A Climate Change Vulnerability Assessment for Resources of the Nez Perce-Clearwater National Forest

A report to the Nez Perce-Clearwater National Forest
and U.S. Forest Service Northern Region

EcoAdapt

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Executive Summary

This vulnerability assessment is an initial science-based effort to identify how and why important resources (ecosystems and species) across the Nez Perce-Clearwater region are likely to be affected by future climate conditions. In this context, vulnerability is a function of the sensitivity of the resource to climate change, its anticipated exposure to those changes, and its capacity to adapt to changes. Specifically, sensitivity is defined as a measure of whether and how a resource is likely to be affected by a given change in climate, or factors driven by climate; exposure is defined as the degree of change in climate or climate-driven factors a resource is likely to experience; and adaptive capacity is defined as the ability of a resource to accommodate or cope with climate change impacts with minimal disruption (Glick et al. 2011). The overarching goal of this assessment is to help resource managers plan their management of these ecosystems and species in light of a changing climate. Specifically, this information can facilitate priority setting for management action and responses, helping to sustain optimal conditions for and productivity of resources. Twenty-eight resources including eight ecosystems and twenty species were identified as important by the Nez Perce-Clearwater National Forest as part of their forest plan revision process and are considered in this assessment. This assessment centers on the Nez Perce-Clearwater region of Idaho (Figure 1).

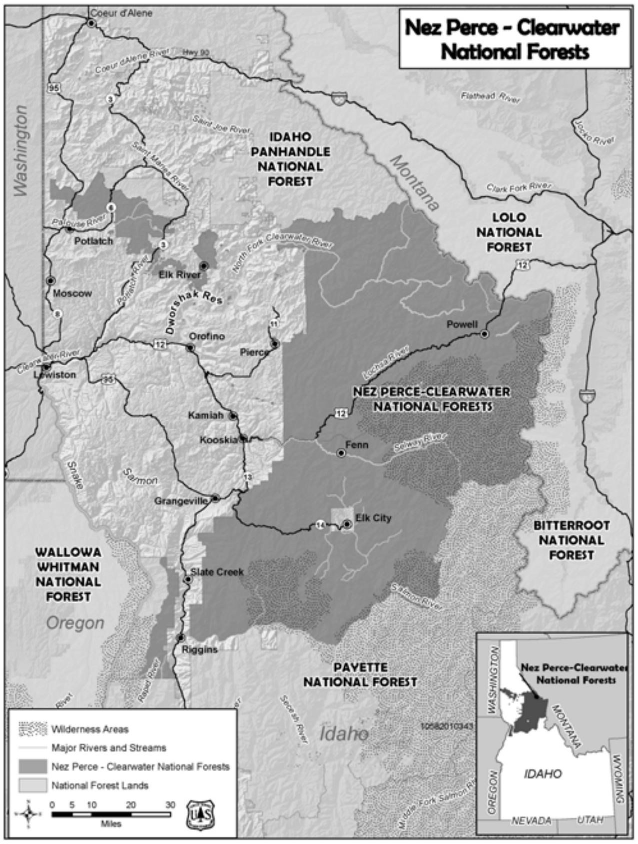

Figure 1. Nez Perce-Clearwater National Forests.

General Overview

Climate change vulnerability of twenty-eight resources was assessed by considering exposure to climate change, sensitivity to climate and non-climate stressors, and adaptive capacity. The twenty-eight resources were identified by the Nez Perce-Clearwater National Forest, and particular attention was given to those resources of management, cultural, or social concern. Downscaled climate exposure information for the Nez Perce-Clearwater (NPCW) region was provided by Sound Science[1] in the form of maps, graphs, and a synthesis section summarizing regional past and projected climate trends and change data, and included information on temperature, precipitation, and hydrology (e.g., low flows, high flows, timing of flows, soil moisture). Section 3 of this report includes the downscaled climate information provided by Sound Science. Additional exposure information for resources was provided by the Template for Assessing Climate Change Impacts and Management Options (TACCIMO)[2].

A vulnerability assessment workshop was convened to evaluate the vulnerability of each resource and included participants from the NPCW National Forest and U.S. Forest Service (USFS) Northern Region. Sensitivity and adaptive capacity were assessed on a 1-7 scale (1 = very low, 7 = very high), while exposure to climate and climate-driven changes was ranked in order of importance; both exercises were based on participant expertise. Each ranking also included a confidence evaluation.

The Report Section-by-Section

Section 1 provides a brief introduction to the project and how the information from vulnerability assessments can be used. Section 2 provides a general summary of past and projected climate trends for the Nez Perce-Clearwater region. Section 3 presents more specific, downscaled climate information – observed and projected – for the NPCW region and discusses trends in temperature, precipitation, and hydrology. Section 4 summarizes uncertainty in terms of climate change projections. Section 5 describes in greater detail the methods used to select resources, as well as the development of the vulnerability assessment model and its application. Section 6 explores the results of the vulnerability assessments for the final suite of resources. Section 7 provides brief concluding remarks and recommendations.

Vulnerability Assessment Summary: Ecosystems and Species

The vulnerabilities for eight ecosystems (aquatic, coastal disjunct, dry forest, grassland, mixed mesic, riparian, subalpine, and wetlands/moist meadows/groundwater-dependent ecosystems (GDEs)) are summarized in Figure 2. This figure is arranged such that ecosystems listed in the upper left region were judged to have less relative vulnerability than those listed in the lower right region. Relative vulnerability does not include an evaluation of future climate exposure, as exposure for each ecosystem was not scored as part of this assessment. Ecosystems assessed as having high sensitivity included dry forest, riparian, and subalpine. Most ecosystems were assessed as having moderate to moderate-high adaptive capacity, with the exception being the dry forest ecosystem (low-moderate adaptive capacity).

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[1] http://www.sound-science.org/

[2] http://www.taccimo.sgcp.ncsu.edu/

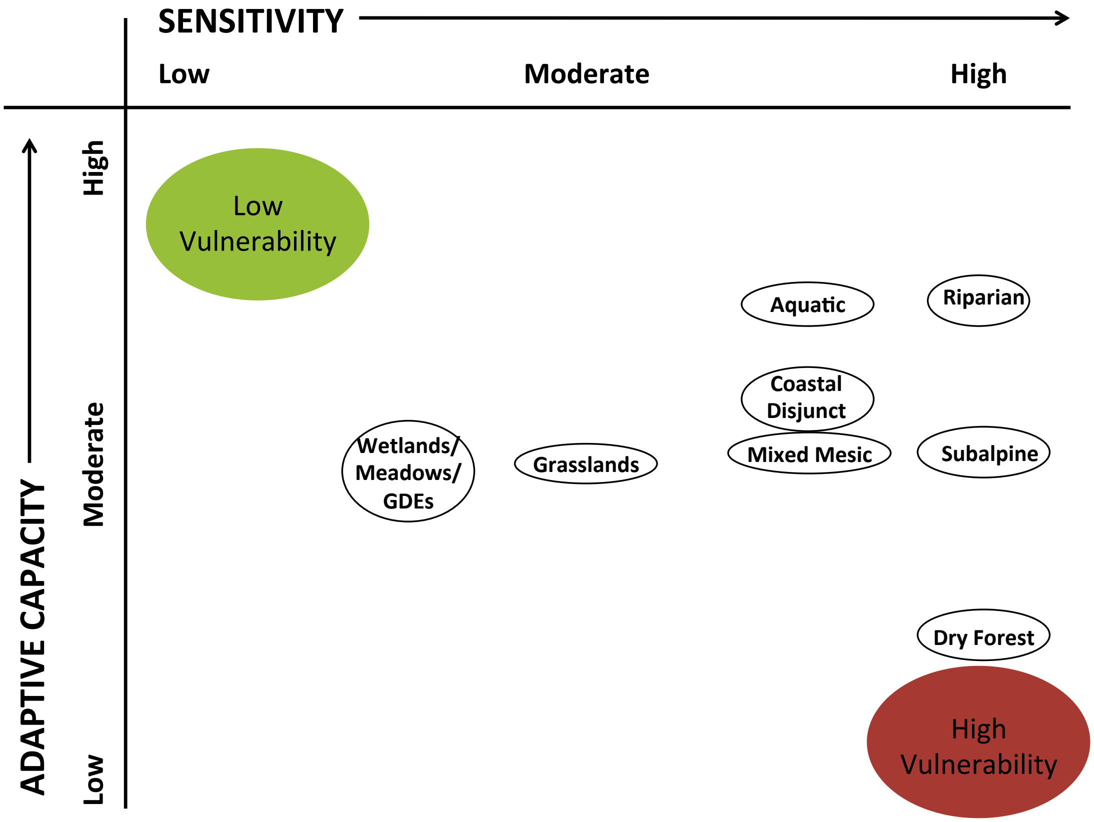

Figure 2. Relative vulnerabilities of eight Nez Perce-Clearwater ecosystems based on the climate change sensitivity and adaptive capacity assessment. Relative vulnerability, which does not include a measure of future climate exposure, increases with increasing sensitivity and decreasing adaptive capacity. Ecosystems listed in the upper left region were judged less vulnerable than those listed in the lower right region. Overall confidence for ecosystem sensitivities and adaptive capacities ranged from moderate to high.

The subalpine, riparian, and dry forest ecosystems were judged to be more sensitive than the other systems. Subalpine and dry forest ecosystems have a combination of high sensitivity to climate and climate-driven changes (temperature, reduced soil moisture and drought), disturbance regimes (insects and disease outbreaks, wildfire), and non-climate stressors (fire suppression). Riparian ecosystems exhibit high sensitivity to climate and climate-driven changes that affect hydrologic regimes and water availability including decreased snowpack, earlier snowmelt and runoff timing, and shifts from snow- to rain-dominant watersheds. Non-climate stressors, particularly transportation corridors (e.g., road networks, construction or maintenance of roads), grazing, invasive species, and dams and water diversions further increase the sensitivity of riparian ecosystems. Conversely, wetlands, moist meadows, and groundwater-dependent ecosystems were assessed with the lowest sensitivity (low-moderate) due to moderate sensitivity to climate-driven changes (e.g., decreased snowpack and earlier snowmelt, reduced soil moisture) and low sensitivity and current exposure to non-climate stressors (e.g., invasive species, grazing) in the NPCW region.

While most ecosystems were judged to have moderate to moderate-high adaptive capacity, dry forest ecosystems were judged to have low-moderate adaptive capacity. This is due to historic dry forest structure and composition occurring only in isolated patches, with the majority of the system considered to be quite degraded. More in-depth explorations of ecosystem vulnerabilities are presented in Table 1 below and in Section 6.

Figure 3 summarizes the vulnerability of nine species or species assemblages considered including aquatic species[3], Canada lynx/wolverine (evaluated together), Coeur D'Alene and Idaho giant salamanders, dry forest birds[4], fisher, mountain goat, red alder, Spalding's catchfly, and whitebark pine. This figure is arranged similarly to Figure 2 in that species listed in the upper left region were judged to be less vulnerable than species listed in the lower right region. Most species and species assemblages were ranked between moderate to high sensitivity, with red alder as the exception (low-moderate sensitivity). Similarly, the majority of species and species assemblages were also judged to have moderate to moderate-high adaptive capacity. The exceptions included Coeur D'Alene and Idaho giant salamanders, Canada lynx and wolverine, and Spalding's catchfly (assessed as having low, low-moderate, or low adaptive capacity, respectively).

Canada lynx and wolverine, Coeur D'Alene and Idaho giant salamanders, dry forest bird species, and whitebark pine were judged to be more sensitive than the other species. Canada lynx and wolverine are sensitive to several climate and climate-driven changes including increased temperatures (wolverine), decreased snowpack (wolverine, lynx), and altered wildfire regimes (lynx), and both species demonstrate high sensitivity to land use changes that result in loss of core habitat and connectivity. Coeur D'Alene and Idaho giant salamanders exhibit high sensitivity to climate-driven changes such as reduced soil moisture, drought, or increased rain-on-snow-events that can affect salamander habitat availability and quality, and reduce foraging and breeding opportunities. Dry forest bird species are moderately sensitive to climate and climate-driven changes, but exhibit high sensitivity to non-climate stressors such as fire suppression practices and timber harvest that have altered or degraded dry forest structure that birds use for breeding and foraging habitat. Whitebark pine exhibits moderate-high sensitivity to climate and climate-driven changes, including warming temperatures and reduced soil moisture, however, insect and disease outbreaks (i.e., mountain pine beetle, white pine blister rust) have led to significant ongoing mortality of whitebark pine. Conversely, red alder was judged to have the lowest sensitivity of evaluated species, as red alder is a pioneer species that responds positively to disturbance and may benefit from future climate changes.

A number of species were assessed as having moderate-high adaptive capacity including aquatic species, dry forest bird species, fisher, red alder, and whitebark pine. Conversely, Coeur D'Alene and Idaho giant salamanders and Spalding's catchfly were ranked as having the lowest adaptive capacity. Salamanders were judged to have low adaptive capacity due to their small, isolated populations, low ability to disperse, and limited genetic exchange. Similarly, Spalding's

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[3] Aquatic species species considered as part of this assessment included: Fall and Spring Chinook salmon, steelhead, cutthroat trout, westslope cutthroat trout, bull trout, and interior redband trout

[4] Dry forest bird species considered as part of this assessment included: flammulated owl, Lewis's woodpecker, pygmy nuthatch, and white-headed woodpecker.

catchfly adaptive capacity was considered low-moderate due to a small, fragmented population size and low genetic diversity. More in-depth explorations of species' vulnerabilities are presented in Table 2 below and in Section 6.

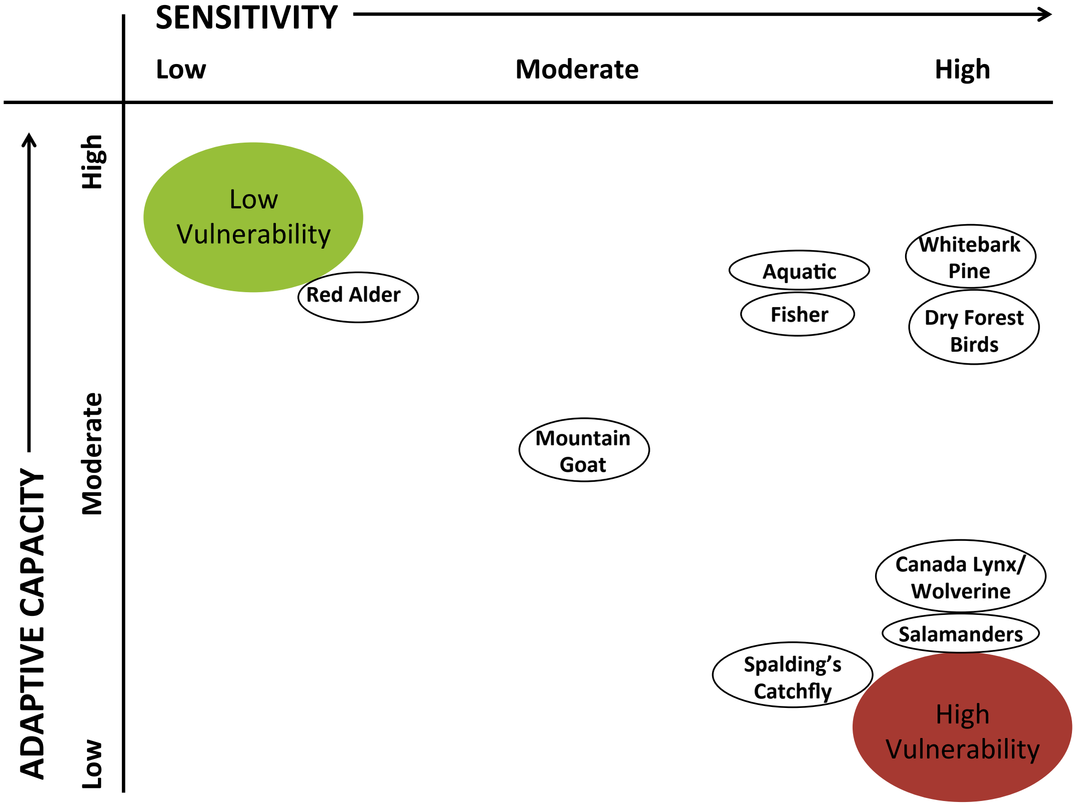


Figure 3. Relative vulnerabilities of nine Nez Perce-Clearwater species or species assemblages based on the climate change sensitivity and adaptive capacity assessment. Relative vulnerability, which does not include a measure of future climate exposure, increases with increasing sensitivity and decreasing adaptive capacity. Species listed in the upper left region were judged less vulnerable than those listed in the lower right region. Overall confidence for species sensitivities and adaptive capacities ranged from moderate to high.

This vulnerability assessment can be used as a foundation from which management and planning can be strengthened by better integrating the effects of climate change. However, it is also important to continue to gather information to better understand local climate, its interactions with non-climate stressors, and the impacts to resources. This assessment is intended to be updatable so that as new information becomes available on ecosystem or species' sensitivity, adaptive capacity, or exposure it can be integrated and used to re-evaluate vulnerability.

The overall vulnerabilities presented above are comparable only within the resources considered here and are not standardized in any way to other climate change vulnerability assessments. The information supporting these results is available in Section 6 and should be referred to before using the overview results in decision-making. The information in this vulnerability assessment is intended to help managers develop and prioritize adaptation strategies to conserve resources in the face of climate change.

Table 1. Key sensitivity, exposure, and adaptive capacity elements summarized for each of eight Nez Perce-Clearwater ecosystems. (+) indicates those factors that contribute positively to adaptive capacity. (-) indicates those factors that contribute negatively to adaptive capacity.

ECOSYSTEM	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
AQUATIC **Relative Vulnerability: Moderate**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (High):** • Increased stream temperatures • Changes that affect hydrologic regimes (e.g., low, high flows) including: ○ Snowpack depth ○ Shifts from snow- to rain-dominant watersheds ○ Snowmelt and runoff timing **Sensitivities to Non-Climate Stressors (Mod-High):** • Transportation corridors • Fire suppression practices • Timber harvest • Dams and water diversions	**Key Exposure Factors:** • Warming air temperatures (leading to increased stream temperatures) • Changes in precipitation type, timing and amount that affect hydrologic regimes: ○ Decreased snowpack ○ Shifts from snow to rain ○ Earlier snowmelt and runoff timing • Altered wildfire regimes	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High physical and topographical diversity • (+) Moderate to highly continuous in the region • (+) Moderate component species and functional group diversity • (-) Features disruptions due to human-related activities (e.g., dams, habitat alteration)
COASTAL DISJUNCT **Relative Vulnerability: Moderate**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Reduced soil moisture • Drought • Extreme temperature events • Wildfire	**Key Exposure Factors:** • Drought • Reduced soil moisture • Extreme hot or cold events • Increased wildfire	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) High component species and functional group diversity • (-) Exists in limited, “patchy” areas due to moist microclimate

ECOSYSTEM	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	Sensitivities to Non-Climate Stressors (High): • Timber harvest • Fire suppression • Grazing • Recreation	frequency and severity	requirements and limited dispersal ability • (-) Barriers to system continuity (e.g., timber harvest, land use conversion) • (-) Degraded structural and functional integrity
DRY FOREST **Relative Vulnerability: High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (High):** • Reduced soil moisture • Drought • Wildfire **Sensitivities to Non-Climate Stressors (High):** • Fire suppression practices • Insect and disease outbreaks	**Key Exposure Factors:** • Altered wildfire regimes • Reduced soil moisture • Drought	**Overall Adaptive Capacity: Low-Mod** **Key Factors Influencing Adaptive Capacity:** • (+) Moderate geographic extent • (+) Moderate physical, topographic, and component species diversity • (-) Historic forest structure and composition exist in isolated patches • (-) Low continuity
GRASSLANDS **Relative Vulnerability: Moderate**	**Overall Sensitivity: Moderate** **Sensitivities to Climate and Climate-Driven Changes (Low):** • Wildfire • Precipitation changes • Drought **Sensitivities to Non-Climate Stressors (Moderate):**	**Key Exposure Factors:** • Altered wildfire regimes	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) Moderate-high physical, topographic, and component species diversity • (+) Moderate-high structural and functional integrity • (-) Exists in isolated patches

ECOSYSTEM	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	• Invasive species • Fire suppression • Grazing		
MIXED MESIC **Relative Vulnerability: Moderate**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Reduced soil moisture • Drought • Wildfire **Sensitivities to Non-Climate Stressors (High):** • Insect and disease outbreaks • Fire suppression practices	**Key Exposure Factors:** • Altered wildfire regimes • Drought • Reduced soil moisture • Increased temperatures	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) High system continuity • (+) High physical, topographical, and component species diversity • (-) Degraded structural and functional integrity • (-) Barriers to dispersal
RIPARIAN **Relative Vulnerability: Moderate to Moderate-High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (High):** • Changes that affect hydrologic regimes and water availability, including: ○ Snowpack depth ○ Snowmelt and runoff timing ○ Shifts from snow- to rain-dominant watersheds • Wildfire **Sensitivities to Non-Climate Stressors**	**Key Exposure Factors:** • Changes in precipitation type, timing, and amount that affect hydrologic regimes and local soil moisture: ○ Decreased snowpack ○ Shifts from snow to rain ○ Earlier snowmelt and runoff timing	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High topographic, component species, and functional group diversity • (+) High system continuity and extent • (+) Moderate structural and functional integrity

ECOSYSTEM	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	(High): • Invasive species • Grazing • Transportation corridors • Dams and water diversions	• Altered wildfire regimes	
SUBALPINE **Relative Vulnerability: Moderate-High to High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (High):** • Temperature increases • Reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors (High):** • Insect and disease outbreaks • Fire suppression practices in lower elevation, non-wilderness areas	**Key Exposure Factors:** • Increased temperatures • Drought • Reduced soil moisture • Altered wildfire regimes	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) High component species diversity • (-) Degraded structural and functional integrity • (-) Low physical, topographic, and functional group diversity
WETLANDS/ MOIST MEADOWS/ GDES **Relative Vulnerability: Low-Moderate to Moderate**	**Overall Sensitivity: Low-Mod** **Sensitivities to Climate and Climate-Driven Changes (Moderate):** • Changes that affect water supply, including: ○ Precipition changes ○ Snowpack depth and snowmelt ○ Shifts from snow- to rain-dominant watersheds	**Key Exposure Factors:** • Precipitation declines • Decreased snowpack and earlier snowmelt • Shifts from snow to rain • Drought • Reduced soil moisture • Altered wildfire regimes	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) High structural and functional integrity • (+) Moderate component species diversity • (-) Low physical, topographical, and functional group diversity

ECOSYSTEM	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	○ Reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors (Low):** • Timber harvest • Grazing • Invasive species • Land use conversion		

Table 2. Key sensitivity, exposure, and adaptive capacity elements summarized for each of nine Nez Perce-Clearwater species, species assemblages, or species grouped together during the vulnerability assessment workshop. (+) indicates those factors that contribute positively to adaptive capacity. (-) indicates those factors that contribute negatively to adaptive capacity.

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
AQUATIC SPECIES **Relative Vulnerability: Moderate**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Increased stream temperatures • Changes that affect hydrologic regimes • Altered flow regimes (e.g., low or high flows, timing of flows) **Sensitivities to Non-Climate Stressors (Mod-High):** • Invasive species • Habitat homeginzation via: ○ Road networks (incl.	**Key Exposure Factors:** • Warming air temperatures (leading to increased stream temperatures) • Changes in precipitation type, timing, and amount that affect hydrologic regimes: ○ Decreased snowpack ○ Shifts from snow to rain	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High connectivity in the NPCW region • (+) High dispersal ability • (+) High intraspecific/life history diversity • (+) High genetic diversity within and among populations • (-) Disperal barriers (e.g., transportation corridors, logging, energy production and mining, dams and water diversions)

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	construction of new roads, road maintenance) ○ Dams and water diversions	○ Earlier snowmelt and runoff timing - Altered wildfire regimes	
CANADA LYNX AND WOLVERINE **Relative Vulnerability: High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (High):** - Temperature (wolverine) - Snowpack (wolverine, lynx) - Wildfire (lynx) **Sensitivities to Non-Climate Stressors (High):** ○ Logging and timber harvest (lynx) ○ Recreation (lynx) ○ Loss of core habitat and connectivity due to land use changes (wolverine, lynx)	**Key Exposure Factors:** - Increased temperatures (wolverine) - Decreased snowpack (wolverine, lynx) - Altered wildfire regimes (lynx)	**Overall Adaptive Capacity: Low-Mod** **Key Factors Influencing Adaptive Capacity:** - (+) Some behavioral diversity - (-) Small, isolated populations - (-) Lack of connectivity/Barriers to dispersal - (-) Low life history and genetic diversity
COEUR D'ALENE AND IDAHO GIANT SALAMANDERS **Relative Vulnerability: High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (High):** - Precipitation changes - Drought and/or reduced soil moisture - Wildfire **Sensitivities to Non-Climate Stressors (Mod-High):**	**Key Exposure Factors:** - Precipitation changes (e.g., more rain-on-snow events, shift from snow- to rain-dominant watersheds) - Reduced soil moisture - Drought (late summer, fall)	**Overall Adaptive Capacity: Low** **Key Factors Influencing Adaptive Capacity:** - (+) IGS ability to recolonize in recovered and/or restored areas - (-) Small, isolated, and declining populations - (-) Low dispersal ability and many natural and anthropogenic dispersal barriers

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	○ Human disturbances (e.g., dams and water diversions, road construction, and timber harvest) ○ Pollution ○ Introduced species	• Altered wildfire regimes	• (-) Limited genetic exchange
DRY FOREST BIRD SPECIES **Relative Vulnerability: Moderate to Moderate-High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (Moderate):** • Temperature increases • Precipitation changes and reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors: (High)** ○ Fire suppression practices ○ Timber harvest ○ Livestock grazing	**Key Exposure Factors:** • Altered wildfire regimes • Precipitation changes affecting soil moisture and drought • Increased temperatures	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High dispersal ability • (+) Some behavioral diversity • (-) Small populations with patchy distribution • (-) Limited reproductive potential
FISHER **Relative Vulnerability: Moderate**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Temperature increases • Snowpack depth • Reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors**	**Key Exposure Factors:** • Increased temperatures • Decreased snowpack • Altered wildfire regimes • Reduced soil moisture	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High current habitat connectivity • (+) Moderate dispersal ability, with some barriers to dispersal (-) • (-) Isolated and small regional population

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	(Moderate): • Timber harvest • Trapping • Disease		• (-) Low-Moderate genetic, behavioral, and life history diversities
MOUNTAIN GOAT **Relative Vulnerability: Moderate**	**Overall Sensitivity: Moderate** **Sensitivities to Climate and Climate-Driven Changes (Moderate):** • Temperature increases • Snowpack depth and snowmelt timing **Sensitivities to Non-Climate Stressors (Moderate):** ○ Hunting ○ Anthropogenic disturbances (e.g., snowmobiling, helicopters, roads)	**Key Exposure Factors:** • Increased temperatures • Decreased snowpack • Earlier snowmelt	**Overall Adaptive Capacity: Moderate** **Key Factors Influencing Adaptive Capacity:** • (+) Mod-High dispersal ability • (+) Moderate behavioral diversity • (-) Small, isolated populations
RED ALDER **Relative Vulnerability: Low**	**Overall Sensitivity: Low-Mod** **Sensitivities to Climate and Climate-Driven Changes (Moderate):** • Snowpack depth or amount • Drought and reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors (Low):** ○ Fire suppression activities ○ Disturbance events (e.g., timber	**Key Exposure Factors:** • Increased temperatures • Precipitation changes leading to drought or reduced soil moisture • Decreased snowpack and earlier snowmelt • Altered wildfire regimes	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) High seed dispersal ability • (+) High behavioral, life history, and phenotypic diversity • (+) Pioneer species that responds positively to disturbance • (-) Some barriers to dispersal • (-) Somewhat isolated

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	harvest, transportation corridors)		
SPALDING'S CATCHFLY **Relative Vulnerability: High**	**Overall Sensitivity: Mod-High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Precipitation changes • Prolonged drought and/or reduced soil moisture • Wildfire **Sensitivities to Non-Climate Stressors (Moderate):** o Livestock grazing (particularly overgrazing) o Invasive species o Land use conversion o Herbicide drift	**Key Exposure Factors:** • Precipitation changes • Prolonged drought • Reduced soil moisture • Altered wildfire regimes	**Overall Adaptive Capacity: Low** **Key Factors Influencing Adaptive Capacity:** • (+) Dormancy stage may buffer catchfly from short-term climate impacts • (-) Small, fragmented population • (-) Low genetic diversity • (-) Low seed dispersal range
WHITEBARK PINE **Relative Vulnerability: Moderate to Moderate-High**	**Overall Sensitivity: High** **Sensitivities to Climate and Climate-Driven Changes (Mod-High):** • Temperature increases • Reduced soil moisture or drought • Wildfire **Sensitivities to Non-Climate Stressors (High):** o Mountain pine beetle outbreaks o White pine blister rust	**Key Exposure Factors:** • Increased temperatures • Reduced soil moisture resulting from decreased snowpack and earlier snowmelt • Altered wildfire regimes	**Overall Adaptive Capacity: Mod-High** **Key Factors Influencing Adaptive Capacity:** • (+) Fairly large dispersal range • (+) Mod-High genetic diversity • (+) Demonstrates potential for life history and behavioral/ phenotypic plasticity • (-) Small and declining population • (-) Limited room to migrate vertically

SPECIES	SENSITIVITY	EXPOSURE	ADAPTIVE CAPACITY
	○ Fire suppression practices in lower subalpine areas		

1. Introduction

This vulnerability assessment is an initial science-based effort to identify how and why resources (ecosystems and species) across the Nez Perce-Clearwater (NPCW) region are likely to be affected by future climate conditions. In this context, vulnerability is a function of the sensitivity of the resource to climate change, its anticipated exposure to those changes, and its capacity to adapt to changes. Specifically, sensitivity is defined as a measure of whether and how a resource is likely to be affected by a given change in climate, or factors driven by climate; exposure is defined as the degree of change in climate or climate-driven factors a resource is likely to experience; and adaptive capacity is defined as the ability of a resource to accommodate or cope with climate change impacts with minimal disruption (Glick et al. 2011). The assessment centers on the NPCW region of Idaho (Figure 1). It focuses on twenty-eight resources including eight ecosystems and twenty species identified as important by the NPCW National Forest as part of their forest plan revision process, and is therefore expected to be relevant to decisions that affect these resources. The analyses and conclusions are based on available information and expert opinion.

Climate change vulnerability assessments provide two kinds of information: (1) they identify which resources are likely to be most affected by changing climate conditions, and (2) they improve understanding as to why these resources are likely to be vulnerable. Knowing which resources are most vulnerable better enables managers to set priorities for conservation action, while understanding why provides a basis for developing appropriate adaptation responses (Glick et al. 2011).

The overarching goal of this assessment is to provide vulnerability information and supporting tools and resources that will help forest managers plan their management of important resources in a changing climate. To meet this goal, the assessment has three main objectives:

1. To use the latest scientific information and expert knowledge to evaluate vulnerabilities of important resources to climate change including assessing sensitivity, exposure, and adaptive capacity.
2. To quantify sensitivities and adaptive capacities of important resources to climate change, and understand how climate exposure for these resources varies spatially across the Nez Perce-Clearwater region.
3. To work with resource managers and planners to increase their institutional knowledge and capabilities to respond to climate change by providing vulnerability assessment resources, support, and tools.

To achieve these objectives, a vulnerability assessment model was developed and applied consistently across the NPCW region that improves understanding of why resources may be vulnerable, how these vulnerabilities may vary across the region, and where and how management could intervene to reduce vulnerabilities. This report describes how this vulnerability model was developed, including how resources (ecosystems and species) were selected, and summarizes the results that were obtained when the model was applied to the

NPCW region. In a separate companion document prepared by Headwaters Economics, the vulnerabilities of important regional ecosystem services are discussed.

We recommend that resource managers and planners refer to the comments and supporting material provided for each assessment rather than only the rankings. While rankings can be a valuable tool, the comments and supporting material more clearly describe resource vulnerabilities, including any sub-regional differences, which can be used to better refine management options for limiting potential impacts.

2. An Overview of Climate Change in the Nez Perce-Clearwater Region

Climate models have projected major changes across the Nez Perce-Clearwater (NPCW) region over the next century. The NPCW National Forests are heterogeneous, spanning over 4 million acres, with jagged peaks of the Bitterroot Mountains, deep canyons of the Salmon, Selway, and Lochsa Rivers, coniferous forests, grasslands, and prairie. Due to its heterogeneous landscape, climate-related changes are expected to vary spatially across the NPCW region. Consequently, exposures and vulnerabilities of ecosystems and species will also vary temporally and geographically over the coming century. An improved understanding of the magnitudes and geographical variations of projected climate-related changes will help mangers be better prepared.

