakley 2010). Canes can grow up to 10 m long and produce over 700 fruits annually (Pojar and MacKinnon 1994). In Hells Canyon, Himalayan blackberry retains its leaves over winter, giving it an additional competitive edge over many native species.

Fire exclusion has affected Hells Canyon, with steep terrain and fast fire spread in dry canyon grasslands, less than other areas in the Blue Mountains. From 1980 to 2013, over 70 percent of all grass and shrublands of the HCNRA were within one or more mapped fire perimeters (S. Mellman-Brown, unpublished data on file at Wallowa-Whitman National Forest, Baker, OR). Many of these fires burned through riparian zones and replaced existing shrub and forest vegetation with early-seral species. Himalayan blackberry, which resprouts readily from its root crown after fire, had covers of 80 percent near Pittsburg Creek one year after fire, an increase of 30 percent compared to measurements one decade earlier. On other sites, white alder, a tree with poor post-fire sprouting abilities, appeared to be replaced by blackberry thickets after high-severity fire. Increasing fire frequencies with climate change will promote fire-adapted species like tree of heaven and Himalayan blackberry, potentially creating widespread novel plant communities with few native elements.

Warming climate with increasing fire frequencies may also facilitate the invasion and dominance of tamarisk in Hells Canyon (Kerns et al. 2009). Major population centers for tamarisk in the northwestern United States are the warmest and driest environments of the northern Basin and Range, Columbia Plateau, and central Snake River Plateau. Tamarisk is currently absent from the HCNRA, but large populations exist nearby at Farewell Bend and with lower frequency along the Brownlee Reservoir. Large areas in the lower Snake River and upper John Day River drainages are also vulnerable to invasion by tamarisk (Kerns et al. 2009). Habitat suitability modeling by Kerns et al. (2009) suggests that tamarisk habitat will expand in the Northwest by the end of this century, which could dramatically change the composition and structure of many riparian corridors in the Blue Mountains.

Figures

Fig. 1.1

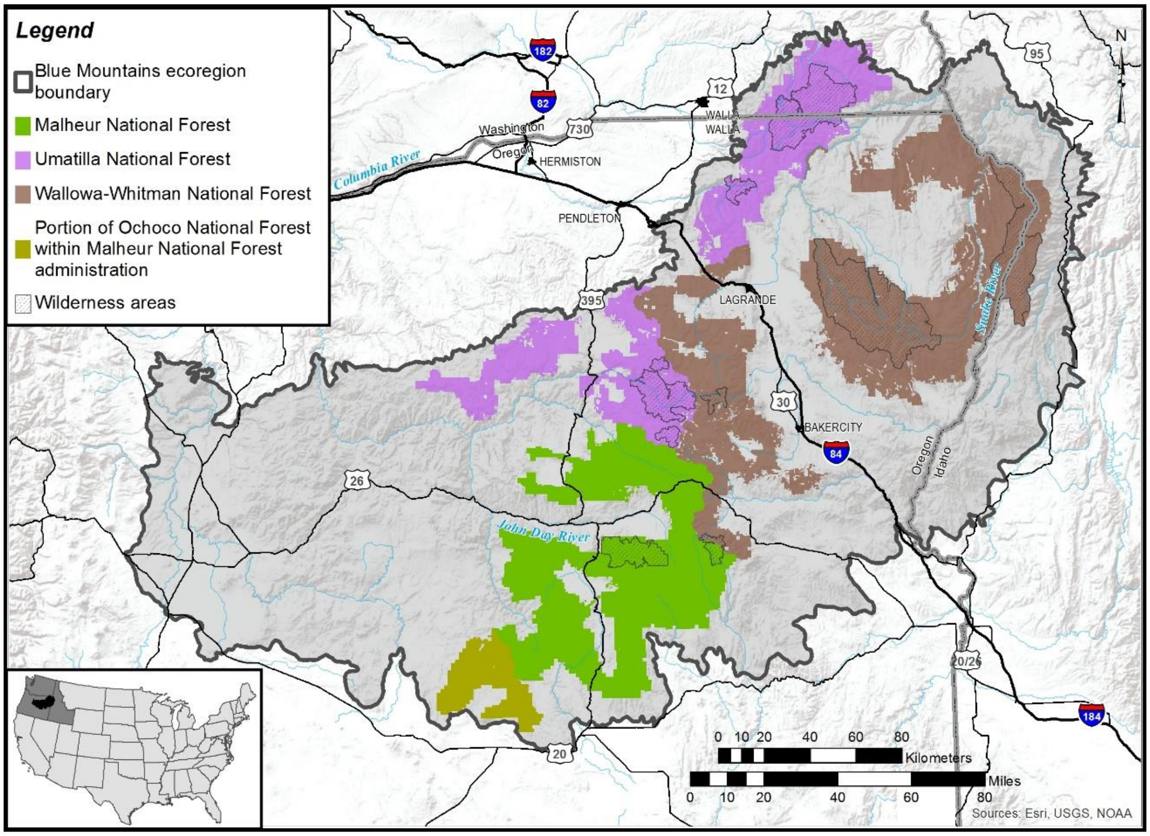


Fig. 2.1

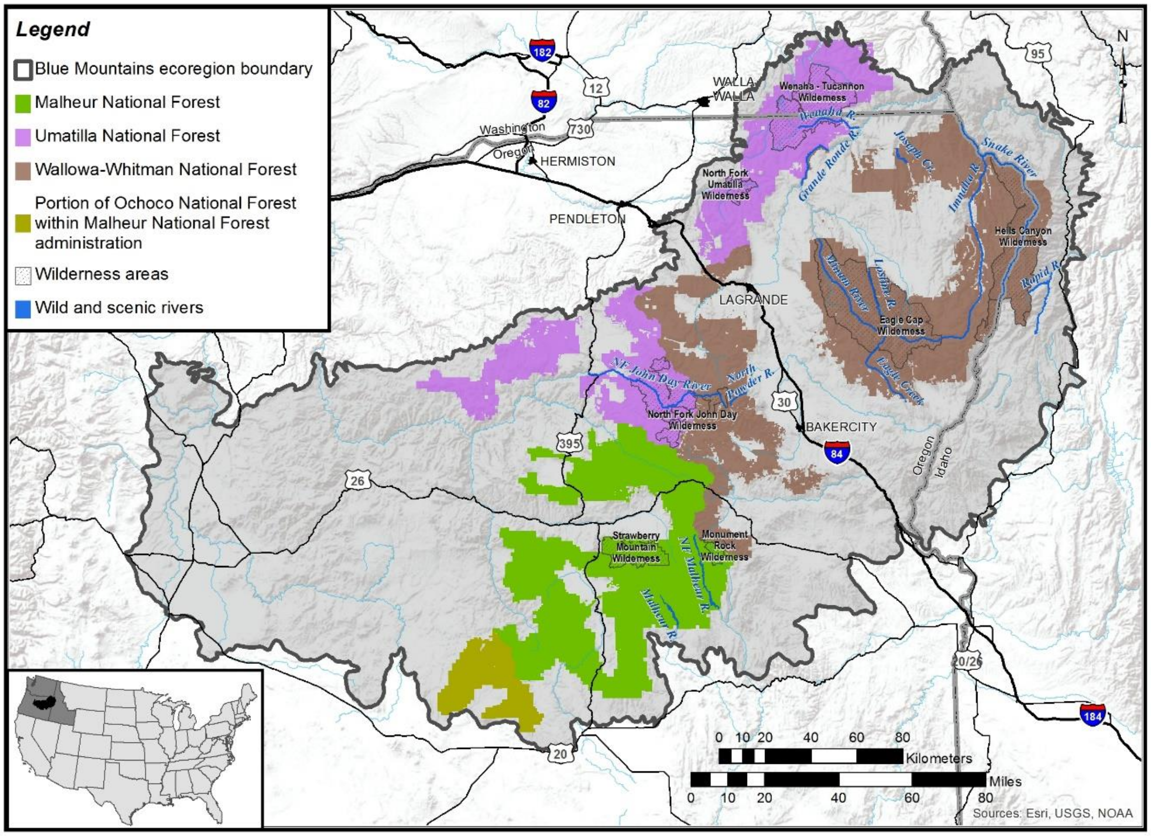

Legend
Blue Mountains ecoregion boundary
Malheur National Forest
Umatilla National Forest
Wallowa-Whitman National Forest
Portion of Ochoco National Forest within Malheur National Forest administration
Wilderness areas
Wild and scenic rivers
WALLA WALLA
Washington
Oregon
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Wenaha - Tucannon Wilderness
North Fork Umatilla Wilderness
Hells Canyon Wilderness
Eagle Cap Wilderness
North Fork John Day Wilderness
Strawberry Mountain Wilderness
Monument Rock Wilderness
Snake River
Grande Ronde R.
Joseph Cr.
Imnaha R.
Lostine R.
Minam River
Eagle Creek
Rapid R.
N.F. John Day River
North Powder R.
N.F. Malheur R.
Malheur R.
Oregon
Idaho
182
82
12
730
95
395
26
30
84
20
20/26
184
N
0 10 20 40 60 80
Kilometers
0 10 20 40 60 80
Miles
Sources: Esri, USGS, NOAA

Fig. 3.1

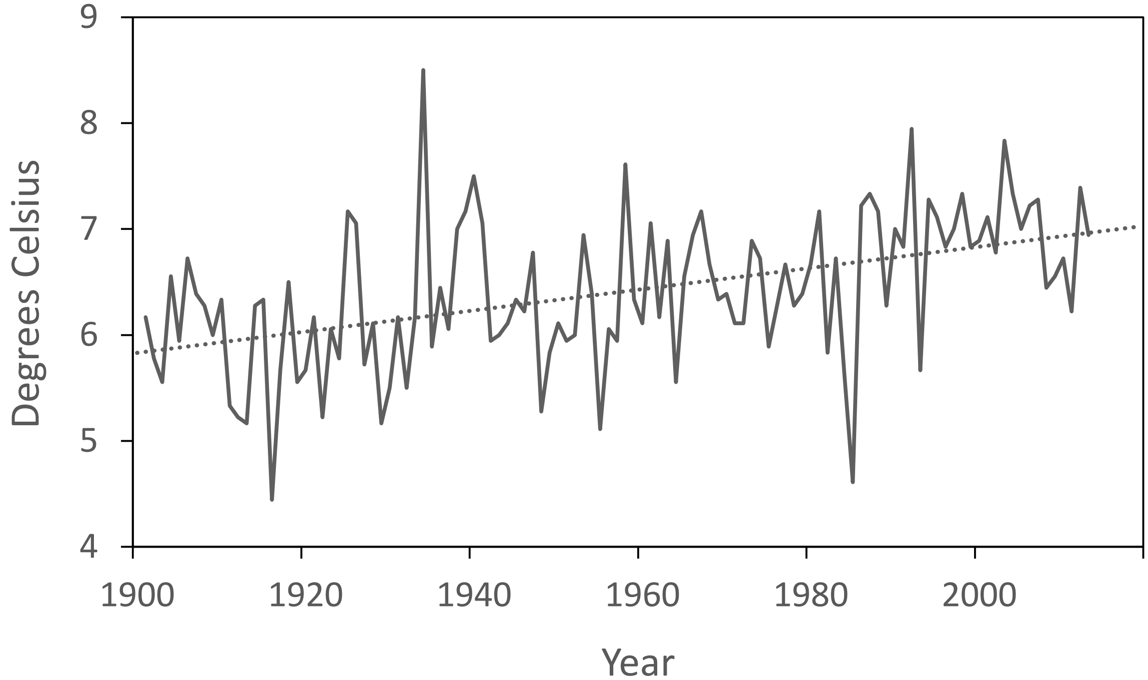


Fig. 3.2

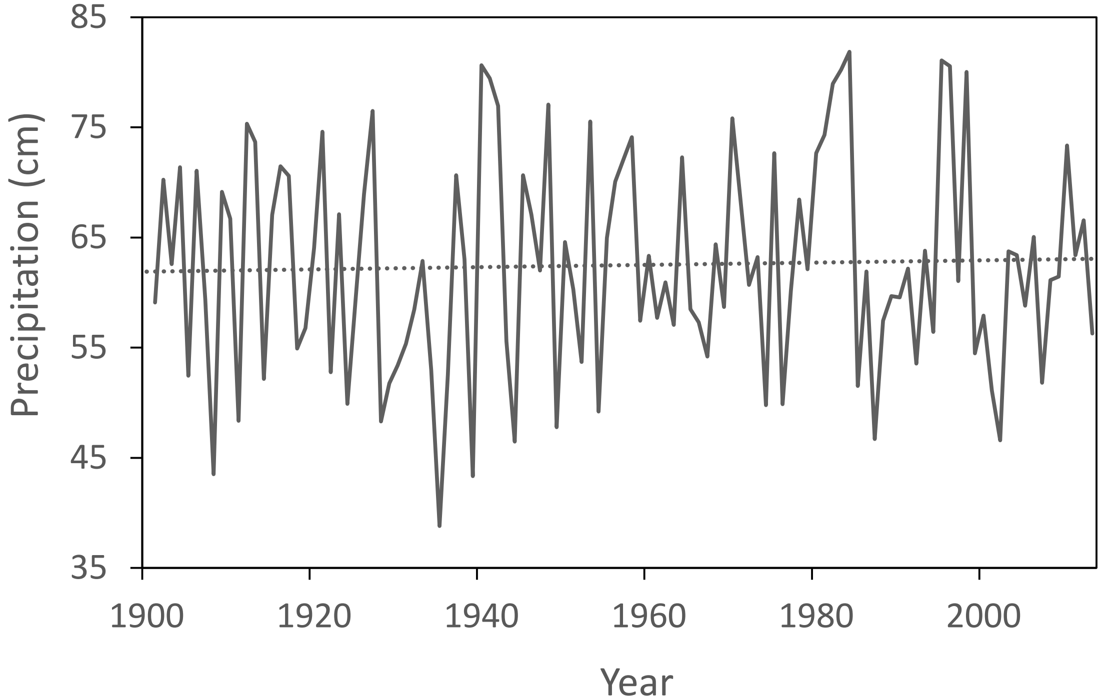

85
75
65
55
45
35
Precipitation (cm)
1900
1920
1940
1960
1980
2000
Year

Fig. 3.3

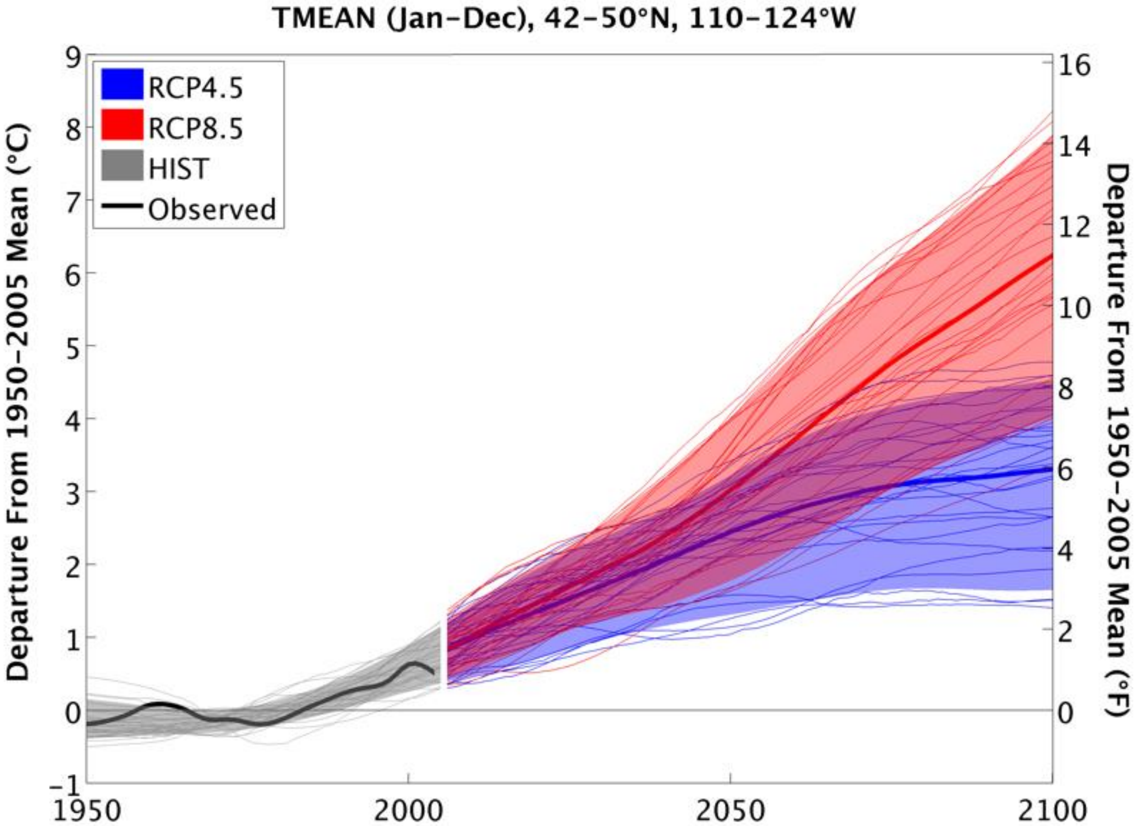

TMEAN (Jan–Dec), 42–50°N, 110–124°W
RCP4.5
RCP8.5
HIST
Observed
Departure From 1950–2005 Mean (°C)
Departure From 1950–2005 Mean (°F)
9
8
7
6
5
4
3
2
1
0
−1
16
14
12
10
8
6
4
2
0
1950
2000
2050
2100

Fig. 3.4a

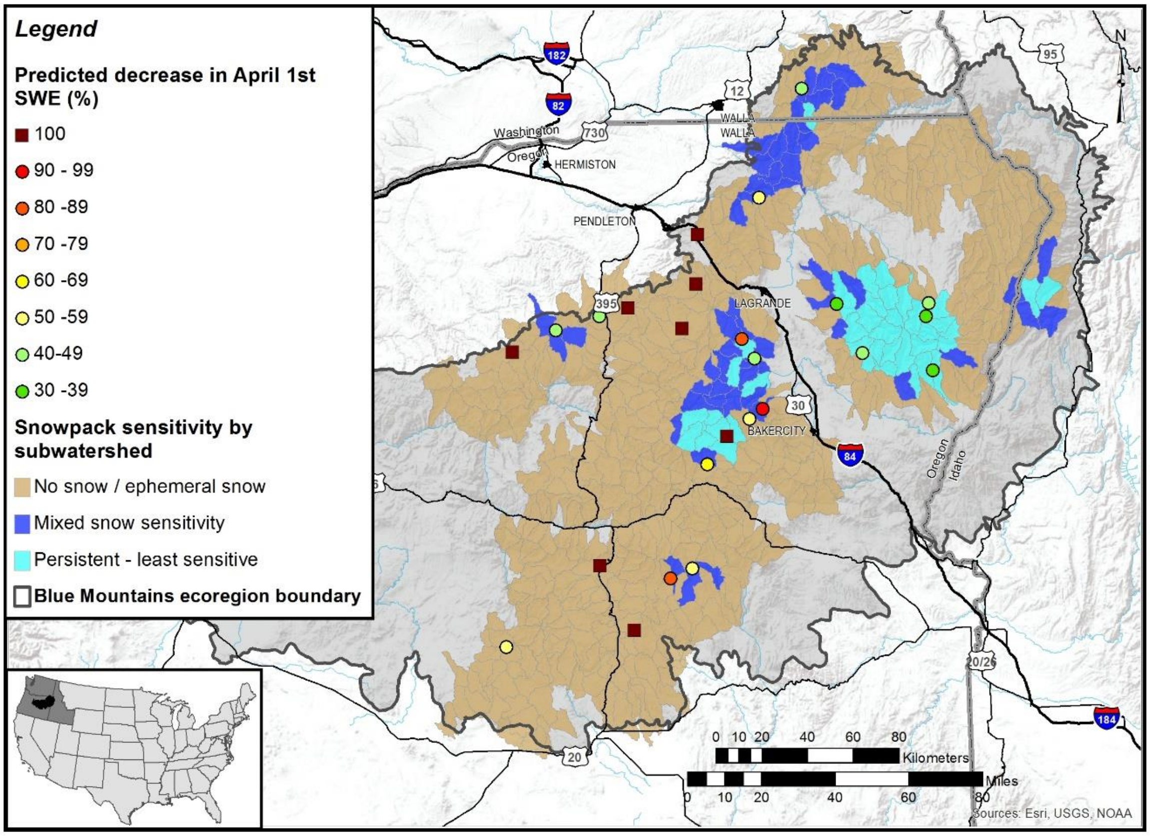

Legend
Predicted decrease in April 1st SWE (%)
100
90 - 99
80 -89
70 -79
60 -69
50 -59
40-49
30 -39
Snowpack sensitivity by subwatershed
No snow / ephemeral snow
Mixed snow sensitivity
Persistent - least sensitive
Blue Mountains ecoregion boundary
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Washington
Oregon
Idaho
0 10 20 40 60 80 Kilometers
0 10 20 40 60 80 Miles
Sources: Esri, USGS, NOAA

Fig. 3.4b

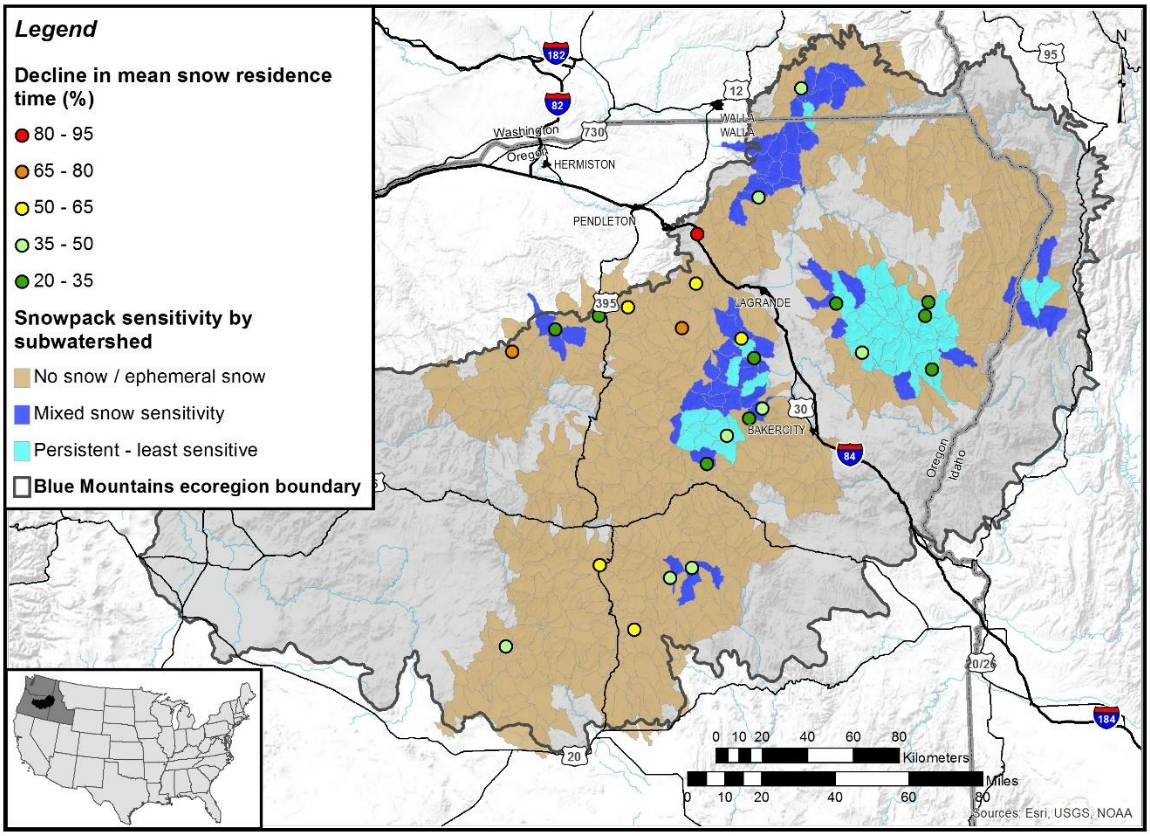

Legend
Decline in mean snow residence time (%)
80 - 95
65 - 80
50 - 65
35 - 50
20 - 35
Snowpack sensitivity by subwatershed
No snow / ephemeral snow
Mixed snow sensitivity
Persistent - least sensitive
Blue Mountains ecoregion boundary
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Washington
Oregon
Idaho
182
82
12
730
395
30
84
95
20
20/26
184
0 10 20 40 60 80
Kilometers
Miles
0 10 20 40 60 80
Sources: Esri, USGS, NOAA

Fig. 3.5a

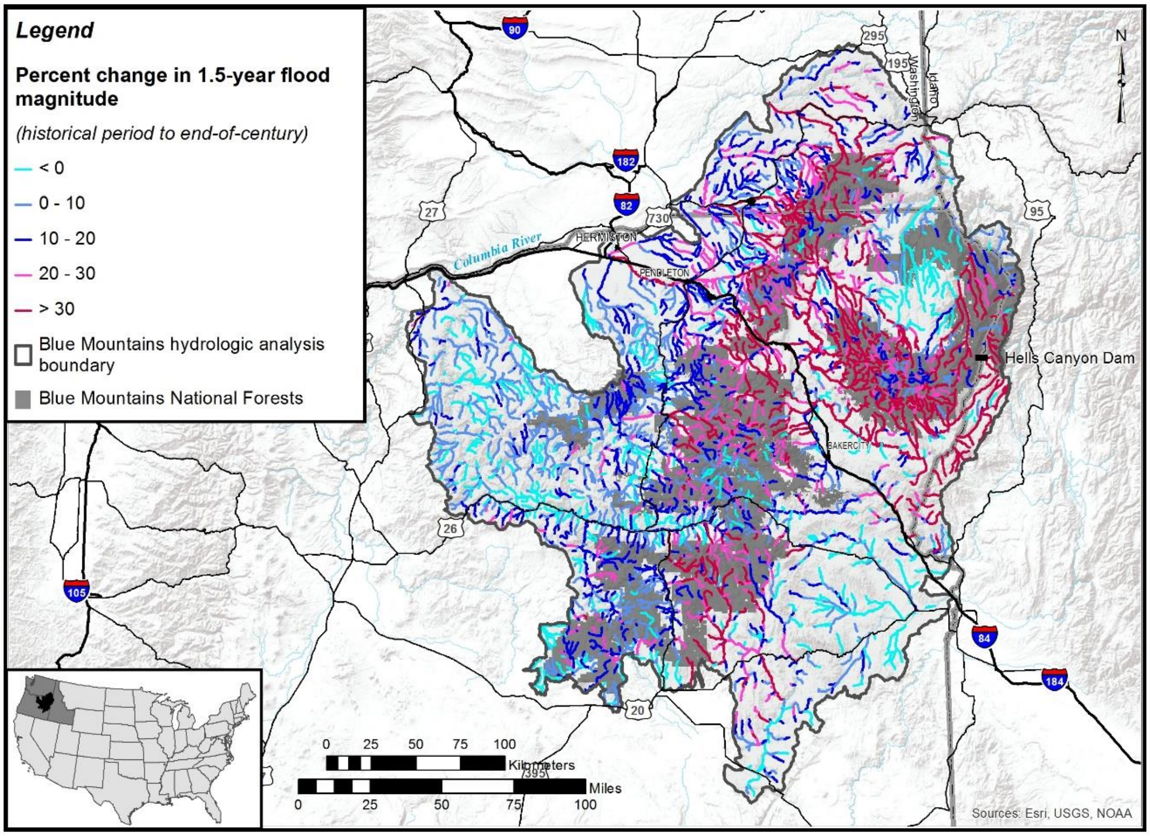

Legend
Percent change in 1.5-year flood magnitude
(historical period to end-of-century)
< 0
0 - 10
10 - 20
20 - 30
> 30
Blue Mountains hydrologic analysis boundary
Blue Mountains National Forests
Columbia River
HERMISTON
PENDLETON
BAKER CITY
Hells Canyon Dam
Washington
Idaho
N
0 25 50 75 100
Kilometers
Miles
0 25 50 75 100
Sources: Esri, USGS, NOAA

Fig. 3.5b and 3.5c

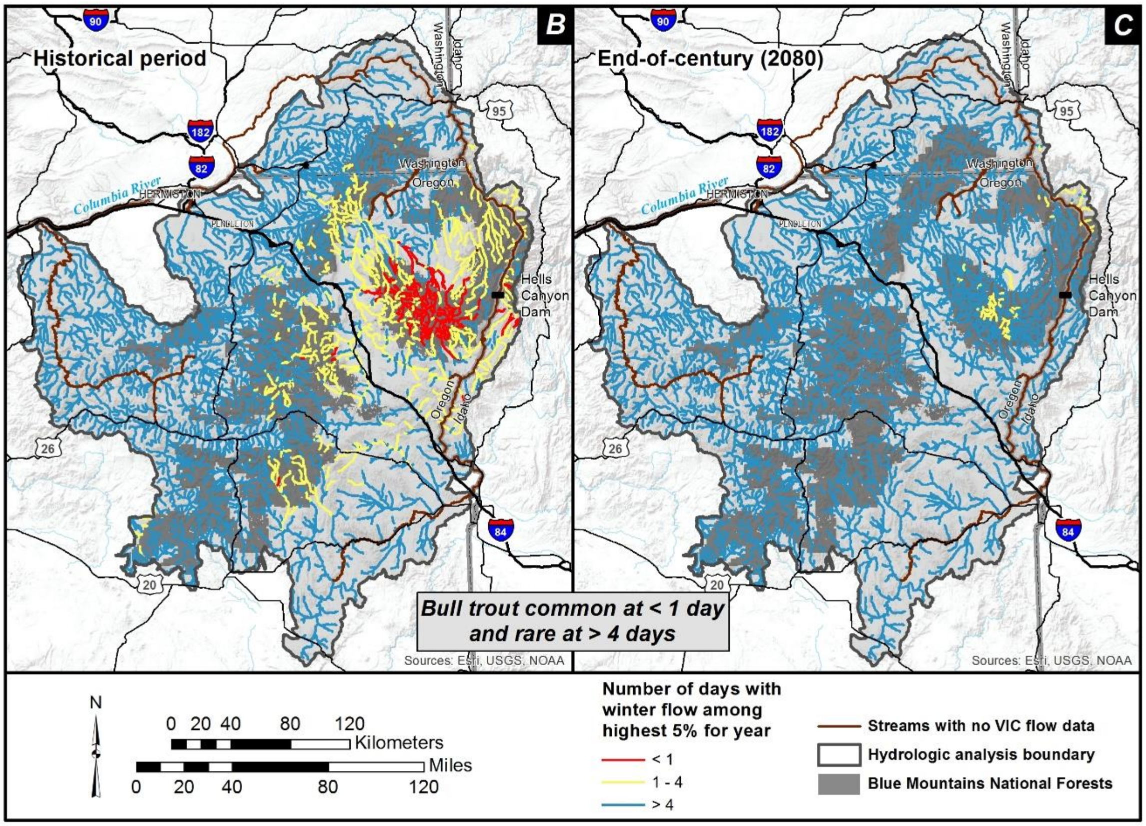


Fig. 3.6

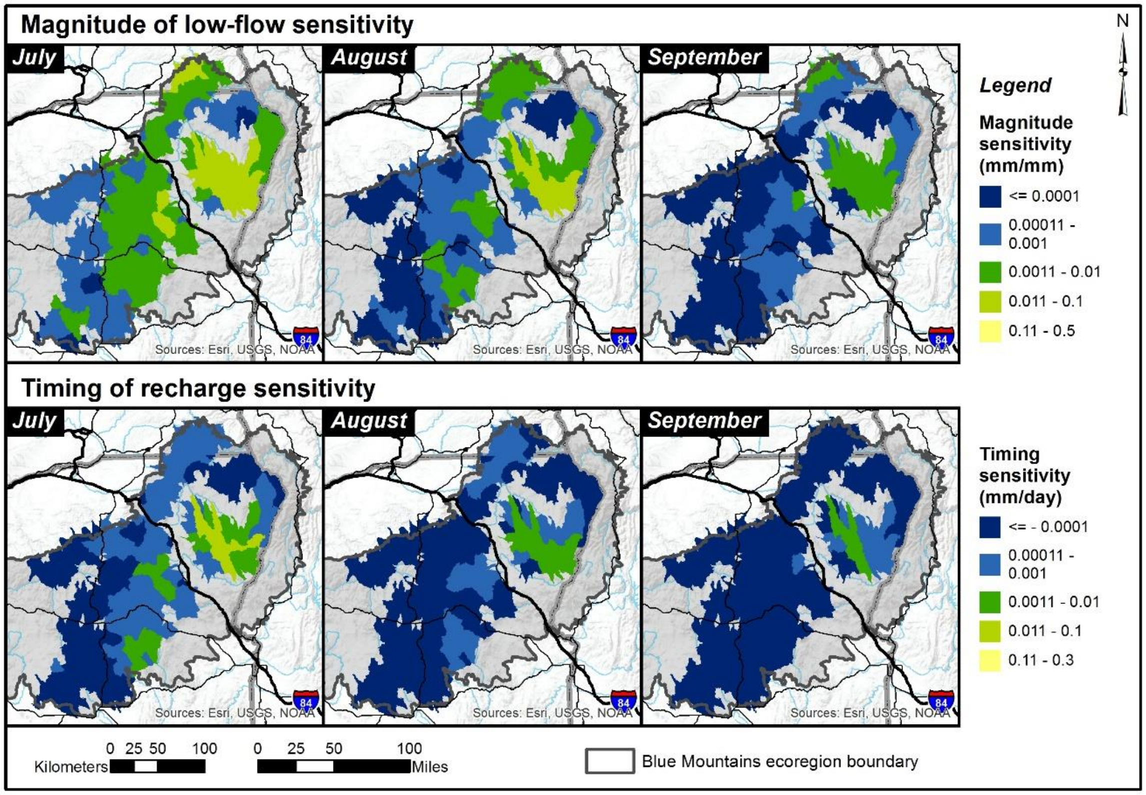

Magnitude of low-flow sensitivity
July
August
September
Sources: Esri, USGS, NOAA
Legend
Magnitude sensitivity (mm/mm)
<= 0.0001
0.00011 - 0.001
0.0011 - 0.01
0.011 - 0.1
0.11 - 0.5
Timing of recharge sensitivity
Timing sensitivity (mm/day)
<= - 0.0001
0.00011 - 0.001
0.0011 - 0.01
0.011 - 0.1
0.11 - 0.3
N
84
0 25 50 100 Kilometers
0 25 50 100 Miles
Blue Mountains ecoregion boundary

Fig. 3.7

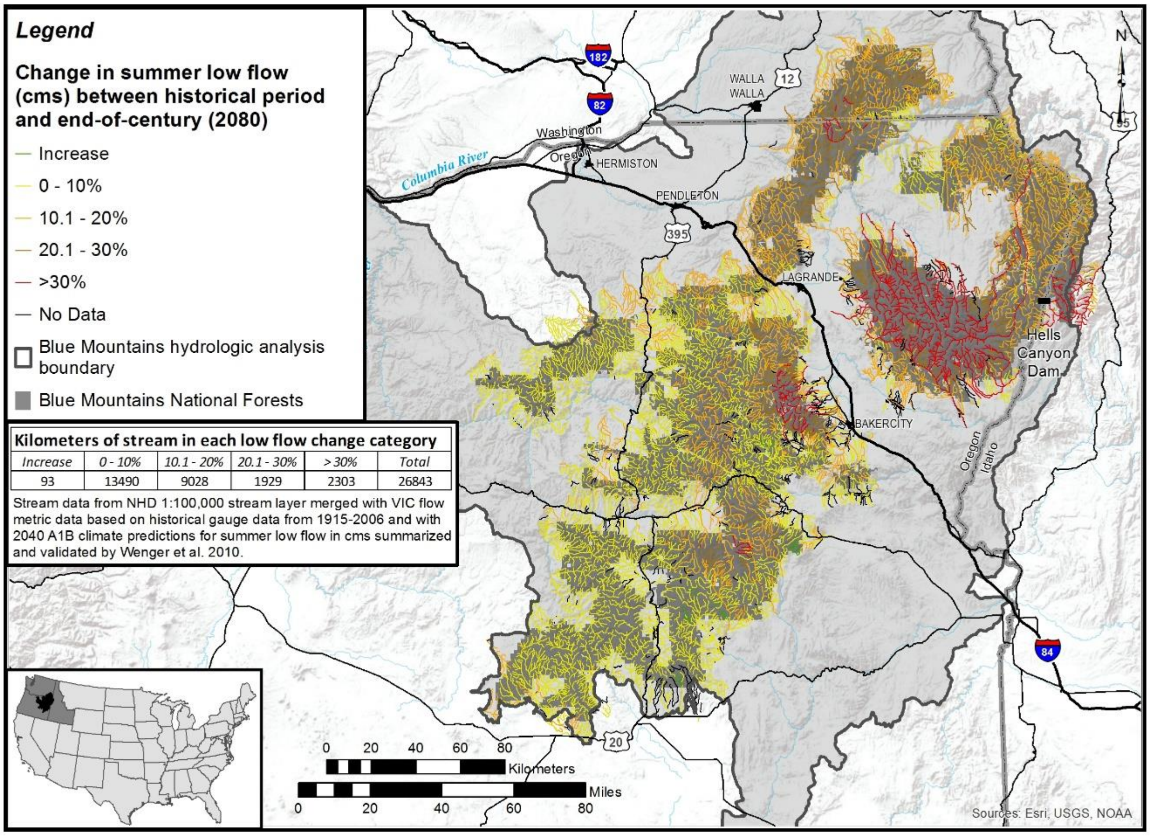

Legend
Change in summer low flow (cms) between historical period and end-of-century (2080)
Increase
0 - 10%
10.1 - 20%
20.1 - 30%
>30%
No Data
Blue Mountains hydrologic analysis boundary
Blue Mountains National Forests
Kilometers of stream in each low flow change category
Increase | 0 - 10% | 10.1 - 20% | 20.1 - 30% | > 30% | Total
93 | 13490 | 9028 | 1929 | 2303 | 26843
Stream data from NHD 1:100,000 stream layer merged with VIC flow metric data based on historical gauge data from 1915-2006 and with 2040 A1B climate predictions for summer low flow in cms summarized and validated by Wenger et al. 2010.
182
82
12
95
395
84
20
WALLA WALLA
Washington
Oregon
HERMISTON
Columbia River
PENDLETON
LAGRANDE
BAKERCITY
Hells Canyon Dam
Oregon
Idaho
N
0 20 40 60 80
Kilometers
Miles
0 20 40 60 80
Sources: Esri, USGS, NOAA

Fig. 3.8

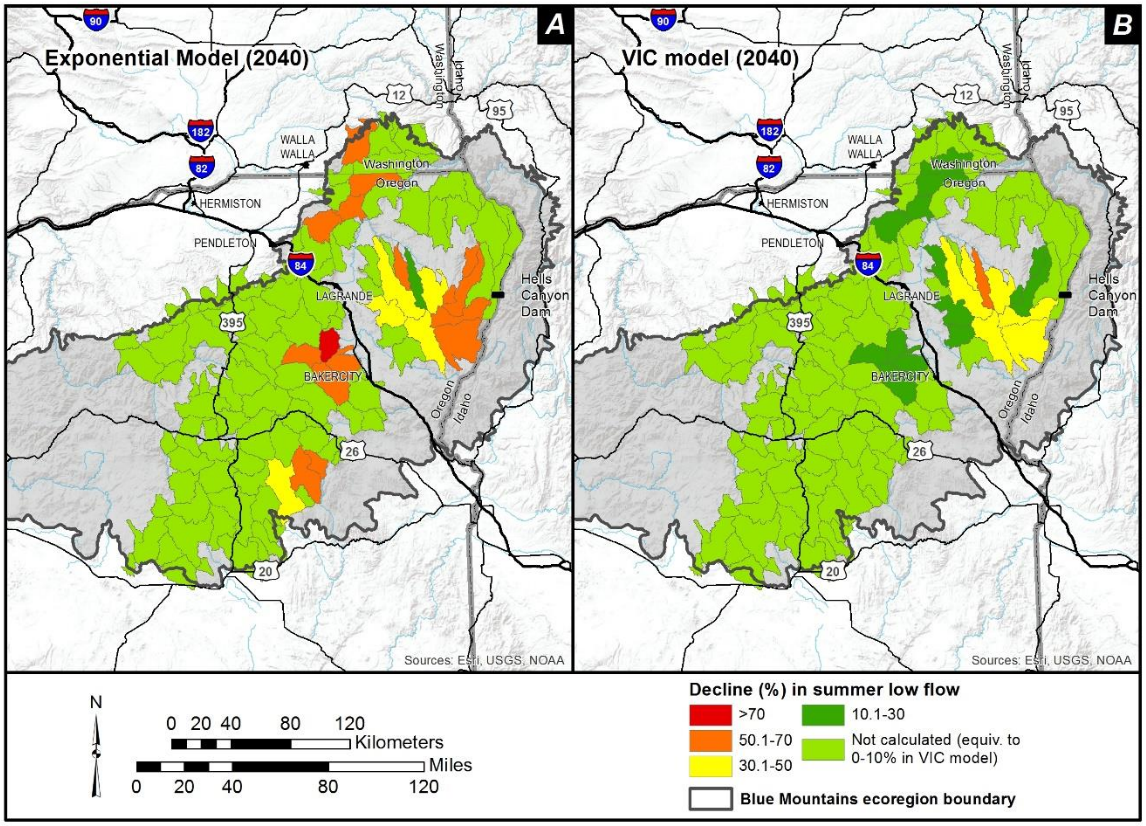

A
Exponential Model (2040)
B
VIC model (2040)
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Hells Canyon Dam
Washington
Oregon
Idaho
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
Decline (%) in summer low flow
>70
50.1-70
30.1-50
10.1-30
Not calculated (equiv. to 0-10% in VIC model)
Blue Mountains ecoregion boundary