This chapter and the following chapter prepared by Sound Science describe current understanding of the changes in climate the NPCW region has and is predicted to experience, and potential influences on major physical and ecological processes such as wildfire and hydrology. This is not an exhaustive quantitative analysis, but an overview from which a better understanding of magnitudes and directions of change can be formed that support improved management decision-making. This overview can be used to begin to spatially and temporally assess relative vulnerabilities of ecosystems and species across the NPCW region. This chapter focuses on general changes – observed and predicted – for the Columbia Plateau and Northern Rockies ecoregions. The subsequent chapter focuses on more specific, downscaled climate data – observed and projected – for the NPCW region.

Climate in the Western U.S., including the NPCW, is strongly influenced by naturally occurring climate cycles such as the 20-30 year Pacific Decadal Oscillation (PDO) and the 1-2 year El Nino-Southern Oscillation (ENSO). These large-scale climate patterns influence the local climate in the NPCW by causing warmer/cooler and drier/wetter conditions depending on the phase of the PDO and ENSO. Currently, there is relatively low ability to predict changes in the PDO and ENSO, and it has proven difficult to understand how climate change may influence these naturally occurring phenomena. Climate models are better at predicting general trends in climate rather than detailing year-to-year variability.

Over the past 30 years, air temperature has been increasing an average of 0.13˚C per decade (Isaak et al. 2011), with annual average minimum temperatures increasing 0.26˚C per decade and annual average maximum temperatures increasing 0.34˚C per decade (Littell et al. 2011). Minimum temperature has increased in all seasons across the NPCW ecoregion, and has been more pronounced at higher elevations (above 800m) and in summer and winter. Recent change in maximum temperature in the region has been seasonally variable (with winter experiencing increases in maximum temperature) and less extreme than change in minimum temperature (see Section 3 of this report for more information). Precipitation trends for the region are mixed, with some areas showing declines in annual precipitation of -1 cm or greater and others showing increases in annual precipitation of +1 cm or greater (Littell et al. 2011). Other historic changes include:

- Increased snow water equivalent (SWE, amount of water contained in snowpack) of 0-0.5% per year from 1916-2003 (Hamlet et al. 2005);
- Little to no shift in timing of snowmelt from 1916-2003 (90% of snowmelt occurred 0-5 days later) (Hamlet et al. 2005);
- Increased average in stream temperatures of 0.01˚C per decade over the past 30 years (Isaak et al. 2011);
- An average decrease in flow of 2.1% per decade over the past 30 years for unregulated streams, whereas regulated streams have seen a decrease in flow of 2.8% (Isaak et al. 2011); and
- Increased wildfire frequency and greatest absolute increase in large wildfires in forests of the Northern Rockies (Westerling et al. 2006).

Over the next century, annual temperatures across the Columbia River Basin are expected to continue to increase by approximately +2˚C by 2040 (Littell et al. 2011; Table 3). In the NPCW region, mid-century conditions are modeled to be about 2.5˚C warmer than the 20th-century baseline. Exact precipitation patterns in the future are uncertain, but in general, summer is projected to be drier while spring, winter and fall will be wetter relative to historic averages (Littell et al. 2011; see Section 3). Precipitation will fall more often in the form of rain rather than snow, decreasing seasonal snowpack and increasing flood risk. Warmer temperatures in the summer and fall will increase evapotranspiration rates causing more severe summer low flows in rivers and reduced soil moisture. Warmer and drier conditions will increase the likelihood of wildfire across the NPCW. Climate models are better at predicting some climate-driven changes, such as higher temperatures and lower snowpack, than others (e.g., precipitation change). Specifically, the following changes are projected for the NPCW:

- By 2040, average annual temperature is expected to increase by 2.5°C (see Section 3), with warmer seasonal temperatures generally occurring in the summers (Littell et al. 2011; see Section 3).
- Minimum and maximum temperatures are also projected to increase, with more significant increases in summer and winter (see Section 3).
- By 2040, precipitation is generally expected to decrease in summer (-13 to -20%) and increase in spring (+5 to +10%), winter (+5%), and fall (+2%) (Littell et al. 2011; see Section 3).
- By 2040, combined flows (runoff + baseflow) are projected to increase in winter (+19%) and decrease in summer (-23%) (Littell et al. 2011). Daily mean stream flow volumes are projected to decrease ~5-10% by 2040 (see Section 3).
- Annual snowpack is projected to decline ~21% by 2040 (Littell et al. 2011).
- Historically snow dominated basins are projected to become transitional (i.e., those basins with between 10-40% of winter precipitation entrained in April 1 snowpack) and transitional basins are projected to become rain dominated by the 2040s (Littell et al.

2011), which has the potential to cause large changes in the timing and magnitude of seasonal hydrographs (Elsner et al. 2010).

- The center of timing of stream flow (i.e., the day when 50% of the year's water has passed) is projected to occur as much as 4-6 weeks earlier in 2040 compared to the historic baseline (see Section 3).
- July 1 soil moisture is projected to decline by up to 35% across the Columbia River Basin by 2040 (Littell et al. 2011). In the NPCW region, soil moisture is modeled to decrease slightly in summer, with increases in spring and fall (see Section 3).
- In the summer, stream temperatures may warm at rates of 0.3-0.45˚C per decade, causing a net increase of 1.2-1.8˚C by mid-century (Isaak et al. 2011). Further, stream isotherms may shift 5-143 km upstream if air temperatures rise by 2˚C (Isaak and Rieman 2013). Lower-order streams in upland areas are projected to have less warming than larger arteries in lower elevations (see Section 3).
- Warming winters with lower snowpack and increased proportion of rain to snow could lead to increases in area burned (Littell et al. 2009), and warming spring and winter conditions could continue to lengthen fire season (e.g., Westerling et al. 2006).

Table 3. Historic climate changes, direction of future change, and confidence for the Pacific Northwest (Hamlet et al. 2005; Westerling et al. 2006; IPCC 2007a and references therein; Littell et al. 2011). Historic changes in temperature and precipitation are from 1950-2006. April 1 Snow Water Equivalent (SWE) is a measure of the amount of water contained in snowpack.

Climate variable	Historic change (1950-2006)	Direction of future change	Confidence in direction of change
Average annual temp	+0.13˚C/decade	Increasing	High, although exact rates and magnitudes of warming are more uncertain.
Precipitation	-3.6 mm	No change to small increases in annual precipitation	Uncertain, due to low skill in predicting precipitation trends in the region.
Snowpack (SWE)	0-0.5%	Decreasing	Uncertain due to low skill in predicting precipitation patterns; temperature-driven declines in snowpack are more likely.
Wildfire	Fire season is longer, increased wildfire frequency	Increasing	Warming air temperatures are likely to lead to more frequent and intense fires, although the locations will vary.

3. Current and Future Climate Space Trend Analysis for the Nez Perce-Clearwater Region

Introduction

Analysis of climate trends provides spatially explicit and visually intuitive metrics of observed and projected change that can inform vulnerability assessments and support adaptation planning efforts by natural resource managers. Using gridded climate data interpolated from weather stations (Daly et al. 2002) and from multiple downscaled global circulation models (Wang et al. 2012), we apply climate space trend analysis in the Nez Perce-Clearwater (NPCW) National Forest to help illuminate the rate, magnitude, spatial and temporal nature of current and forecasted climate change. Using time-series gridded climate data, we generate 20th century baseline values of a range of seasonal climate variables, quantify their interannual variability, and map the degree of climate change between baseline values and recent decades. We also analyze future projected change forecasted by downscaled global climate models, quantify values that exceed natural variability of 20th century climate, quantify the degree of model agreement in projected changes, and map the spatial distribution of modeled change in two future time slices: near-term (2010-2039) and mid-century (2040-2069) for a suite of 6-7 climate models run under two greenhouse gas emissions scenarios (A2 and A1B; Pachauri et al. 2007).

The analysis boundary includes a buffer around the NPCW National Forest to support understanding of the landscape context of climate change. The boundary is based on hydrologic unit code sub-basin level 4 (8-digit). Level 4 hydrologic unit codes surrounding the NPCW National Forest were selected and merged to create a regional boundary.

The descriptive statistics chosen for this analysis are non-parametric due to the nature of some variables in the climate data, which are non-normally distributed. Traditional parametric techniques such as mean and standard deviation are inappropriate for describing non-normally distributed data because outliers may critically affect their estimation (Lanzante 1996; Von Storch and Zwiers 2001). For this reason, the median value is the statistic used to represent central tendency of a climate variable across the years in a given timeslice.

Current Trends in Climate Space – PRISM 800m

The objective of analysis of current trends in climate space is to identify the rate, magnitude, spatial, and temporal nature of climate change *that is already occurring* across the NPCW region. The climate variables analyzed were seasonal minimum temperature, seasonal maximum temperature, and seasonal precipitation for the period 1901-2012. We used PRISM 800m gridded climate data, a widely used spatial climate time-series dataset that is the official climatology of the USDA (Daly et al. 2002). This is the finest scale gridded climate data available for a historical time series and it supports an understanding of the interactions and patterns of climate and topography across the NPCW region.

Methods Description

1) Baseline: We calculated a median value representing the 20th century baseline for each 800m pixel, for each variable, for each season (winter, spring, summer, fall) using the years 1901-1980. This 80-year baseline was chosen in order to capture a wide range of historic variability, and it represents climatic conditions prior to clearly discernible influences of anthropogenic greenhouse gas emissions (Lee et al. 2006; Solomon et al. 2007).
2) Recent: We calculated median values for each pixel, for each variable, for each season, for each of 3 timeslices representing recent periods: 1981-2012 ("30-year"), 1991-2012 ("20-year"), and 2001-2012 ("10-year").
3) We calculated the delta (the change in value) between the three recent timeslice medians and the baseline median for each pixel, for each variable, for each season. For temperature, delta is calculated as a difference (future – baseline). For precipitation, delta is expressed as a ratio (future/baseline), so a delta ratio of 1.0 indicates no change, less than 1.0 is decreasing precipitation, and greater than 1.0 is increasing precipitation.
4) We calculated "climate novelties" for each variable, for each 30-year, 20-year, and 10-year recent time slice, to identify the relationship between recent climate conditions relative to the range of historic variability (Figure 4). This is a continuous surface from 0-1, which represents the percentile at which the median value of the recent timeslice falls within the distribution of 80-year baseline values. Our color scheme for map scales highlights the pixels that fall outside of the interquartile range of the baseline, identifying "anomalous" climate values. The Interquartile range (IQR) is our metric of normal historic variability, which is the first quartile subtracted from the third quartile of the data (Figure 5). In other words, this statistic estimates the middle 50% of the data, which defines the range of "normal" baseline conditions, and trims the outer 50% (25% on both sides of the distribution) (Lanzante 1996).

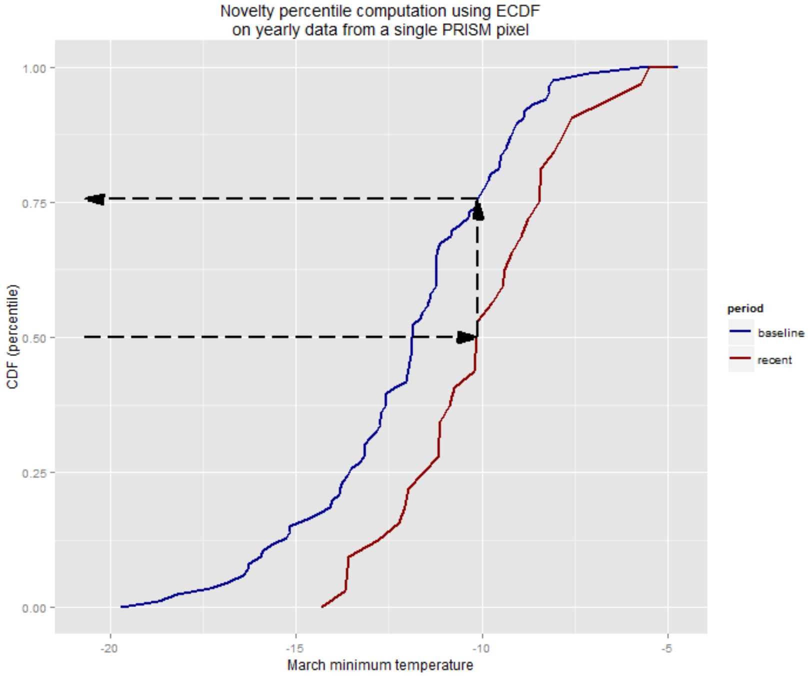


Figure 4. This figure demonstrates the method for identifying climate novelties by plotting an empirical cumulative distribution function (ECDF) for a single pixel across baseline and recent years. To determine novelty, we ask:

where does the recent median of minimum temperature fall within the values of the baseline period? Following the logic of the black arrows, the figure shows that the recent median falls in the 75th percentile of the baseline. This means that the value of March minimum temperature in the recent timeslice is higher than 75% of the values that occurred in the 80-year baseline period. This calculation is done for every pixel across the region.

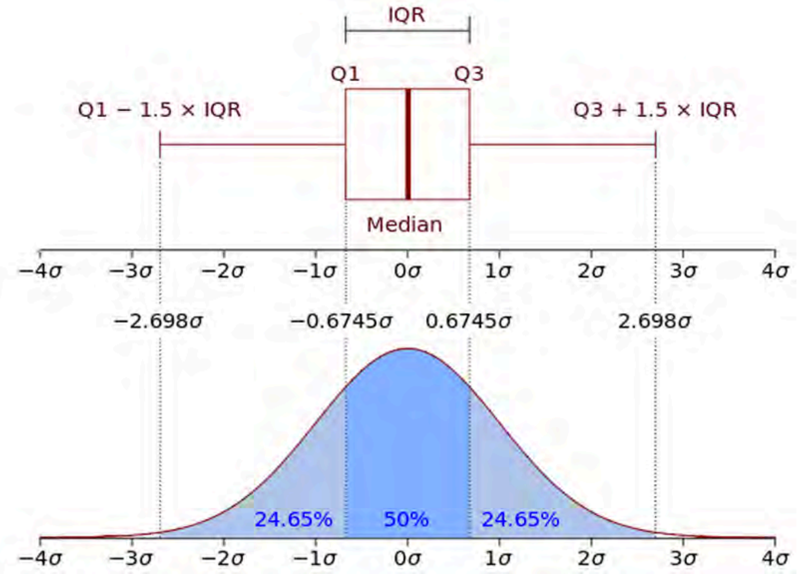


Figure 5. A visualization of the interquartile range (IQR) for a normal distribution is shown below. IQR is the first quartile subtracted from the third quartile resulting in the middle 50% of the data.

Current Trends in Climate Space – Results Description

Current Climate Space Scatter Plots*:* Climate space scatter plots visualize the shifts in raw climate values for each pixel, for each recent timeslice, for each variable and season, for the NPCW forests as well as the greater HUC4 region.

Current Climate Deltas: Maps showing the spatial distribution of the deltas (change in values) between recent and baseline time periods, for each variable, for each season, for each recent time slice.

Current Deltas by Elevation*:* Charts that show deltas between recent timeslices and the baseline in relation to 100m bins of elevation and total land area for the regional NPCW boundary.

Current Climate Novelties*:* Maps of the location and magnitude of climate novelties (pixels that are outside the 20th century interquartile range).

Projected Future Trends in Climate Space – Climate Western North America 4km

Climate data for projected future changes in the NPCW region is based on downscaled global circulation model (GCM) forecasts from the ‘Climate Western North America’ (CWNA) dataset (Wang et al. 2012), a 4km gridded climate time series dataset offering a suite of directly calculated and derived variables. The dataset includes observations from 1900-2010 and projected futures from a range of GCM and emissions scenario combinations for three 30-year timeslices. Here, we present analyses of forecasted climate change for two future timeslices: 2010-2039 (near-term, here labeled as “2020s”), and 2040-2069 (mid-century, labeled “2050s”). Trend analysis was conducted between a 1961-1990 baseline and an ensemble of future models for seasonal minimum temperature (tmin), seasonal maximum temperature (tmax), seasonal precipitation (ppt), and annual climatic moisture deficit (CMD). CMD is the sum of the monthly difference between evaporative demand and precipitation (Wang et al. 2012), and is a useful indicator of drought stress.

All analyses of future change are based on the "2020s" and "2050s" timeslices for the A2 and A1B emissions scenarios. The A1B emission scenario reflects a future with low population growth, rapid economic growth, and new efficient technologies (Pachauri et al. 2007). The A2 emission scenario reflects a future in which there is high population growth, regional economic development, and slow technological changes (Pachauri et al. 2007). There are a total of 6 GCMs available in the CWNA dataset for the A2 emissions scenario and 7 GCMs available for the A1B emissions scenario. There are issues with the source data for the GCMs that are carried through in the CWNA dataset and cause some limitations with certain variables. Specifically, for the A1B scenario GCMs, tmin and tmax were derived from tmean, making tmin and tmax, and also CMD (which is derived from these), unusable under this scenario. **Therefore, our analysis includes mean temperature and precipitation for the A1B scenario, and minimum temperature, maximum temperature, precipitation, and CMD for the A2 scenario.**

Methods Description

1) We calculated a mean value representing the 20th century baseline for each 4km pixel, for each variable, for each season (winter, spring, summer, fall) using the years 1961-1990, which is the baseline with which the CWNA future GCMs were downscaled (Wang et al. 2012).
2) We calculated the delta (the value of change) between each future timeslice and the baseline climate, for each pixel, for each variable, for each season. This was done for each GCM available for a given emissions scenario. For calculations where a single ensemble projection is required, the median value across GCMs was calculated. For temperature, delta is calculated as a difference (future – baseline). For precipitation, delta is expressed as a ratio (future/baseline), so a delta ratio of 1.0 indicates no change, less than 1.0 is decreasing precipitation, and greater than 1.0 is increasing precipitation.
3) Methods for synthesizing GCM projections for temperature and precipitation differ, because for precipitation, GCMs disagree on the direction of change, whereas for temperature, GCMs agree on the direction and only vary in magnitude of change (Girvetz 2009).
    a. For seasonal temperature variables and climatic moisture deficit, deltas were calculated for each GCM and the final "delta" output is the median delta value across GCMs for each scenario.
    b. For precipitation, divergent climate model projections were synthesized using an ensemble quantile analysis. This was conducted by overlaying delta ratios for the 7 GCMs and calculating the quantile range across GCMs, following the method of Girvetz et al. (2009). For each pixel, projections are only displayed if a predefined percentage of models agree about the direction of change. In those cases, the delta ratio displayed is that projected by the most conservative of this subset of agreeing models. Thus in the resulting charts, the stated percentage of models agree there will be at least the mapped amount of change. Maps were created showing both 60% and 80% model agreement.
4) We calculated "climate novelties" for each variable, for each season, for each pixel, to identify where future climate falls within the range of historic variability. This was done for each GCM in

the ensemble, and then the median across GCMs was calculated to represent the final ensemble projection. This variable ranges from 0 to 1, and measures the percentile at which the mean value of the future timeslice falls within the distribution of historic baseline values. The color scheme for map scales highlights the pixels that fall outside of the interquartile range of the baseline, identifying “anomalous” climate values. Pixels with a value of 0 or 1 indicate locations where the mean future climate is projected to fall completely outside the range of historic climate variability.

Projected Future Trends in Climate Space – Results Desciption

Projected Future Climate Space Scatter Plots*:* Climate space scatter plots that show shifts in raw climate values for each pixel, for each variable, for each season, for each future timeslice (2020s and 2050s), for the forests and the greater HUC4 region.

Projected Future Climate Deltas*:* Maps of geographic variation in deltas between future and baseline periods for each variable, for each season, for each future timeslice (2020s and 2050s). Maximum and minimum temperatures were only available for the A2 emissions scenario. A1B was used for mean temperature and precipitation.

Projected Future Climate Deltas – Precipitation Model Agreement*:* Maps of geographic variation in deltas between future and baseline periods for precipitation showing 60% and 80% model agreement in a direction of change (increasing/decreasing), for each season.

Projected Future Climate Novelties*:* Maps of the location and magnitude of climate novelties (pixels with mean future values that are outside the baseline interquartile range) for each variable, for each season, for each future timeslice (2020s and 2050s).

Current and Projected Future Stream Temperature and Flow Metrics

Stream Temperature Dataset Description

The NorWest stream temperature dataset (Hamlet et al. 2013) is a composite of daily stream temperature recordings at more than 15,000 locations. These temperature records were used with spatial statistical stream network models to develop a comprehensive regional database of historic and recent stream temperatures. Historical stream temperature estimates were modeled based on mean August air temperature and stream discharge values, and matched to those observed for a historical period (1993-2011). Future stream temperature values for the “2040s” (2030-2059) and the “2080s” (2070-2099) were estimated with climate model projections from the A1B scenario with an ensemble of 10 GCMs downscaled using a spatially explicit delta method (Hamlet et al. 2013) and adjusted for differential stream sensitivity.

Stream Flow Metrics Dataset Description

The VIC (Variable Infiltration Capacity) dataset (Wenger et al. 2010) was used to analyze projected future changes in stream flow regimes. The University of Washington Climate Impacts Group, Trout Unlimited, and the U.S. Forest Service Rocky Mountain Research Station used the VIC model to estimate stream flows for historical and future timeslices. The historical baseline used is a 1915-2006 average; future projections are based on the A1B emissions scenario using an ensemble of 10 GCMs.

Methods Description

Stream temperature data was linked to the National Hydrography Dataset (NHD) for the Clearwater Basin to show maps of stream temperature values for historic (1993-2011) and mid-century modeled future (2040s). Classes for degrees Celsius are symbolized in equal intervals of 2 degrees. The temperature change between historic and future was calculated by subtracting the historic values from the modeled future values per stream segment.

To display future change in flow metrics the VIC dataset was joined with stream segment shapefiles at 1:100,000 scale from the NHD. Deltas were calculated to show the change between historic flow metrics and projections for the future mid-century timeslice 2030-2059 ("2040s"). Deltas are expressed as a ratio (future/historic) as well as a difference (future-historic) in cubic feet/second. Flow metrics are symbolized with equal intervals for values within one standard deviation of the mean, and larger intervals for values outside two standard deviations of the mean.

Stream Temperature and Flow Metrics – Results Description

Stream Temperature*:* Maps showing historic average stream temperatures (1993-2011) and future modeled temperatures for the mid-century ("2040s").
Stream Flow Metrics*:* Maps showing various metrics of flow for historic and mid-century future (2040s) periods, and delta maps showing the projected changes between these two timeslices.

Flow metrics:

1. Daily Mean (DM) – mean daily flow averaged over a year (cubic feet/second).
2. Channel Flow (Q1.5) – the 1.5-year flow, also called channel-forming flows (cubic feet/second).
3. Center of Timing (CT) – timing of the center of mass of flow (i.e. day of the water year at which 50% of the year's flow has passed). A water year is the period between October 1st of one year and September 30th of the next.
4. Flow7q10 (7Q10) – the 7-day low flow with a 10-year return interval (cubic feet/second).

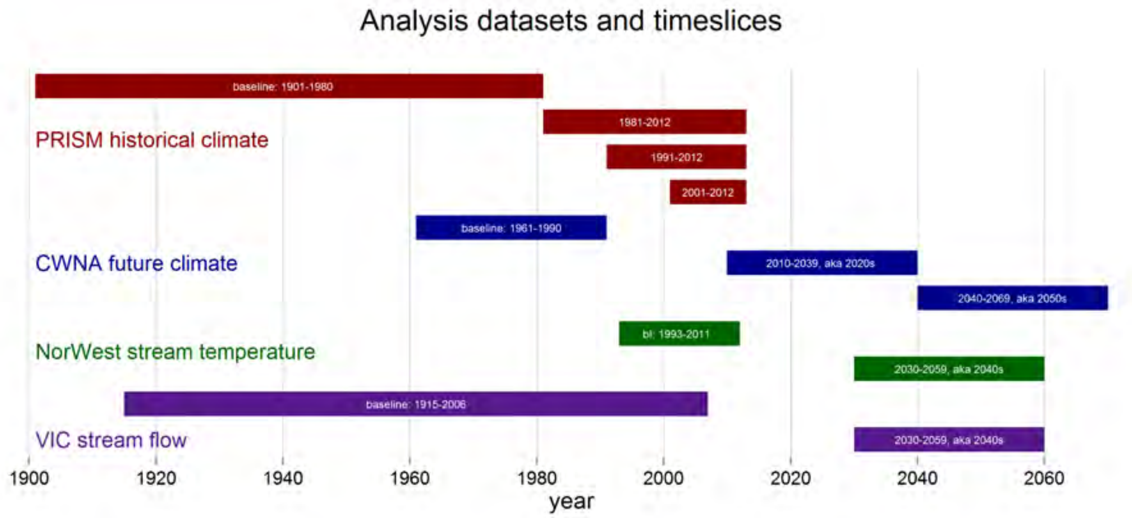


Figure 6. Climate datasets and timeslices.

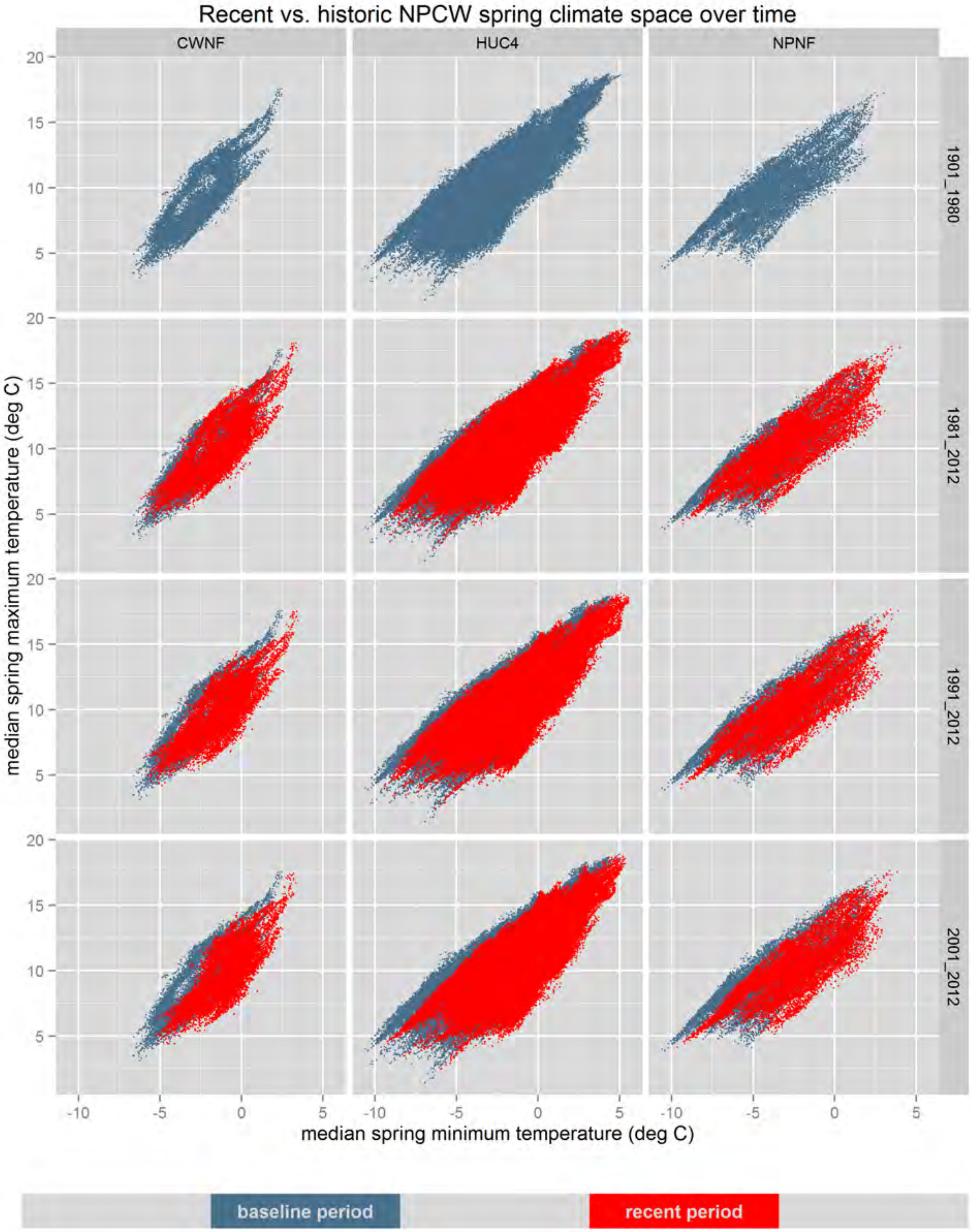
Maps of Current Trends in Climate Space – PRISM 800m

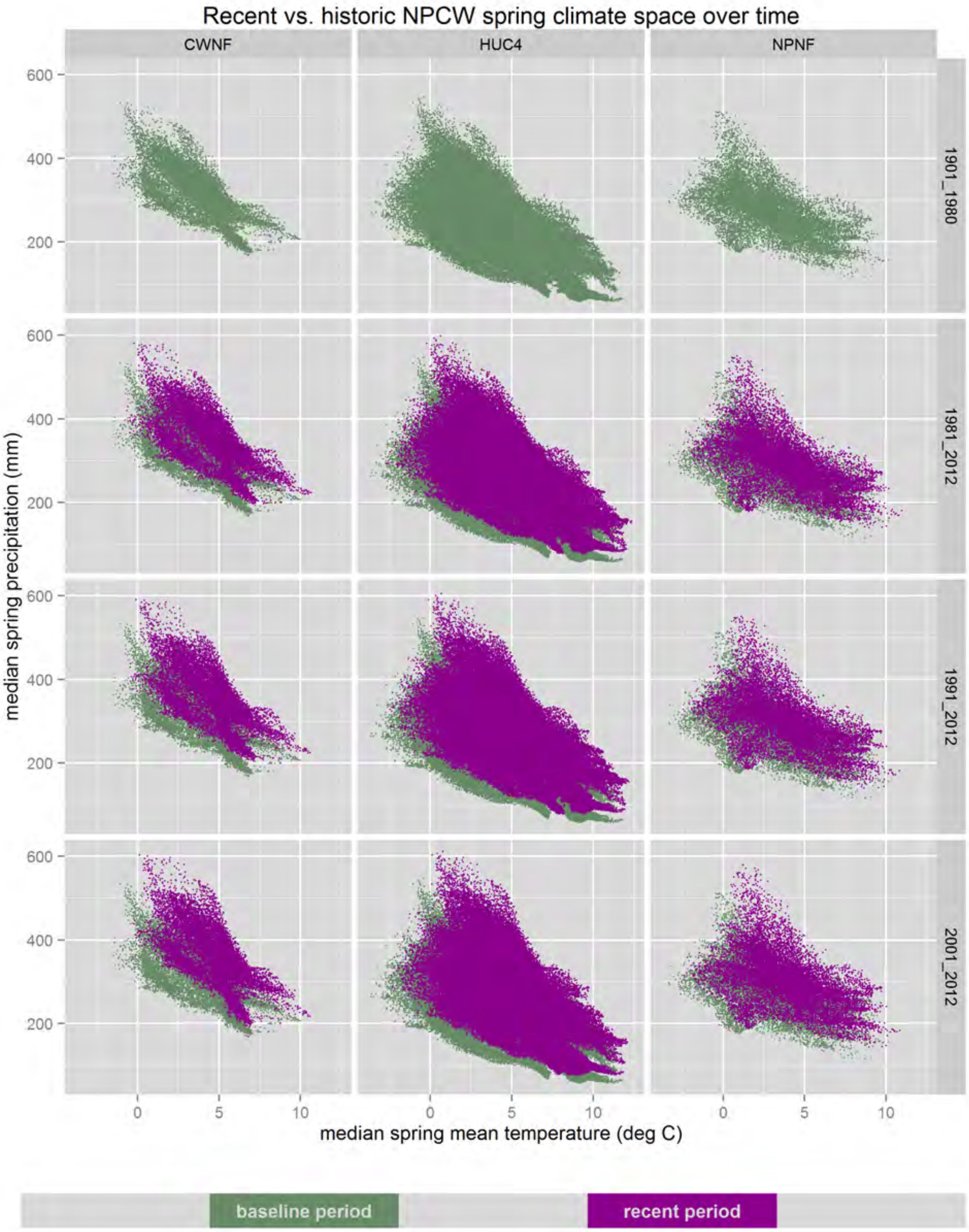
Current Climate Space Scatter Plots

These climate space scatter plots show shifts in raw climate values for each pixel, for each recent timeslice, for the NPCW forests as well as the greater HUC4 region. Each point on the climate space chart represents a single 800m pixel in geographic space, and the offset between baseline and recent points indicates the magnitude of recent climate change that has occurred. Points that have shifted into a space not previously occupied by any baseline point represent locations in the NPCW where the recent climate is unlike the baseline climate anywhere in the region.