Fig. 3.9

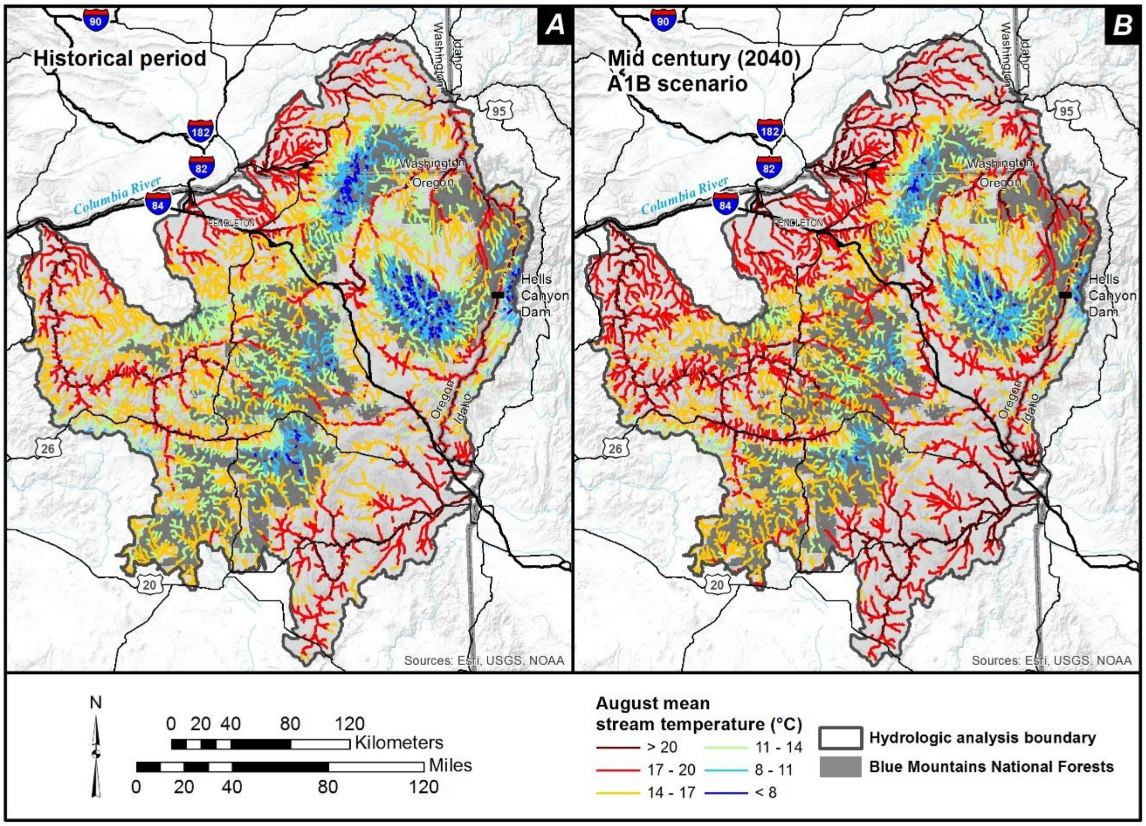
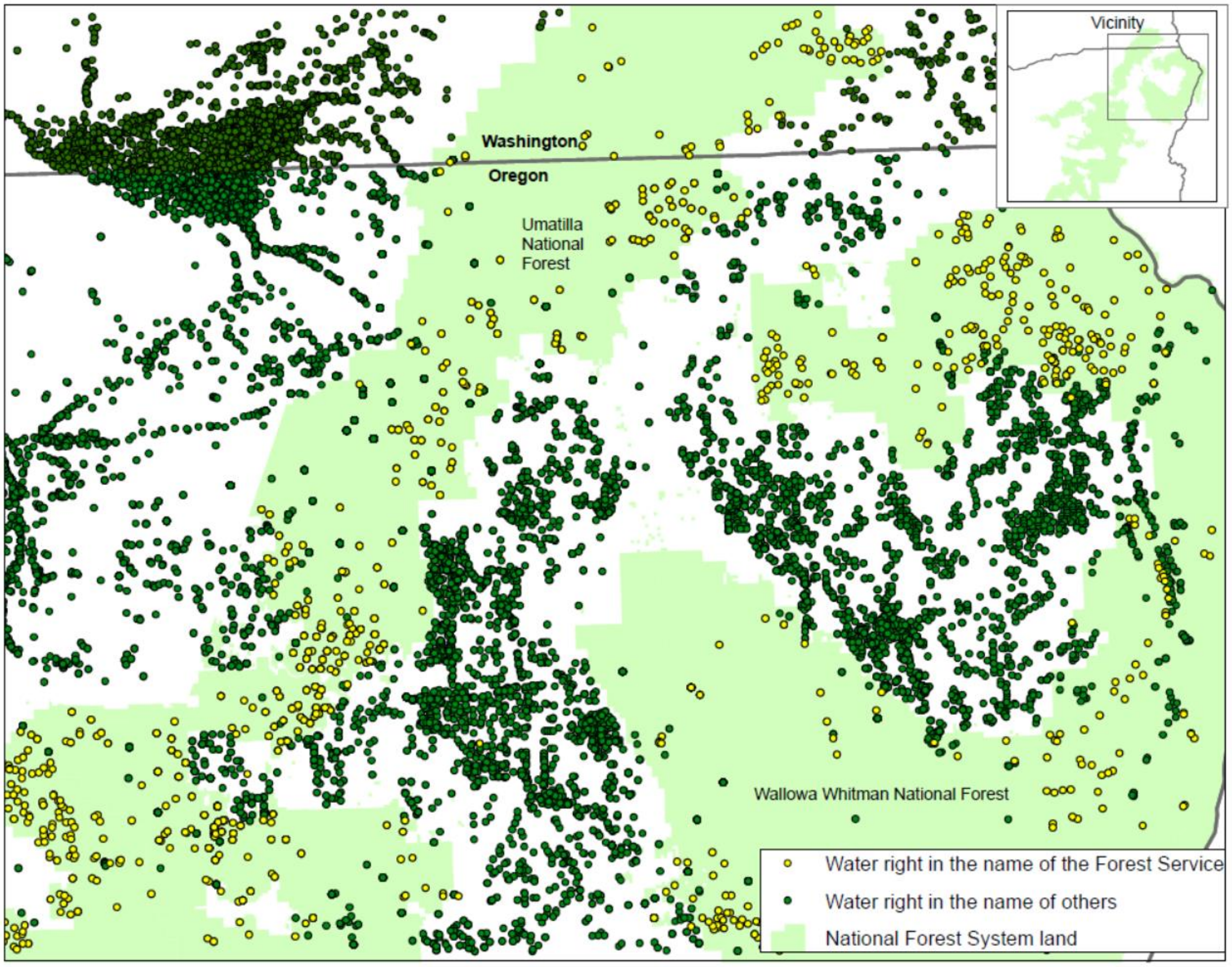

A
Historical period
B
Mid century (2040)
A1B scenario
Columbia River
Washington
Oregon
Idaho
Hells
Canyon
Dam
PENDLETON
90
182
82
84
95
26
20
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
August mean
stream temperature (°C)
> 20
17 - 20
14 - 17
11 - 14
8 - 11
< 8
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 4.1



Vicinity
Washington
Oregon
Umatilla National Forest
Wallowa Whitman National Forest
Water right in the name of the Forest Service
Water right in the name of others
National Forest System land

Fig. 4.2

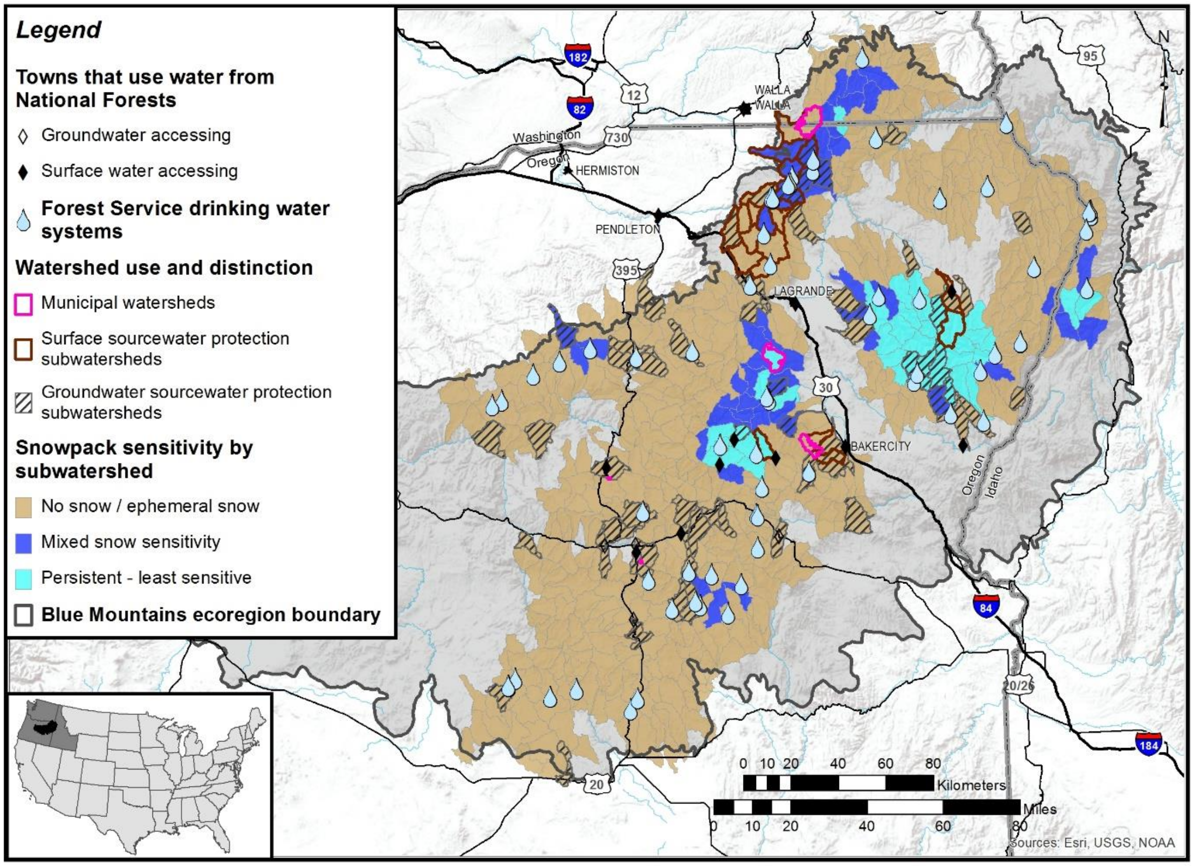

Legend
Towns that use water from National Forests
Groundwater accessing
Surface water accessing
Forest Service drinking water systems
Watershed use and distinction
Municipal watersheds
Surface sourcewater protection subwatersheds
Groundwater sourcewater protection subwatersheds
Snowpack sensitivity by subwatershed
No snow / ephemeral snow
Mixed snow sensitivity
Persistent - least sensitive
Blue Mountains ecoregion boundary
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Washington
Oregon
Idaho
0 10 20 40 60 80
Kilometers
0 10 20 40 60 80
Miles
Sources: Esri, USGS, NOAA

Fig. 4.3

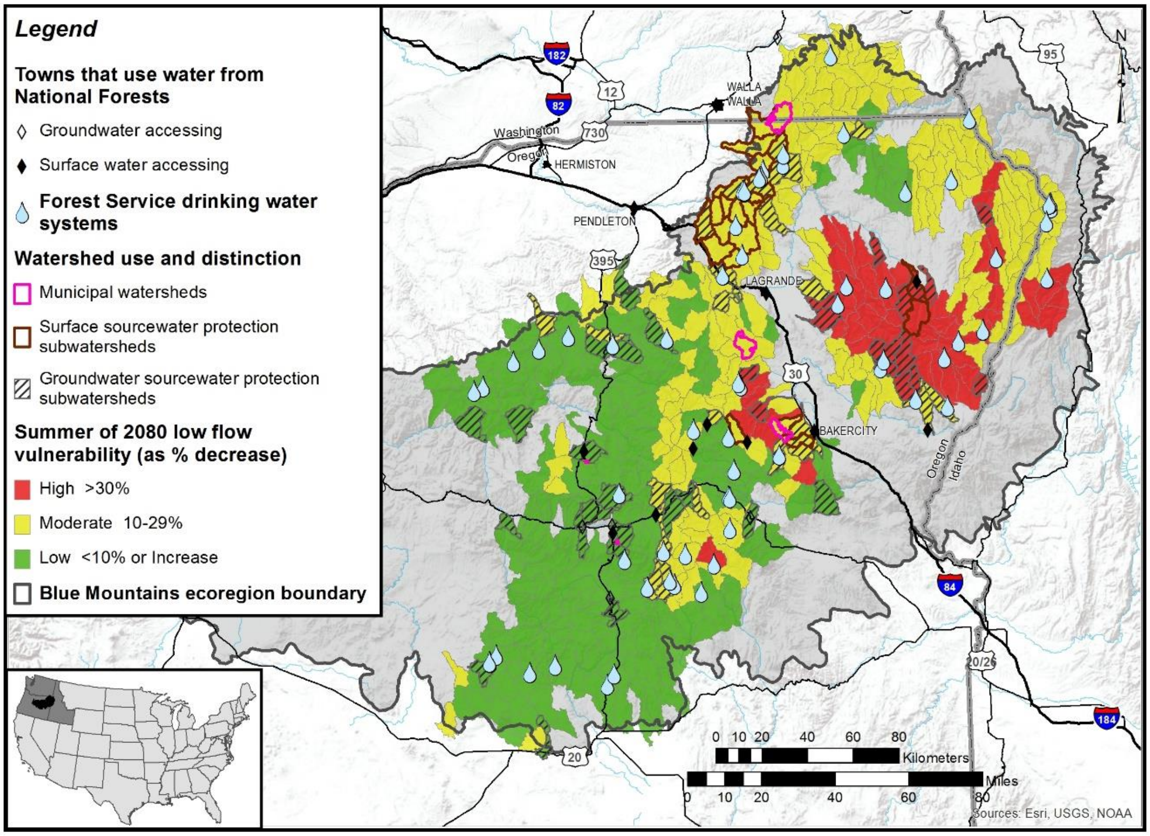
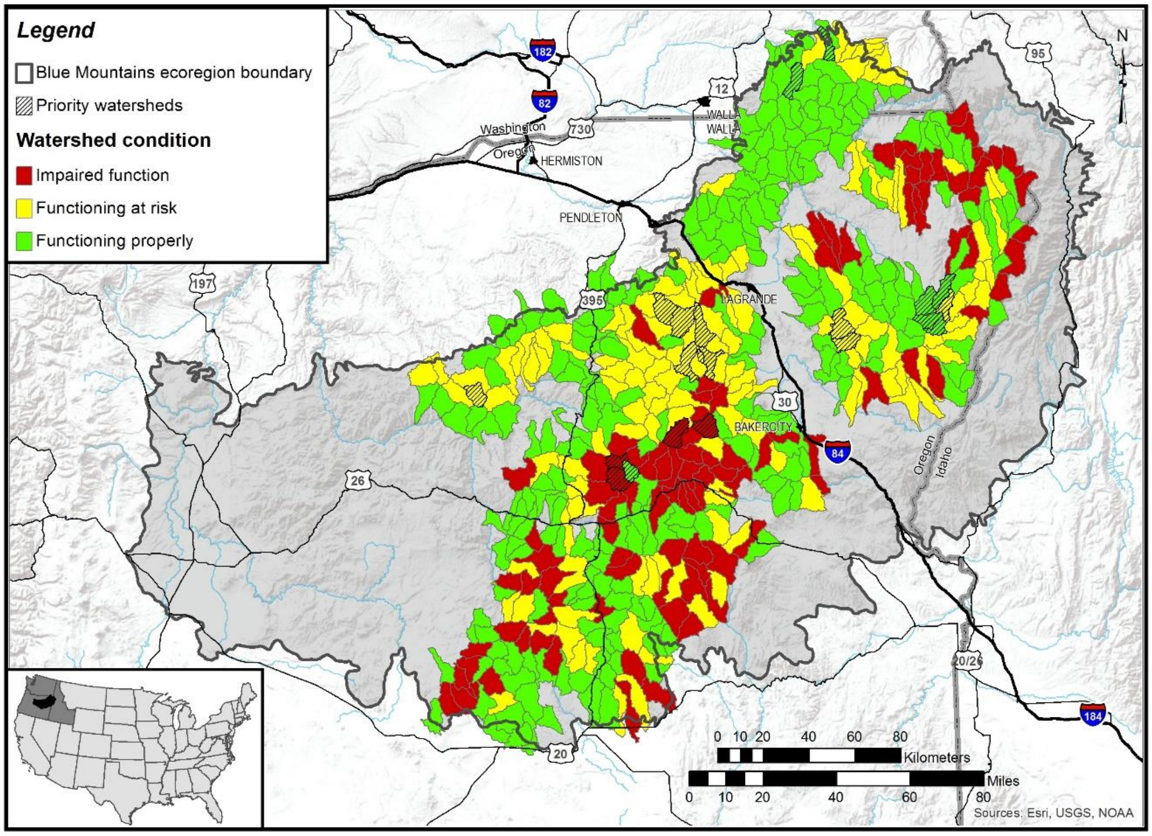

Legend
Towns that use water from National Forests
Groundwater accessing
Surface water accessing
Forest Service drinking water systems
Watershed use and distinction
Municipal watersheds
Surface sourcewater protection subwatersheds
Groundwater sourcewater protection subwatersheds
Summer of 2080 low flow vulnerability (as % decrease)
High >30%
Moderate 10-29%
Low <10% or Increase
Blue Mountains ecoregion boundary
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Washington
Oregon
Idaho
182
82
12
730
395
30
95
84
20/26
184
20
0 10 20 40 60 80
Kilometers
0 10 20 40 60 80
Miles
Sources: Esri, USGS, NOAA

Fig. 4.4



Legend
Blue Mountains ecoregion boundary
Priority watersheds
Watershed condition
Impaired function
Functioning at risk
Functioning properly
Washington
Oregon
HERMISTON
PENDLETON
WALLA WALLA
LAGRANDE
BAKER CITY
Oregon
Idaho
N
182
82
12
730
95
197
395
30
84
26
20
20/26
184
0 10 20 40 60 80
Kilometers
0 10 20 40 60 80
Miles
Sources: Esri, USGS, NOAA

Fig. 4.5

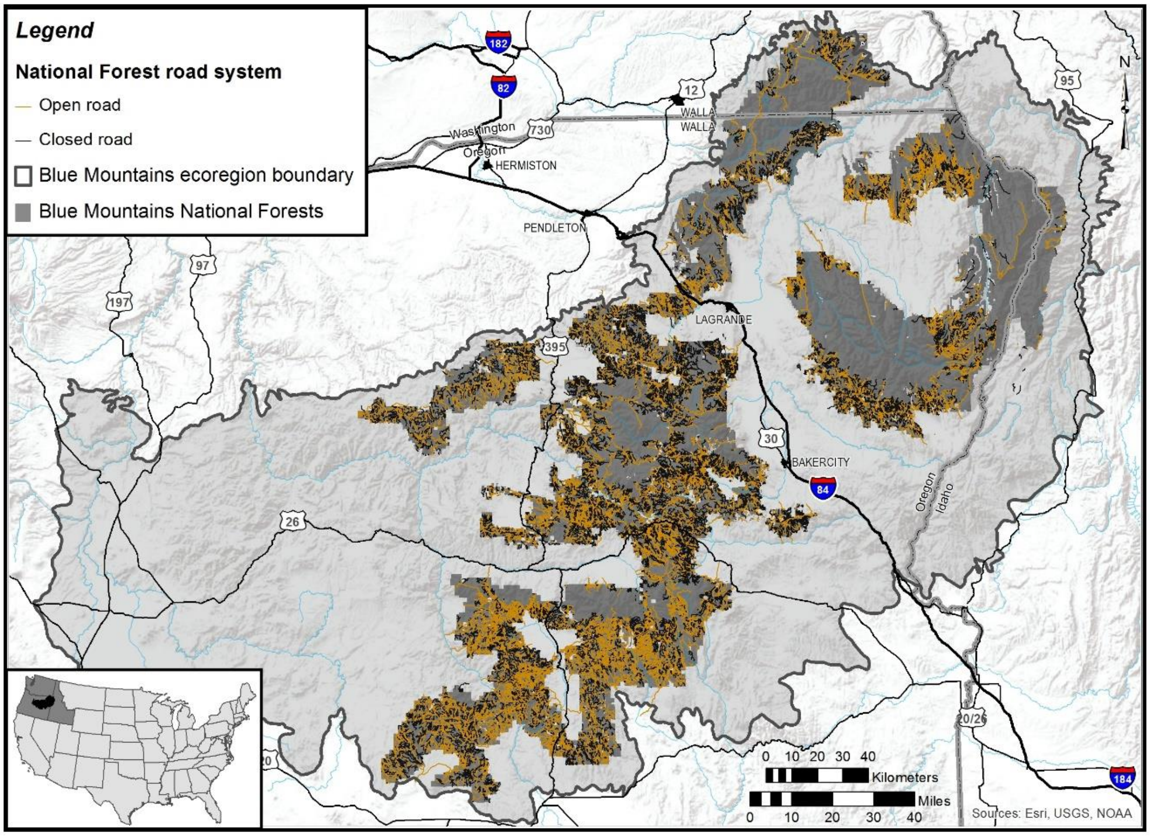

Legend
National Forest road system
Open road
Closed road
Blue Mountains ecoregion boundary
Blue Mountains National Forests
182
82
12
95
N
WALLA
WALLA
Washington
730
Oregon
HERMISTON
PENDLETON
97
197
LAGRANDE
395
30
BAKERCITY
84
Oregon
Idaho
26
20/26
0 10 20 30 40
Kilometers
Miles
0 10 20 30 40
184
Sources: Esri, USGS, NOAA