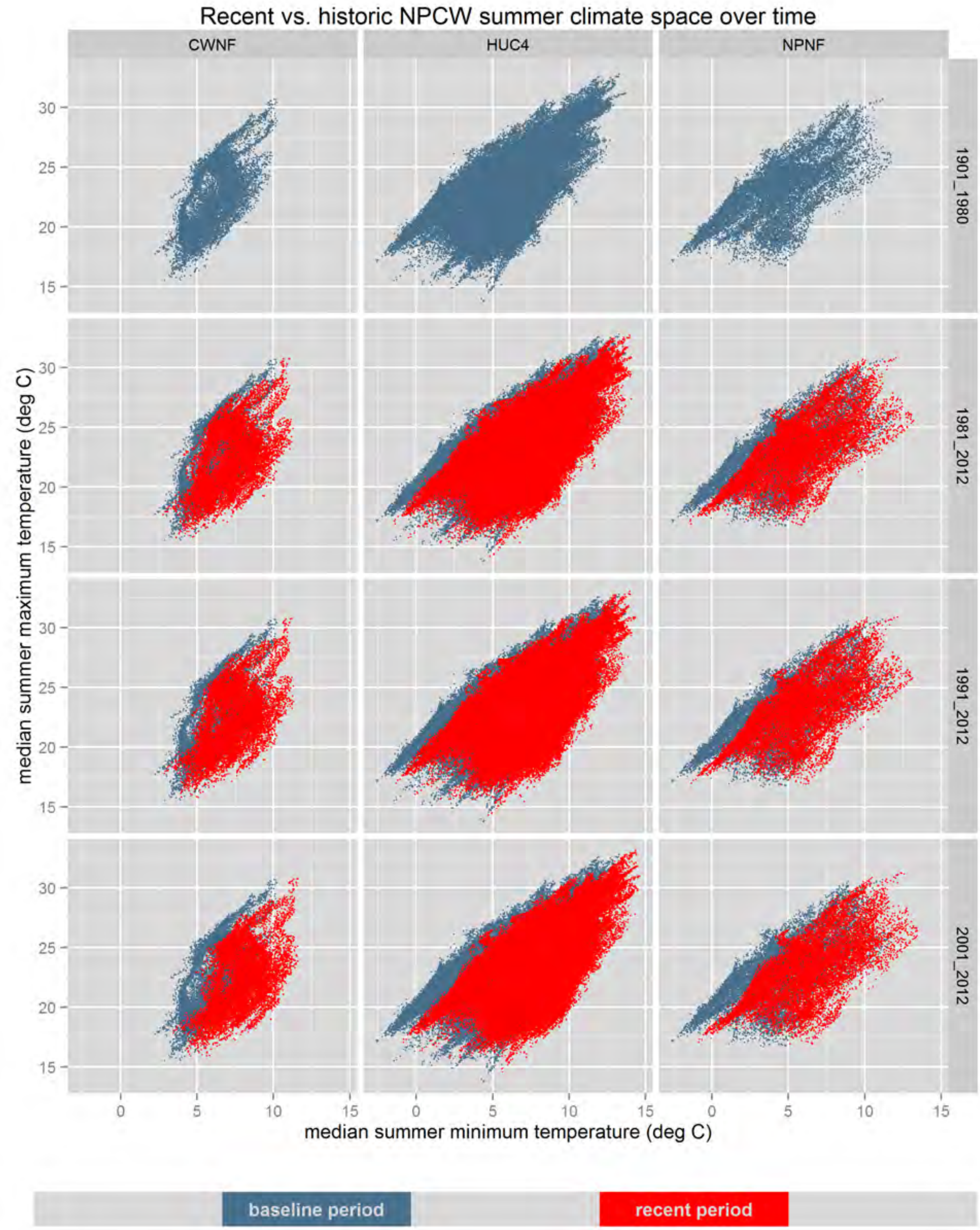
For each season there are two graphs: 1) maximum temperature vs. minimum temperature, and 2) mean temperature vs. precipitation. Across the top of the figure is the region (CWNF = Clearwater National Forest; HUC4 = regional boundary; NPNF = Nez Perce National Forest). The right side panel shows the baseline (1901-1980) and the three recent timeslices (1981-2012, 1991-2012, 2001-2012).

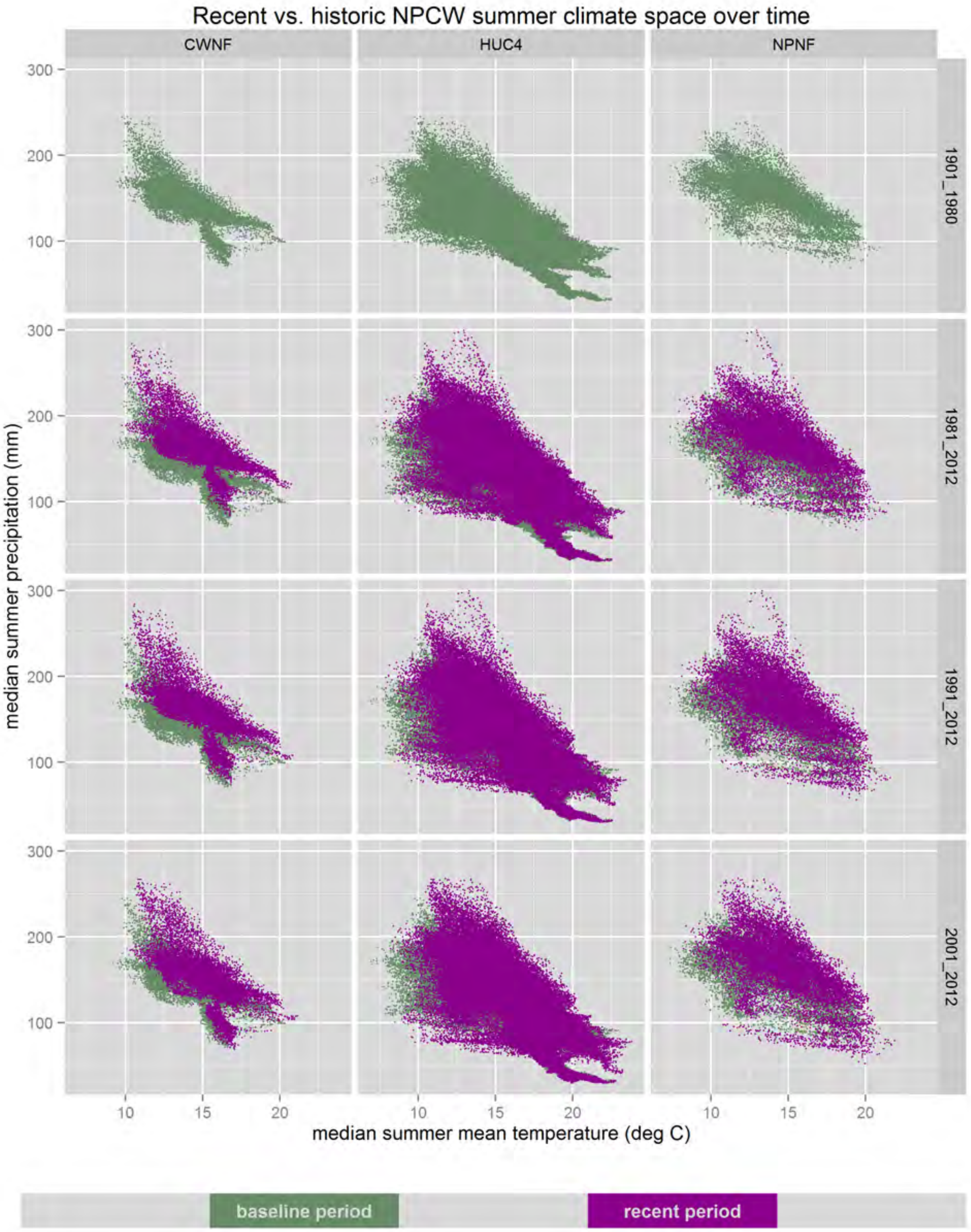
Current Spring Climate Space




Recent vs. historic NPCW spring climate space over time
CWNF
HUC4
NPNF
1901_1980
1981_2012
1991_2012
2001_2012
600
400
200
median spring precipitation (mm)
0
5
10
median spring mean temperature (deg C)
baseline period
recent period

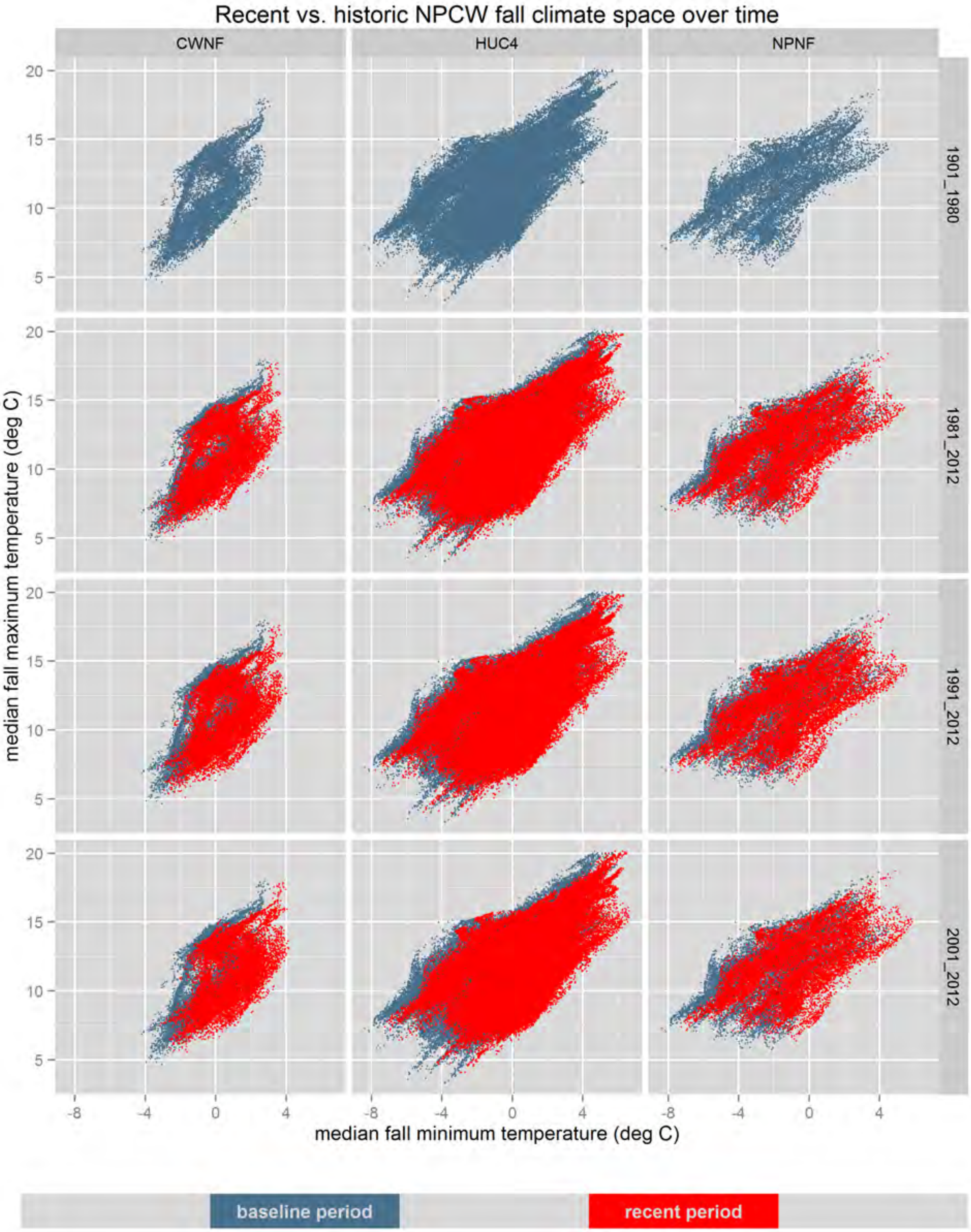
Current Summer Climate Space

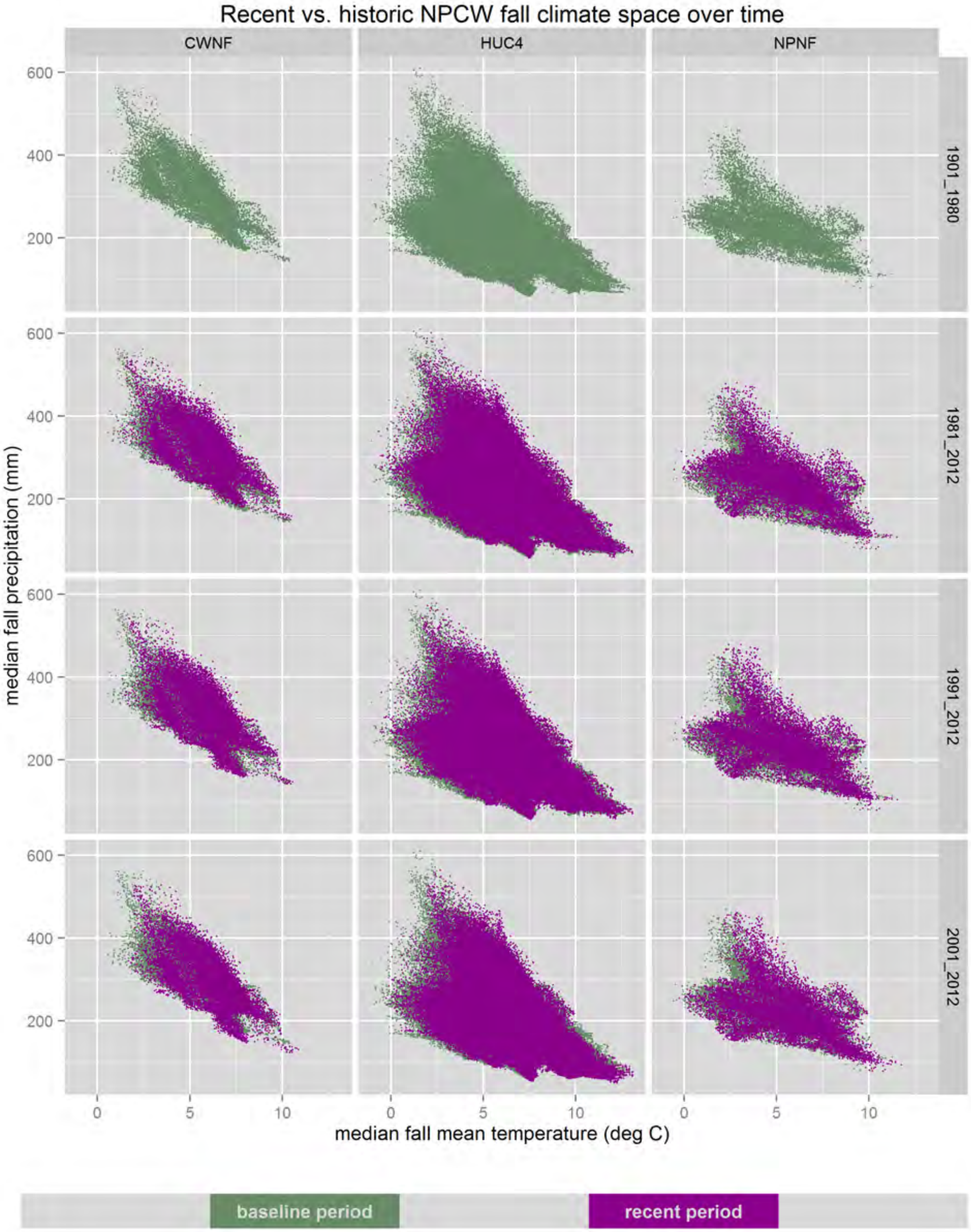




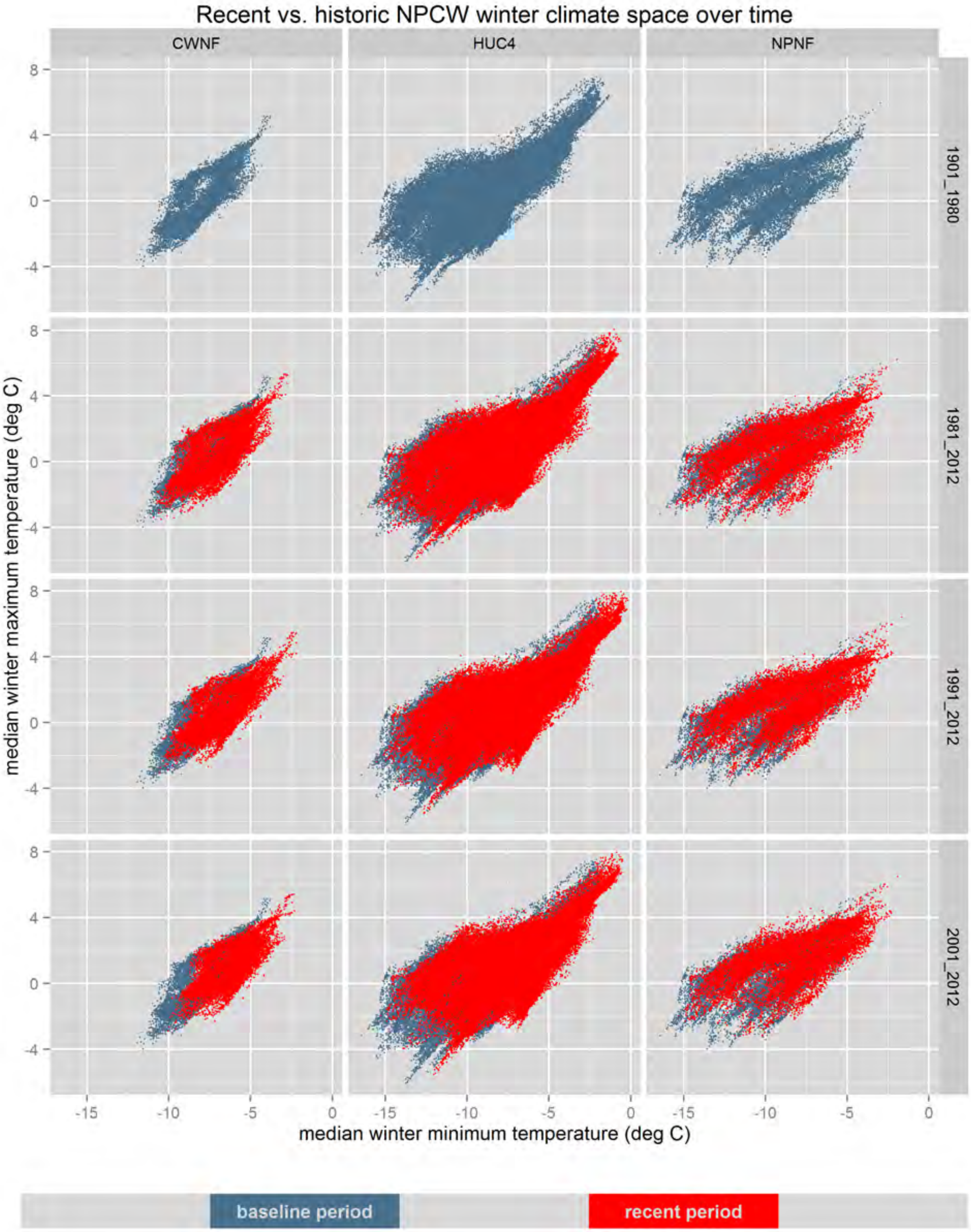
Recent vs. historic NPCW summer climate space over time
CWNF
HUC4
NPNF
1901_1980
1981_2012
1991_2012
2001_2012
300
200
100
10
15
20
median summer precipitation (mm)
median summer mean temperature (deg C)
baseline period
recent period

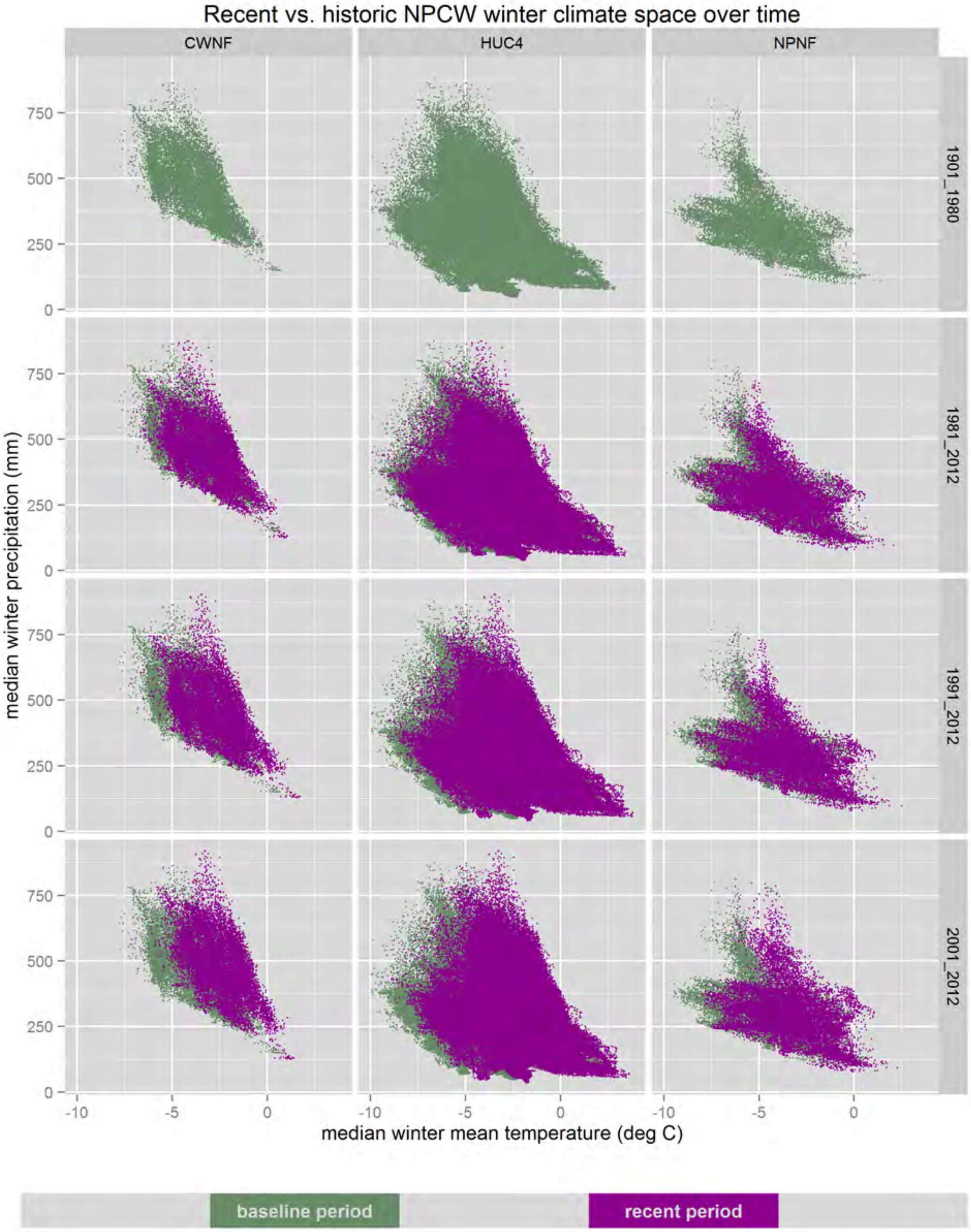
Current Fall Climate Space




Recent vs. historic NPCW fall climate space over time
CWNF
HUC4
NPNF
1901_1980
1981_2012
1991_2012
2001_2012
600
400
200
0
5
10
median fall precipitation (mm)
median fall mean temperature (deg C)
baseline period
recent period

Current Winter Climate Space

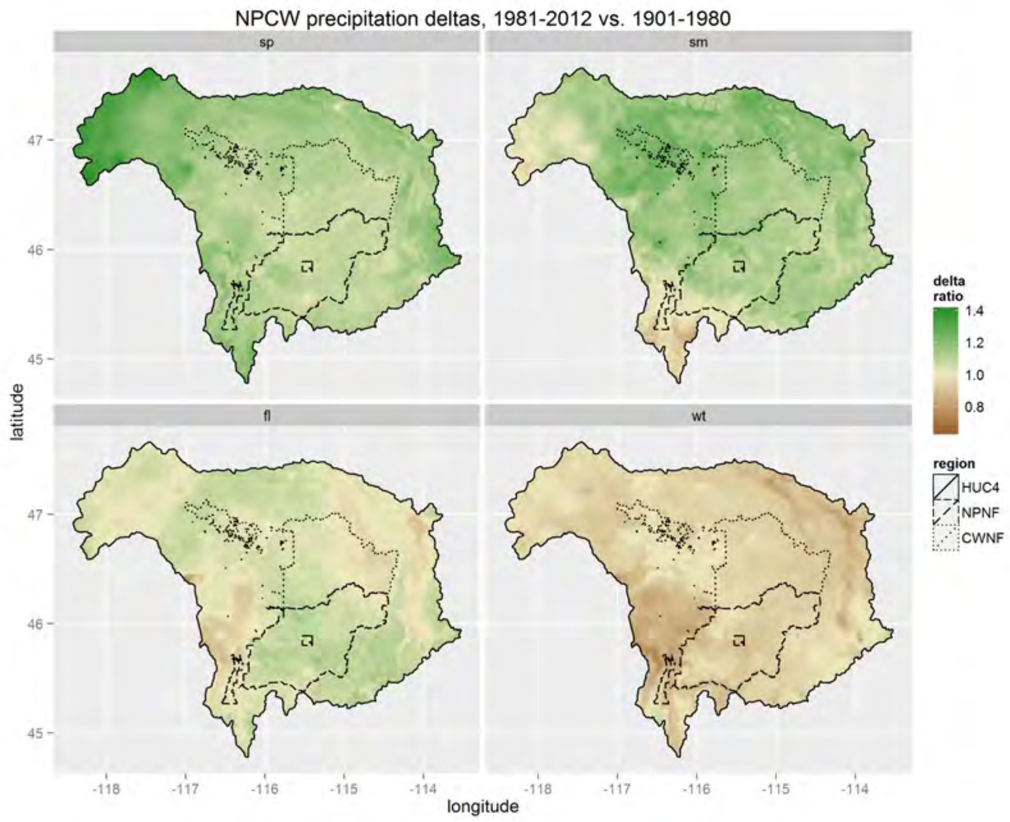


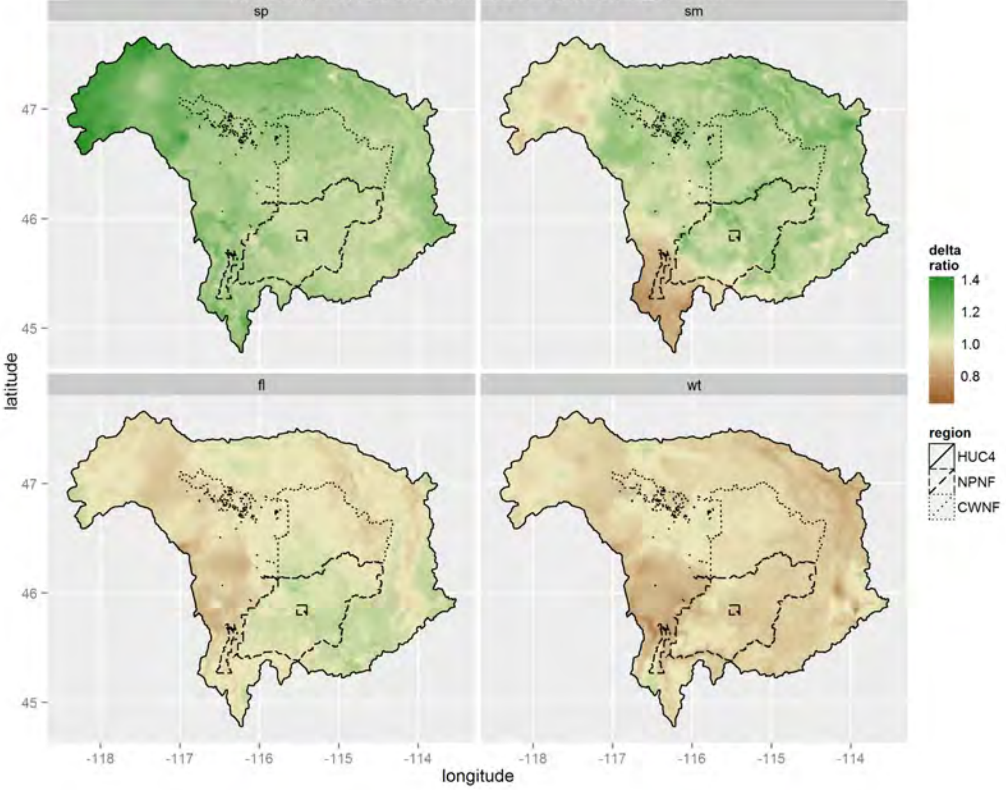

Recent vs. historic NPCW winter climate space over time
CWNF
HUC4
NPNF
1901_1980
1981_2012
1991_2012
2001_2012
0
250
500
750
median winter precipitation (mm)
-10
-5
0
median winter mean temperature (deg C)
baseline period
recent period

Current Climate Deltas

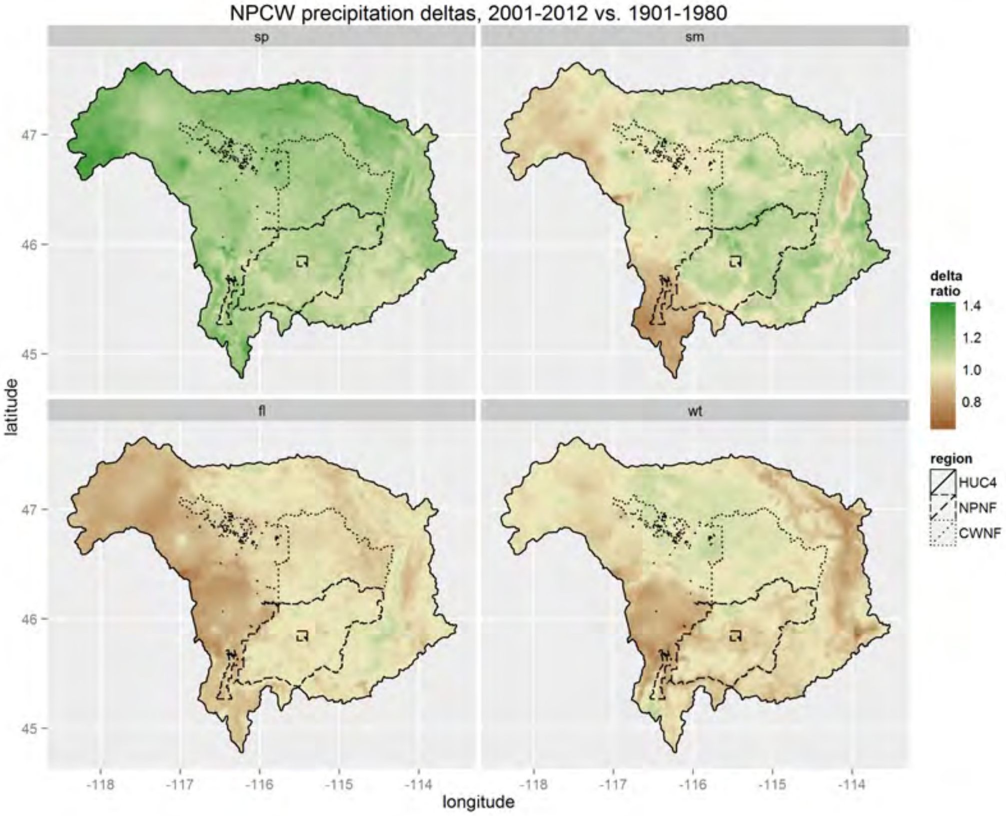
The figures below show differences between the 80-year baseline (1901-1980) and the three recent timeslices (30-year, 20-year, and 10-year) for seasonal maximum temperature, minimum temperature, and precipitation. Temperature deltas are expressed as a difference (future – baseline) in degrees C, with 0 representing no change, positive values representing warming (red areas), and negative values representing cooling (blue areas). Precipitation deltas are expressed as a unitless ratio (future/baseline), with 1.0 representing no change, values greater than 1 representing increasing precipitation (green areas), and values less than 1 representing decreasing precipitation (brown areas). The top of the map images are labeled by season: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7, 8), **fl** = fall (months 9, 10, 11), **wt** = winter (months 12, 1, 2).

Current Precipitation Deltas



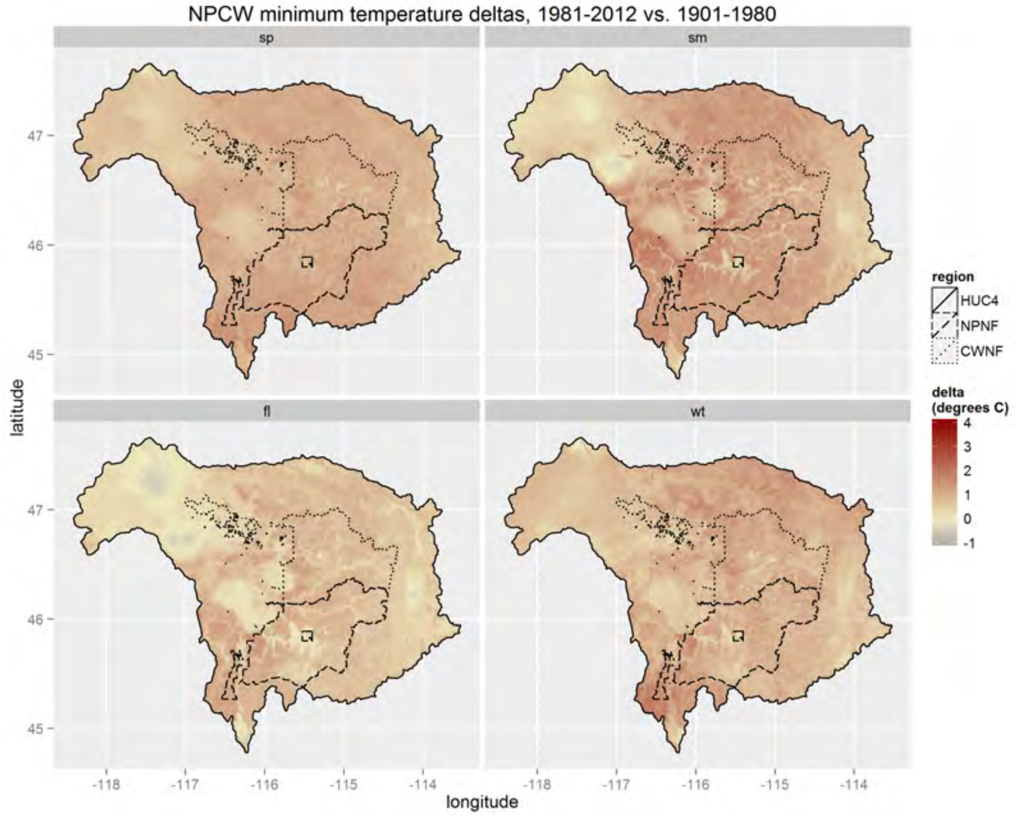


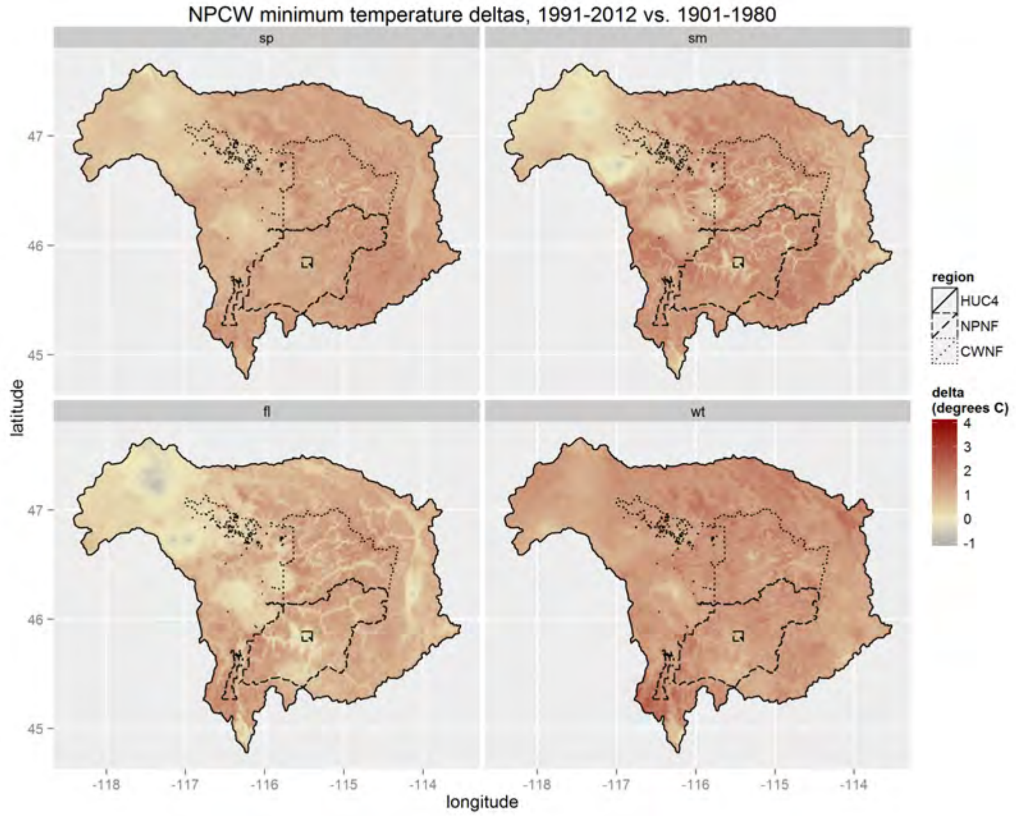
NPCW precipitation deltas, 1991-2012 vs. 1901-1980

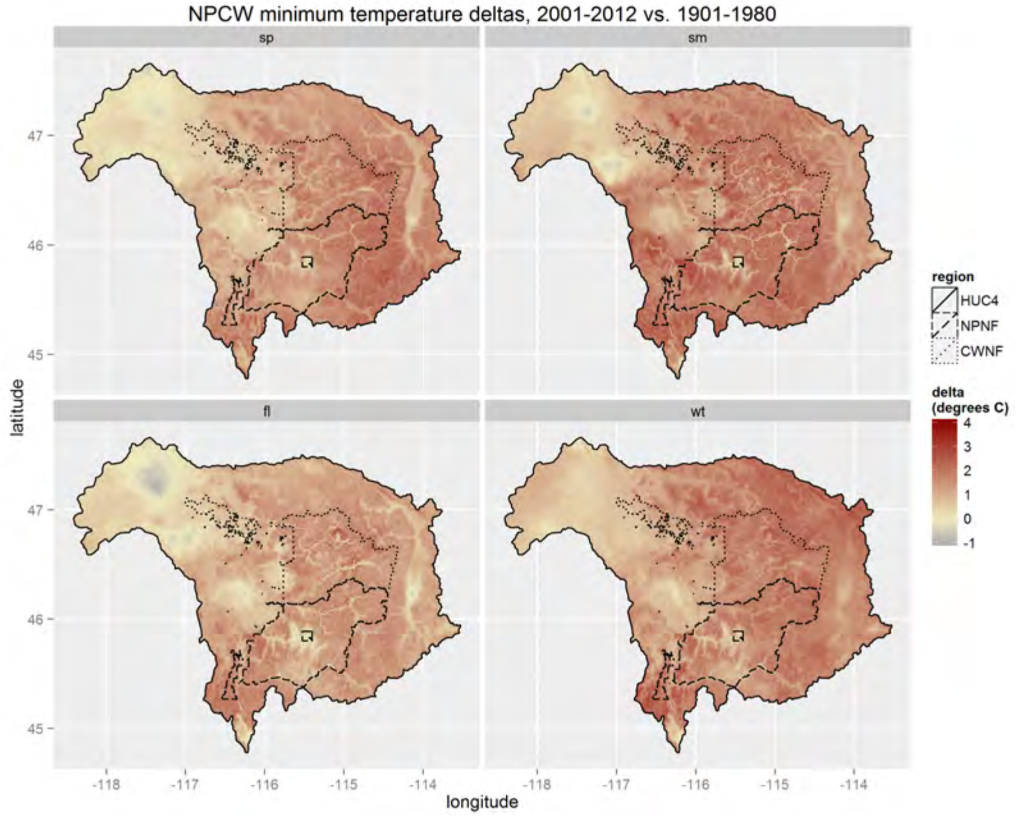


NPCW precipitation deltas, 2001-2012 vs. 1901-1980

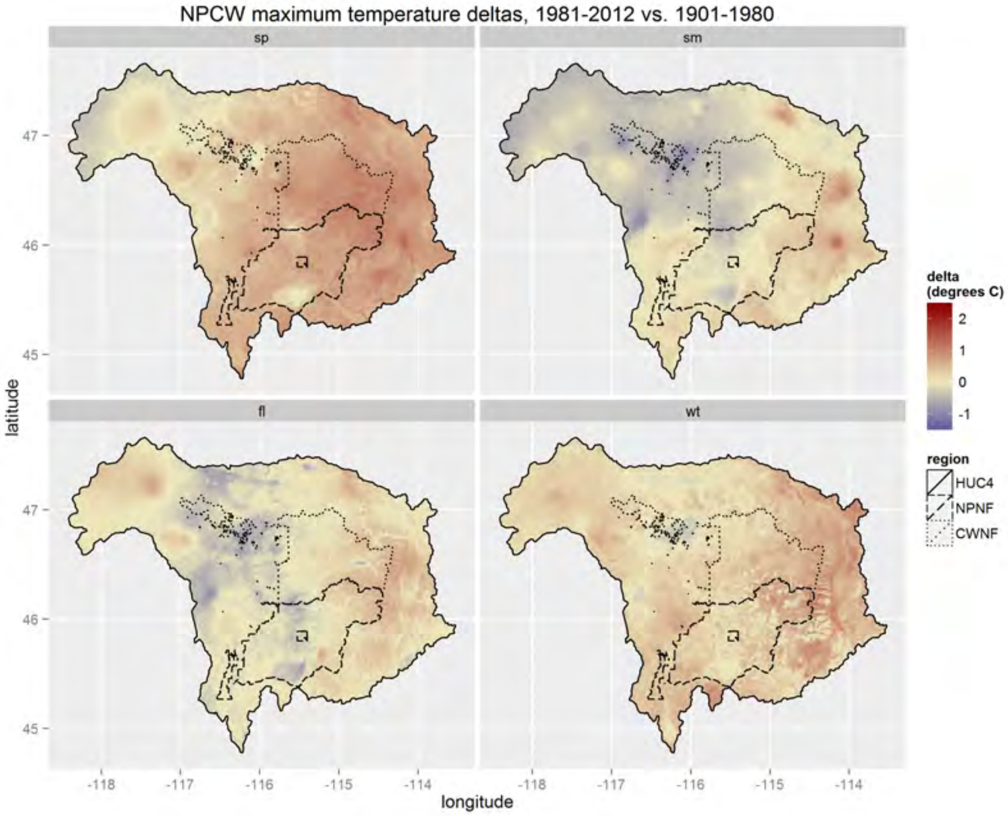
Current Minimum Temperature Deltas

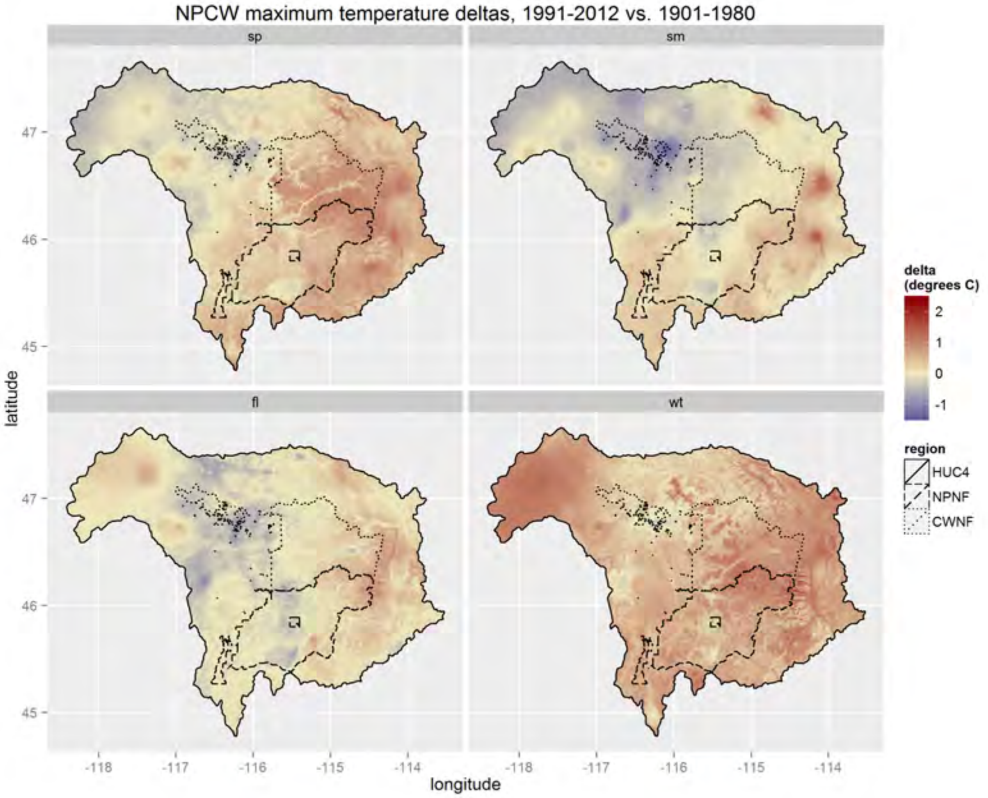


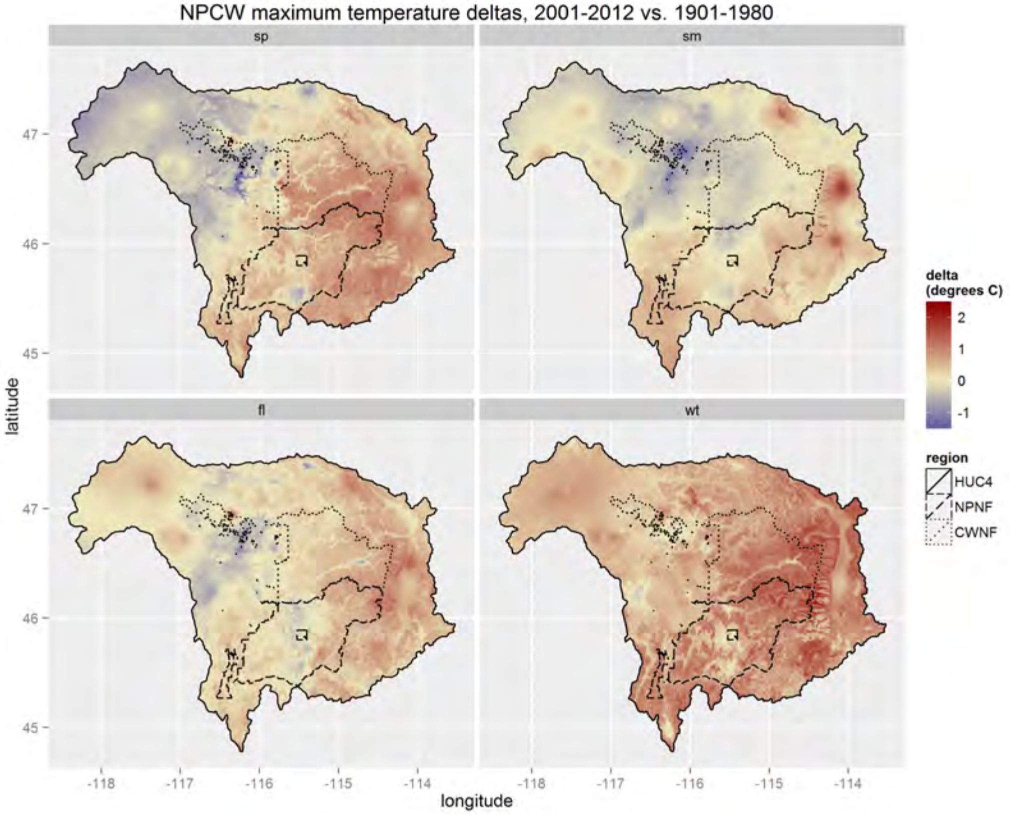



NPCW minimum temperature deltas, 2001-2012 vs. 1901-1980
sp
sm
fl
wt
latitude
longitude
47
46
45
-118
-117
-116
-115
-114
region
HUC4
NPNF
CWNF
delta (degrees C)
4
3
2
1
0
-1

Current Maximum Temperature Deltas

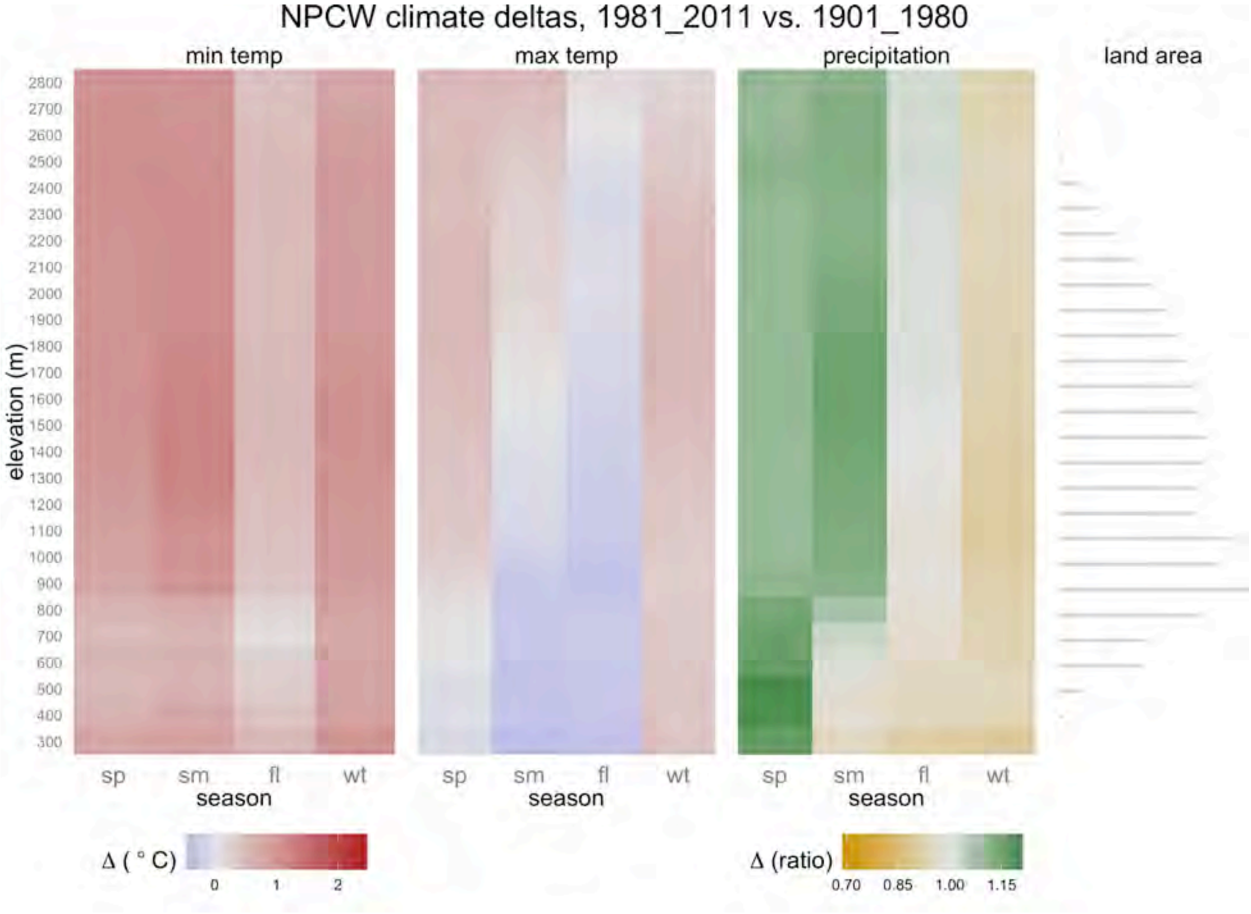


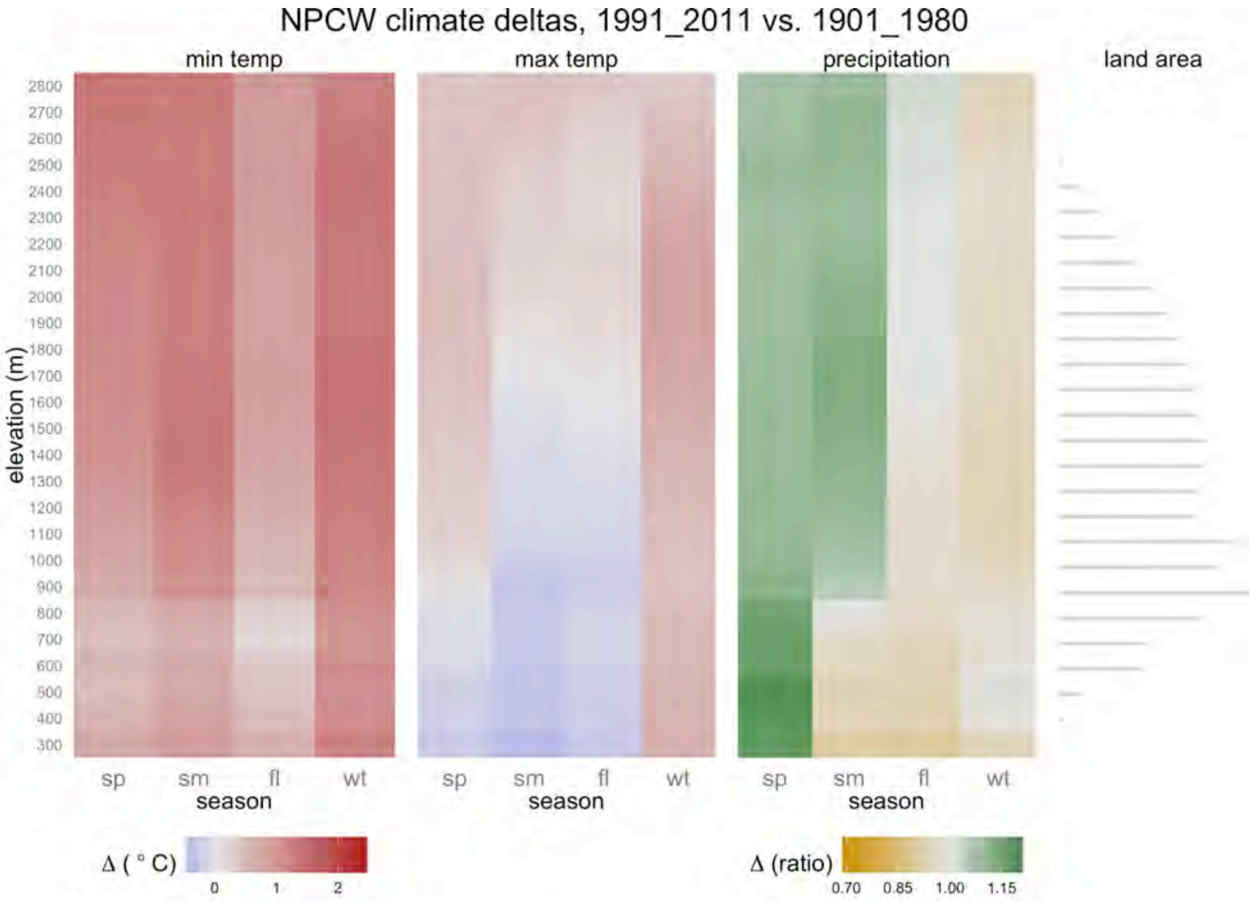

NPCW maximum temperature deltas, 1991-2012 vs. 1901-1980

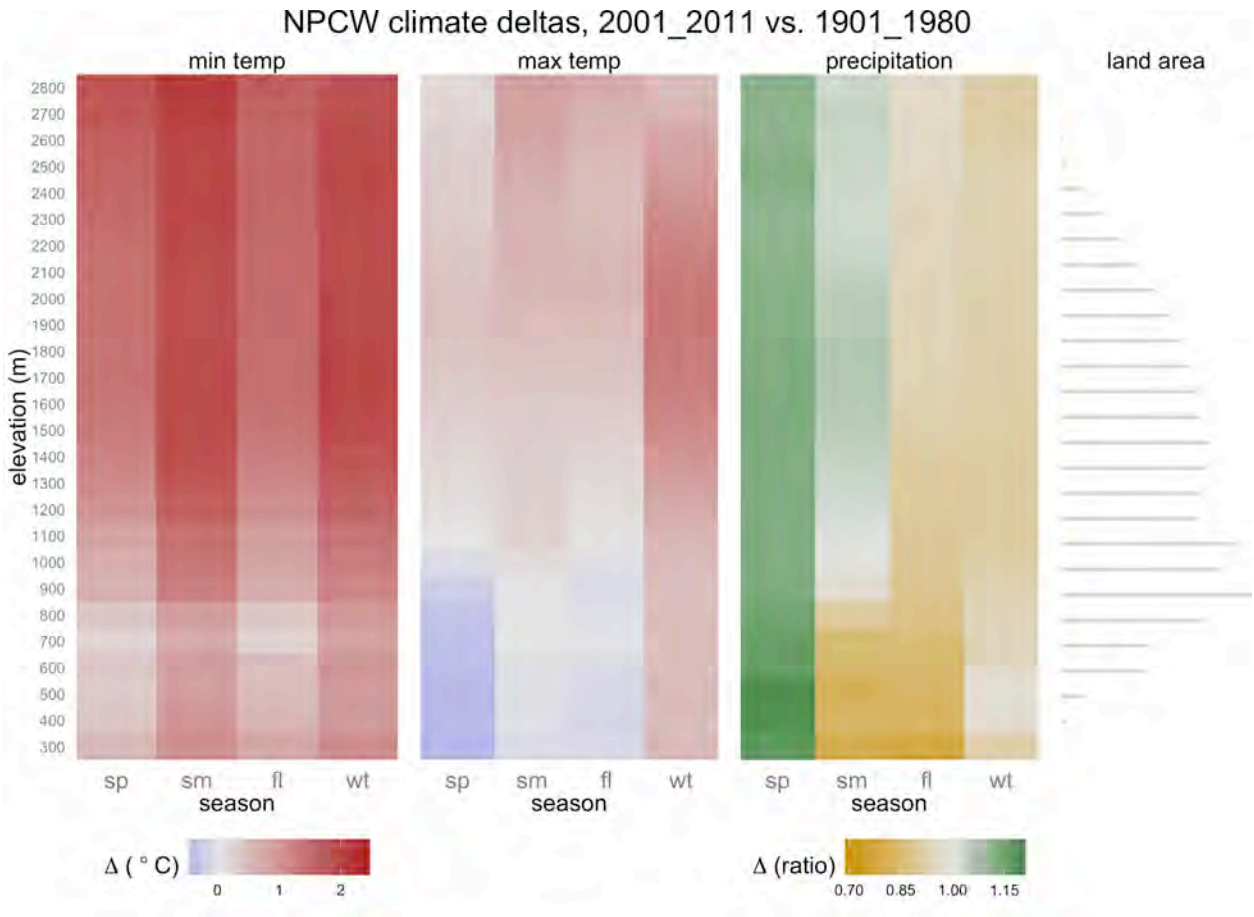

NPCW maximum temperature deltas, 2001-2012 vs. 1901-1980

Current Climate Deltas by Elevation

These charts summarize the same data as the above delta maps, but add the perspective of elevational distribution of change and the total area at that elevation experiencing the change. Temperature deltas are expressed as a difference (future – baseline) in degrees C, with 0 representing no change, positive values representing warming (red areas), and negative values representing cooling (blue areas). Precipitation deltas are expressed as a unitless ratio (future/baseline), with 1.0 representing no change, values greater than 1 representing increasing precipitation (green areas), and values less than 1 representing decreasing precipitation (brown areas). Each chart shows deltas for all three seasonal variables (minimum temperature, maximum temperature, precipitation) for a single recent timeslice vs. the 80-year baseline, aggregated by elevation (mean delta of all pixels within each elevation band) and land area (right side legend). Each column represents a season (labeled along the bottom: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7, 8), **fl** = fall (months 9, 10, 11), **wt** = winter (months 12, 1, 2). Temperature deltas are expressed as a difference in degrees (future – baseline) and precipitation is expressed as a ratio (future/baseline). The bars on the right-hand side show the relative amount of land area (within the greater NPCW HUC4 region) found at each elevation.