Fig. 4.6

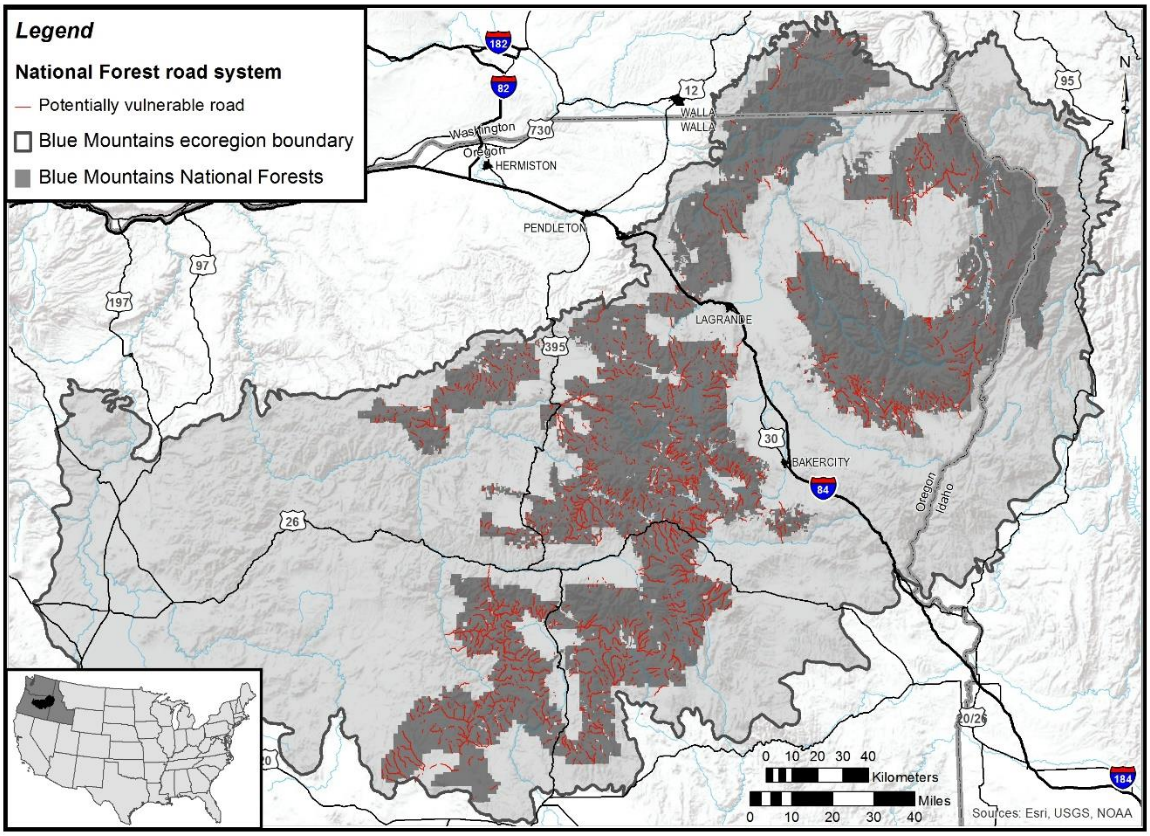
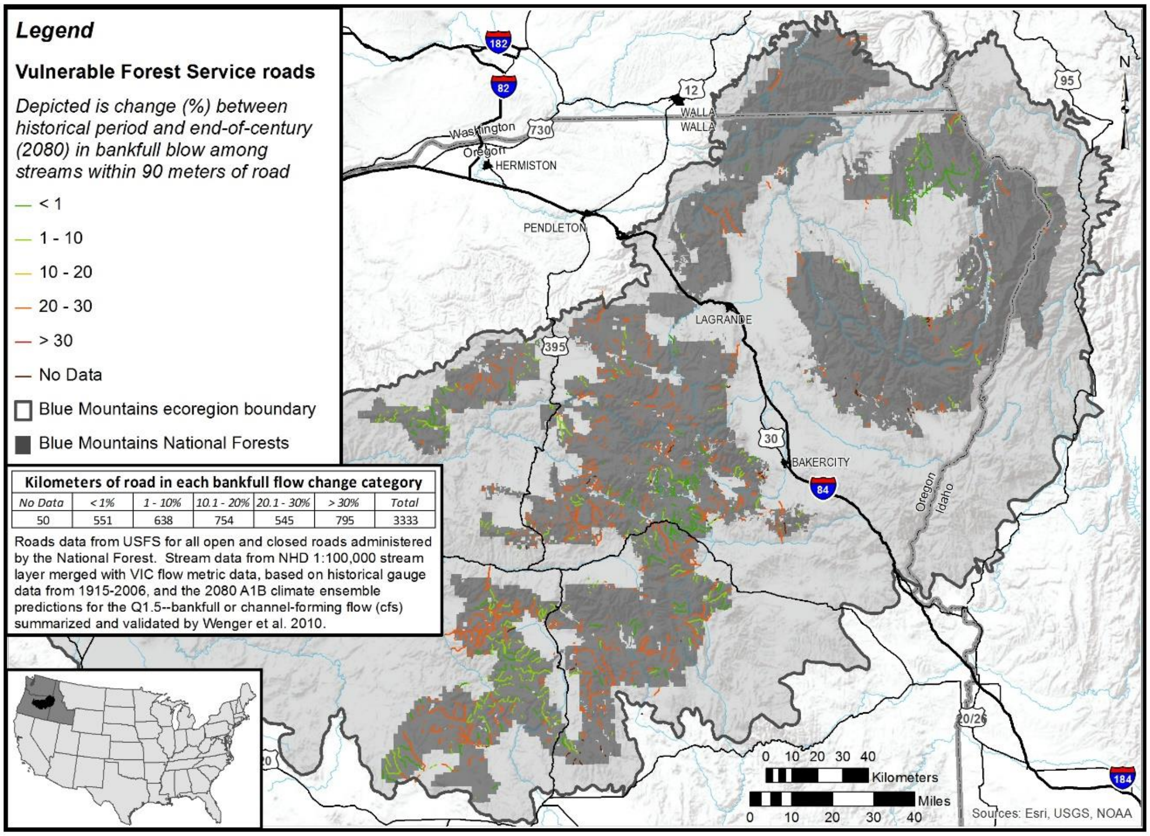

Legend
National Forest road system
Potentially vulnerable road
Blue Mountains ecoregion boundary
Blue Mountains National Forests
WALLA WALLA
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Washington
Oregon
Idaho
0 10 20 30 40 Kilometers
0 10 20 30 40 Miles
Sources: Esri, USGS, NOAA

Fig. 4.7



Kilometers of road in each bankfull flow change category						
No Data	*< 1%*	*1 - 10%*	*10.1 - 20%*	*20.1 - 30%*	*> 30%*	*Total*
50	551	638	754	545	795	3333

Fig. 4.8

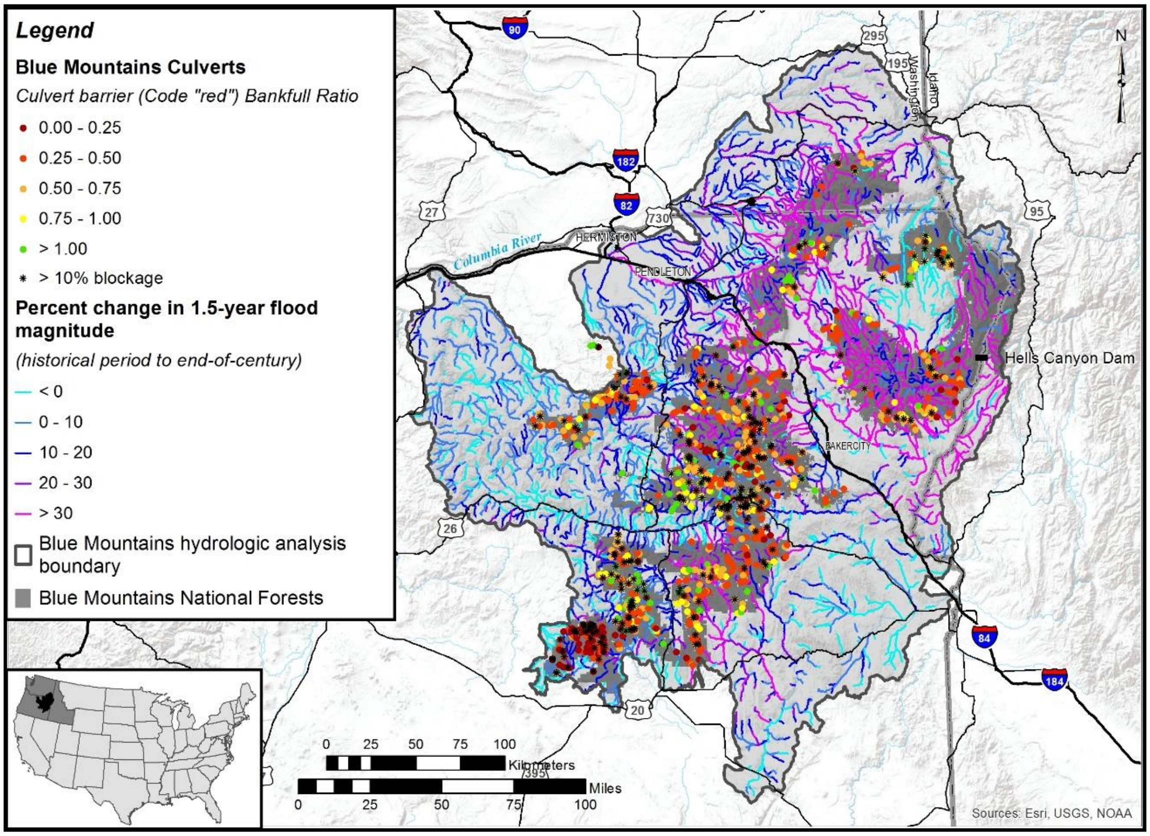
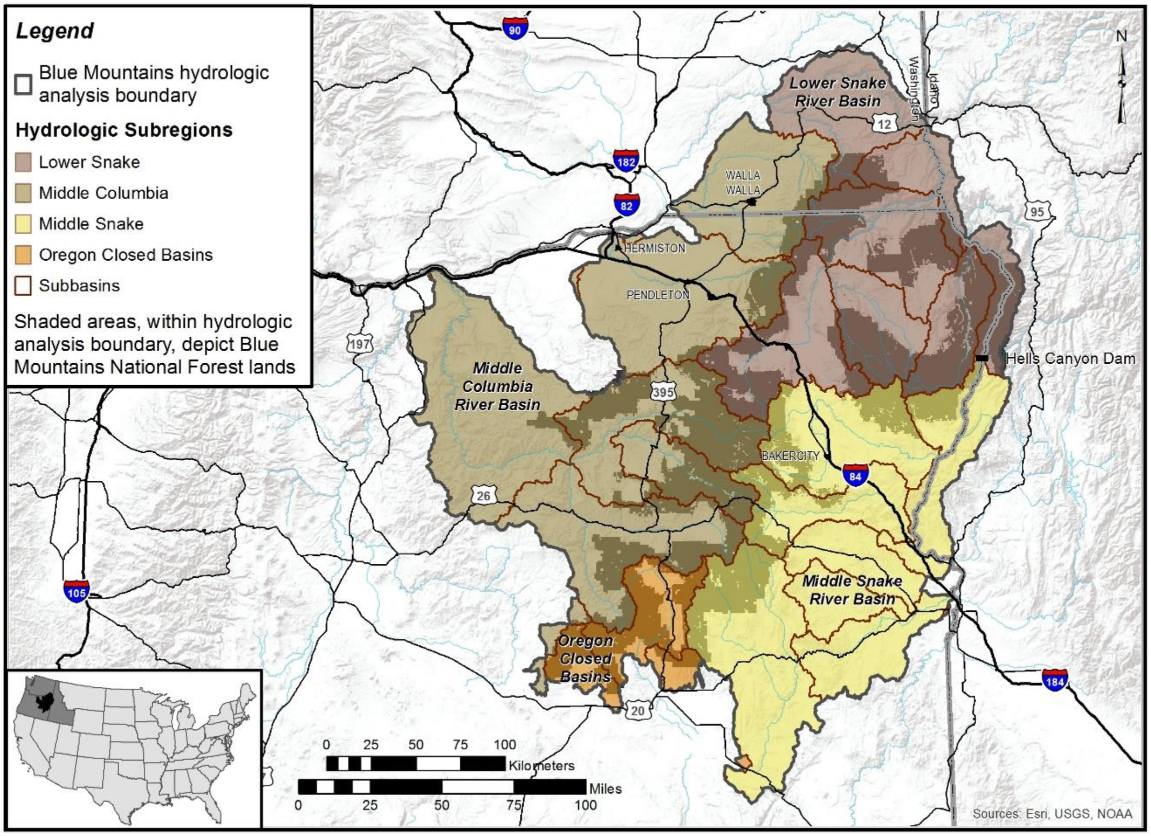

Legend
Blue Mountains Culverts
Culvert barrier (Code "red") Bankfull Ratio
0.00 - 0.25
0.25 - 0.50
0.50 - 0.75
0.75 - 1.00
> 1.00
> 10% blockage
Percent change in 1.5-year flood magnitude
(historical period to end-of-century)
< 0
0 - 10
10 - 20
20 - 30
> 30
Blue Mountains hydrologic analysis boundary
Blue Mountains National Forests
Columbia River
HERMISTON
PENDLETON
BAKERCITY
Hells Canyon Dam
Washington
Idaho
N
0 25 50 75 100 Kilometers
0 25 50 75 100 Miles
Sources: Esri, USGS, NOAA

Fig. 5.1



Legend
Blue Mountains hydrologic analysis boundary
Hydrologic Subregions
Lower Snake
Middle Columbia
Middle Snake
Oregon Closed Basins
Subbasins
Shaded areas, within hydrologic analysis boundary, depict Blue Mountains National Forest lands
Lower Snake River Basin
Middle Columbia River Basin
Middle Snake River Basin
Oregon Closed Basins
WALLA WALLA
HERMISTON
PENDLETON
BAKERCITY
Hells Canyon Dam
Washington
Idaho
N
0 25 50 75 100 Kilometers
0 25 50 75 100 Miles
Sources: Esri, USGS, NOAA

Fig. 5.2

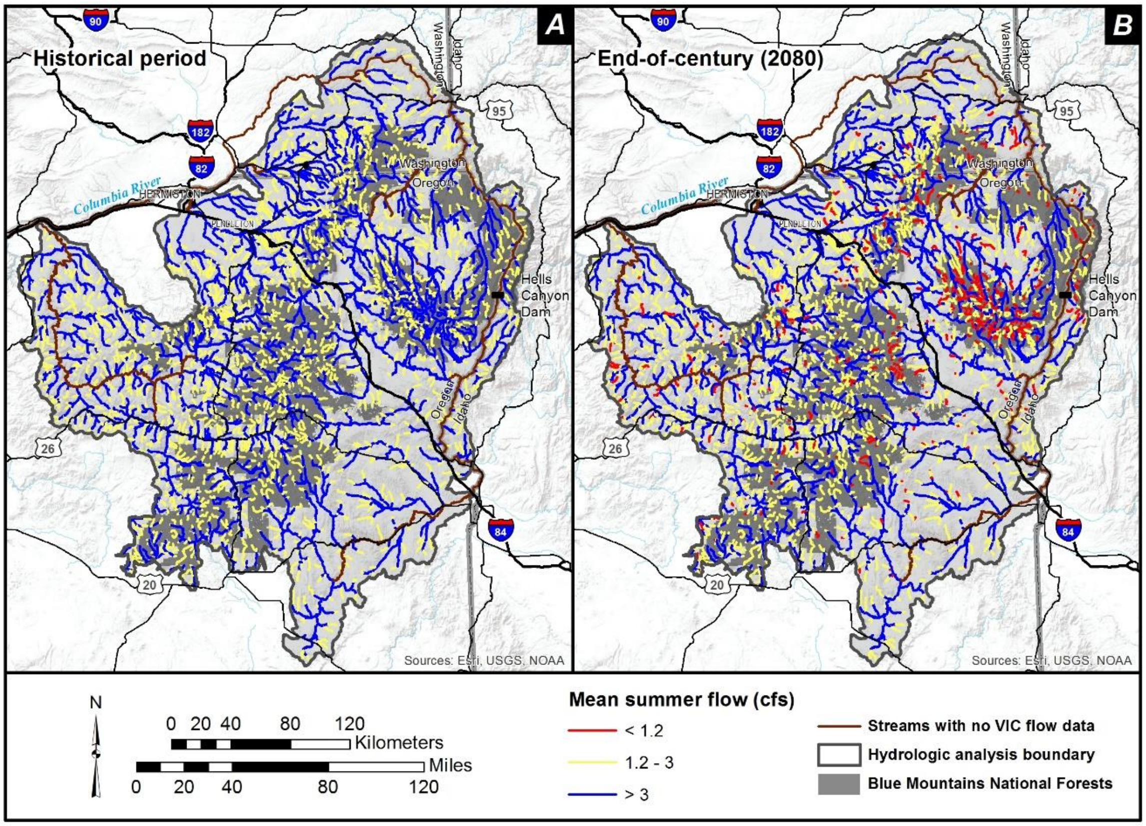

A
Historical period
B
End-of-century (2080)
Columbia River
HERMISTON
Washington
Oregon
Idaho
Hells
Canyon
Dam
90
182
82
95
26
84
20
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
Mean summer flow (cfs)
< 1.2
1.2 - 3
> 3
Streams with no VIC flow data
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.3

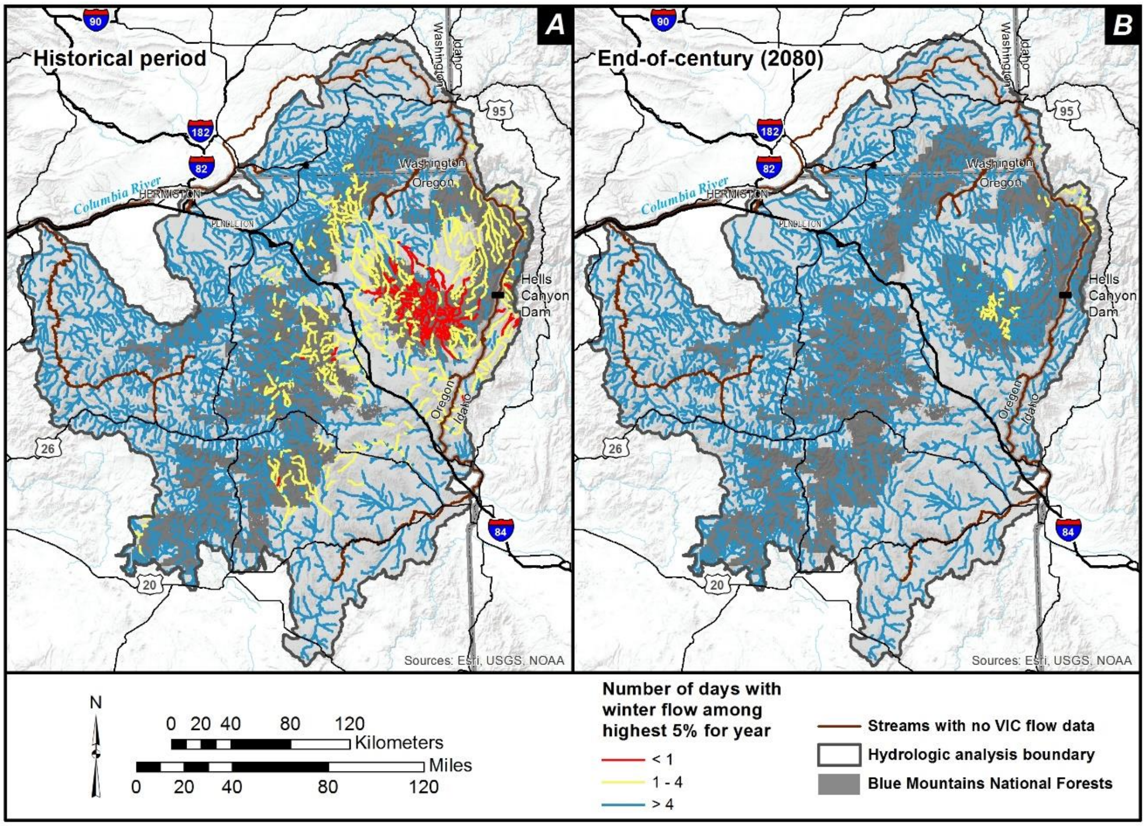

A
Historical period
B
End-of-century (2080)
Columbia River
Washington
Oregon
Idaho
Hells Canyon Dam
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
Number of days with winter flow among highest 5% for year
< 1
1 - 4
> 4
Streams with no VIC flow data
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.4

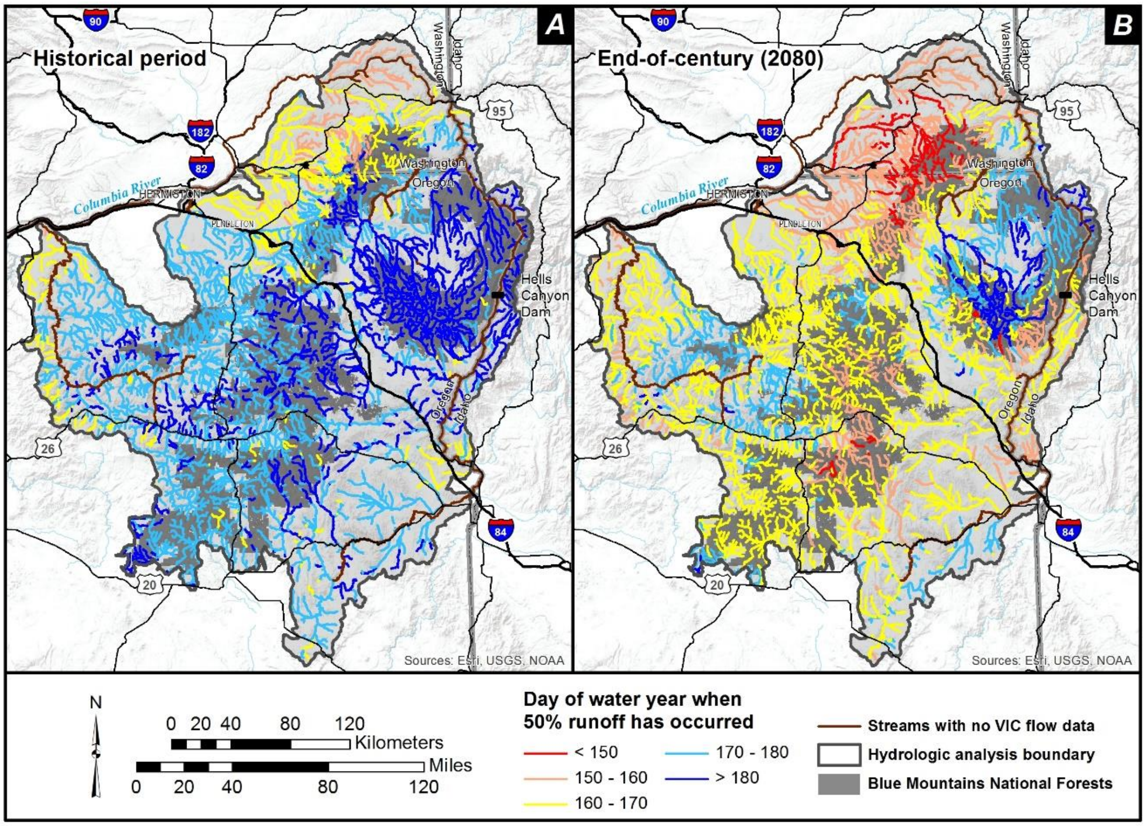

A
Historical period
B
End-of-century (2080)
Washington
Idaho
Oregon
Columbia River
HERMISTON
PENDLETON
Hells
Canyon
Dam
90
182
82
95
26
20
84
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
Day of water year when
50% runoff has occurred
< 150
150 - 160
160 - 170
170 - 180
> 180
Streams with no VIC flow data
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.5

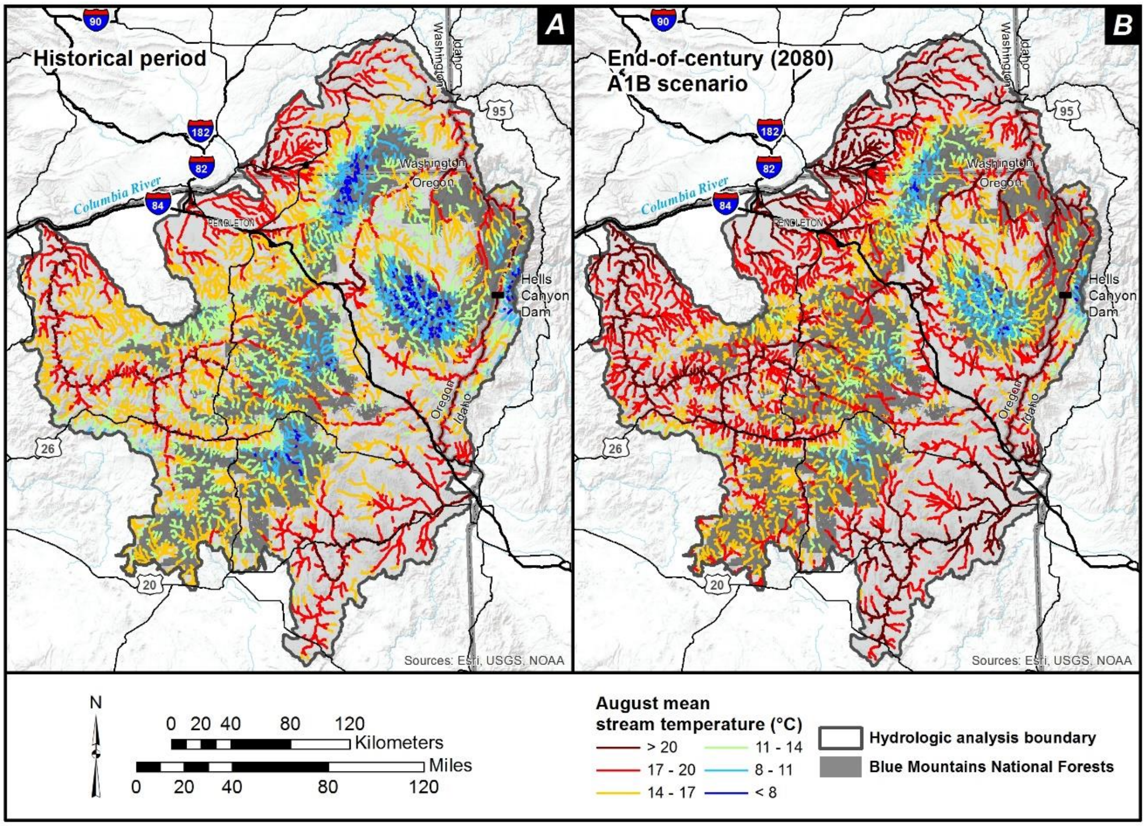

A
Historical period
B
End-of-century (2080)
A1B scenario
Columbia River
Washington
Idaho
Oregon
PENDLETON
Hells
Canyon
Dam
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
August mean
stream temperature (°C)
> 20
17 - 20
14 - 17
11 - 14
8 - 11
< 8
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.6

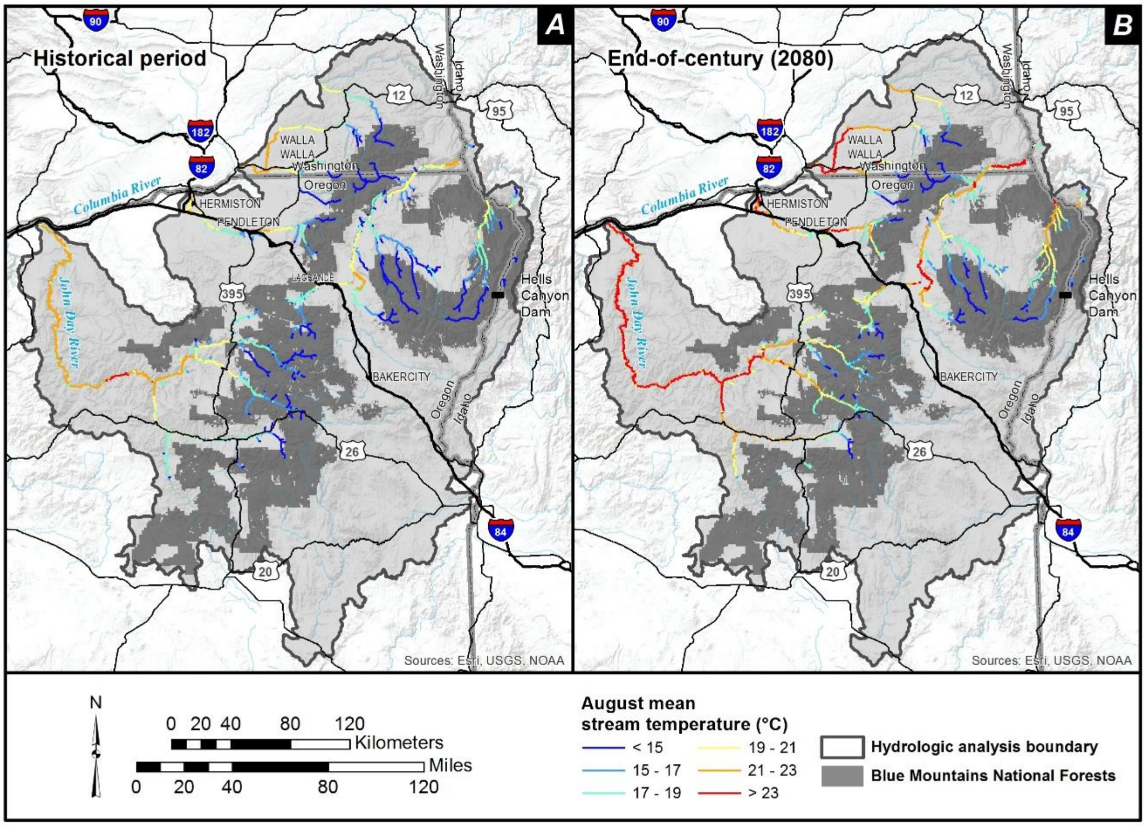

A
Historical period
B
End-of-century (2080)
Columbia River
John Day River
WALLA
WALLA
Washington
Oregon
HERMISTON
PENDLETON
LA GRANDE
BAKER CITY
Hells
Canyon
Dam
Washington
Idaho
Oregon
Idaho
90
182
82
12
95
395
26
84
20
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
August mean
stream temperature (°C)
< 15
15 - 17
17 - 19
19 - 21
21 - 23
> 23
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.7

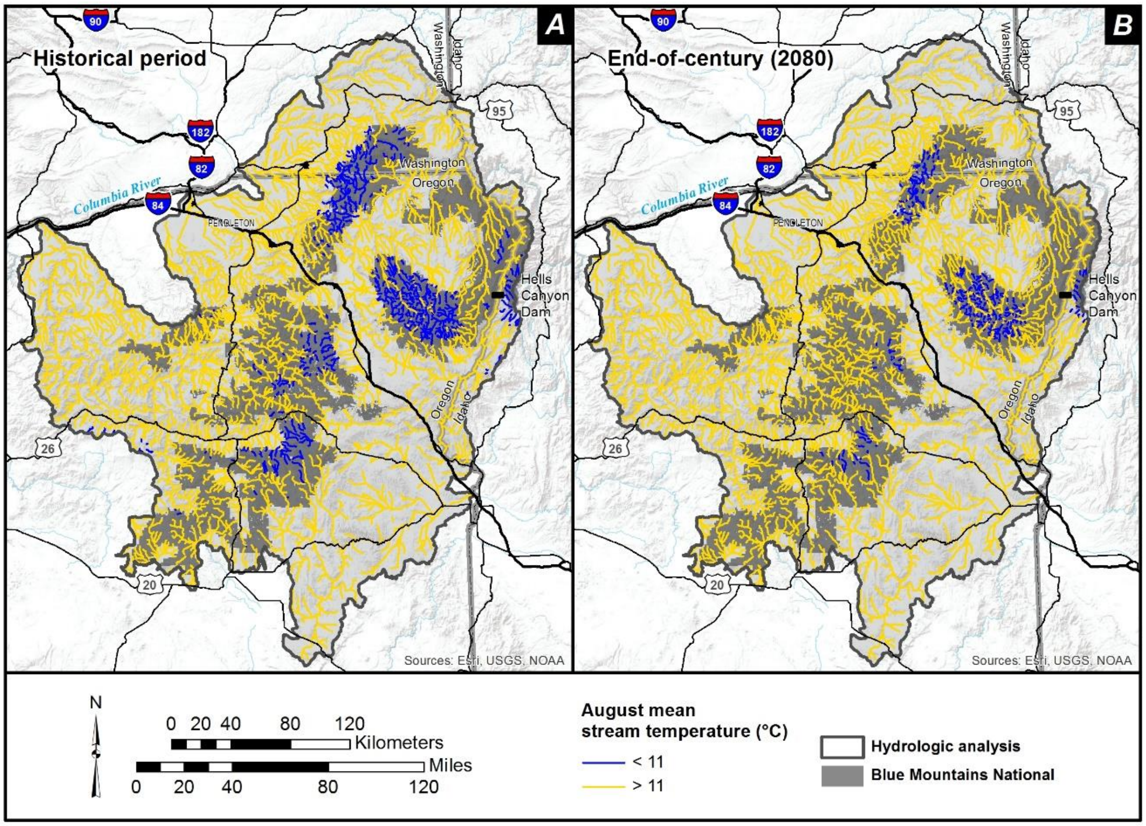

A
Historical period
B
End-of-century (2080)
Columbia River
Washington
Idaho
Oregon
Pendleton
Hells Canyon Dam
90
182
82
84
95
26
20
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
August mean
stream temperature (°C)
< 11
> 11
Hydrologic analysis
Blue Mountains National

Fig. 5.8

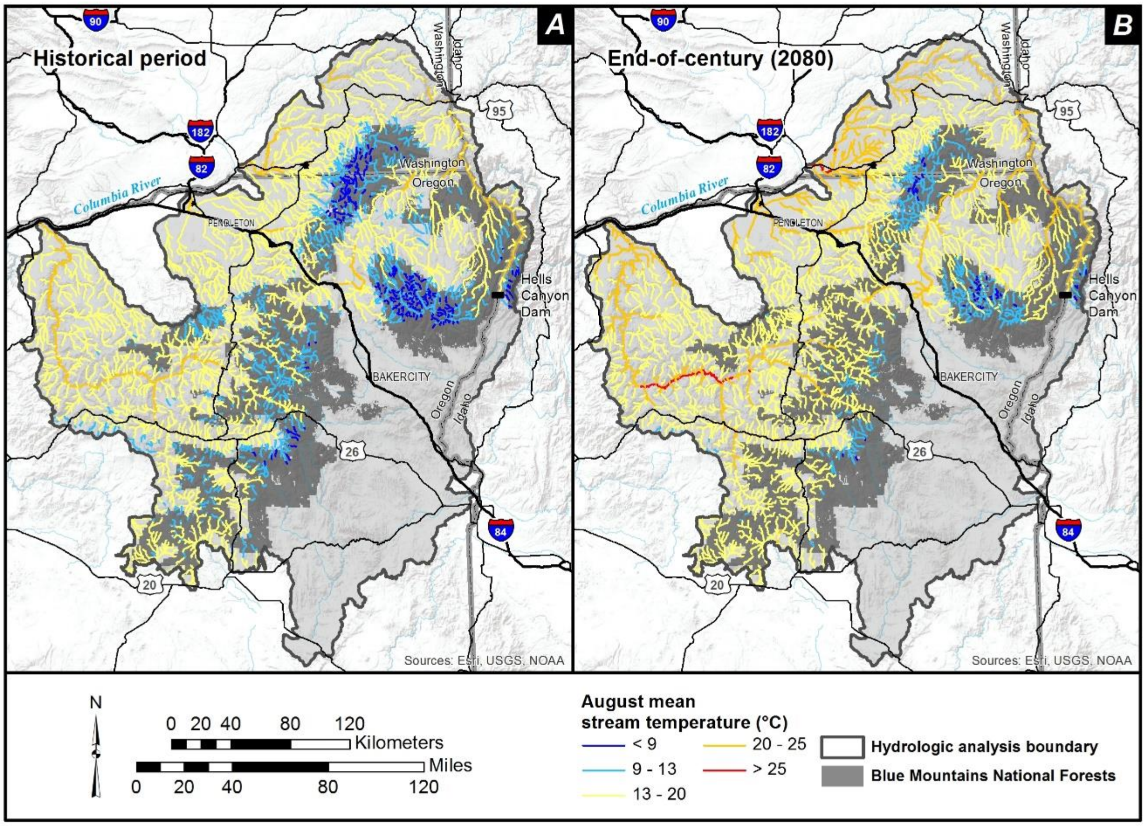

A
Historical period
B
End-of-century (2080)
Columbia River
Washington
Oregon
Idaho
PENDLETON
BAKERCITY
Hells Canyon Dam
90
182
82
95
26
20
84
Sources: Esri, USGS, NOAA
N
0 20 40 80 120
Kilometers
Miles
0 20 40 80 120
August mean
stream temperature (°C)
< 9
9 - 13
13 - 20
20 - 25
> 25
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 5.9

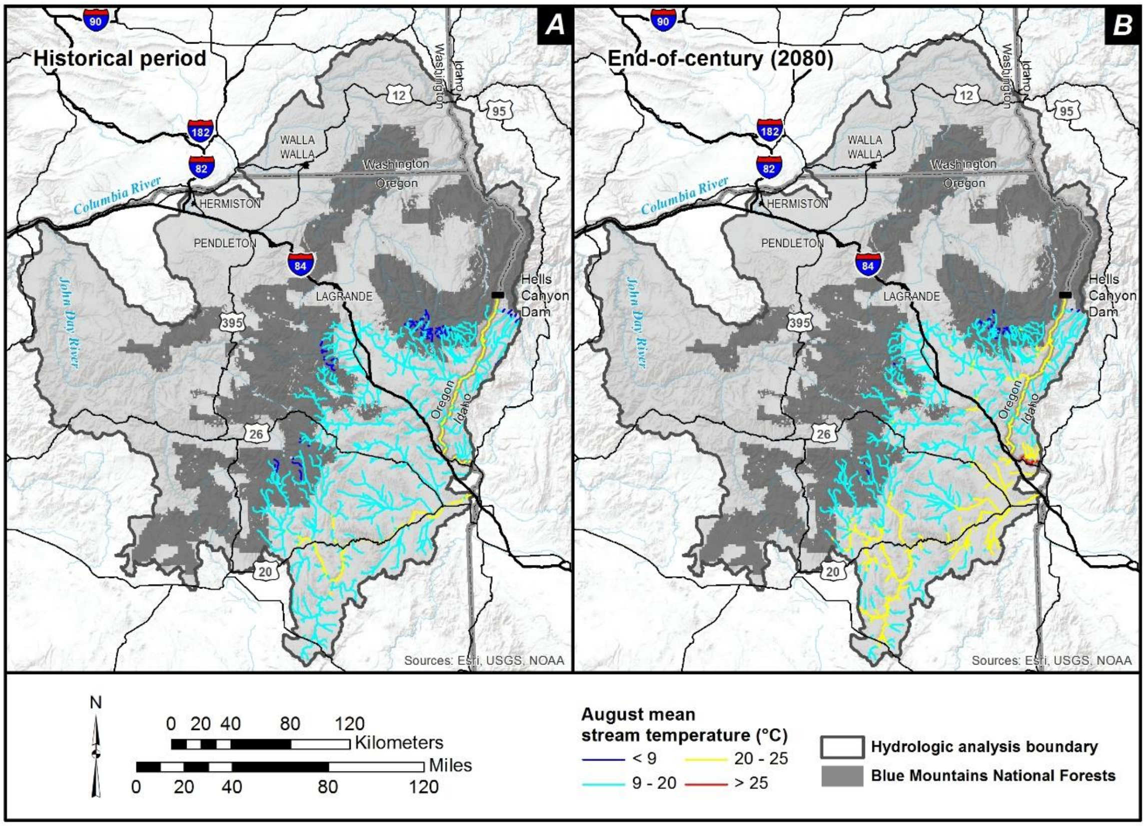

A
Historical period
B
End-of-century (2080)
Columbia River
John Day River
WALLA WALLA
Washington
Oregon
Idaho
HERMISTON
PENDLETON
LAGRANDE
Hells Canyon Dam
90
182
82
84
12
95
395
26
20
Sources: Esri, USGS, NOAA
N
0 20 40 80 120 Kilometers
0 20 40 80 120 Miles
August mean stream temperature (°C)
< 9
9 - 20
20 - 25
> 25
Hydrologic analysis boundary
Blue Mountains National Forests

Fig. 6.1

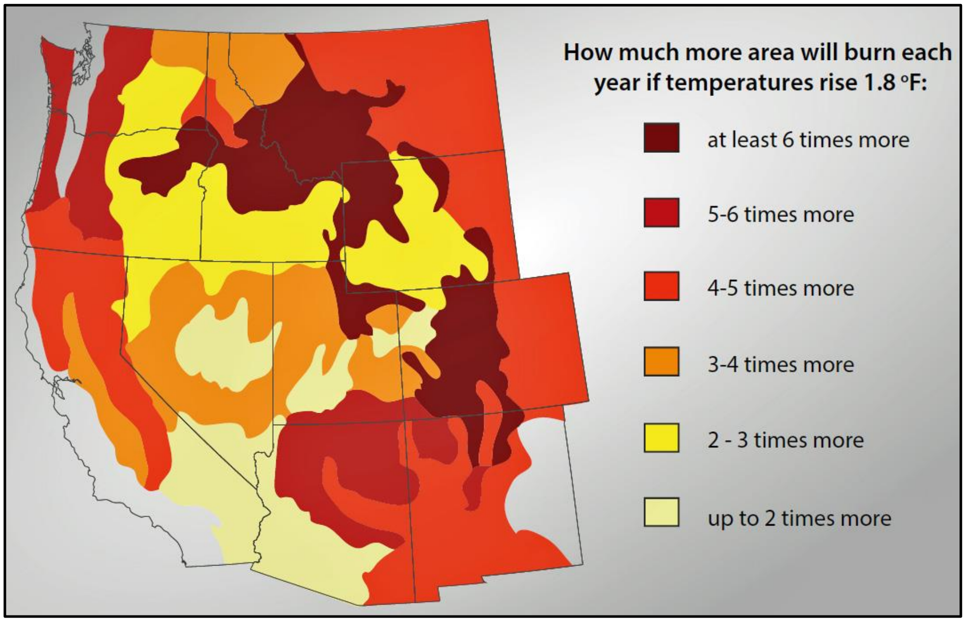

How much more area will burn each year if temperatures rise 1.8 °F:
at least 6 times more
5-6 times more
4-5 times more
3-4 times more
2 - 3 times more
up to 2 times more

Fig. 6.2

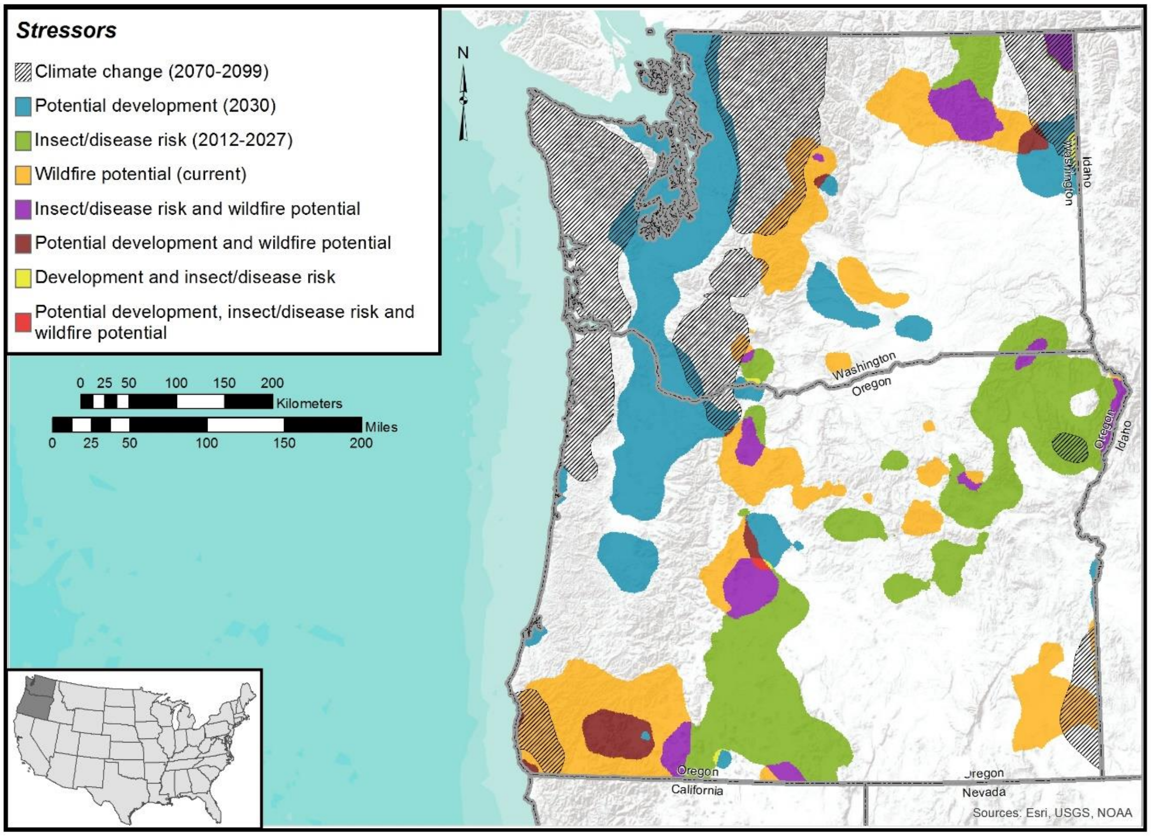

Stressors
Climate change (2070-2099)
Potential development (2030)
Insect/disease risk (2012-2027)
Wildfire potential (current)
Insect/disease risk and wildfire potential
Potential development and wildfire potential
Development and insect/disease risk
Potential development, insect/disease risk and wildfire potential
0 25 50 100 150 200
Kilometers
Miles
0 25 50 100 150 200
N
Washington
Oregon
Idaho
California
Nevada
Sources: Esri, USGS, NOAA