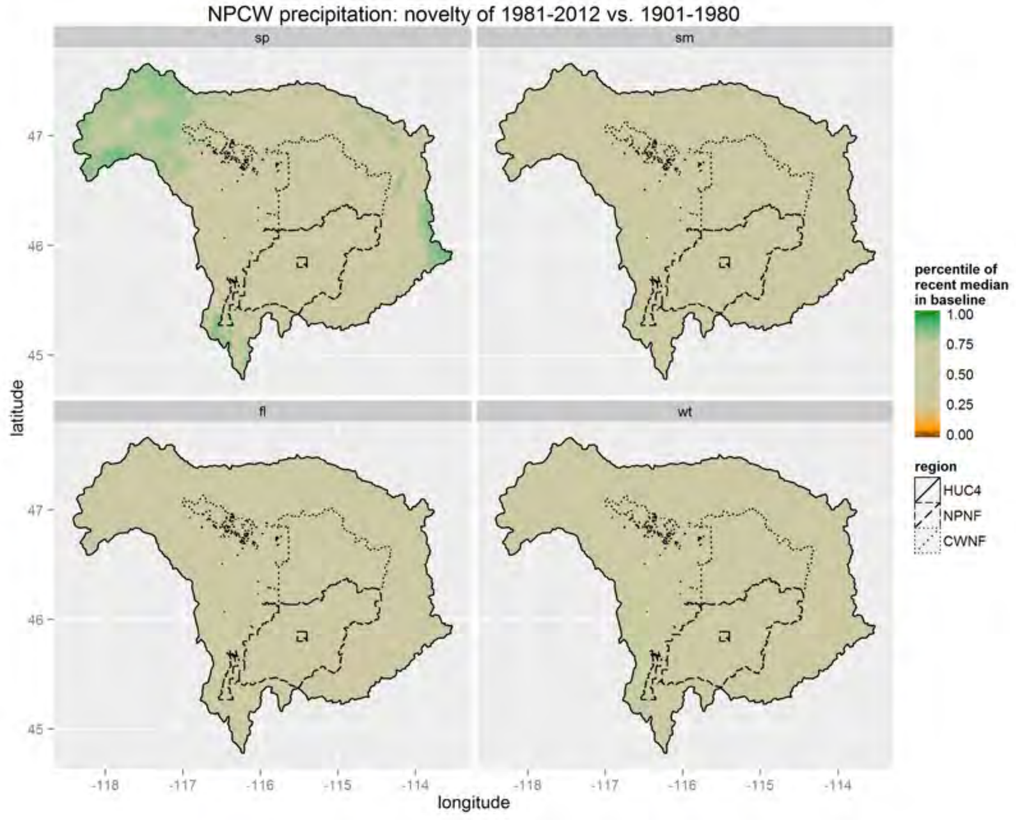

NPCW climate deltas, 1991_2011 vs. 1901_1980
min temp
max temp
precipitation
land area
elevation (m)
2800
2700
2600
2500
2400
2300
2200
2100
2000
1900
1800
1700
1600
1500
1400
1300
1200
1100
1000
900
800
700
600
500
400
300
sp
sm
fl
wt
season
sp
sm
fl
wt
season
sp
sm
fl
wt
season
Δ (° C)
0
1
2
Δ (ratio)
0.70
0.85
1.00
1.15

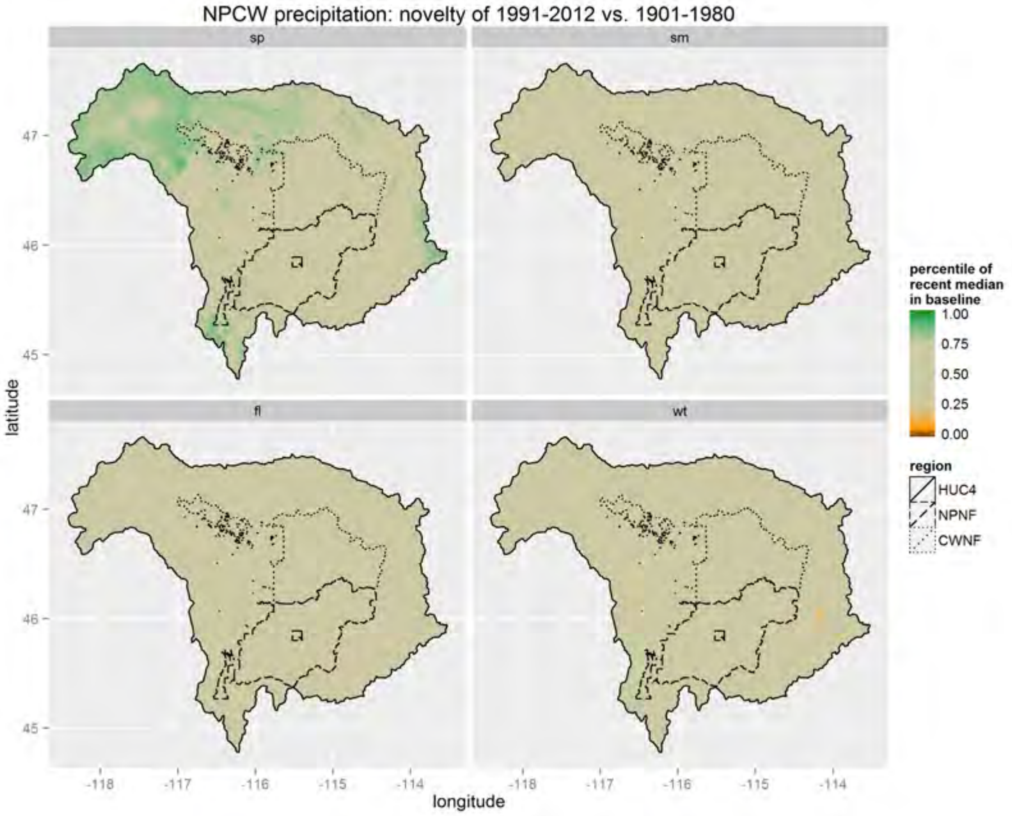

NPCW climate deltas, 2001_2011 vs. 1901_1980
min temp
max temp
precipitation
land area
elevation (m)
2800
2700
2600
2500
2400
2300
2200
2100
2000
1900
1800
1700
1600
1500
1400
1300
1200
1100
1000
900
800
700
600
500
400
300
sp
sm
fl
wt
season
Δ (° C)
0
1
2
Δ (ratio)
0.70
0.85
1.00
1.15

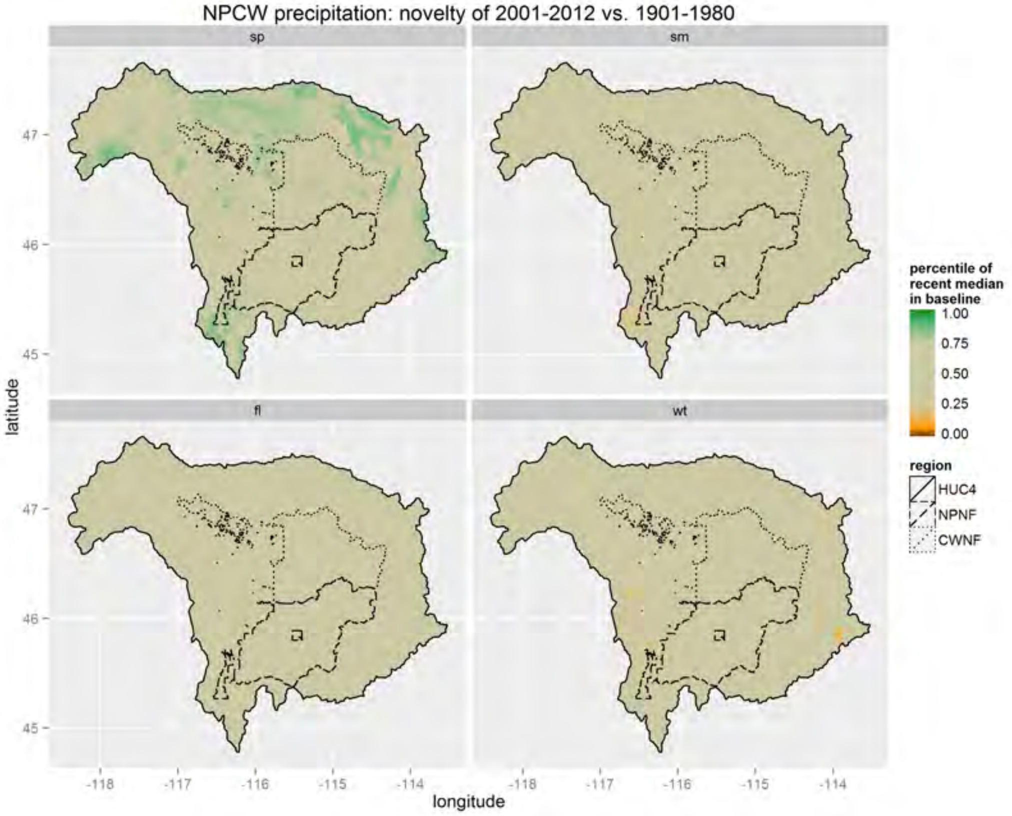
Current Climate Novelties

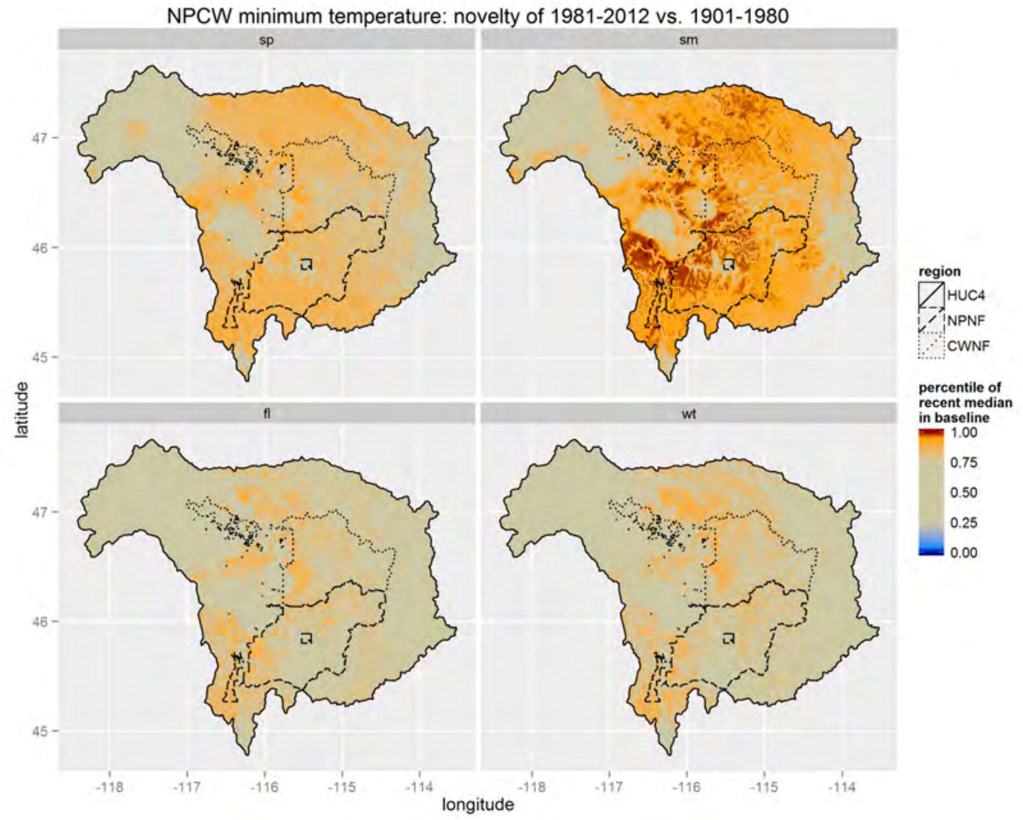
While the above delta maps depict the *magnitude* of recent climate change, additional perspective on the interpretation of these changes can be derived from comparison against the backdrop of historic year-to-year *variability*: a 2-degree C increase in average winter temperature might mean an unprecedented new climate for a location with low historic variability, while the same change could be historically common in an area with high climate variability. Map figures below show temperature and precipitation “novelties”, indicating where recent climate falls compared to the range of historic variability. Values range from 0 to 1, and the value per pixel represents the percentile where the median year in the recent timeslice falls within the 80-year baseline. Areas in grey/tan are within the interquartile range of historic variability. For precipitation, green indicates areas of increased precipitation and orange/red indicates areas of decreased precipitation, while for temperature, blue indicates areas of decreased temperature and orange/red indicates areas of increased temperature. A value of .85 would represent a pixel where half of recent years were warmer (or wetter) than 85% of baseline years. Values of 1 and 0 indicate places where at least half of recent years are completely outside the range of historic variability. The top of the map images are labeled by season: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7 & 8), **fl** = fall (months 9, 10, 11), **wt** = winter (12, 1, 2).

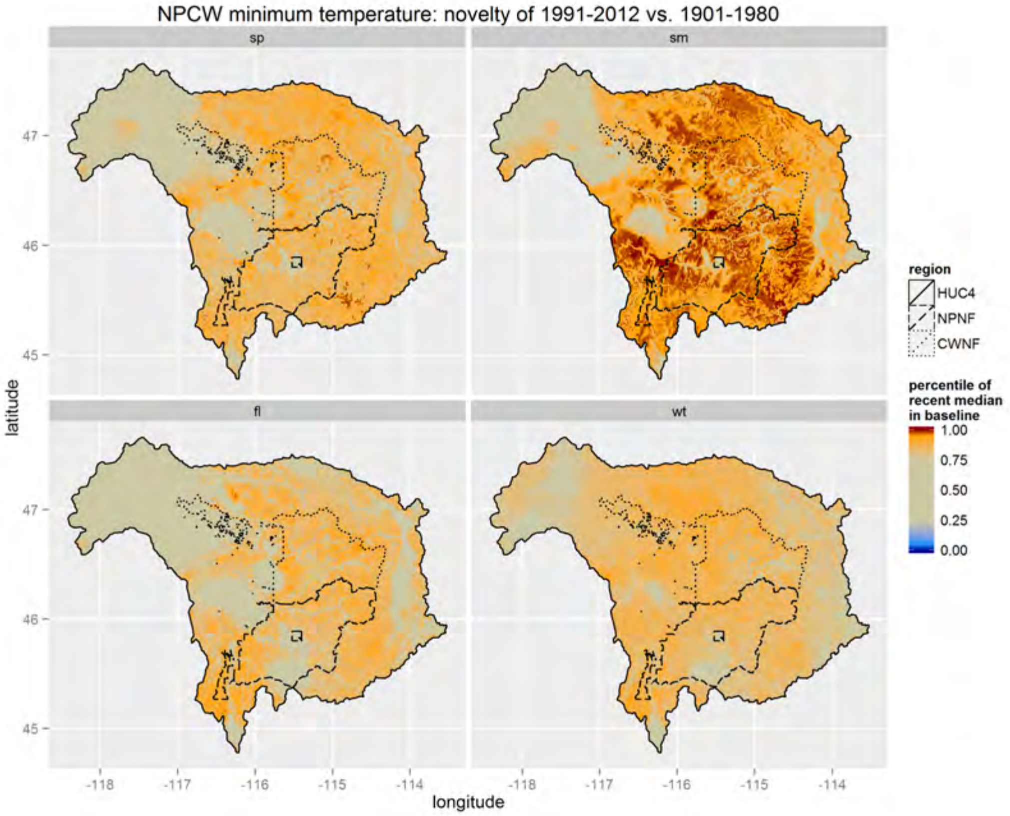
Current Precipitation Novelties



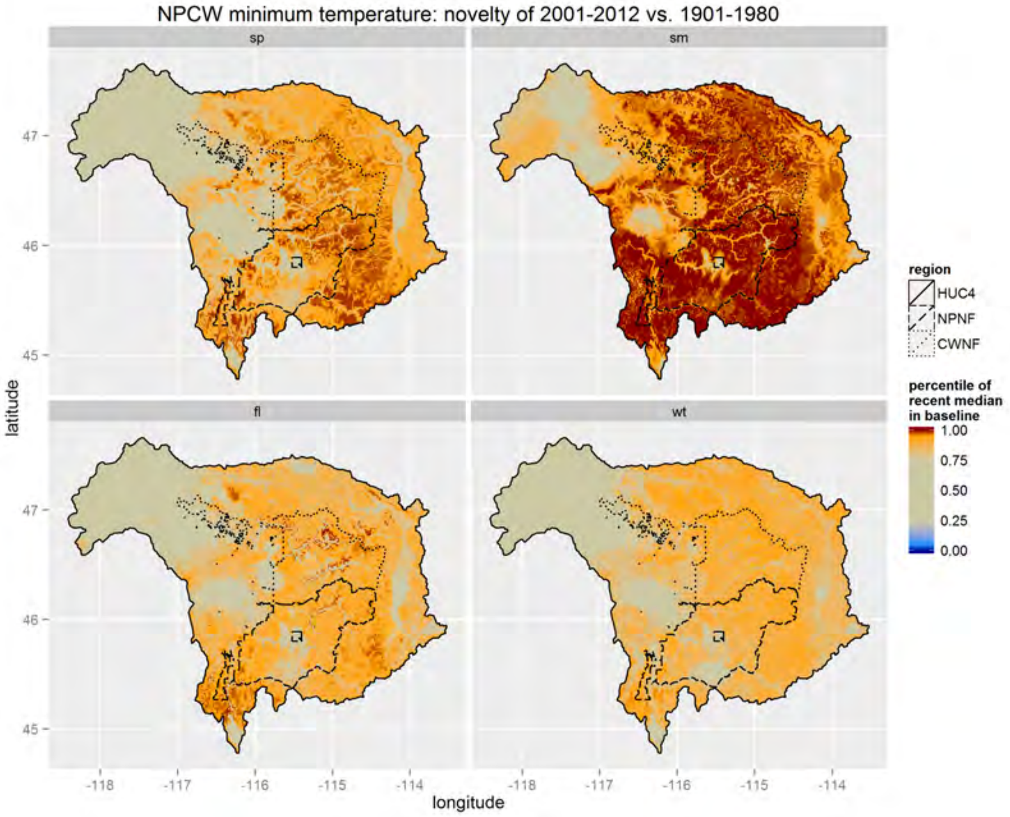

NPCW precipitation: novelty of 1991-2012 vs. 1901-1980
sp
sm
fl
wt
latitude
47
46
45
longitude
-118
-117
-116
-115
-114
percentile of recent median in baseline
1.00
0.75
0.50
0.25
0.00
region
HUC4
NPNF
CWNF


NPCW precipitation: novelty of 2001-2012 vs. 1901-1980
sp
sm
fl
wt
latitude
47
46
45
longitude
-118
-117
-116
-115
-114
percentile of recent median in baseline
1.00
0.75
0.50
0.25
0.00
region
HUC4
NPNF
CWNF


NPCW minimum temperature: novelty of 1981-2012 vs. 1901-1980
sp
sm
fl
wt
latitude
47
46
45
longitude
-118
-117
-116
-115
-114
region
HUC4
NPNF
CWNF
percentile of recent median in baseline
1.00
0.75
0.50
0.25
0.00

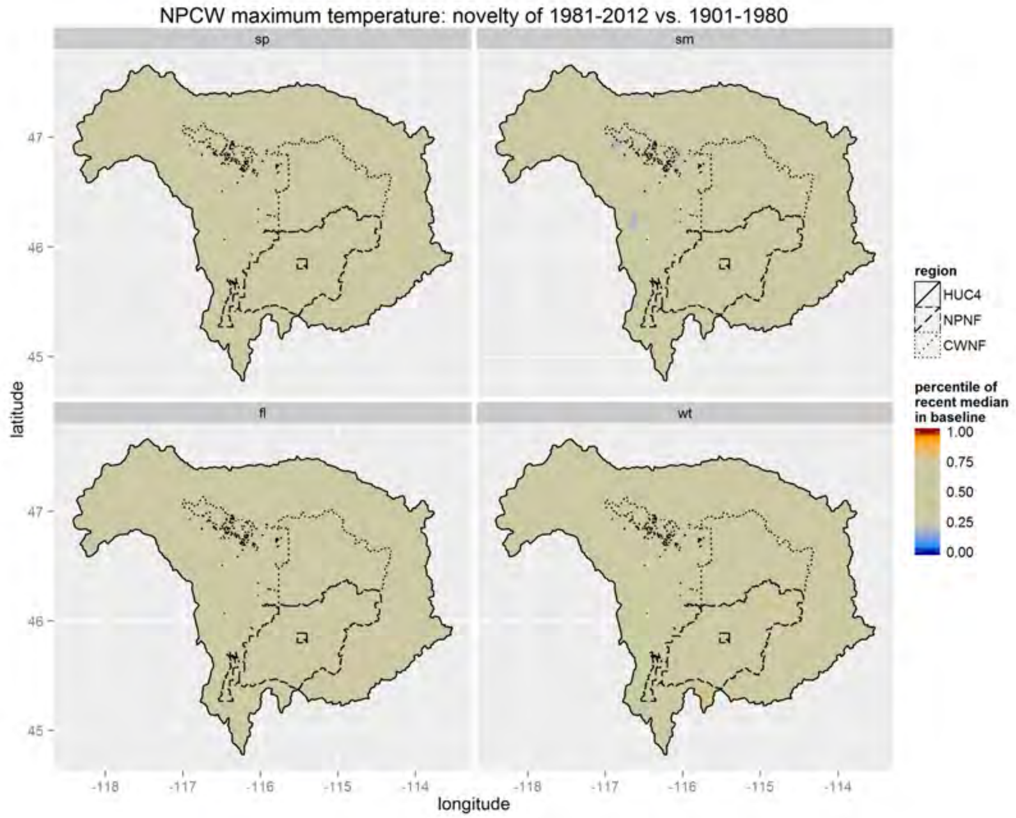


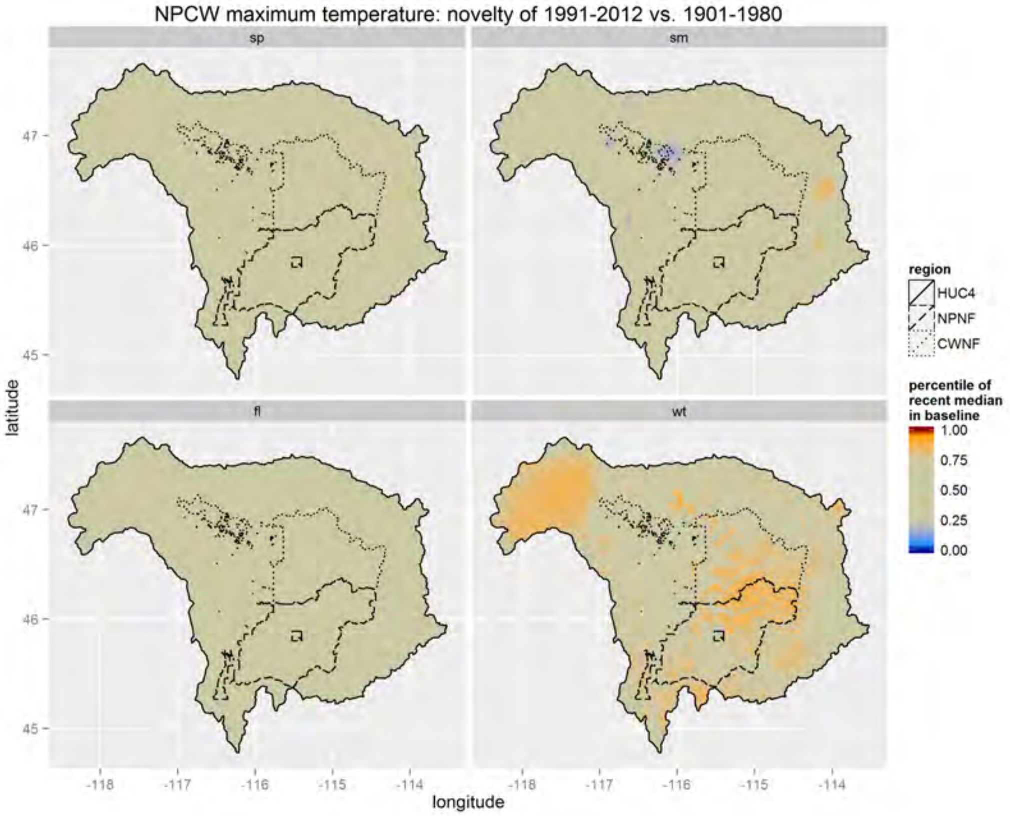
NPCW minimum temperature: novelty of 1991-2012 vs. 1901-1980

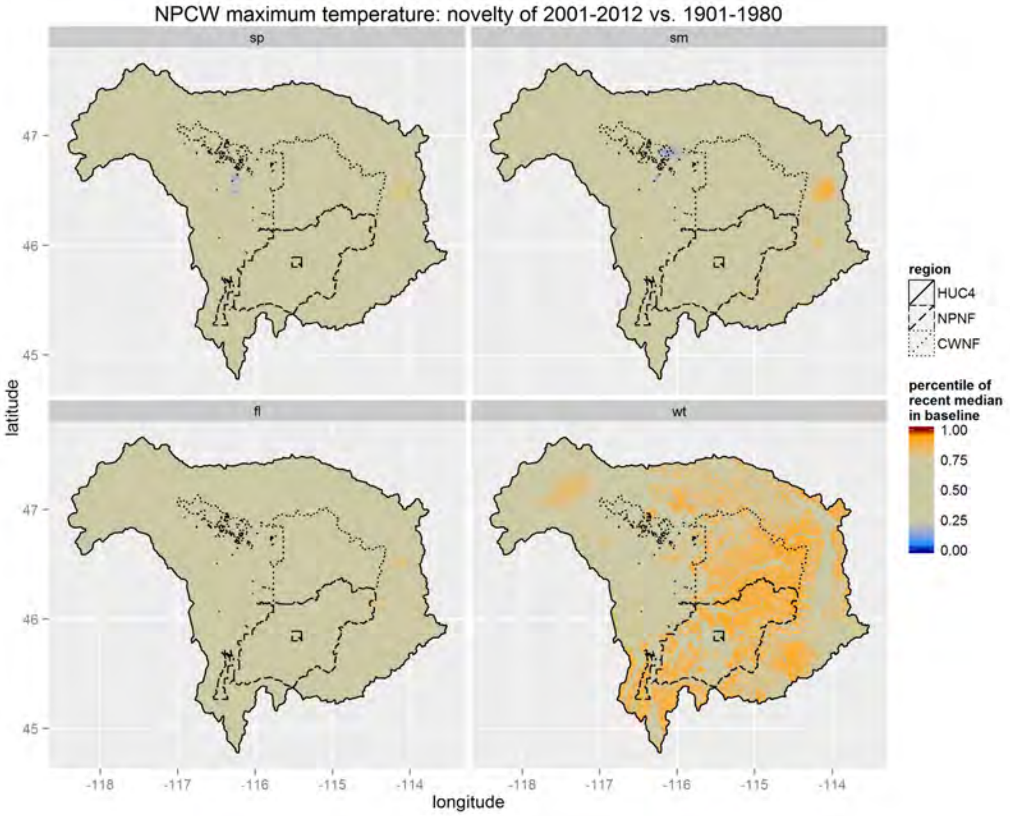


NPCW minimum temperature: novelty of 2001-2012 vs. 1901-1980

Current Maximum Temperature Novelties




NPCW maximum temperature: novelty of 1991-2012 vs. 1901-1980


NPCW maximum temperature: novelty of 2001-2012 vs. 1901-1980

Summary of Recent Climate Trends

On average, climate across the NPCW ecoregion has been notably warmer but not notably different in precipitation during recent years compared to the 1901-1980 baseline period, as indicated by the progressively rightward-shifting climate space clouds in the scatter plots – but there is considerable seasonal and geographic deviation from this average. Below we summarize the results across various dimensions.

Climate Variables

Precipitation has decreased throughout the region in winter, and increased in spring (these two seasons together account for a sizeable majority of total annual precipitation, with winter the wettest overall). While differing in direction, precipitation deltas in these two seasons do share an elevational gradient, with the lowest-elevation areas experiencing larger increases (spring) or smaller decreases (winter) than higher-elevation locations (see Current Deltas by Elevation). The novelty maps (see Current Climate Novelties) show that spring is the only season in which recent precipitation patterns lie outside the interquartile range of historic variability – for many areas across the ecoregion, median recent spring precipitation levels have been higher than 80% to 90% of historic years; however, these areas of abnormally high recent spring precipitation are mainly surrounding the NPCW forests rather than within the forest boundaries.

Precipitation change in the summer (the driest of seasons overall) and fall has been more geographically variable. Looking at the entire 30-year recent period, summer precipitation has increased noticeably compared to the baseline in all but the lowest-elevation areas in the extreme southwestern and western corners of the watershed, which have dried slightly (see Current Climate Deltas and Current Climate Novelties); fall precipitation has changed less than any other season, and has increased in some areas while decreasing in others. Fall and summer share a relationship between elevation and precipitation change that is opposite that of the other seasons: higher elevations have experienced larger increases or smaller decreases than lower elevations.

Minimum temperature has increased in all seasons and recent timeslices across the NPCW ecoregion, with deltas as high as 4 degrees C in some areas (see Current Climate Deltas). Increases in minimum temperatures have been more pronounced at higher elevations (above 800m or 2,600ft), and are most pronounced in the summer and winter (see Current Climate Deltas by Elevation). The northwest corner of the watershed has been more resistant to the predominant increases in minimum temperature than has most of the region, with a few localities even experiencing modest decreases across most of the year. Minimum temperature novelties (see Current Climate Novelties) show that this variable has deviated more from the range of historic variability than either of the other two – this departure is strongest and most widespread in the summer, when median minimum temperatures across a large portion of the region have exceeded the highest value observed during the baseline period.

Recent change in maximum temperature has been more seasonally variable and less extreme than change in minimum temperature. The one season in which *increases* in recent maximum temperature have clearly predominated is winter, with the largest increases concentrated in mid-to-high elevation areas in the western half of the watershed (see Current Climate Deltas). Spring maximum temperature changes largely mirror those of winter, except for some pockets of cooling in low-elevation areas in the northwestern corner of the watershed (see Current Climate Deltas). In the summer and fall, decreased maximum temperature has been the norm over even greater portions of the low-elevation areas in the northern and western parts of the ecoregion, while maximum temperatures have increased in the higher western and southern parts of the watershed. It is therefore unsurprising that novelty maps (see Current Climate Novelties) show maximum temperatures have remained largely inside the interquartile range of baseline variability (exceptions being some higher-elevation areas in the winter and fall, in which half of the most recent years saw maximum temperatures warmer than 80% to 90% of historic years.

"Thermal continentality" is a concept that measures the degree to which a location has an inland (as opposed to coastal) climate. Although it was not mathematically calculated for this study, it is based on the idea that coastal climates have smaller daily or annual temperature ranges than inland climates – and in both these respects, the data suggest that the continentality of the NPCW region has decreased in recent years. Minimum temperatures have increased more quickly than maximum temperatures, and winter temperatures somewhat more quickly than summer temperatures, leading to decreases in the amplitude of temperature fluctuations on both the daily and annual timescales.

Seasonal Patterns

Region-wide, no single season has consistently outpaced the others in its degree of climate change across variables and analyses, particularly when the directionality of change is disregarded. However, it is possible to make certain generalizations about the seasonality of observed recent climate change. Winter has become warmer and drier, while spring has become warmer and wetter. Recent summers were also warmer and wetter than in historic years, but with considerably more variability than spring. Fall climate has remained much more stable in recent years compared to other seasons, with increased minimum temperatures but decreased maximum temperatures in many areas, and relatively unchanged precipitation. It is important to remember that there are localized exceptions to many of these generalizations.

Geographic Patterns

While this region is relatively small and homogenous in a global sense, recent climate change has nevertheless unfolded differently in different parts of the NPCW ecoregion. The higher-elevation, western portions of the watershed have experienced the greatest temperature increases (including clear increases in both minimum and maximum temperatures). Northwestern, lower-elevation parts of the region have seen increased minimum, but decreased maximum temperature, and have experienced a greater increase in precipitation than the rest of the region.

Recent Timeslices

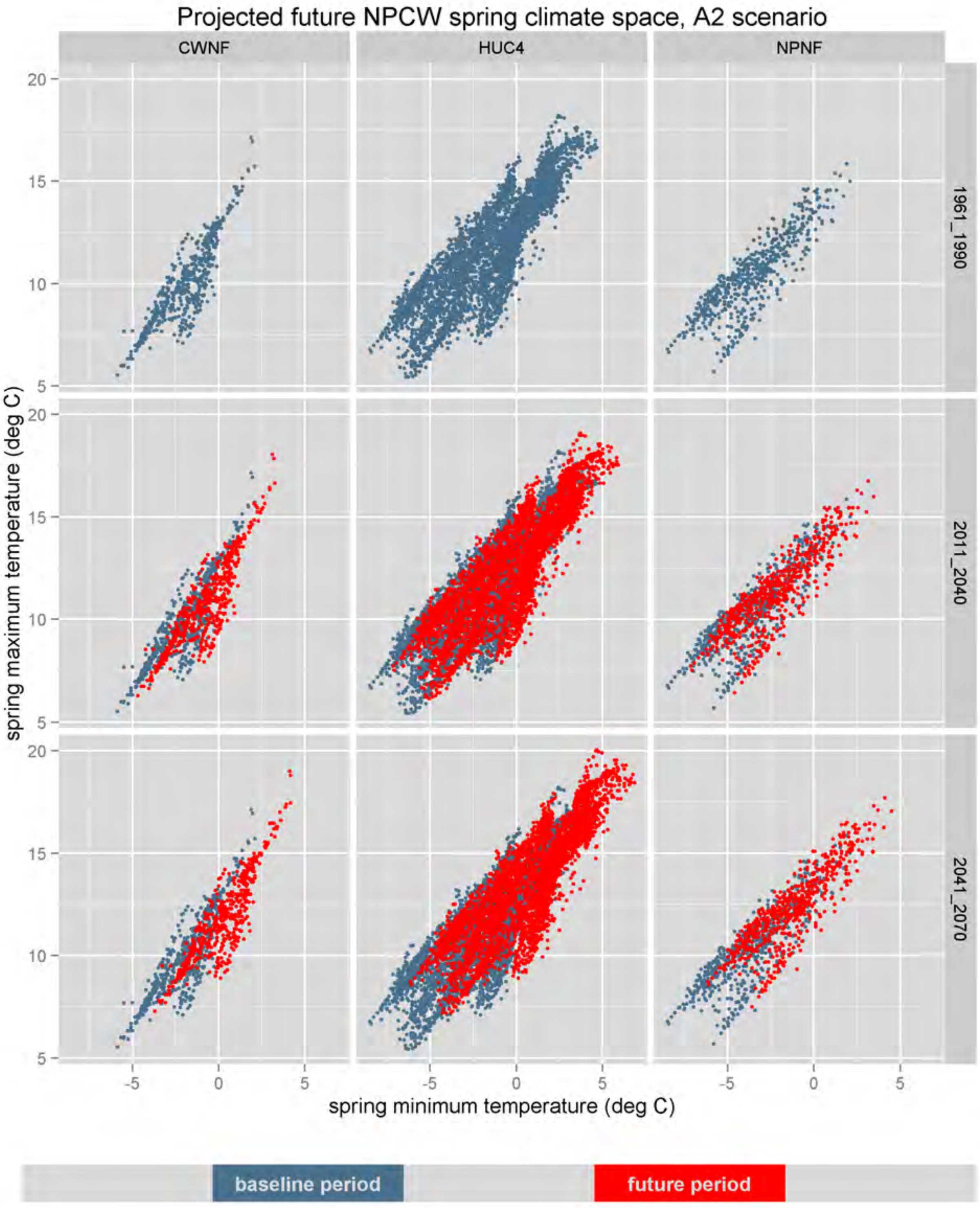
Results were presented for each of the most recent 30-, 20-, and 10-year nested time periods. While results from periods shorter than 30 years must be interpreted cautiously, as they are influenced by climate oscillations like ENSO and PDO in addition to more fundamental climate trends, comparing these three timeslices is still instructive. Across most of the seasons and geographies discussed above, the 30-, 20- and 10-year periods are progressively warmer and drier. This suggests that, *separate* from the longer-term climate changes observed between the historic baseline and recent timeslices discussed above, the region’s climate became warmer and drier overall between 1981 and 2012.

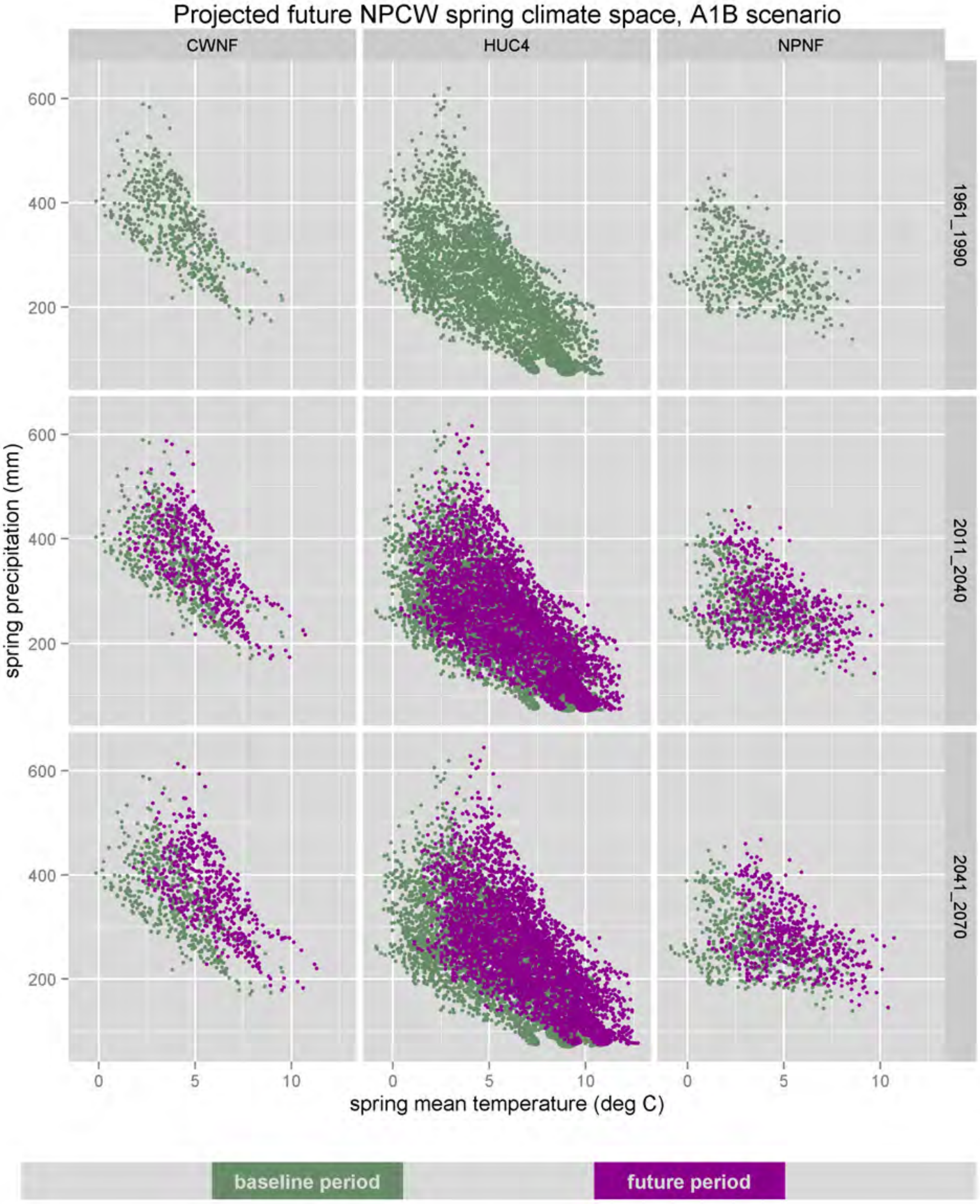
Maps of Future Trends in Climate Space – CWNA 4km

Projected Future Climate Space Scatter Plots

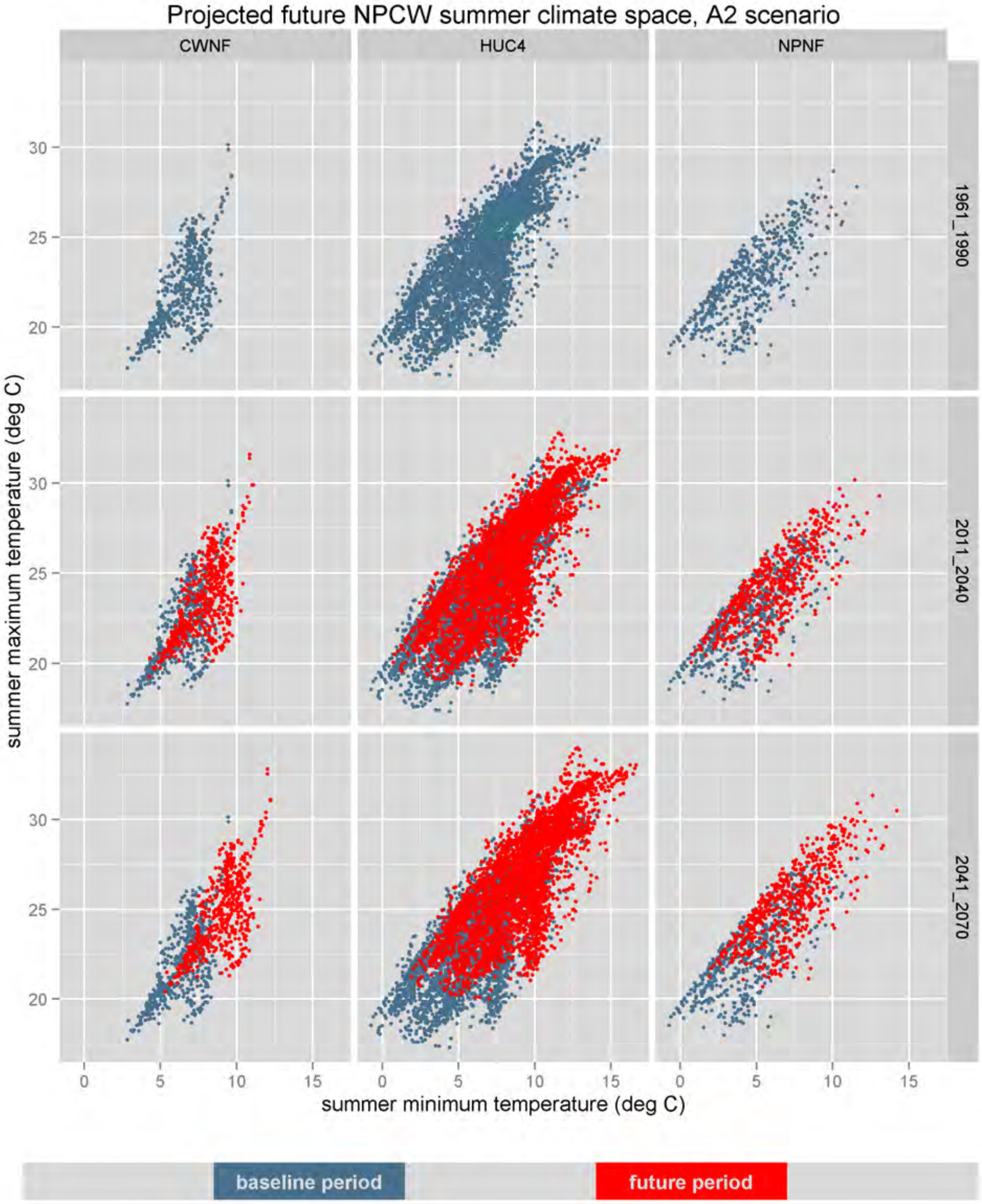
These climate space scatter plots show shifts in raw climate values for each pixel, for each season, for the NPCW forests as well as the larger HUC4 region. Each point on the climate space chart represents a single 4km pixel in the geographic space of the analysis area, and the offset between baseline and future points indicates the magnitude of future climate change that is projected to occur. For each season there are two charts: 1) precipitation vs. mean temperature for A1B emissions scenario, and 2) maximum temperature vs. minimum temperature for A2 emissions scenario. Labels for the three timeslices are shown on the right-hand side of the image: baseline (1961-1990), 2020s (2011-2040), and 2050s (2041-2070). Across the top of the figure is the region (CWNF = Clearwater National Forest; HUC4 = regional boundary; NPNF = Nez Perce National Forest).

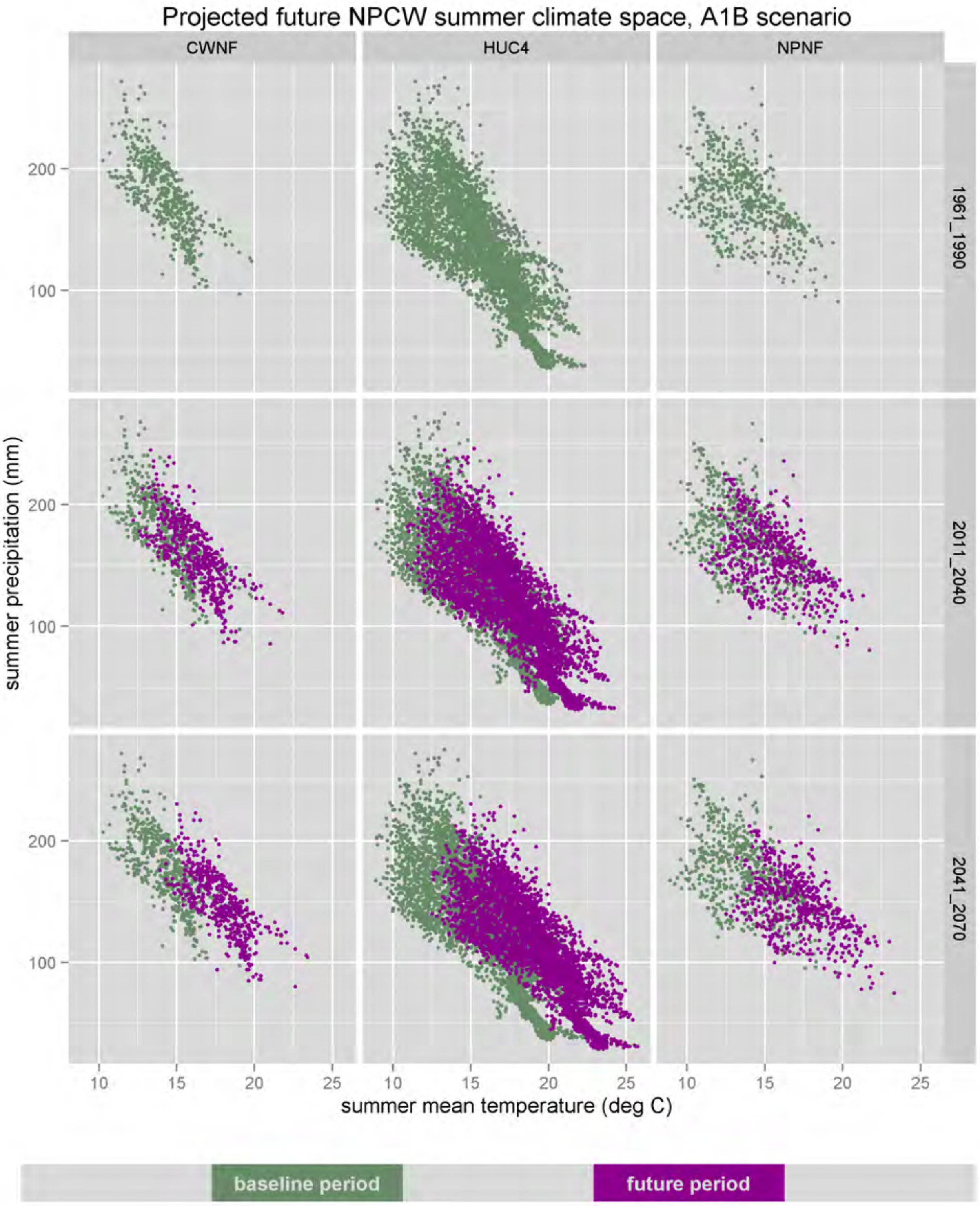
Projected Future Spring Climate Space



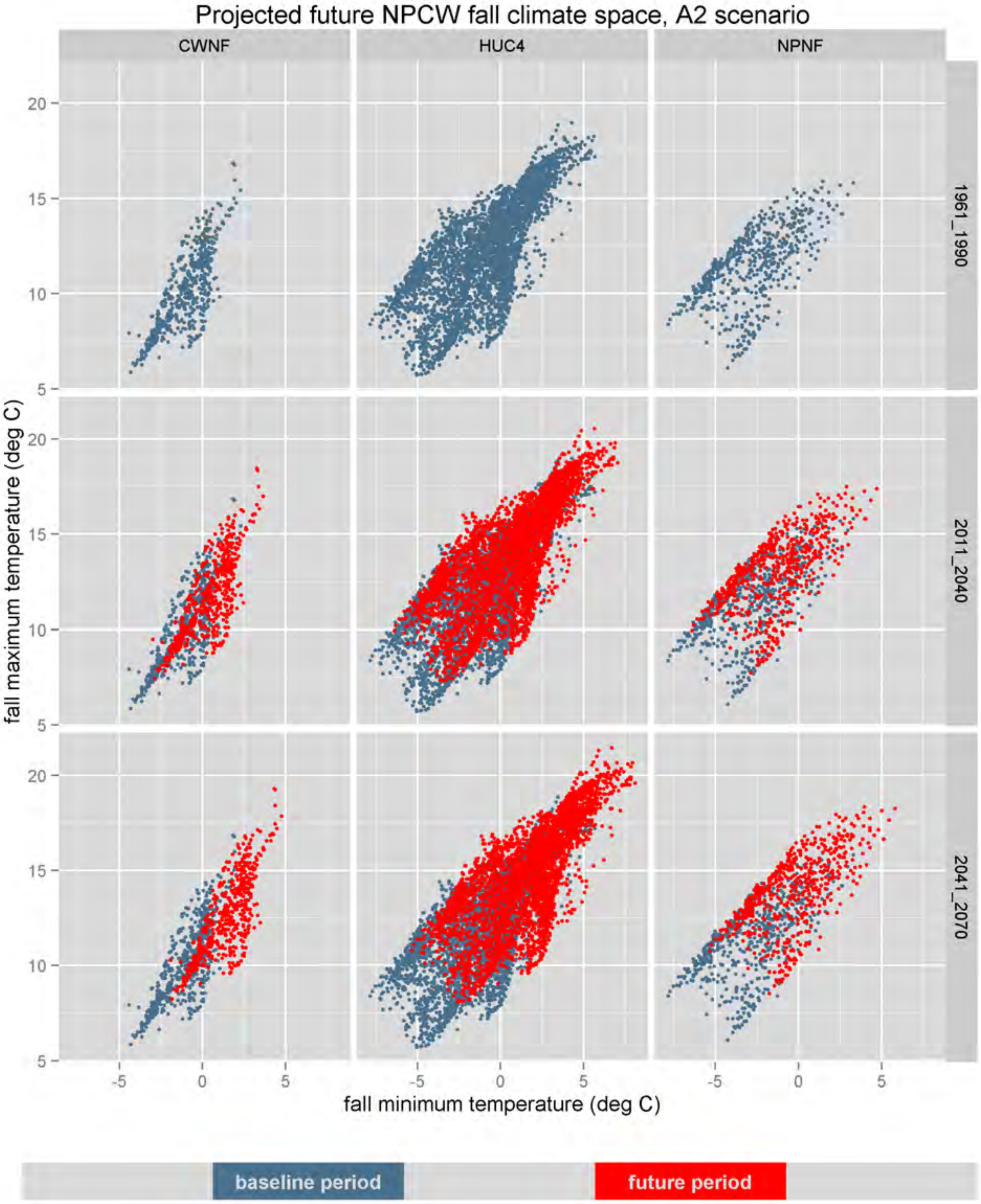

Projected future NPCW spring climate space, A1B scenario
CWNF
HUC4
NPNF
1961_1990
2011_2040
2041_2070
spring precipitation (mm)
200
400
600
spring mean temperature (deg C)
0
5
10
baseline period
future period

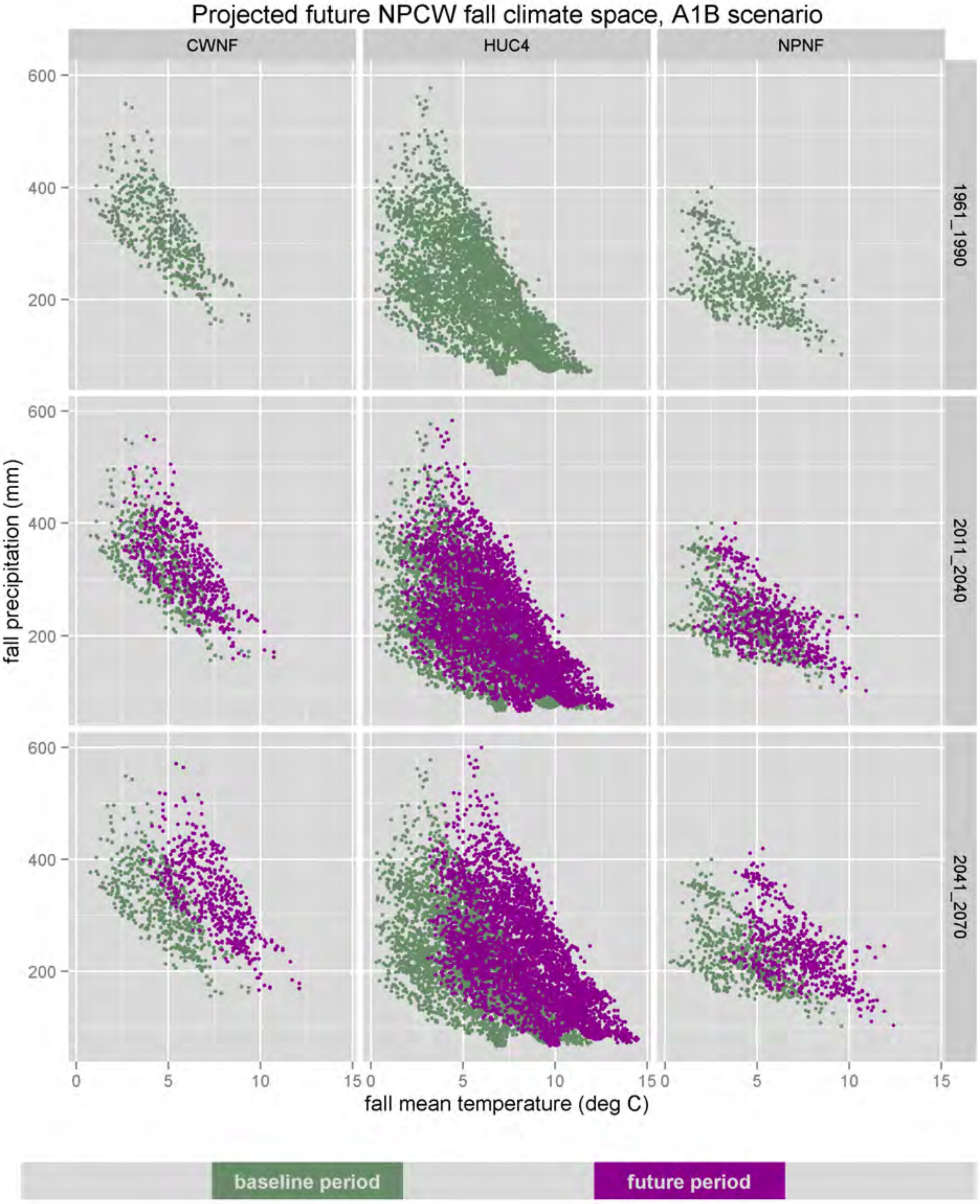
Projected Future Summer Climate Space



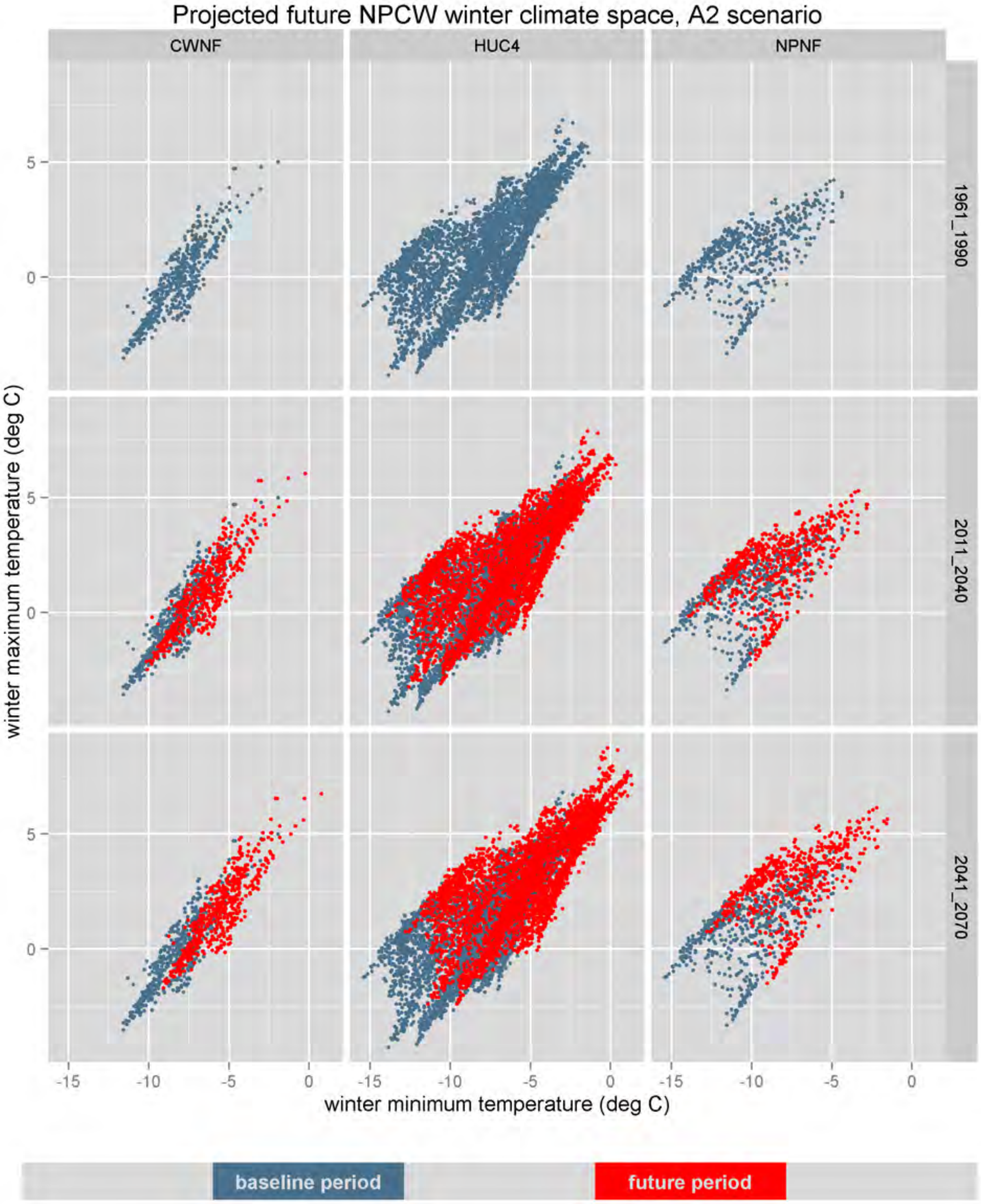

Projected future NPCW summer climate space, A1B scenario
CWNF
HUC4
NPNF
1961_1990
2011_2040
2041_2070
summer precipitation (mm)
200
100
200
100
200
100
10
15
20
25
10
15
20
25
10
15
20
25
summer mean temperature (deg C)
baseline period
future period

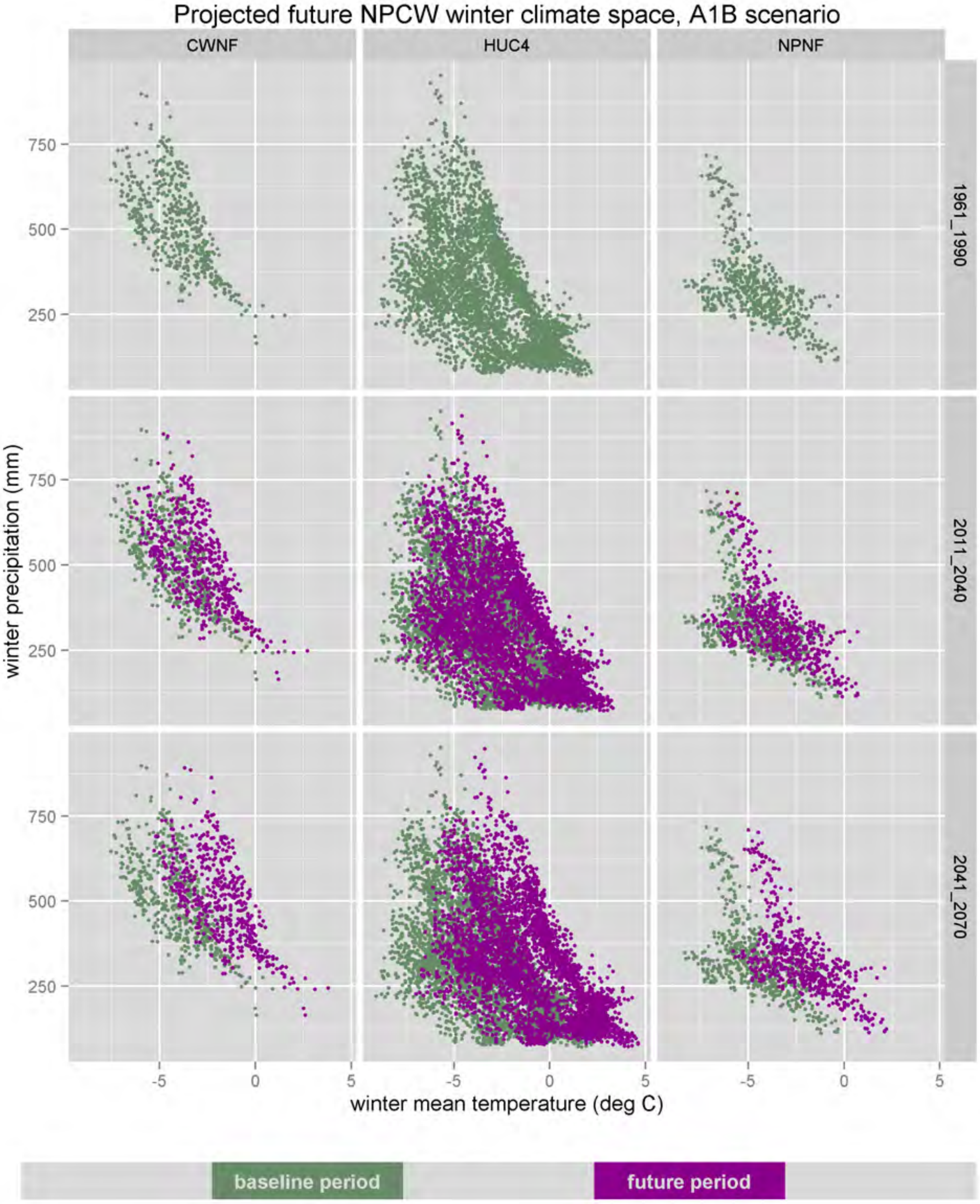
Projected Future Fall Climate Space




Projected future NPCW fall climate space, A1B scenario
CWNF
HUC4
NPNF
1961_1990
2011_2040
2041_2070
fall precipitation (mm)
600
400
200
fall mean temperature (deg C)
0
5
10
15
baseline period
future period