Fig. 6.3

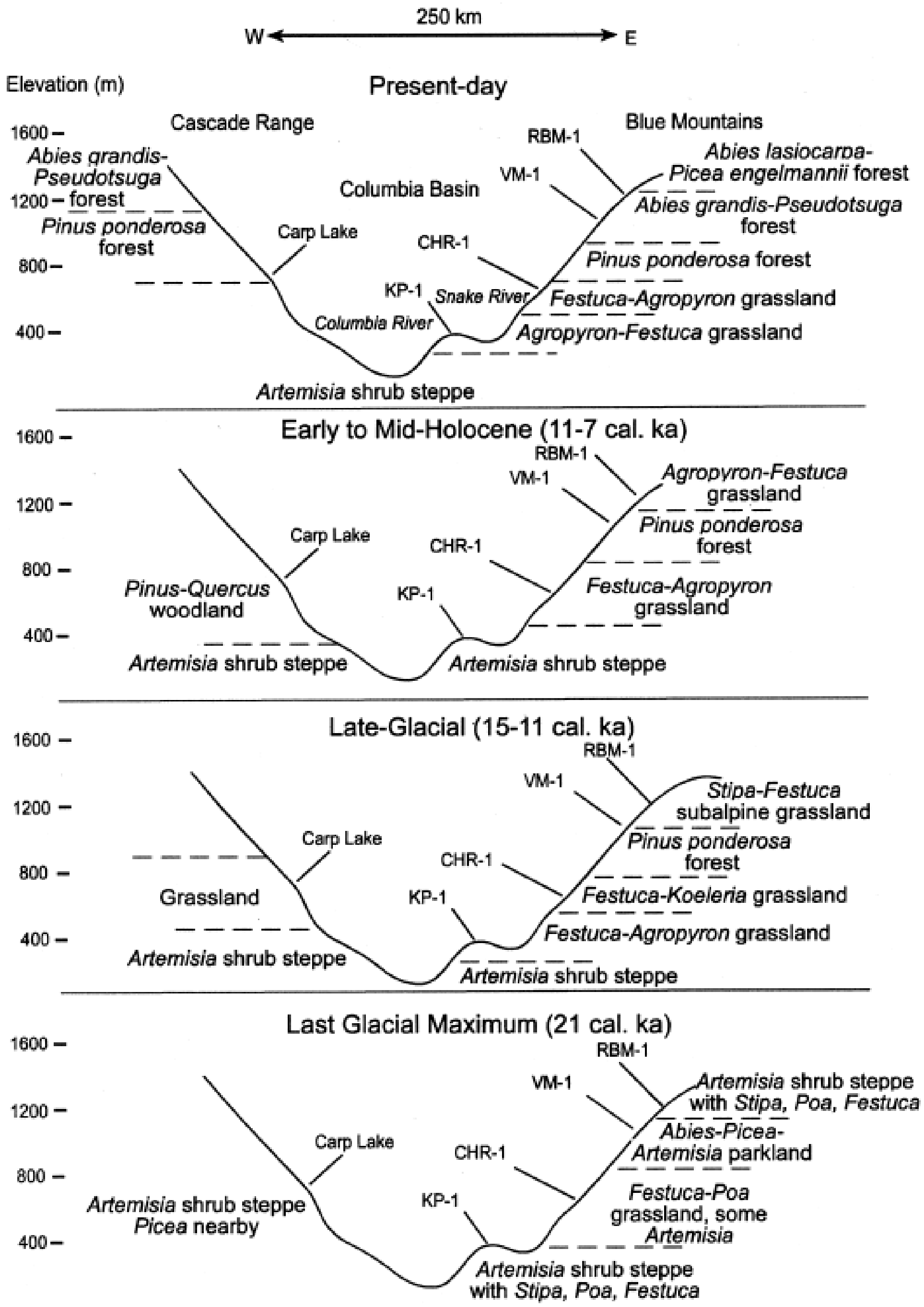

250 km
W
E
Elevation (m)
Present-day
1600
1200
800
400
Cascade Range
Blue Mountains
RBM-1
Abies grandis-
Pseudotsuga
forest
Abies lasiocarpa-
Picea engelmannii forest
VM-1
Columbia Basin
Abies grandis-Pseudotsuga
forest
Pinus ponderosa
forest
Carp Lake
CHR-1
Pinus ponderosa forest
KP-1
Snake River
Festuca-Agropyron grassland
Columbia River
Agropyron-Festuca grassland
Artemisia shrub steppe
Early to Mid-Holocene (11-7 cal. ka)
RBM-1
VM-1
Agropyron-Festuca
grassland
Pinus ponderosa
forest
Carp Lake
CHR-1
Festuca-Agropyron
grassland
Pinus-Quercus
woodland
KP-1
Artemisia shrub steppe
Artemisia shrub steppe
Late-Glacial (15-11 cal. ka)
RBM-1
VM-1
Stipa-Festuca
subalpine grassland
Pinus ponderosa
forest
Carp Lake
CHR-1
Grassland
KP-1
Festuca-Koeleria grassland
Festuca-Agropyron grassland
Artemisia shrub steppe
Artemisia shrub steppe
Last Glacial Maximum (21 cal. ka)
RBM-1
VM-1
Artemisia shrub steppe
with Stipa, Poa, Festuca
Abies-Picea-
Artemisia parkland
Carp Lake
CHR-1
Festuca-Poa
grassland, some
Artemisia
Artemisia shrub steppe
Picea nearby
KP-1
Artemisia shrub steppe
with Stipa, Poa, Festuca

Fig. 6.4

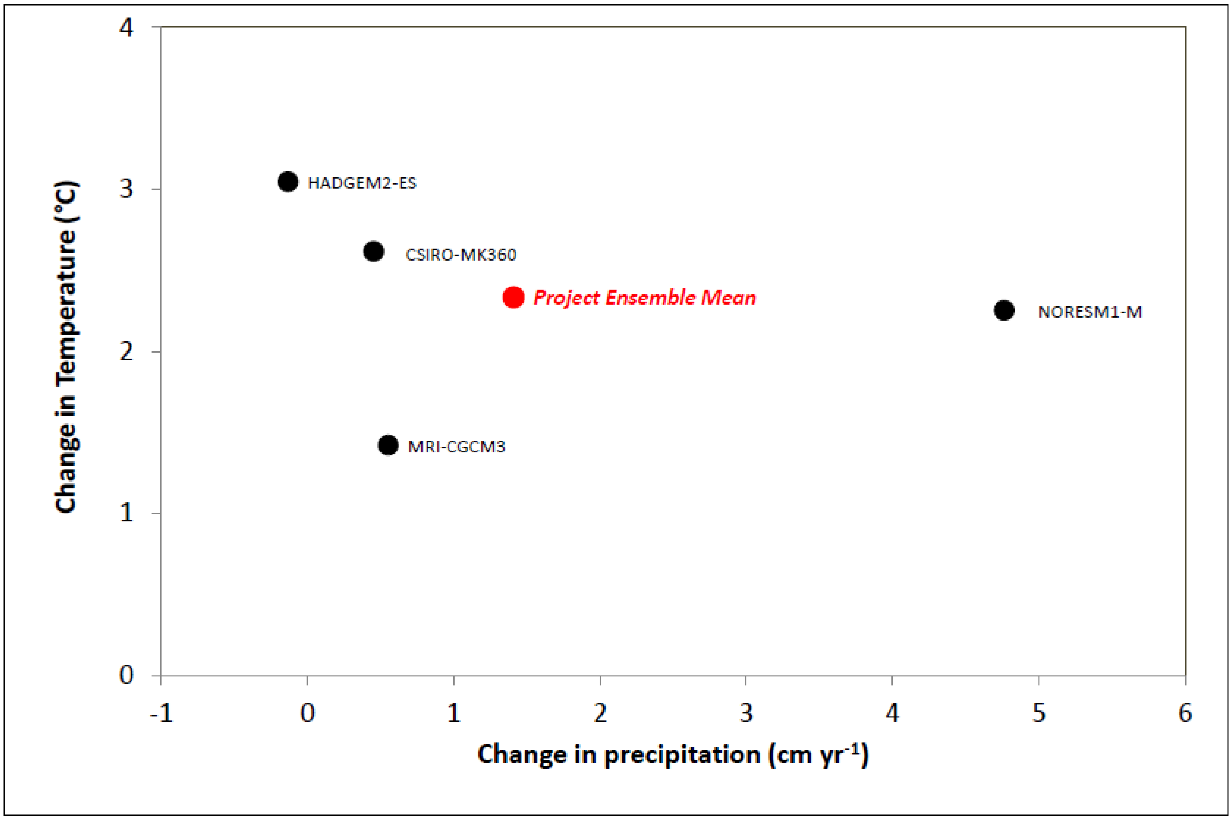

HADGEM2-ES
CSIRO-MK360
Project Ensemble Mean
NORESM1-M
MRI-CGCM3
Change in Temperature (°C)
Change in precipitation (cm yr-1)
0
1
2
3
4
-1
0
1
2
3
4
5
6

Fig. 6.5

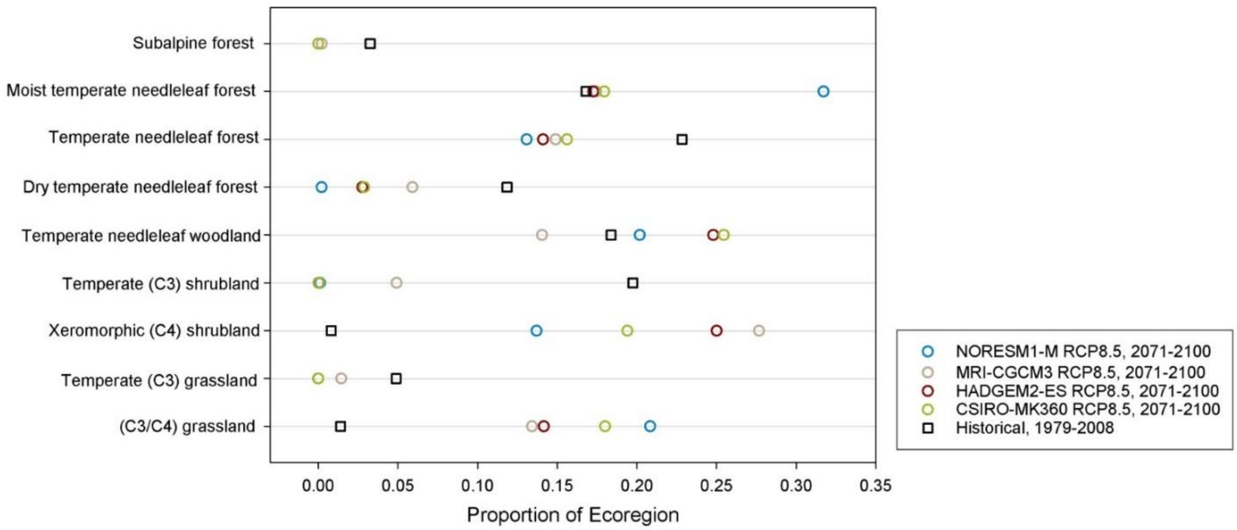
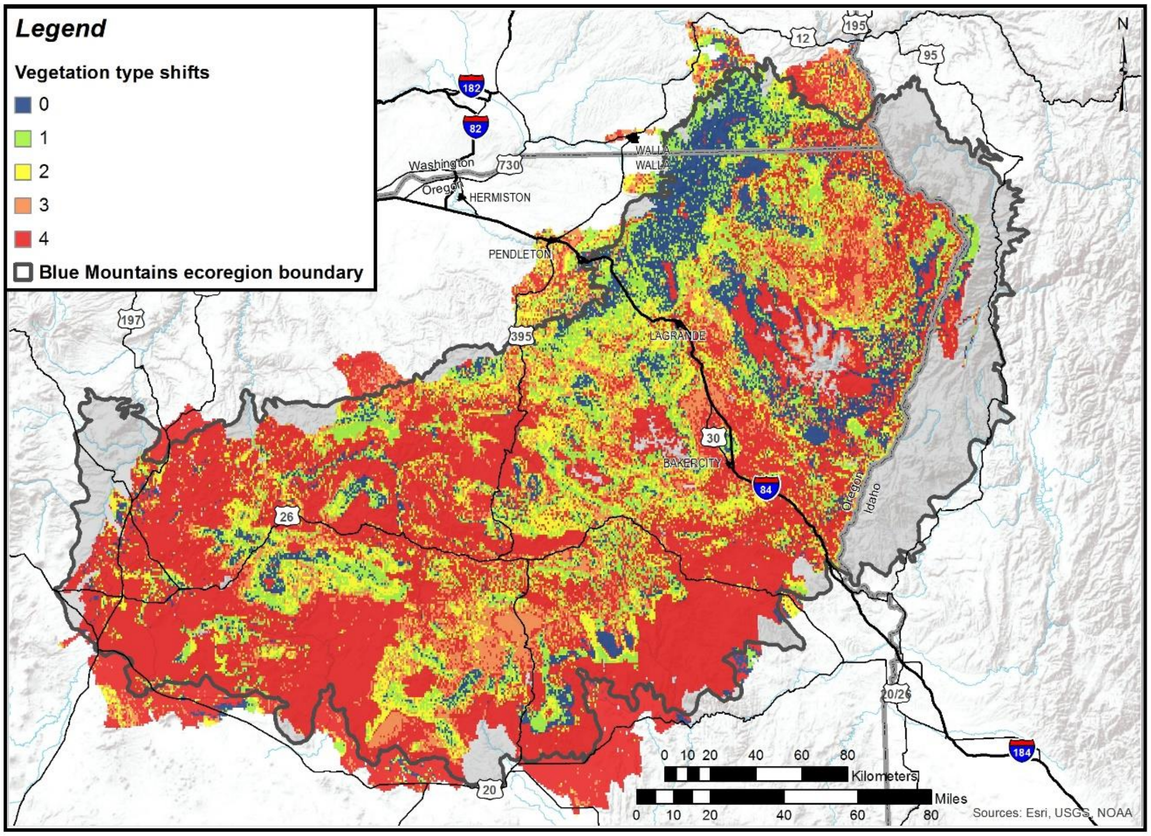

Subalpine forest
Moist temperate needleleaf forest
Temperate needleleaf forest
Dry temperate needleleaf forest
Temperate needleleaf woodland
Temperate (C3) shrubland
Xeromorphic (C4) shrubland
Temperate (C3) grassland
(C3/C4) grassland
0.00
0.05
0.10
0.15
0.20
0.25
0.30
0.35
Proportion of Ecoregion
NORESM1-M RCP8.5, 2071-2100
MRI-CGCM3 RCP8.5, 2071-2100
HADGEM2-ES RCP8.5, 2071-2100
CSIRO-MK360 RCP8.5, 2071-2100
Historical, 1979-2008

Fig. 6.6



Legend
Vegetation type shifts
0
1
2
3
4
Blue Mountains ecoregion boundary
Washington
Oregon
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Oregon
Idaho
0 10 20 40 60 80
Kilometers
Miles
0 10 20 40 60 80
Sources: Esri, USGS, NOAA

Fig. 6.7

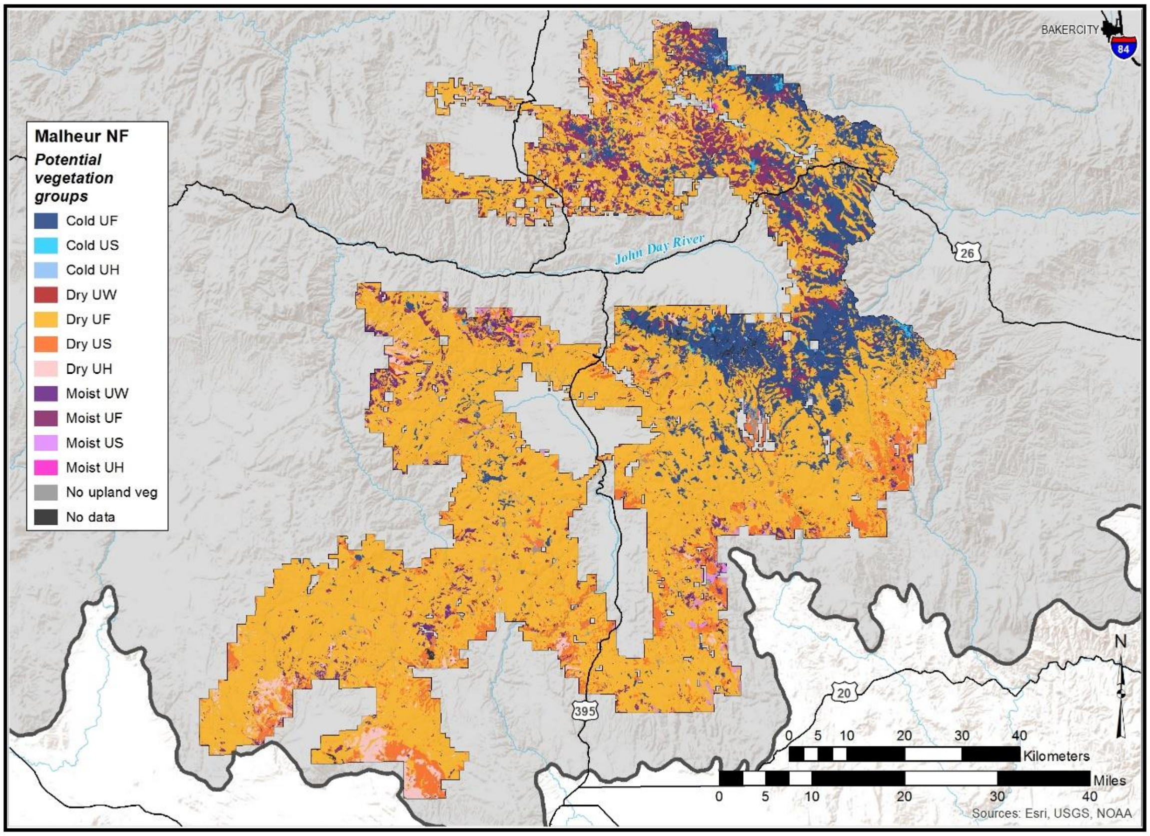

Malheur NF
Potential vegetation groups
Cold UF
Cold US
Cold UH
Dry UW
Dry UF
Dry US
Dry UH
Moist UW
Moist UF
Moist US
Moist UH
No upland veg
No data
John Day River
BAKERCITY
84
26
20
395
N
0 5 10 20 30 40
Kilometers
0 5 10 20 30 40
Miles
Sources: Esri, USGS, NOAA

Fig. 6.8

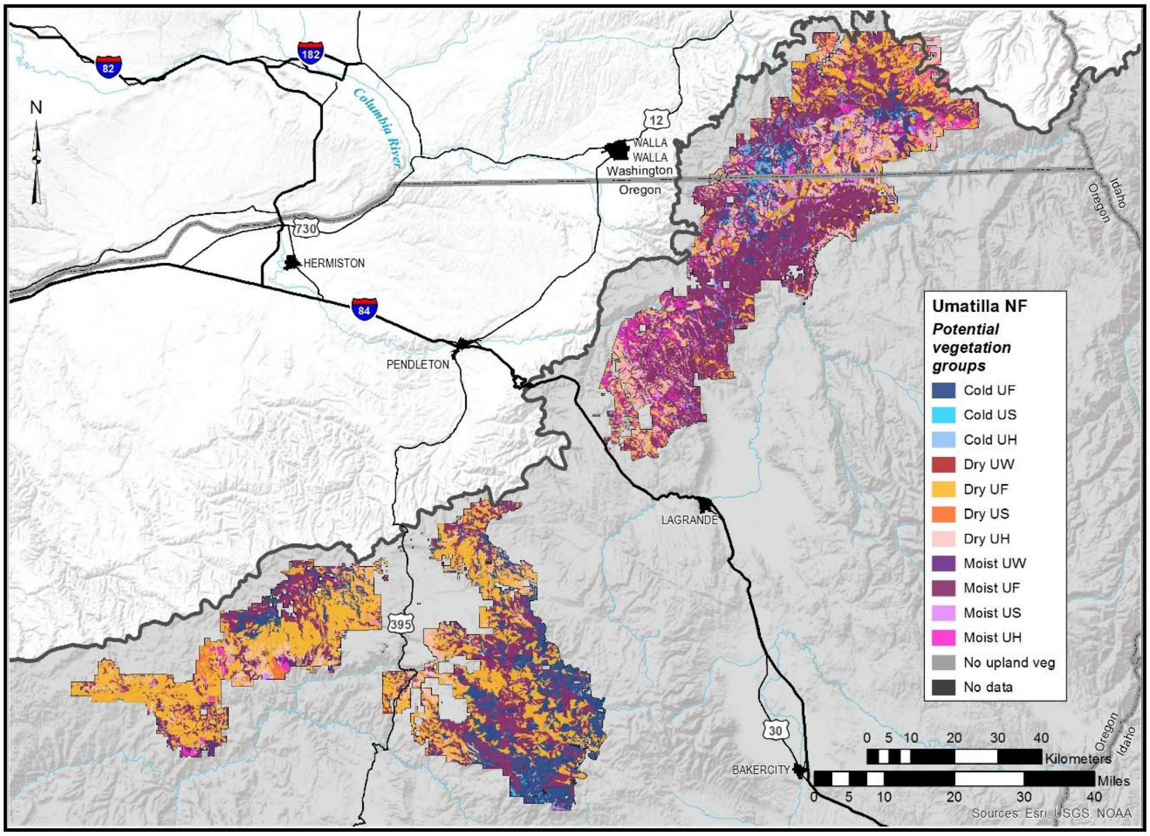
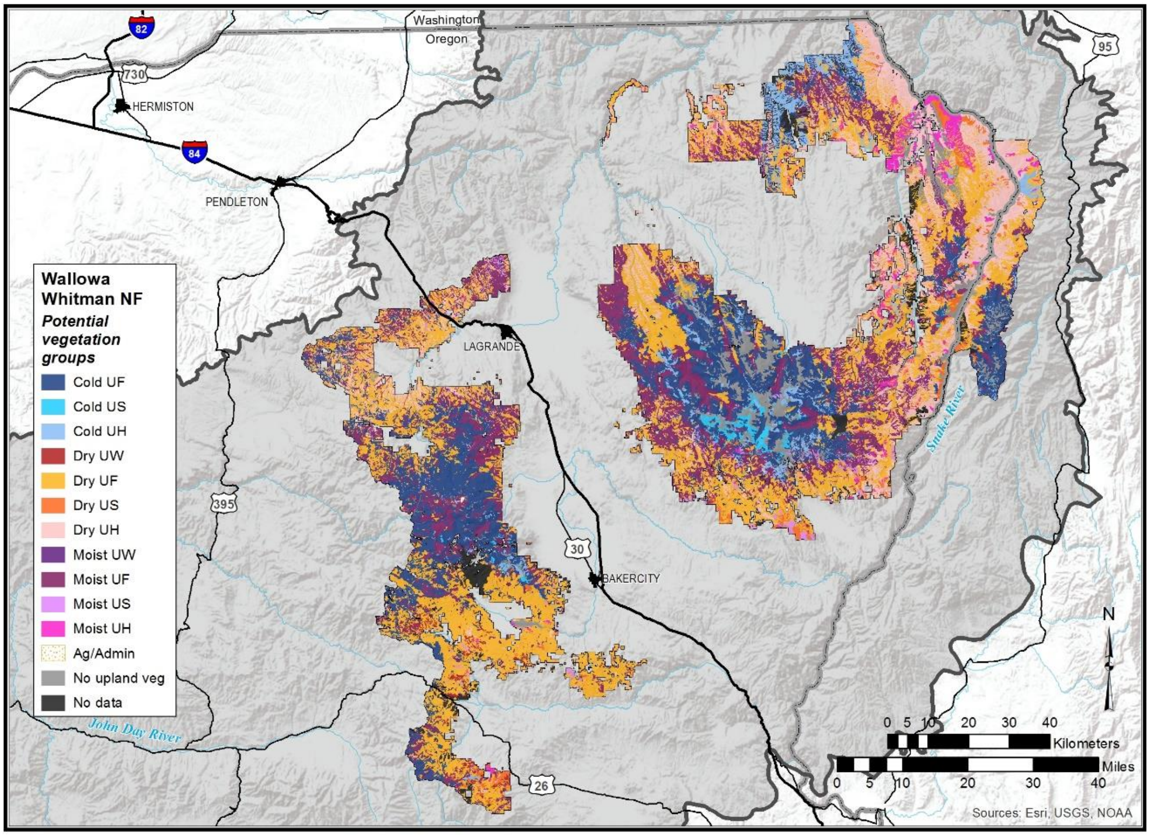