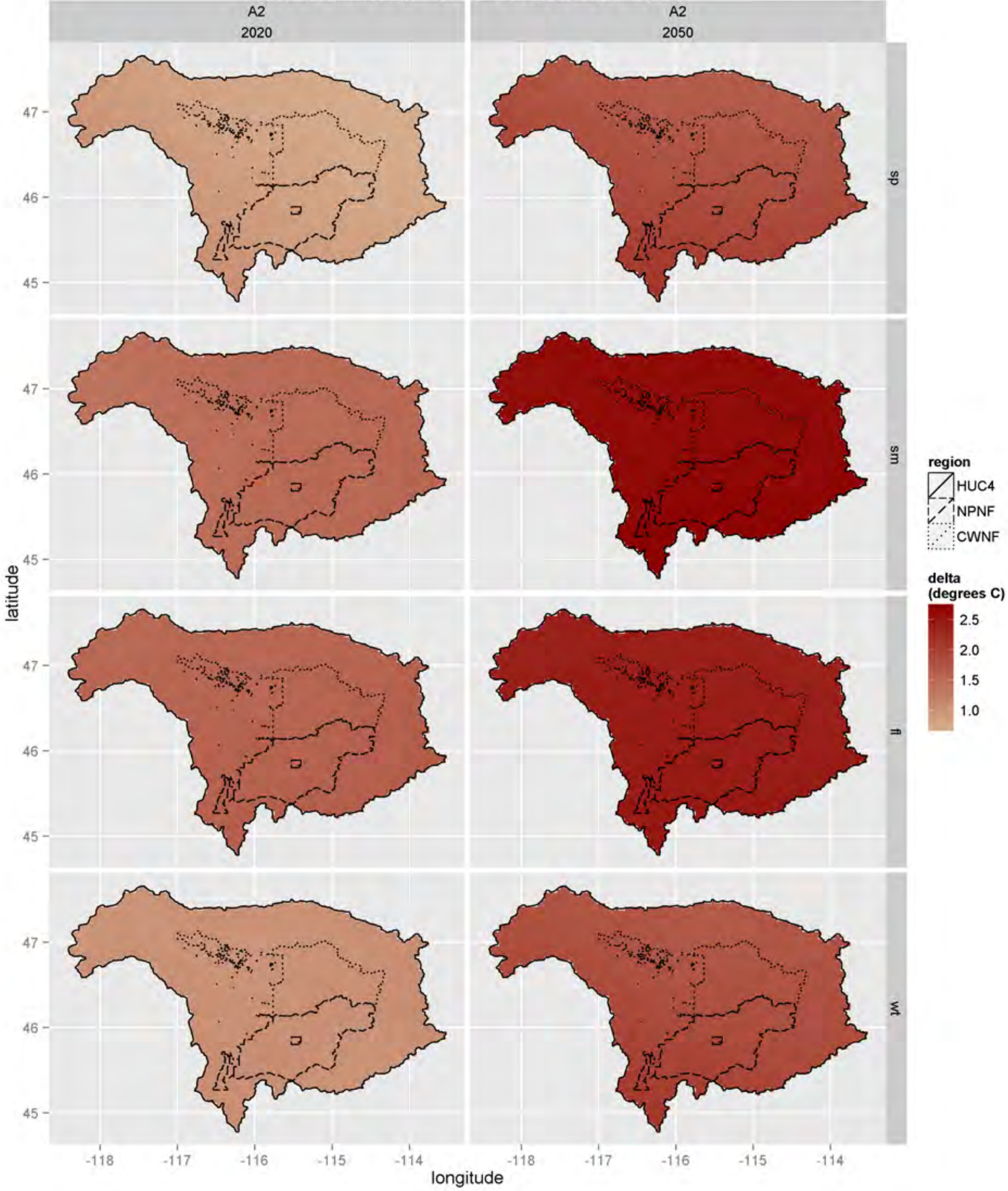
Projected Future Winter Climate Space

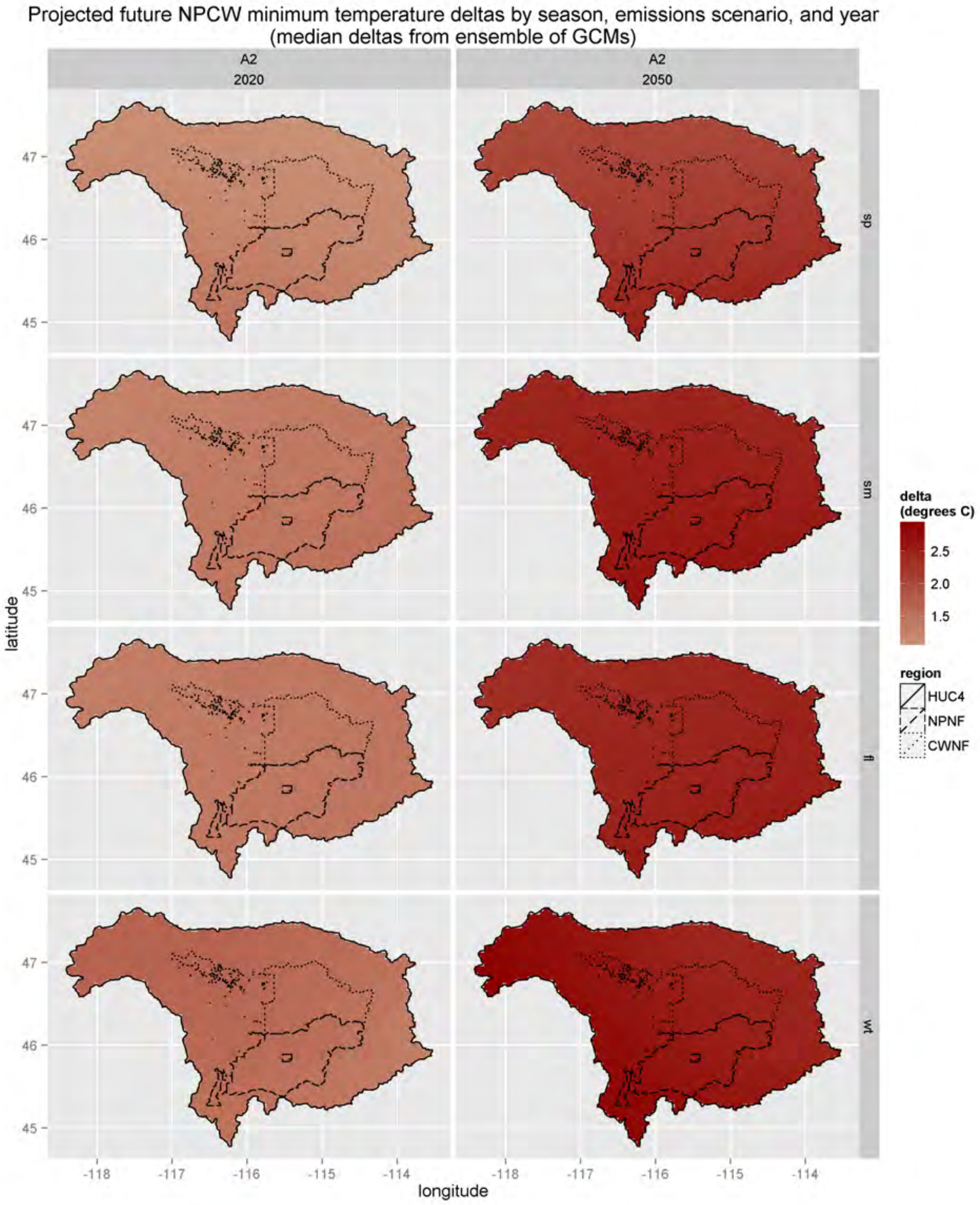


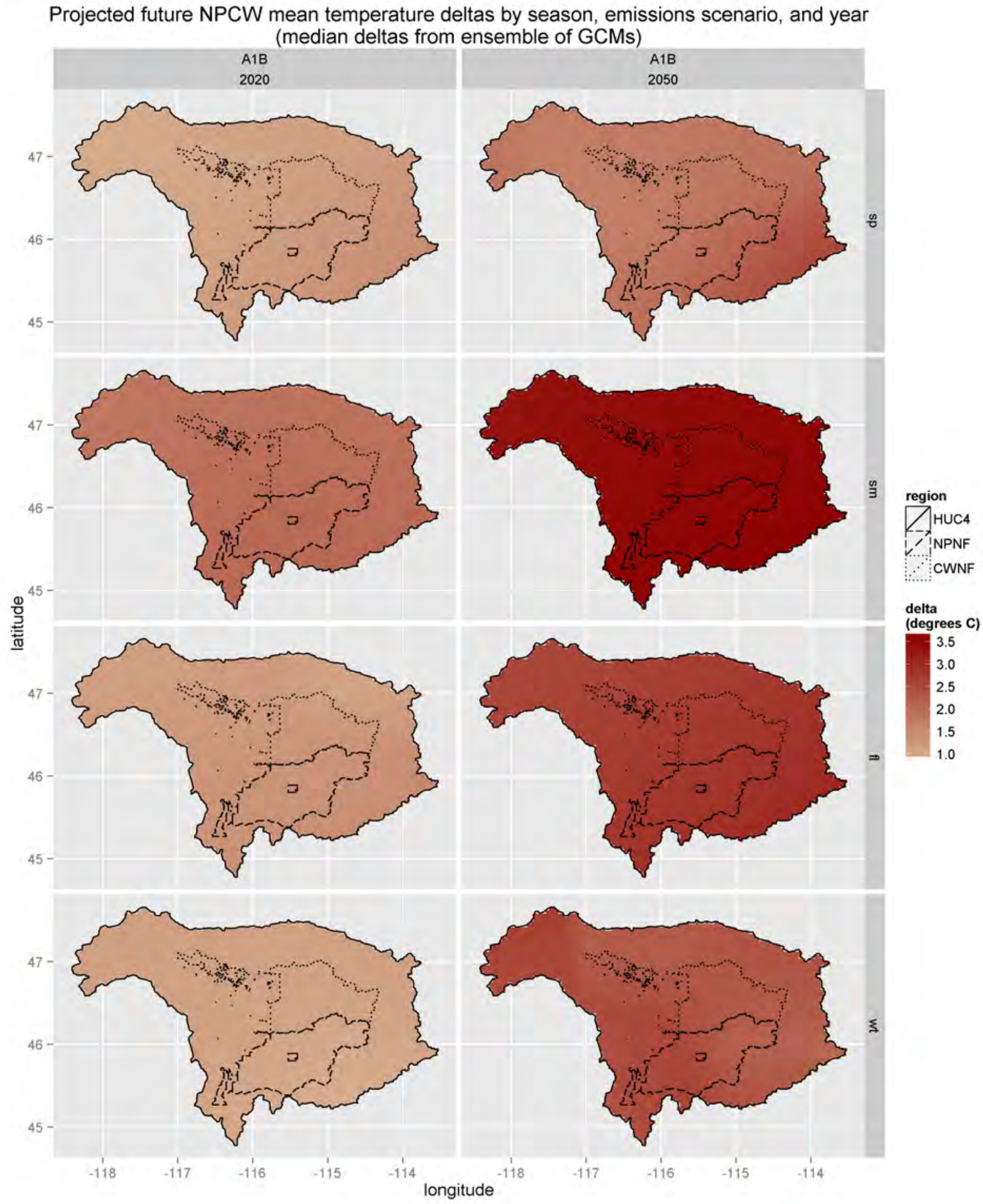

Projected future NPCW winter climate space, A1B scenario
CWNF
HUC4
NPNF
1961_1990
2011_2040
2041_2070
winter precipitation (mm)
750
500
250
winter mean temperature (deg C)
-5
0
5
baseline period
future period

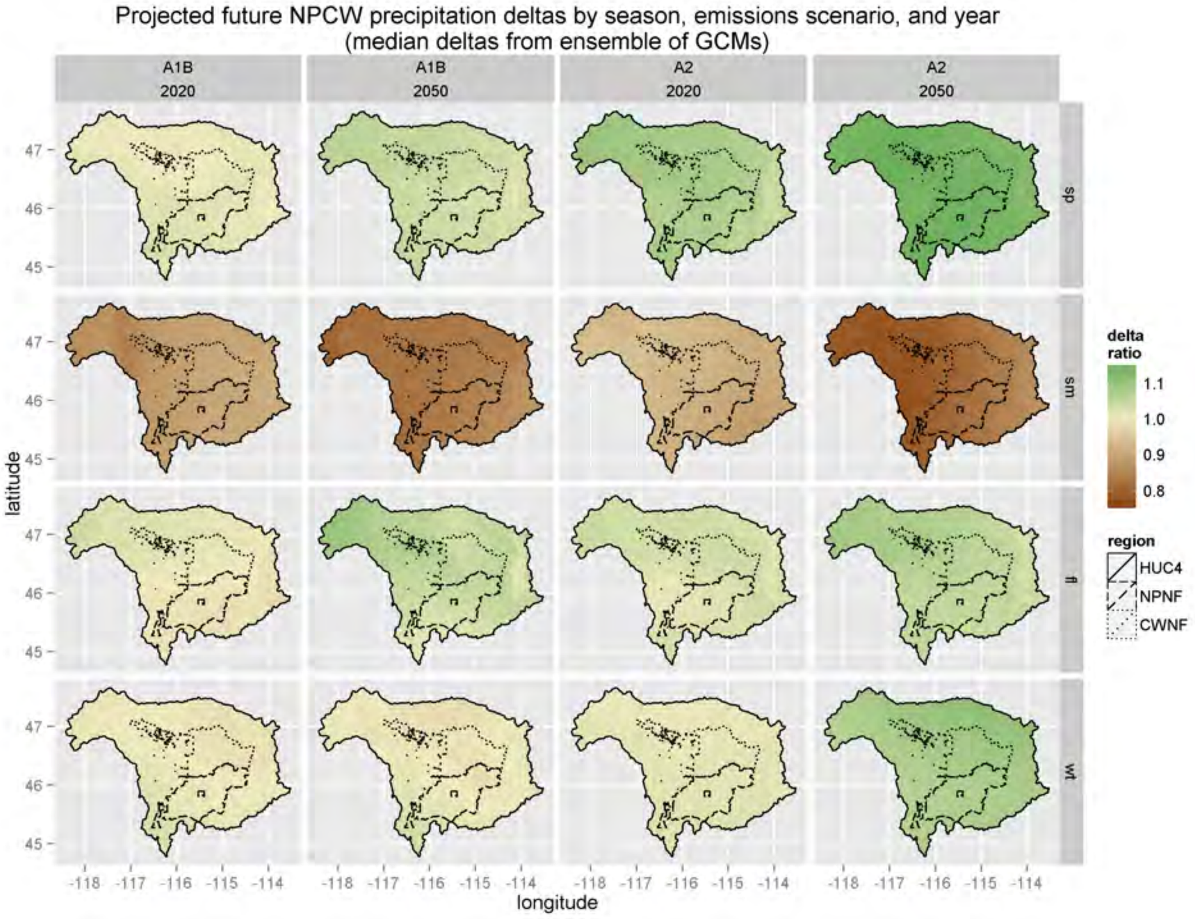
Projected Future Climate Deltas

The figures below show differences between the baseline (1961-1990) and two future timeslices (2020s and 2050s) for seasonal maximum temperature, minimum temperature, mean temperature, and precipitation, and for annual CMD. Temperature deltas are expressed as a difference (future – baseline) in degrees C, with 0 representing no change, positive values representing warming (red areas), and negative values representing cooling (blue areas). Precipitation deltas are calculated as a unitless ratio (future/baseline), with 1.0 representing no change, values greater than 1 representing increasing precipitation (green areas), and values less than 1 representing decreasing precipitation (brown areas). The same is true for CMD, except the color scale is inverted, as increasing CMD indicates increasing moisture stress. (Note that Tmax, Tmin, and CMD are presented for the A2 scenario, Tmean for the A1B scenario, and precipitation for both scenarios; see methodology for details.) Across the top of the figures is the emissions scenario and future timeslice. The right-hand side panels show the season: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7, 8), **fl** = fall (months 9, 10, 11), **wt** = winter (months 12, 1, 2).

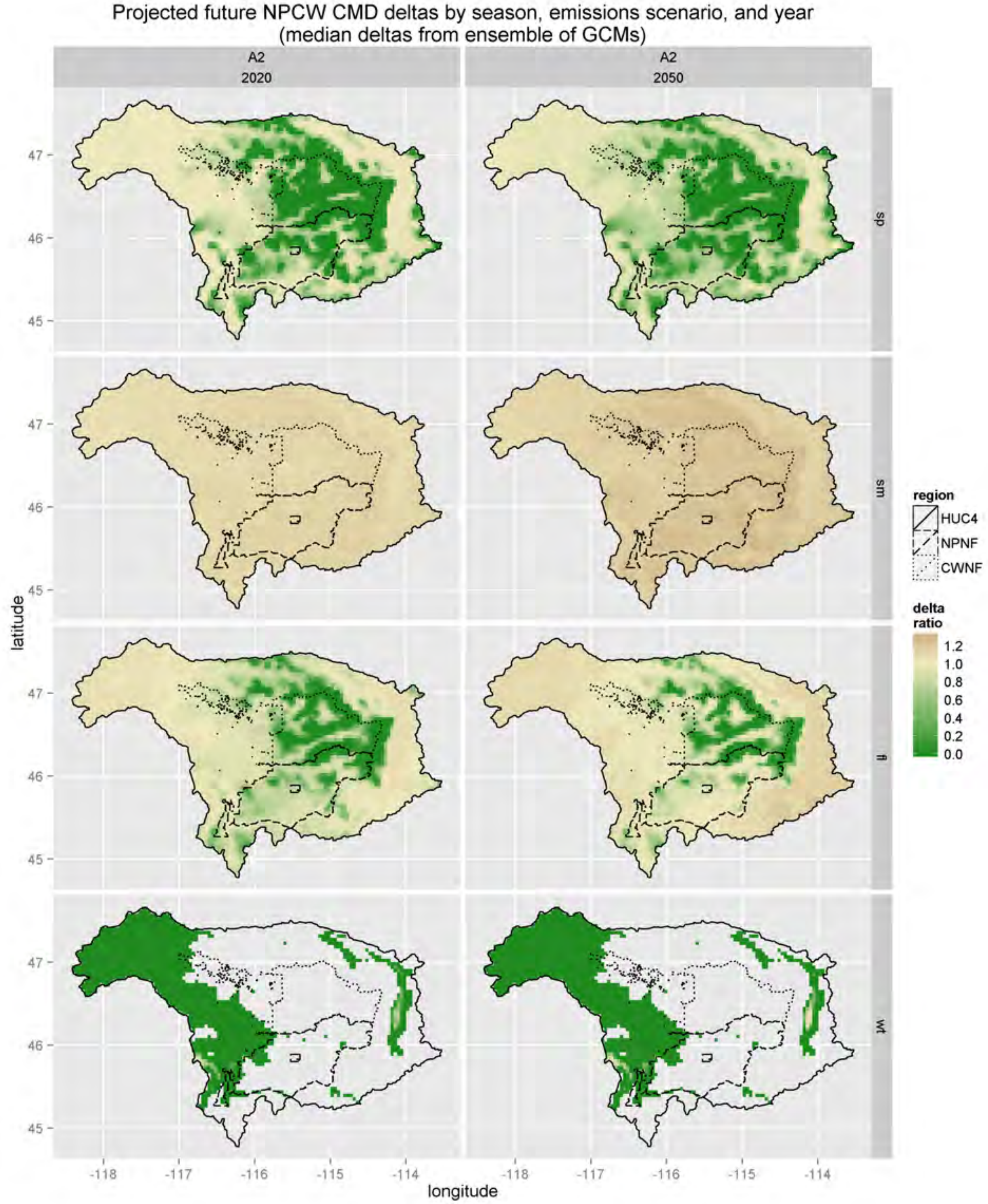

Projected future NPCW maximum temperature deltas by season, emissions scenario, and year (median deltas from ensemble of GCMs)


Projected future NPCW minimum temperature deltas by season, emissions scenario, and year (median deltas from ensemble of GCMs)


Projected future NPCW mean temperature deltas by season, emissions scenario, and year
(median deltas from ensemble of GCMs)

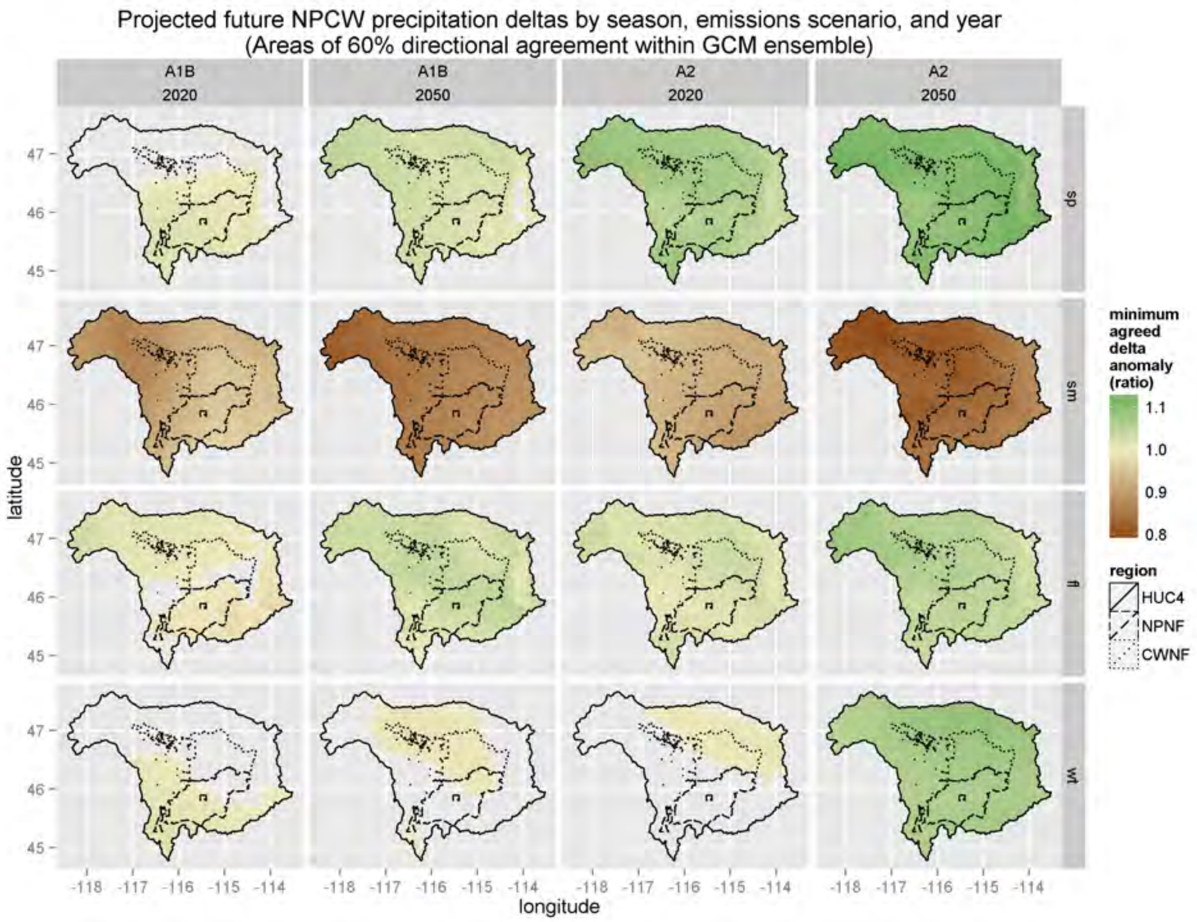


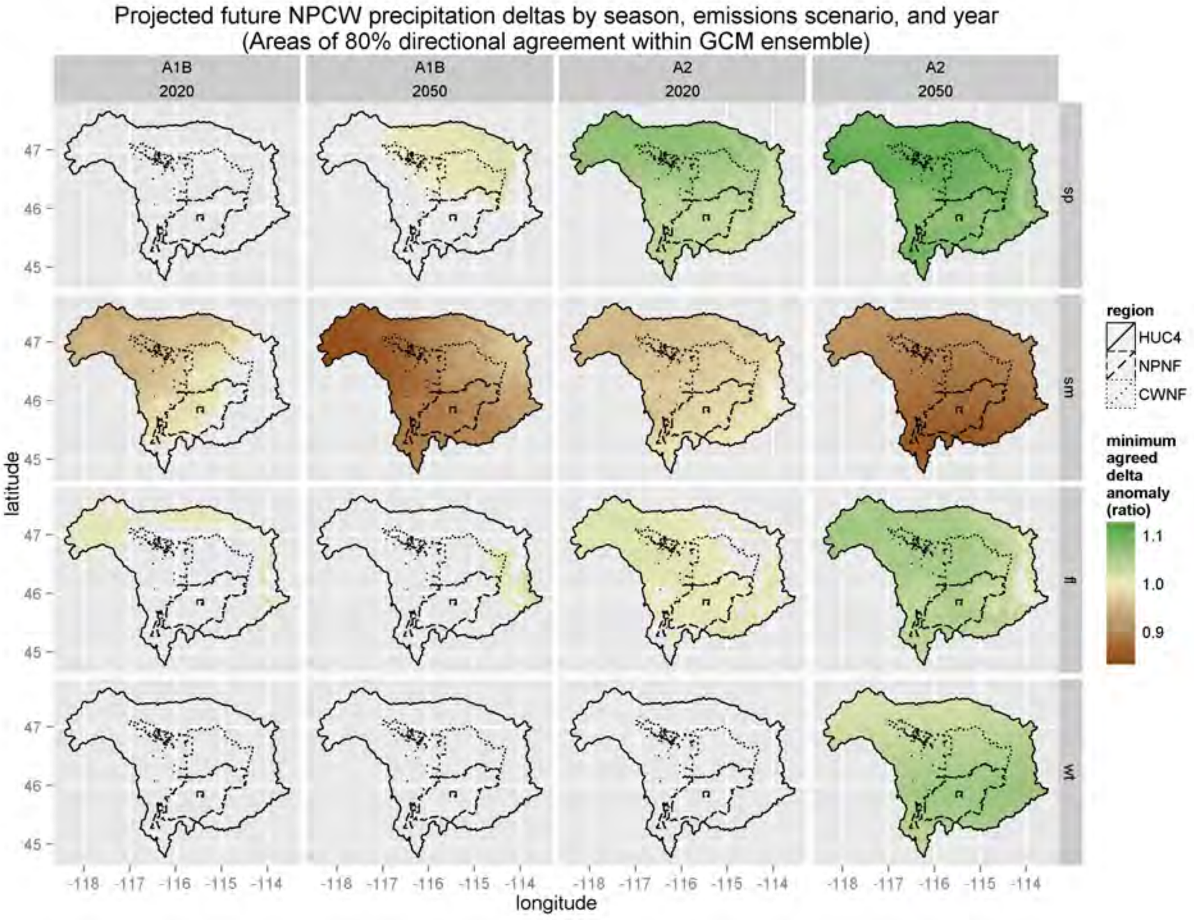
Projected future NPCW precipitation deltas by season, emissions scenario, and year
(median deltas from ensemble of GCMs)


Projected future NPCW CMD deltas by season, emissions scenario, and year
(median deltas from ensemble of GCMs)
A2
2020
A2
2050
sp
sm
fl
wt
latitude
45
46
47
longitude
-118
-117
-116
-115
-114
region
HUC4
NPNF
CWNF
delta ratio
1.2
1.0
0.8
0.6
0.4
0.2
0.0

Projected Future Climate Deltas – Precipitation Model Agreement

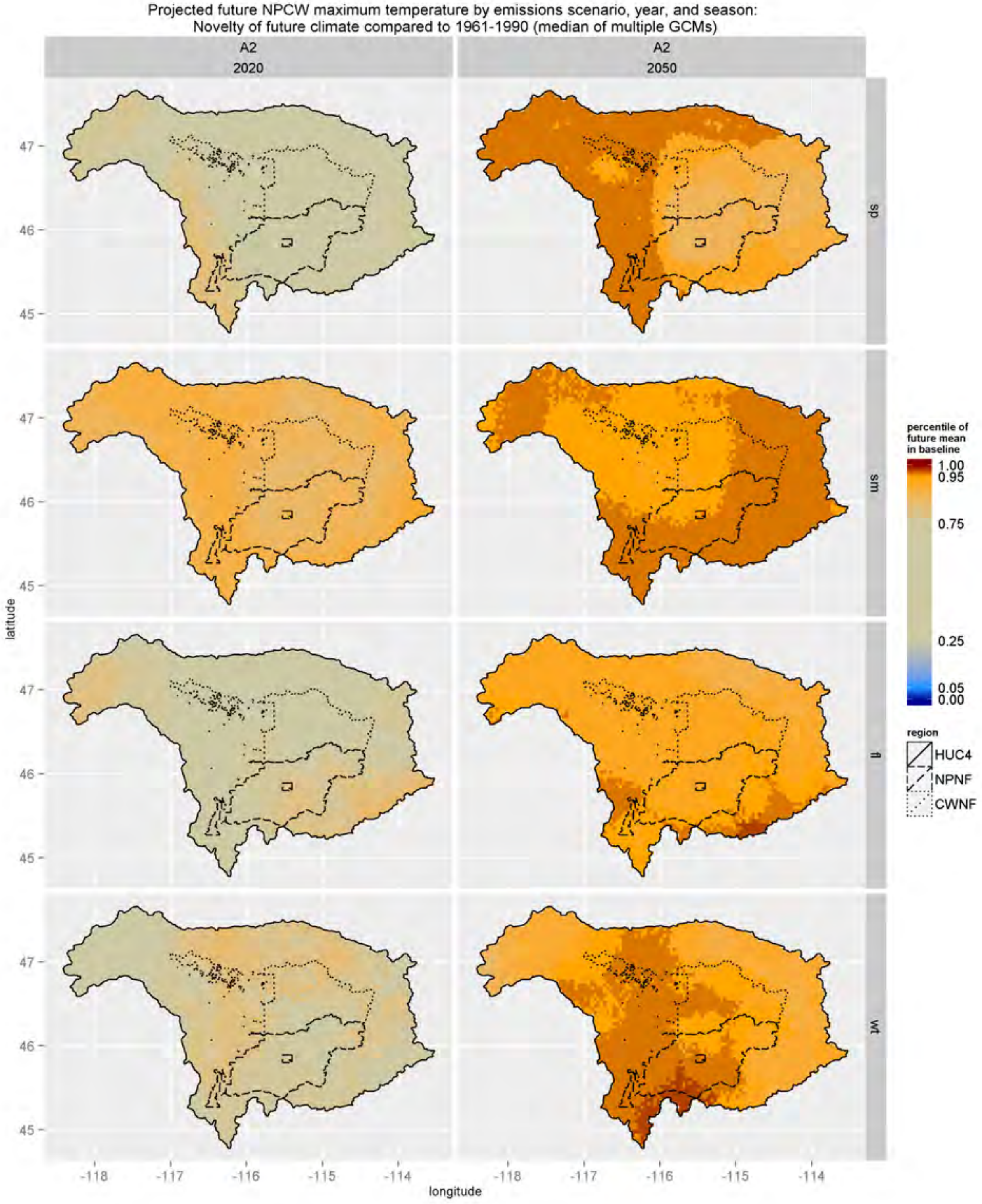
In the charts below, the stated percentage (60% or 80%) of models agree there will be *at least* the mapped amount of change. This is calculated across 7 models for A1B and 6 models for A2 emissions scenarios. Areas in green are where models agree that precipitation is increasing, areas in brown are where models agree that precipitation is decreasing, and hollow areas are pixels where less than the stated percentage of models agree on the direction of change. Across the top of the figures is the emission scenario (A2 and A1B) and future timeslice. The right-hand side panels show the season: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7, 8), **fl** = fall (months 9, 10, 11), **wt** = winter (months 12, 1, 2).



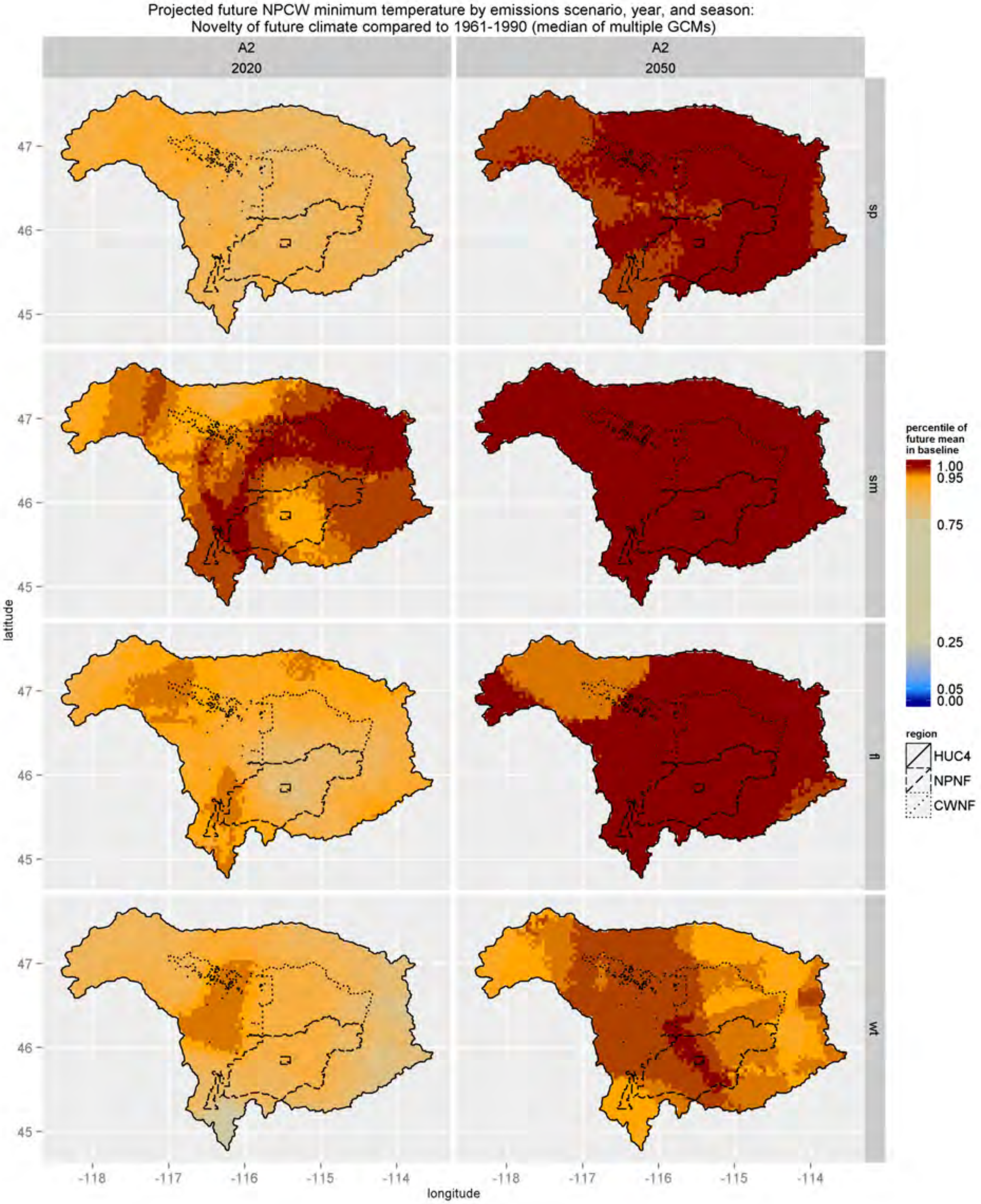

Projected future NPCW precipitation deltas by season, emissions scenario, and year
(Areas of 80% directional agreement within GCM ensemble)
A1B 2020
A1B 2050
A2 2020
A2 2050
sp
sm
fl
wt
latitude
47
46
45
longitude
-118
-117
-116
-115
-114
region
HUC4
NPNF
CWNF
minimum agreed delta anomaly (ratio)
1.1
1.0
0.9

Projected Future Climate Novelties

Future climate novelty values, which represent where recent climate falls within the range of historic variability for a particular location, are shown below for each variable (maximum temperature, minimum temperature, mean temperature, precipitation, CMD). Values range from 0 to 1, and represent the percentile at which the mean climate of future years falls within the 80-year baseline. Areas in grey/tan (values of .25 to .75) are within the interquartile range of historic variability. For precipitation and CMD, green indicates areas of increased precipitation or decreased moisture deficit and orange/red indicates areas of decreased precipitation or increased moisture deficit. For temperature, blue indicates areas of decreased temperature and orange/red indicates areas of increased temperature. A value of .85 would represent a pixel where half of recent years were warmer/wetter than 85% of baseline years. Values of 1 and 0 indicate places where at least half of recent years are completely outside the range of historic variability. Across the top of the figure is the emission scenario (A2 and A1B) and future timeslice. The right-hand side panel shows the season: **sp** = spring (months 3, 4, 5), **sm** = summer (months 6, 7 & 8), **fl** = fall (months 9, 10, 11), **wt** = winter (12, 1, 2).