N
82
182
12
Columbia River
WALLA
WALLA
Washington
Oregon
Idaho
Oregon
730
HERMISTON
84
PENDLETON
LAGRANDE
395
30
BAKERCITY
Umatilla NF
Potential vegetation groups
Cold UF
Cold US
Cold UH
Dry UW
Dry UF
Dry US
Dry UH
Moist UW
Moist UF
Moist US
Moist UH
No upland veg
No data
0 5 10 20 30 40
Kilometers
0 5 10 20 30 40
Miles
Oregon
Idaho
Sources: Esri, USGS, NOAA

Fig. 6.9



Wallowa Whitman NF
Potential vegetation groups
Cold UF
Cold US
Cold UH
Dry UW
Dry UF
Dry US
Dry UH
Moist UW
Moist UF
Moist US
Moist UH
Ag/Admin
No upland veg
No data
Washington
Oregon
82
730
HERMISTON
84
PENDLETON
LAGRANDE
395
30
BAKERCITY
26
95
John Day River
Snake River
N
0 5 10 20 30 40
Kilometers
0 5 10 20 30 40
Miles
Sources: Esri, USGS, NOAA

Fig. 6.10

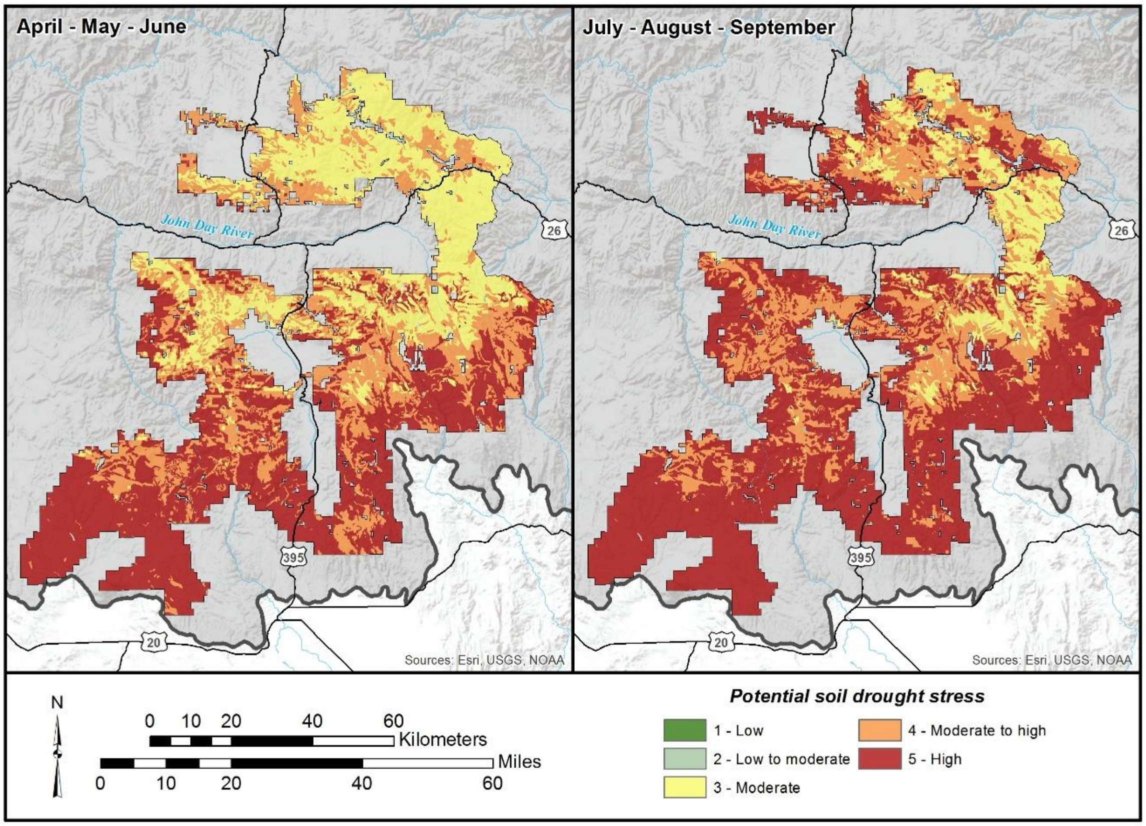
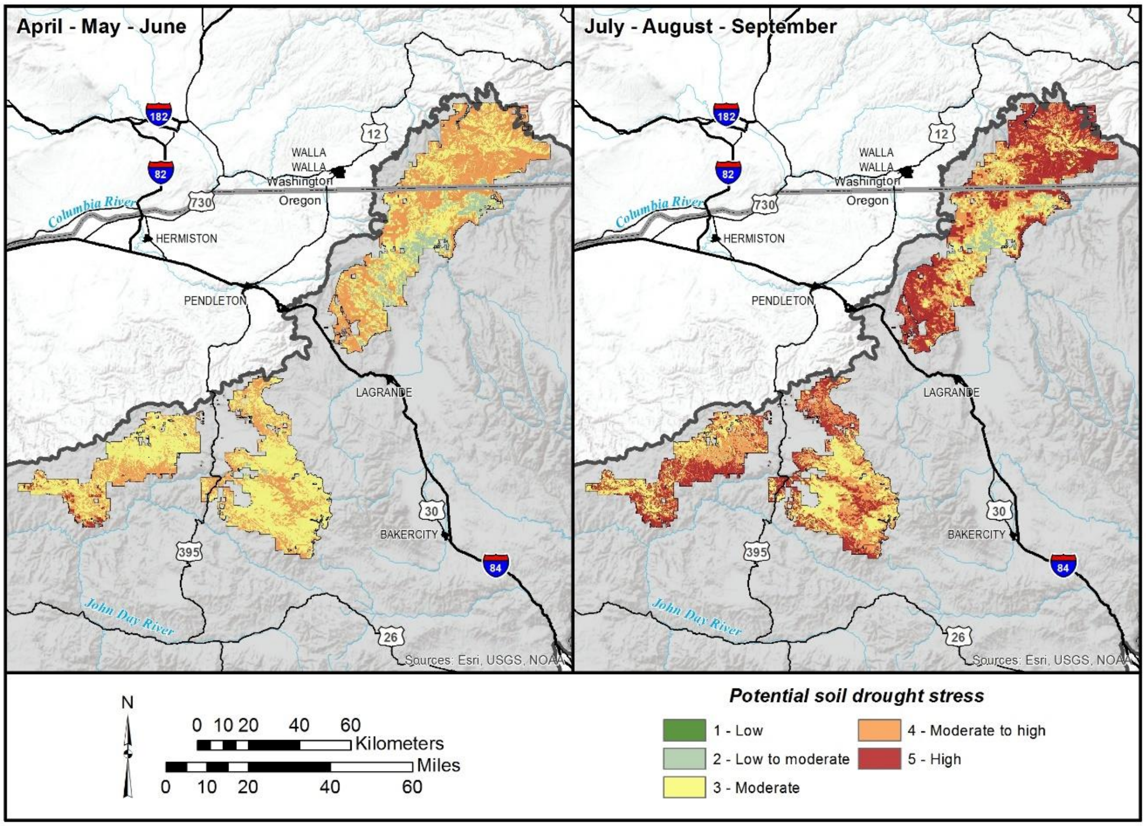

April - May - June
July - August - September
John Day River
26
395
20
Sources: Esri, USGS, NOAA
N
0 10 20 40 60 Kilometers
0 10 20 40 60 Miles
Potential soil drought stress
1 - Low
2 - Low to moderate
3 - Moderate
4 - Moderate to high
5 - High

Fig. 6.11



April - May - June
July - August - September
WALLA
WALLA
Washington
Oregon
Columbia River
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
John Day River
182
82
12
730
30
395
84
26
Sources: Esri, USGS, NOAA
N
0 10 20 40 60
Kilometers
Miles
0 10 20 40 60
Potential soil drought stress
1 - Low
2 - Low to moderate
3 - Moderate
4 - Moderate to high
5 - High

Fig. 6.12

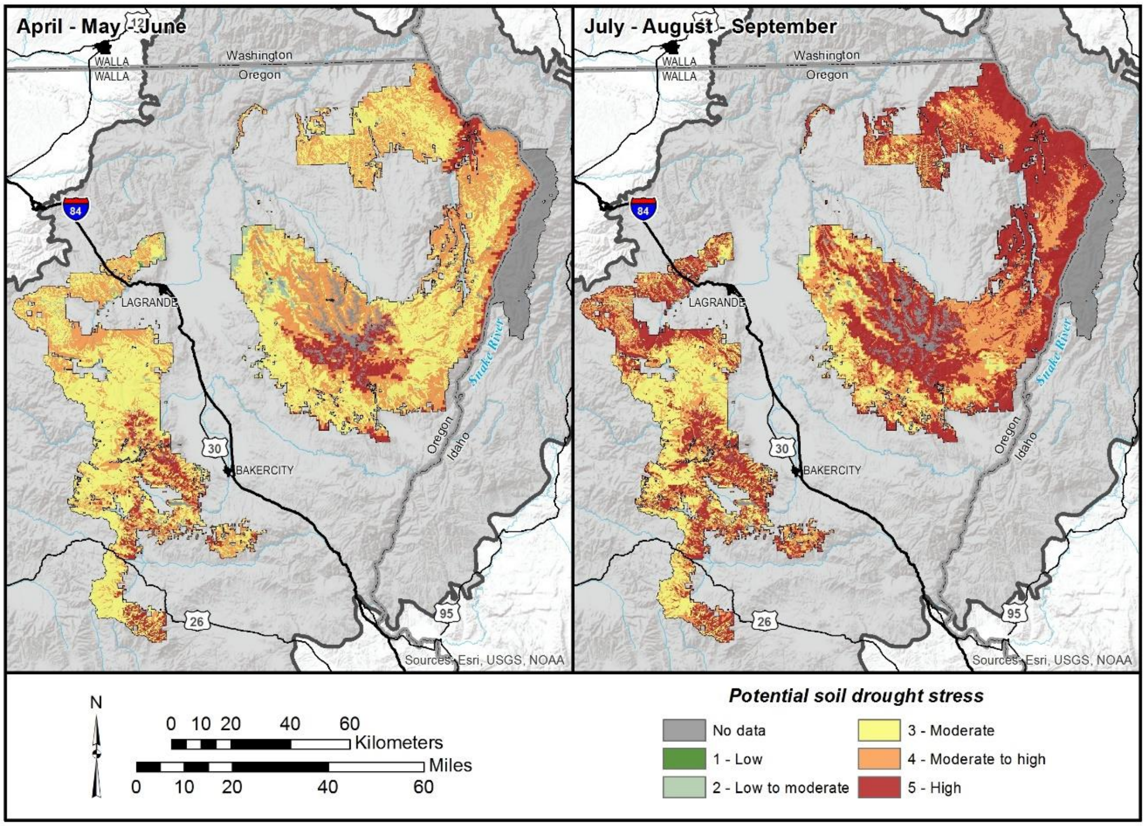

April - May - June
July - August - September
WALLA WALLA
Washington
Oregon
LAGRANDE
BAKERCITY
Snake River
Oregon
Idaho
84
30
26
95
Sources: Esri, USGS, NOAA
N
0 10 20 40 60
Kilometers
Miles
0 10 20 40 60
Potential soil drought stress
No data
1 - Low
2 - Low to moderate
3 - Moderate
4 - Moderate to high
5 - High

Fig. 6.13

Figure 6.14

Figure 6.15

Figure 6.16

Figure 6.17

Figure 6.18

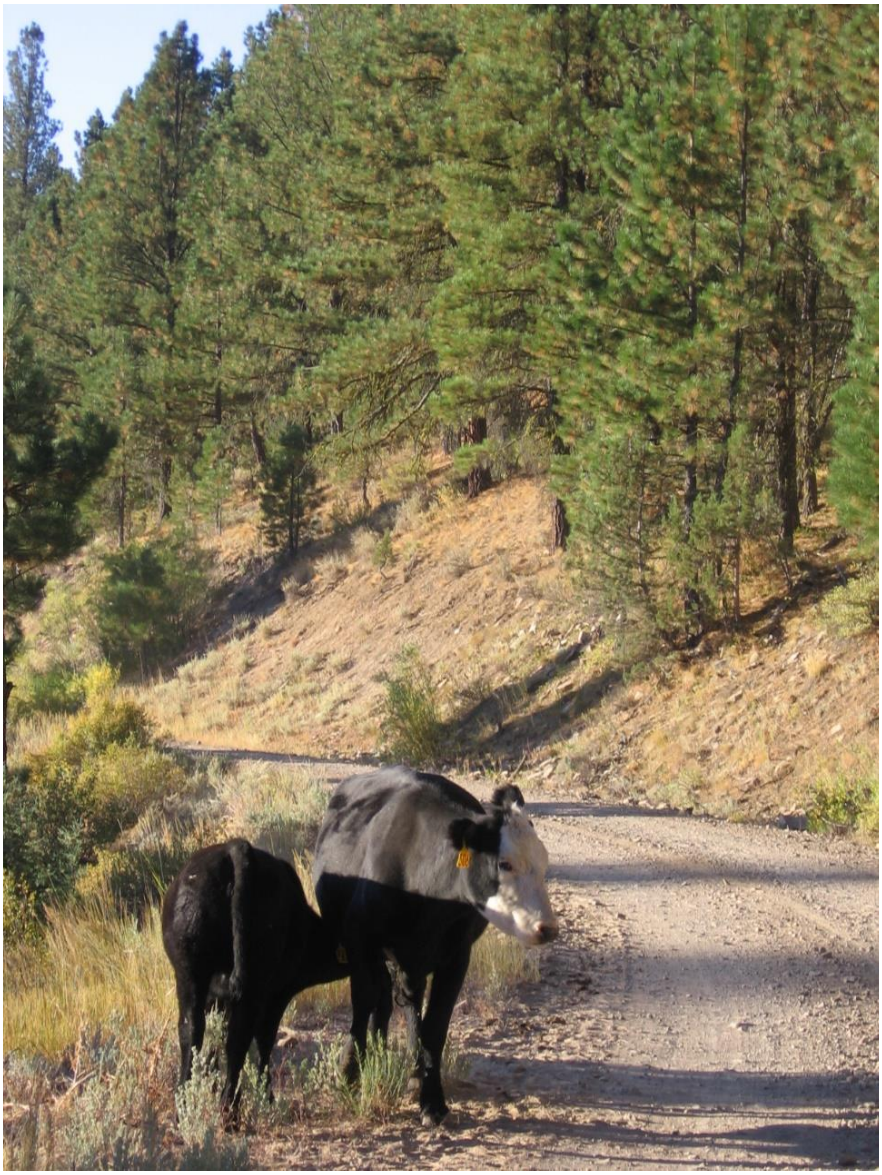


Figure 6.19

Figure 6.20

Figure 6.21

Figure 6.22

Fig. 7.1

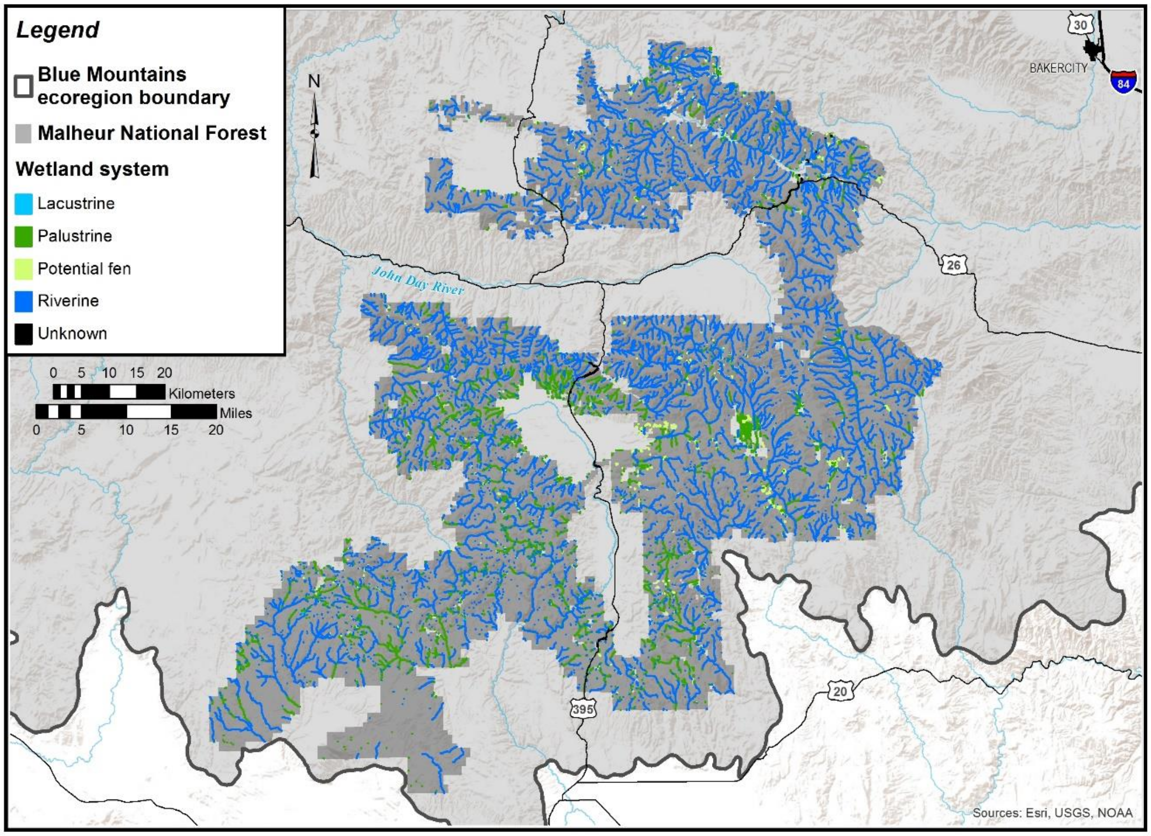

Legend
Blue Mountains ecoregion boundary
Malheur National Forest
Wetland system
Lacustrine
Palustrine
Potential fen
Riverine
Unknown
0 5 10 15 20
Kilometers
Miles
0 5 10 15 20
N
John Day River
BAKERCITY
30
84
26
20
395
Sources: Esri, USGS, NOAA

Fig. 7.2

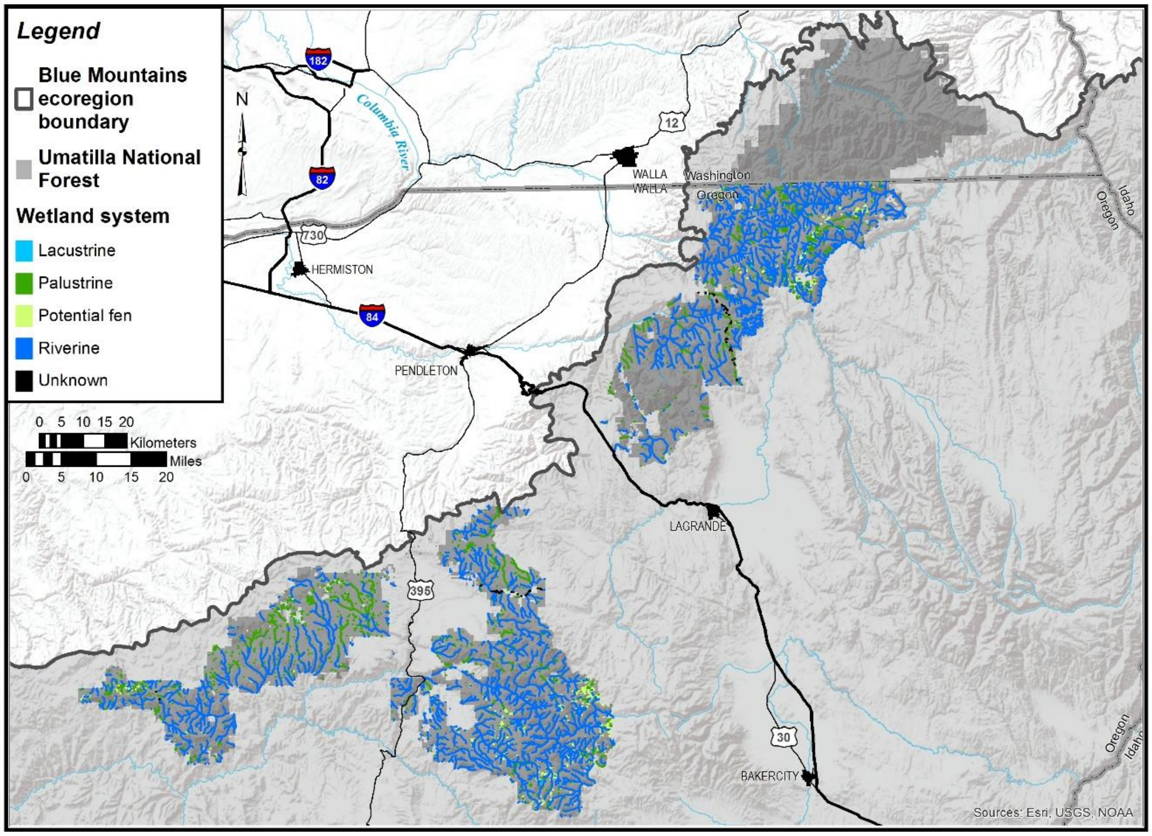

Legend
Blue Mountains ecoregion boundary
Umatilla National Forest
Wetland system
Lacustrine
Palustrine
Potential fen
Riverine
Unknown
0 5 10 15 20
Kilometers
Miles
0 5 10 15 20
N
182
82
12
730
84
395
30
Columbia River
WALLA WALLA
Washington
Oregon
Idaho
HERMISTON
PENDLETON
LAGRANDE
BAKERCITY
Sources: Esri, USGS, NOAA

Fig. 7.3

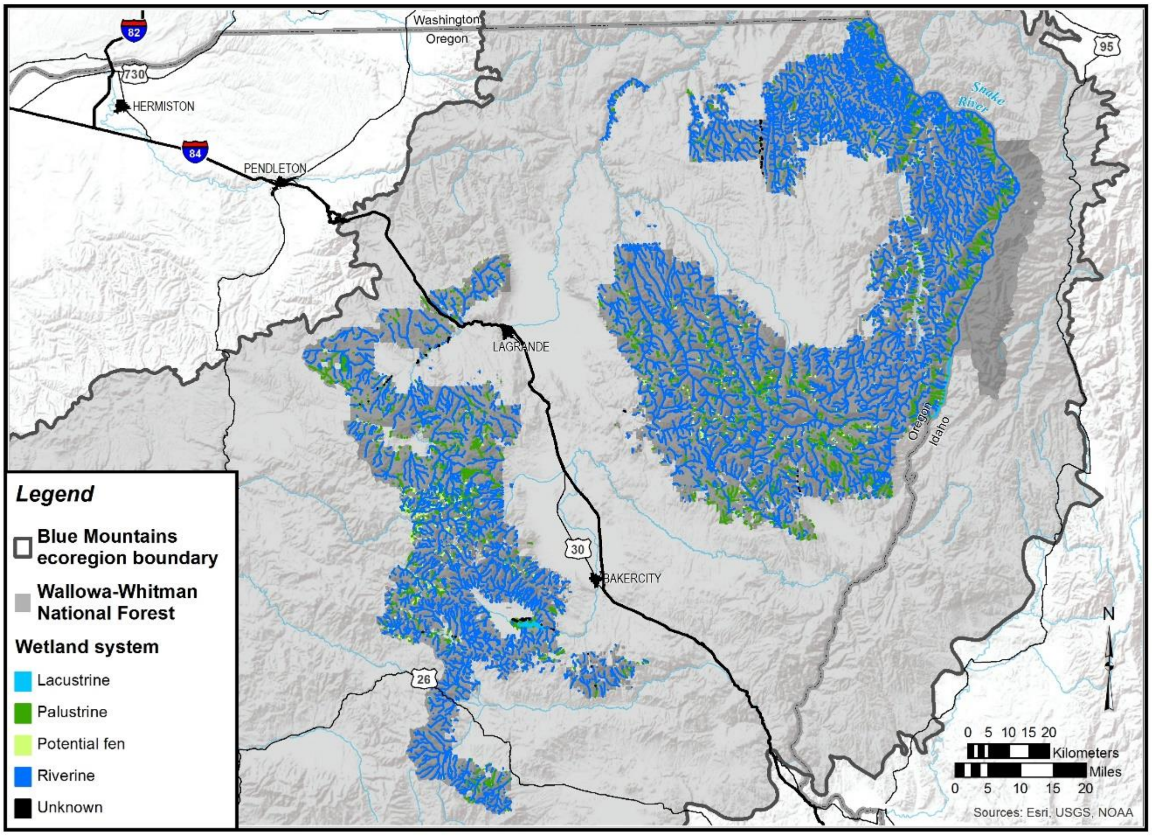
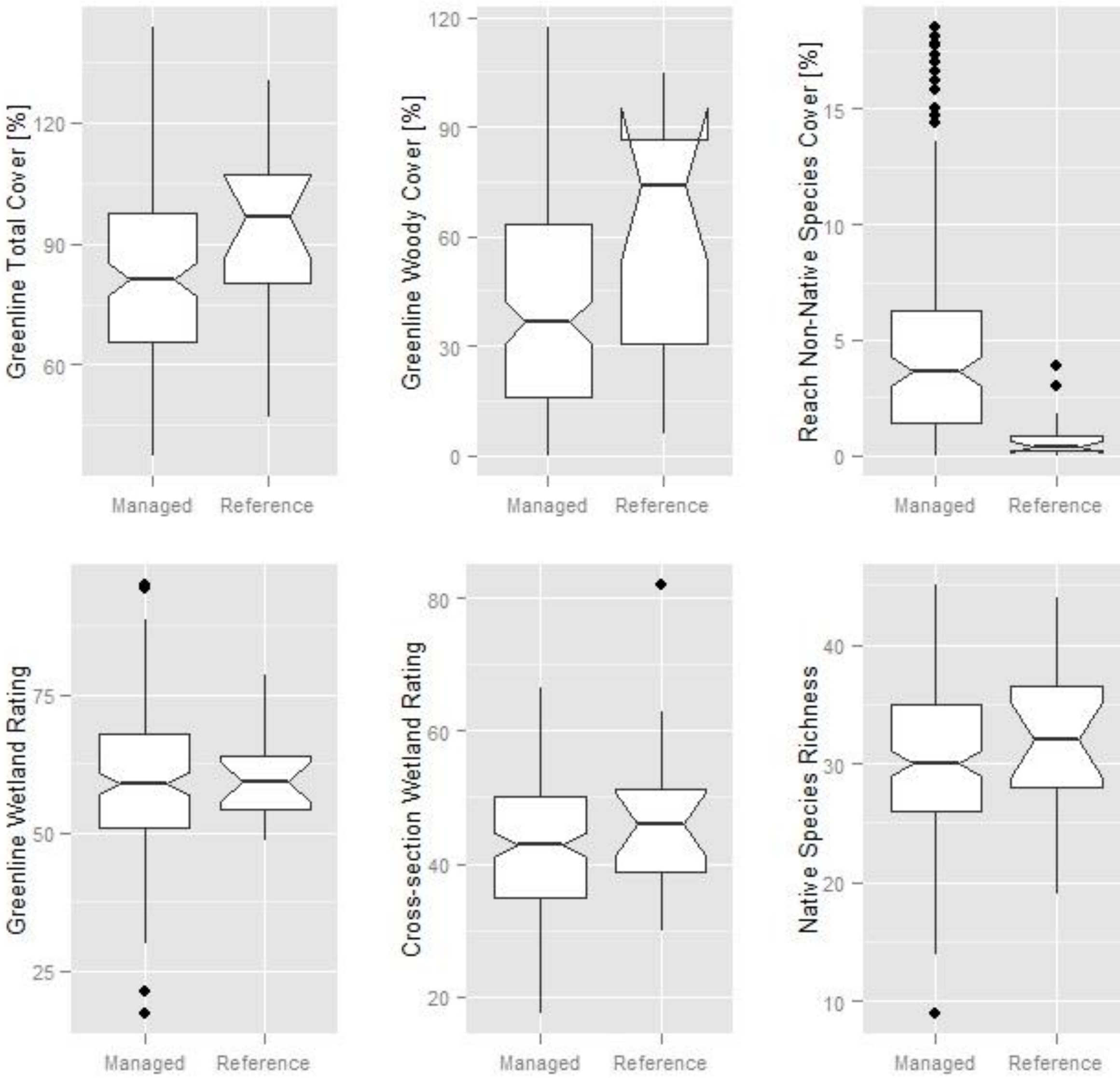

Washington
Oregon
82
730
HERMISTON
84
PENDLETON
LAGRANDE
30
BAKERCITY
26
95
Snake River
Oregon
Idaho
Legend
Blue Mountains ecoregion boundary
Wallowa-Whitman National Forest
Wetland system
Lacustrine
Palustrine
Potential fen
Riverine
Unknown
N
0 5 10 15 20
Kilometers
Miles
0 5 10 15 20
Sources: Esri, USGS, NOAA

Fig. 7.4



Greenline Total Cover [%]
Greenline Woody Cover [%]
Reach Non-Native Species Cover [%]
Greenline Wetland Rating
Cross-section Wetland Rating
Native Species Richness
Managed
Reference

Figure Captions

Figure 1.1 – Project area for the Blue Mountains Adaptation Partnership.

Figure 2.1 – Wilderness areas and Wild and Scenic Rivers in the project area for the Blue Mountains Adaptation Partnership.

Figure 3.1—Annual historical temperature for Oregon Climate Division 8. Data are from the National Oceanic and Atmospheric Administration National Centers for Environmental Information (http://www.ncdc.noaa.gov/cag/time-series/us).

Figure 3.2—Annual historical precipitation for Oregon Climate Division 8. Data are from the National Oceanic and Atmospheric Administration National Centers for Environmental Information (http://www.ncdc.noaa.gov/cag/time-series/us).

Figure 3.3—Observed (1950-2011) and simulated (1950-2100) Pacific Northwest regional mean temperature for selected global climate models under Representative Concentration Pathways (RCP) 4.5 and 8.5. The gray (historical), red (RCP 8.5), and blue (RCP 4.5) envelopes represent the range of model projections. Each individual narrow line represents a model projection, and bold lines are means of the model projections. Figure from Mote et al. (2013).

Figure 3.4—Projected change in snow water equivalent (SWE) (a) and mean snow residence time (b) for a 3 ^{o}C increase in temperature in the Blue Mountains. Point data are projections at snowpack telemetry (SNOTEL) stations from Luce et al. (2014). Snowpack sensitivity classes (the same in both figures) reflect the amount of shift in snowmelt timing seen in two contrasting historical years (Kramer and Snook, 2014; summarized in table 3.3).

Figure 3.5—(a) Percent change in the 1.5-year flood magnitude (approximately bankfull) between 2080 and the historical period (1970 to 1999) for the Blue Mountains region; brown lines are state boundaries. (b) Historical mid-winter flooding potential. (b) Projected mid-winter flooding potential for 2080. Flooding potential is shown as the number of days that winter flow is among the highest 5 percent for the year for streams in the Blue Mountains. Comparing the historical (1970-1999) to 2080s model runs shows that although much of the Blue Mountains has frequent midwinter flooding now, it is rare in some high-elevation areas. In the future, midwinter flooding is expected to be widespread. All projections are from the Variable Infiltration Capacity model, using data from Wenger et al. (2010).

Figure 3.6—Spatial distribution of July, August, and September streamflow sensitivities to a change in magnitude (mm mm-1) and timing (mm day-1) of recharge from snowmelt or rainfall in the Blue Mountains analysis area. See text for further explanation of streamflow sensitivity. From Safeeq et al. (2014).

Figure 3.7—Percent decreases in mean summer streamflow from historic time period (1970-1999) to 2080 for streams in the Blue Mountains region. Projections are from the Variable Infiltration Capacity hydrologic model using data from Wenger et al. (2010).

Figure 3.8—Comparison of percent decline in summer low flows for HUC-10 watersheds, calculated as percent of average daily flow predicted by the Variable Infiltration Capacity model (Wenger et al. 2010) and the exponential model (Safeeq et al. 2014) using historic (1915-2006) data and the A1B emission scenario for the 2040s. Low-flow calculations using the exponential model were calculated only when streamflow decline was ≥0.01 mm day^{-1}.

Figure 3.9—Recent historic (1970-1999) (a) and projected future (b) (2040, A1B Scenario) August mean temperatures for streams in the Blue Mountains region. Projections are from NorWeST Regional Stream Temperature Database and Model (http://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html).

Figure 4.1—Water right points of diversions (PODs) in the name of the Forest Service and in the name of others on and near the Wallowa Whitman and northern Umatilla National Forest. PODs in the name of the Forest Service represent a minor percentage of consumptive water rights. Consumptive water rights in the name of others are concentrated off-forest in the Umatilla, Walla Walla, Grande Ronde, and Wallowa valleys.

Figure 4.2—Historical snowpack sensitivity and different water uses in the Blue Mountains region. Snowpack sensitivity was classified as “no snow/ephemeral snow” if April 1 SWE was less than 3.8 cm during dry years (no snow) and greater than 3.8 cm during wet years (snow cover) in ≥80 percent of the subwatershed; “mixed snow sensitivity” if the timing of peak snowmelt in the warmest, driest years (e.g., 2003, El Niño year) occurred more than 30 days earlier than the coldest, wettest year (2011, La Niña year) in ≥50 percent of the subwatershed; and “persistent-least sensitive” if timing of peak snowmelt differed by less than 30 days between the warmest, driest years and the coldest, wettest years in ≥30 percent of the subwatershed. Locations in the mixed snow vulnerability category will likely see the greatest decrease in snowpack, but even the most persistent snowpacks (persistent least sensitive) will likely decline (see chapter 3). Areas of concern include municipal watersheds, locations with Forest Service drinking water systems, and national forest lands that are over-allocated downstream. Snowpack sensitivity information was adapted from Kramer and Snook (2014).

Figure 4.3—Projections of risk of summer water shortage associated with low streamflows in summer for 2080. Projections were calculated using flow data from the Variable Infiltration Capacity model, based on historical data for 1915-2006 and summer flow simulated for a global climate model ensemble for the A1B emission scenario (from Wenger et al. 2010). The Burnt, Powder, Upper Grande Ronde, Silver, Silvies, Upper John Day, Wallowa, and Willow Creek watersheds are at highest risk of summer water shortage.

Figure 4.4—Magnitude of flow alteration by dams and diversions in the Blue Mountains region. Resource specialists in each national forest rated relative flow alteration of subwatersheds as part of the watershed condition classification in the Watershed Condition Framework (Potyondy and Geier 2011). A relatively high percentage of subwatersheds within the Burnt, Powder, Upper Grande Ronde, and Wallowa subbasins were rated as having impaired function for this indicator of watershed condition.

Figure 4.5—Distribution of roads and trails within the three national forests in the Blue Mountains region. The national forests cover a contiguous area of over 2 million ha and contain 37,567 km of roads. The density of roads is higher at low elevations.

Figure 4.6—National forest roads located within 90 m of major rivers and streams. These roads, which are considered vulnerable to increased flooding comprise 3,300 km in the Blue Mountains (940 km in maintenance level (ML) 1 [basic custodial care; closed], 1,915 km in ML2 [high clearance cars and trucks], 377 km in ML3 [suitable for passenger cars], 21 km in ML4 [passenger cars; moderate comfort], and 98 km in ML5 [passenger cars; high comfort]). Note that not all vulnerable roads are represented; some roads also interrupt smaller intermittent streams and vice versa.

Figure 4.7—Projected percent change in bankfull flow in 2080 for roads within 90 m of a major river or stream (bankfull flow refers to the flow that just fills the channel to the top of its banks and at a point where the water begins to overflow onto a floodplain. Projections were calculated using flow data from the Variable Infiltration Capacity model, based on historical data for 1915-2006 and the Q1.5-bankfull or channel-forming flow simulated for a global climate model ensemble for the A1B emission scenario (from Wenger et al. 2010). Note that not all vulnerable roads are represented; some roads also interrupt smaller intermittent streams and vice versa.

Figure 4.8—Projected percent change in Q1.5-bankfull flow in 2080, with culvert barriers indicated based on the ratio of culvert width to bankfull width. Projections were calculated using flow data from the Variable Infiltration Capacity model, based on historical data for 1915-2006 and the Q1.5-bankfull, or channel-forming flow, simulated for a global climate model ensemble under the A1B emission scenario (from Wenger et al. 2010).

Figure 5.1—Analysis area and hydrologic subdomains for fisheries climate change assessment in the Blue Mountains.

Figure 5.2—Maps showing stream reaches with mean summer flows sufficient to support fish populations for the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory. Red stream reaches depict locations where summer flows are projected to drop below 0.034 m3s-1 and become intermittent fish habitat.

Figure 5.3—Maps showing the frequency of days when winter high flows are among the highest 5 percent of the year for the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory.

Figure 5.4—Maps showing dates of the center of annual flow mass (50 percent of annual flows) for the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory. Cool colors indicate streams with snowmelt-dominated hydrographs, and warm colors indicate rainfall-dominated hydrographs.

Figure 5.5—Summer stream temperature map for the 1980s (panel a) and 2080s (panel b) based on NorWeST scenarios and the A1B emissions trajectory.

Figure 5.6—Distribution of thermally suitable habitat for spring Chinook salmon downstream of Hells Canyon during the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory.

Figure 5.7—Distribution of thermally suitable habitat for bull trout during the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory.

Figure 5.8—Distribution of thermally suitable habitat for steelhead and redband trout downstream of Hells Canyon in the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory.

Figure 5.9—Distribution of thermally suitable habitat for redband trout upstream of Hells Canyon and in the Oregon Closed Basins in the 1980s (panel a) and 2080s (panel b) based on the A1B emissions trajectory.

Figure 6.1—Projected increase in area burned by wildfire as associated with a mean annual temperature increase of 1 °C, shown as the percentage change relative to the median annual area burned during 1950-2003 (Littell [n.d.] cited in Ojima et al. [2014]). Results are aggregated to ecoprovinces of the western United States based on Bailey (1995).

Figure 6.2—Results from a neighborhood analysis for concentrated exposure across Oregon and Washington to four stressors and their combinations.

Figure 6.3—Inferred vegetation in the Columbia Basin during the last 21,000 years based on the pollen record at Carp Lake and phytoliths in four loess sections in the Columbia Basin. Figure from Blinnikov et al. 2002.

Figure 6.4—Projected annual change in precipitation and temperature compared to average annual historical Pacific Northwest climate for the global climate model scenarios used as input to the MC2 model. All global climate models were run under Representative Concentration

Pathway (RCP) 8.5, and the ensemble of all global climate models runs under RCP 8.5 is shown for comparison.

Figure 6.5—Projected changes in potential vegetation functional types for the end of the 21st century. Data are based on output from the MC2 model and four future climate scenarios (see inset box). Historical data were generated from simulated MC2 data and may be different than other potential vegetation maps.

Figure 6.6—Number of projections resulting in vegetation type shift by the MC2 model from the historical period (1979-2008) to the late 21st century (2071-2100). The modeled ecoregion boundary is slightly different than the U.S. EPA Level II Ecoregion (shown by the heavy black line).

Figure 6.7—Potential vegetation groups for the Malheur National Forest. Forest boundaries are proclaimed and PVG data from small inholdings were removed.

Figure 6.8—Potential vegetation groups (PVGs) for the Umatilla National Forest. Forest boundaries are proclaimed and PVG data from small inholdings were removed.

Figure 6.9—Potential vegetation groups for the Wallowa-Whitman National Forest. Forest boundaries are proclaimed and PVG data from small inholdings were removed.

Figure 6.10—Potential soil drought stress in the spring and summer for the Malheur National Forest.

Figure 6.11—Potential soil drought stress in the spring and summer for the Umatilla National Forest.

Figure 6.12—Potential soil drought stress in the spring and summer for the Wallowa-Whitman National Forest.

Figure 6.13—Windthrow disturbance in a cold upland forest.

Figure 6.14—An example of a Cold Upland Shrubland community from the south slope of the Greenhorn Mountains.

Figure 6.15—Examples of rare and locally endemic species found in alpine and subalpine environments. On the left is Potentilla sp. from serpentine substrates in the Greenhorn Mountains. On the right is Greenman's biscuitroot from Mount Howard, Wallowa County, Oregon.

Figure 6.16—Moist Upland Forest in the Blue Mountains.

Figure 6.17—View of Dry Upland Forest from Lone Rock, Malheur National Forest.

Figure 6.18—Domestric livestock grazing is common in the Blue Mountains, particularly in dry upland ponderosa pine forests (Emigrant Creek Ranger District, Malheur National Forest).

Figure 6.19—Pine white defoliation is extentsive in a ponderosa pine stand in the southern Blue Mountains (Malheur National Forest, autumn 2011).

Figure 6.20—A stand with a mixture of western juniper and ponderosa pine.

Figure 6.21—An example of a Dry Upland Herbland dominated by Idaho fescue, Umatilla National Forest. This site is considered to be in good condition. (Photo by Mark Darrach)

Figure 6.22—Cheatgrass invasion in a ponderosa pine stand after a prescribed fire that caused substantial overstory mortality (Malheur National Forest). (Photo by Becky Kerns)

Figure 7.1—Wetlands in Malheur National Forest. Source: Oregon Wetlands Geodatabase.

Figure 7.2—Wetlands in the Oregon portion of Umatilla National Forest. Source: Oregon Wetlands Geodatabase.

Figure 7.3—Wetlands in Wallowa-Whitman National Forest. Source: Oregon Wetlands Geodatabase.

Fig. 7.4—Box-and-whisker plots for selected vegetation indicators of managed sites (n=164) and reference sites (n=18) in the Blue Mountains (2007-2011 monitoring data). Greenline (densely-vegetated streamside zone) total cover and woody cover are calculated as the sum of covers above and below 1 meter for a maximum value of 200 percent. Non-native species cover is the combined cover of non-native species on the greenline and cross-section for the sampled reach. Wetland ratings are derived from Coles-Richie et al. (2007). Native species richness is the number of native species inventoried in each site. See sampling protocols for vegetation parameters for further explanation of variables (Archer et al. 2012b). Horizontal lines in boxes indicate medians, boxes indicate inter-quartile ranges, whiskers indicate ranges, and dots indicate outliers.