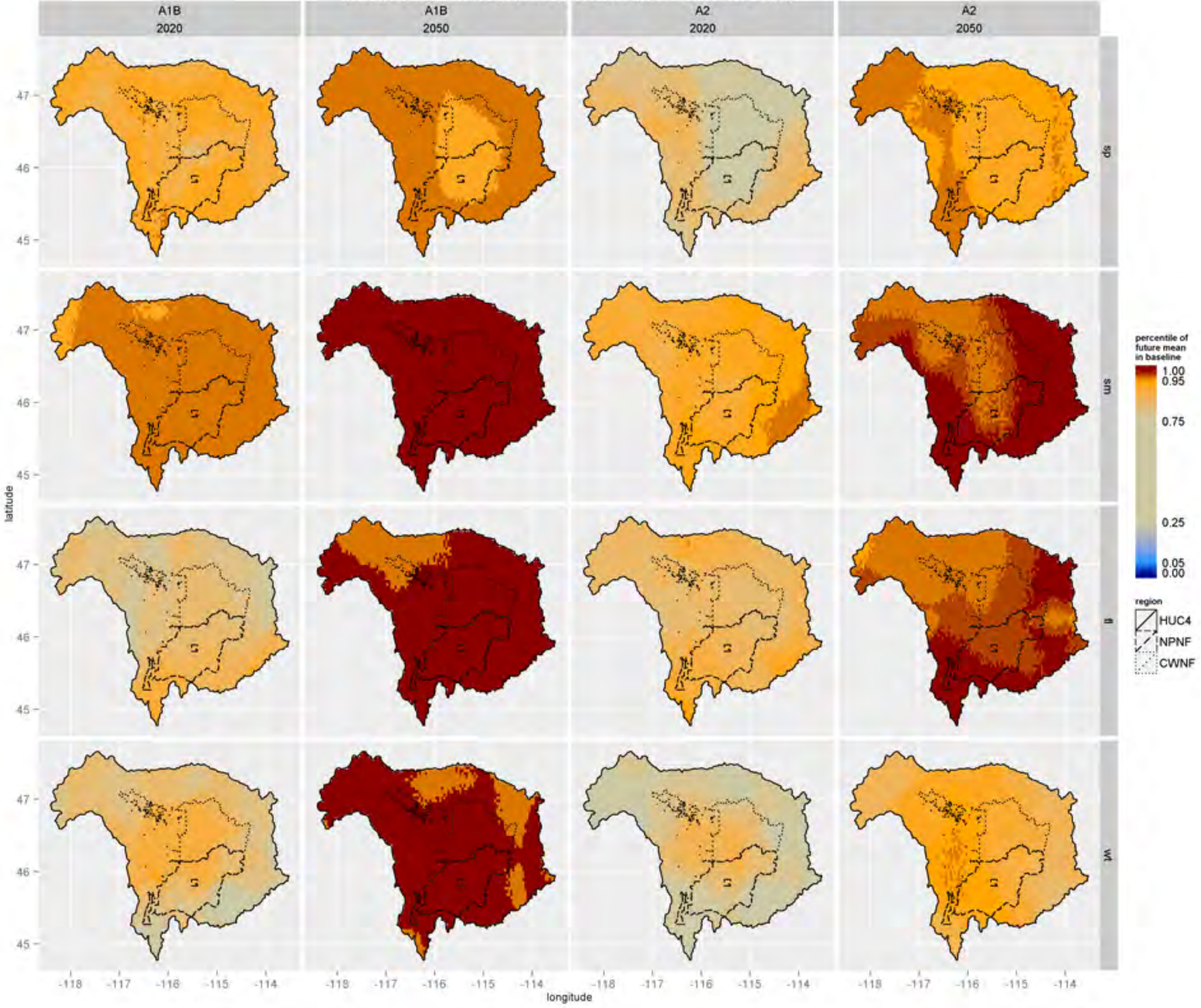


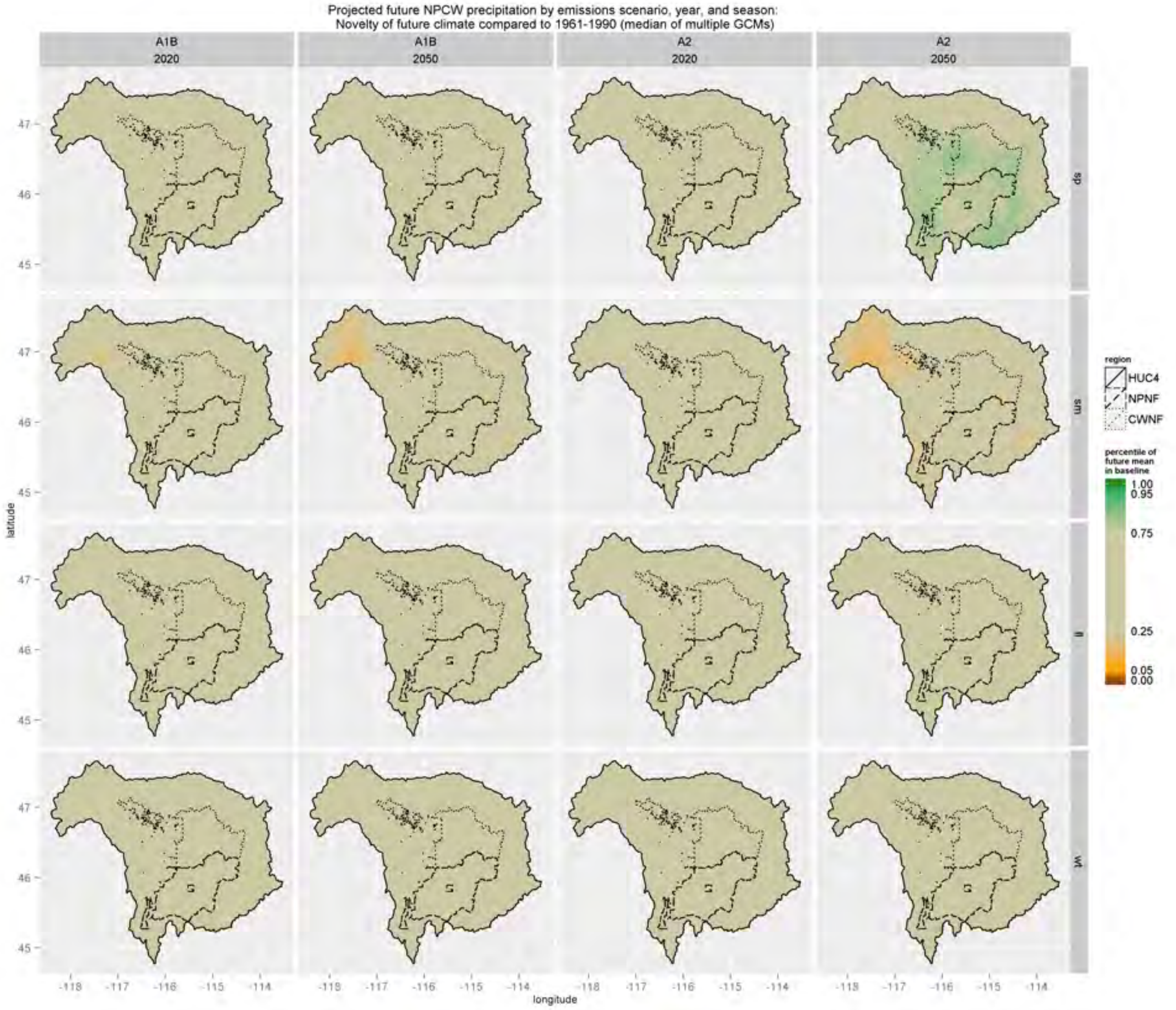
Projected future NPCW maximum temperature by emissions scenario, year, and season:
Novelty of future climate compared to 1961-1990 (median of multiple GCMs)

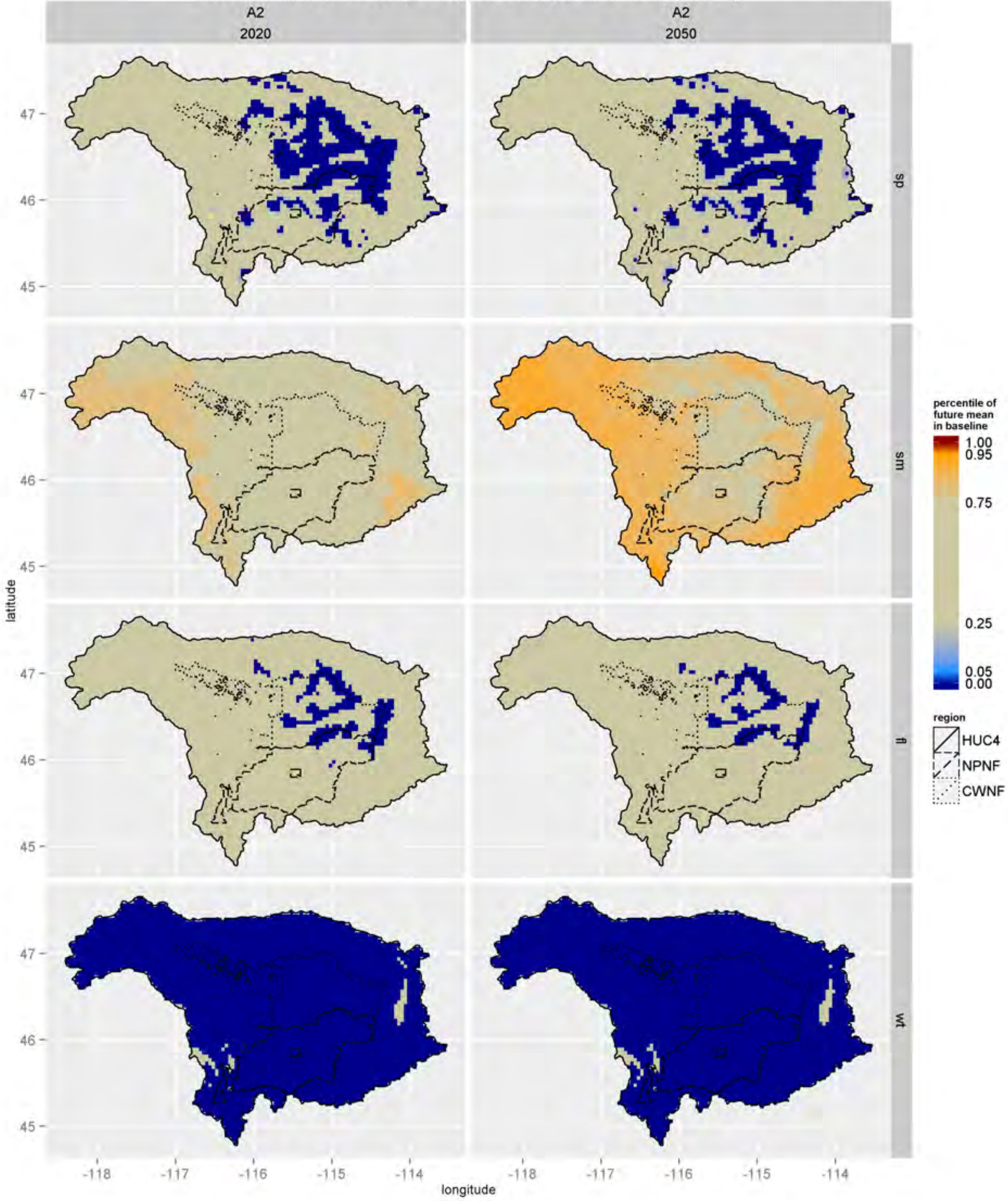


Projected future NPCW minimum temperature by emissions scenario, year, and season:
Novelty of future climate compared to 1961-1990 (median of multiple GCMs)

Projected future NPCW mean temperature by emissions scenario, year, and season:
Novelty of future climate compared to 1961-1990 (median of multiple GCMs)




Projected future NPCW precipitation by emissions scenario, year, and season:
Novelty of future climate compared to 1961-1990 (median of multiple GCMs)
A1B
2020
A1B
2050
A2
2020
A2
2050
sp
sm
fl
wt
latitude
47
46
45
longitude
-118
-117
-116
-115
-114
region
HUC4
NPNF
CWNF
percentile of future mean in baseline
1.00
0.95
0.75
0.25
0.05
0.00


Projected future NPCW CMD by emissions scenario, year, and season:
Novelty of future climate compared to 1961-1990 (median of multiple GCMs)

Summary of Projected Future Climate Trends

This discussion will focus on climate model results for the A2 emissions scenario (this scenario is much closer to the latest projected greenhouse gas emission rates than is the more optimistic A1B scenario), for the year 2050 (2020 projections follow similar patterns but are smaller in magnitude). Where applicable we discuss concordance between recent trends and future projections, as one might have increased confidence in GCM results that project trends similar to those currently being observed.

Climate models agree that temperatures across the NPCW ecoregion are projected to increase significantly by mid-century. Under the A2 emissions scenario, mid-century conditions are modeled to be about 2.5 degrees C (5.5 degrees F) warmer than the 20th-century baseline (see Projected Future Climate Deltas). Minimum temperature increases are projected to be relatively uniform across seasons, a pattern that is consistent with recent climate trends, with the highest values in the summertime at nearly 3 degrees C (see Projected Future Climate Deltas). These increases would make the average mid-century summer minimum temperature warmer than 100% of observed baseline summers, and the average mid-century winter minimum temperature warmer than at least 80% of baseline winters, with spring and fall at intermediate percentiles (see Projected Future Climate Novelties).

Maximum temperature is also projected to increase, but less rapidly and with more seasonal variability than minimum temperature. Again, this forecast is consistent with 20th-century trends. Maximum temperature deltas for 2050 are 1.5-2 degrees C in winter and more than 2.5 degrees in summer, with intermediate values in spring and fall (see Projected Future Climate Deltas). Across seasons and geographies, this would mean mid-21st-century average maximum temperatures are higher than roughly 95% of 20th-century years (see Projected Future Climate Novelties).

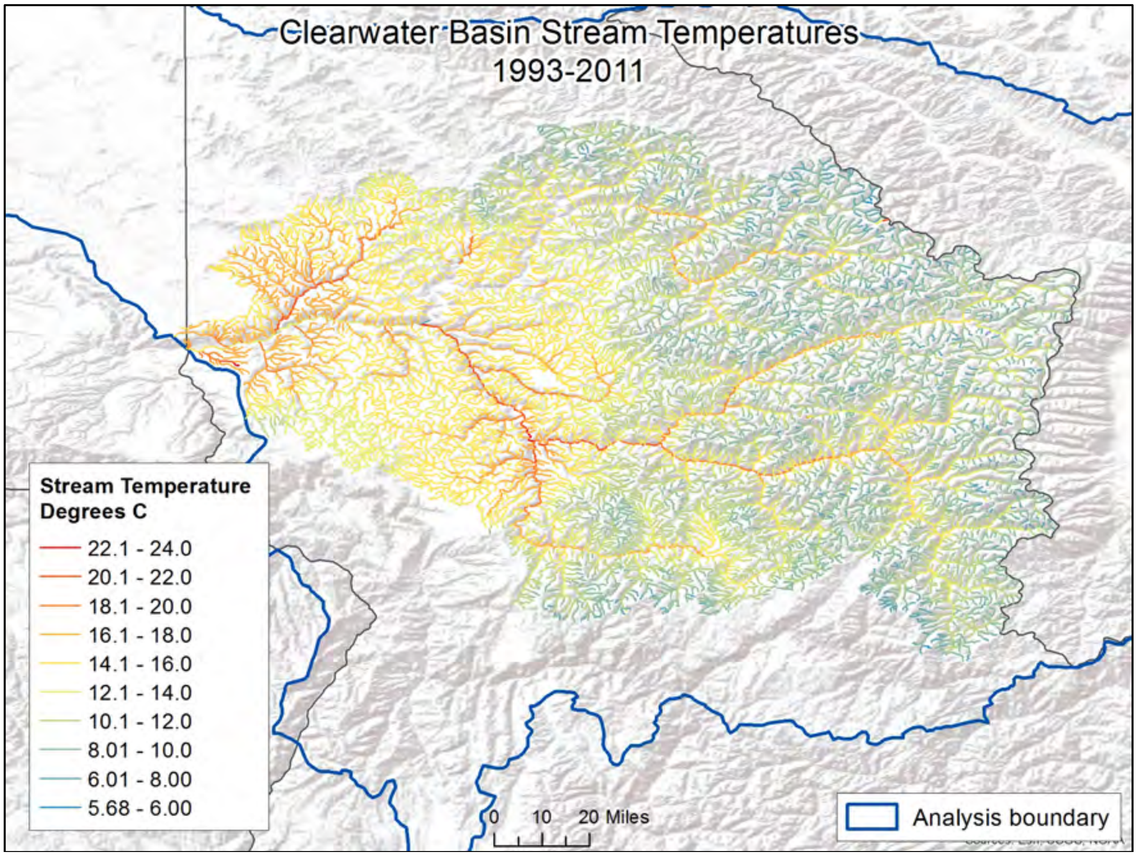
Precipitation projections are more variable among climate models, with some disagreeing even on the direction of change. However, the A2 scenario for 2050 shows relatively good model agreement for this region (see Projected Future Climate Deltas – Precipitation Model Agreement). Eighty percent of models agree that precipitation will increase by *at least* 10% in spring, decrease by *at least* 10% in summer, and see more modest increases in fall and winter (see Projected Future Climate Deltas – Precipitation Model Agreement). The largest projected precipitation change is in summertime, when the GCM ensemble median indicates a roughly 20% decline – this change would remain well within the range of historic variability for most parts of the region, but would be drier than all but the driest 15-20% of summers in the northwest part of the watershed. Projected mid-century precipitation increases in the other three seasons would make spring wetter than 75-80% of historic springs across many central and southern parts of the region, but would not put fall or winter climate outside the historic interquartile range (see Projected Future Climate Novelties). For precipitation, the seasonality of future projections partially aligns with that of recent trends – both analyses show precipitation increasing in spring and remaining relatively unchanged in fall, but for summer and winter the directions of observed 20th-century change are opposite those of projected 21st-century change (see Current Climate Deltas and Projected Future Climate Deltas).

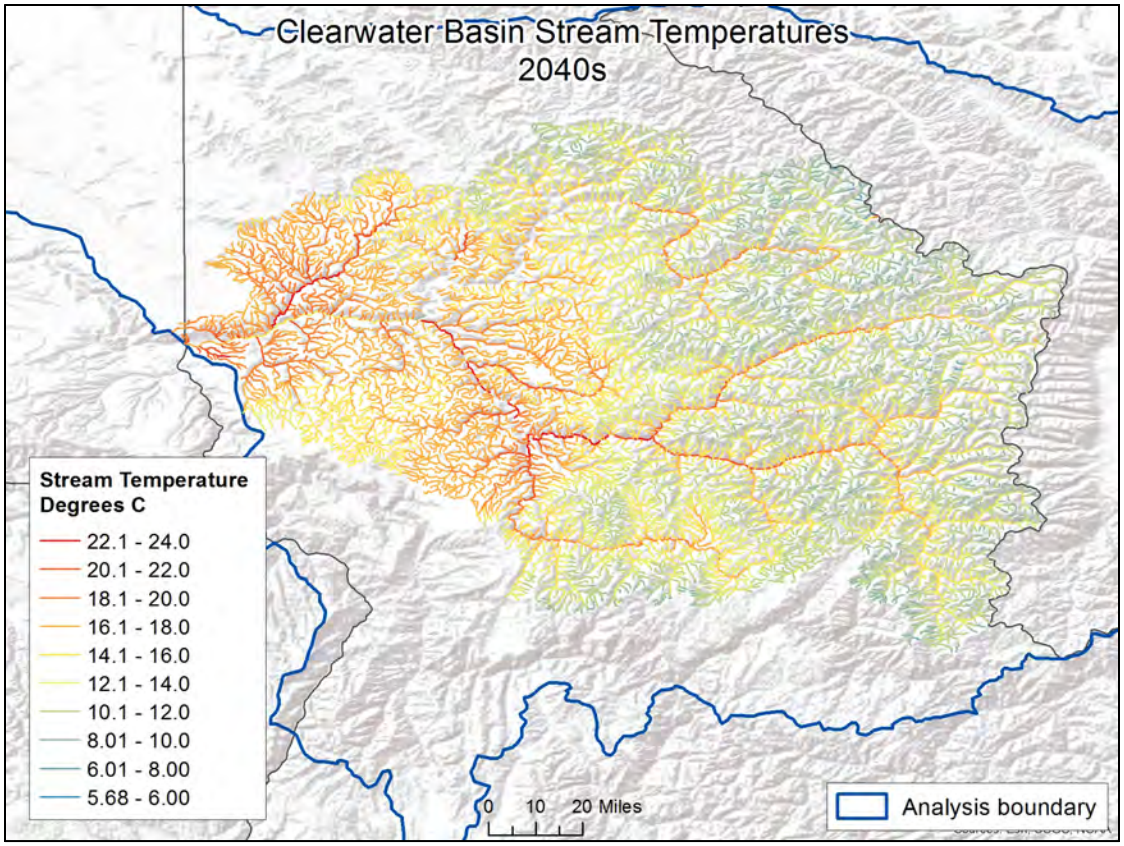
Climatic moisture deficit (CMD) can react in complex and nonlinear ways to changes in temperature and precipitation. CMD is modeled to increase slightly in summer (the season with the largest projected temperature increases and precipitation decreases), making for historically high but not unprecedented drying conditions across most of the NPCW ecoregion (see Projected Future Climate Deltas and Projected Future Climate Novelties). In the other three seasons, CMD changes vary spatially, ranging from little change to dramatic and unprecedented decreases – in winter these decreases are widespread, while in spring and fall they are confined to the core of the watershed (see Projected Future Climate Novelties). Rising temperatures releasing frozen moisture in historically colder months and locations may drive these projected increases in moisture availability.

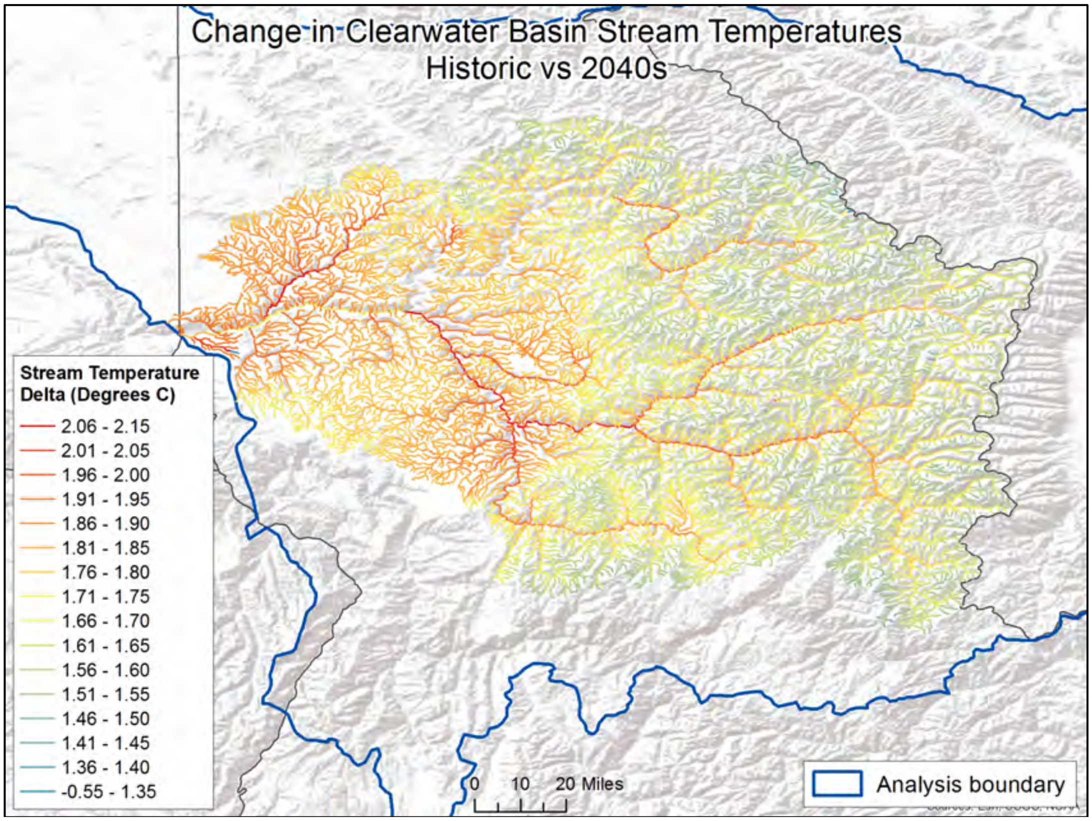
Stream Temperature and Flow Metrics

Stream Temperature

Below are maps of historic stream temperatures (1993-2011), modeled future stream temperatures (A1B 2040s), and the change between future and historic expressed as a difference (future – historic) in degrees C. Classes are symbolized in equal intervals of 2 degrees C.

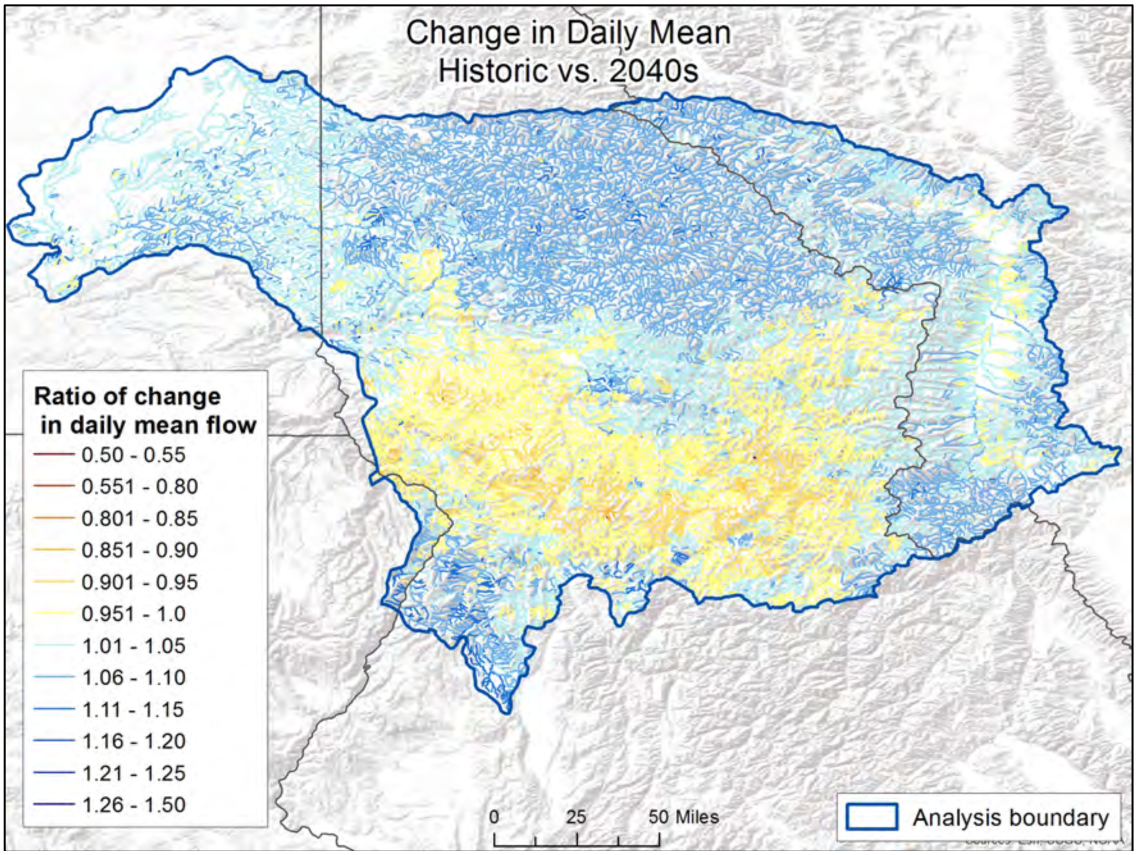


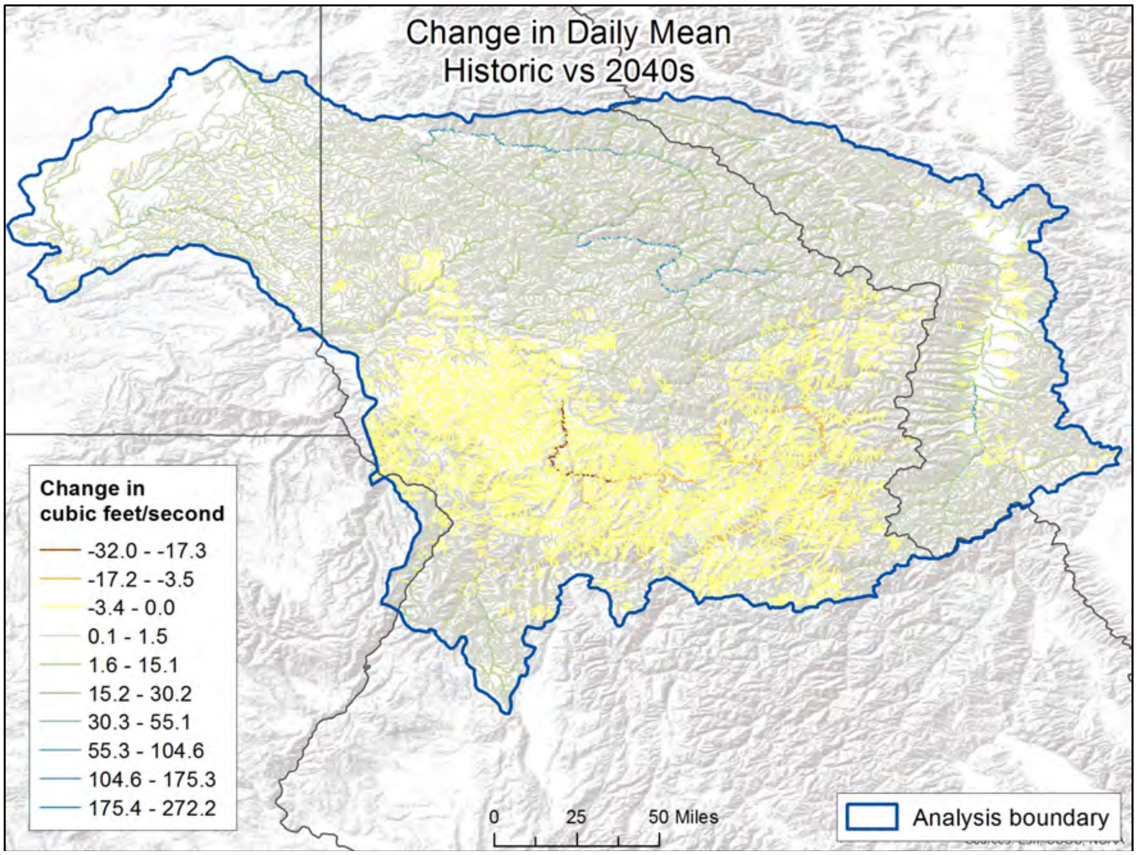

Clearwater Basin Stream Temperatures
2040s
Stream Temperature
Degrees C
22.1 - 24.0
20.1 - 22.0
18.1 - 20.0
16.1 - 18.0
14.1 - 16.0
12.1 - 14.0
10.1 - 12.0
8.01 - 10.0
6.01 - 8.00
5.68 - 6.00
0 10 20 Miles
Analysis boundary

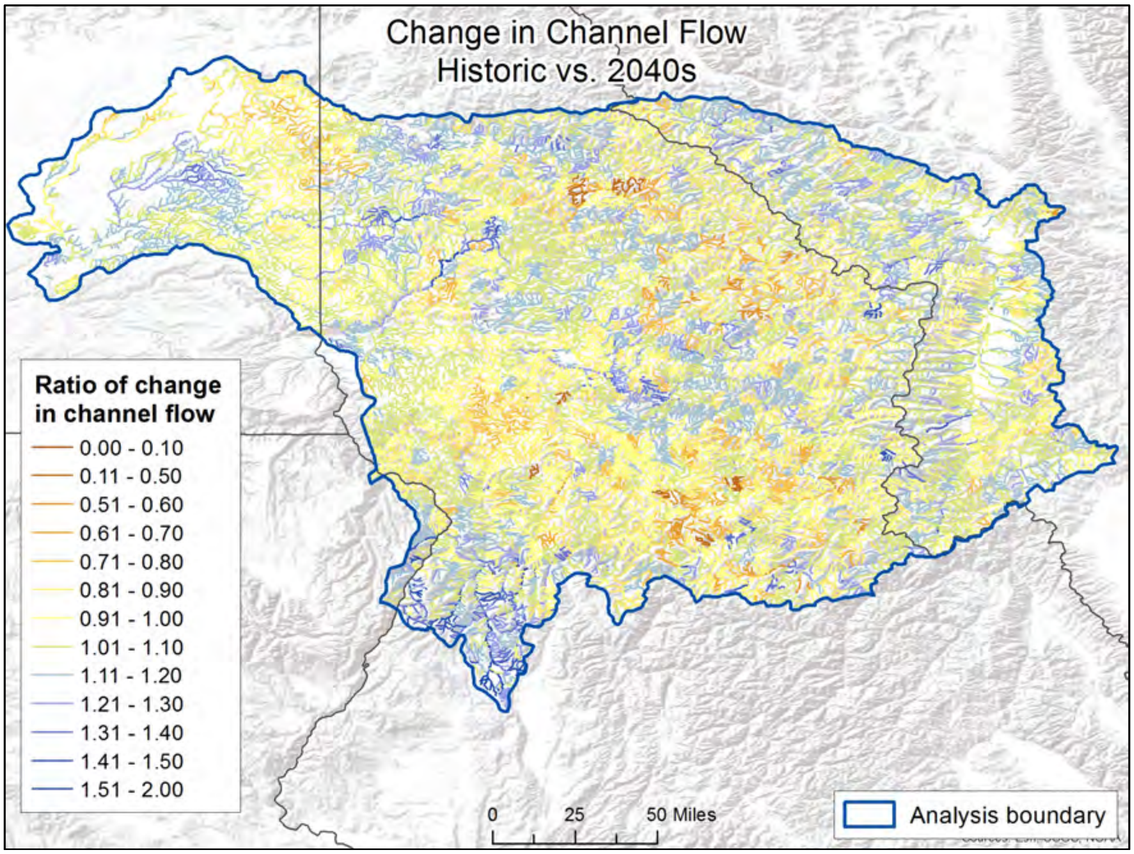

Change in Clearwater Basin Stream Temperatures
Historic vs 2040s
Stream Temperature
Delta (Degrees C)
2.06 - 2.15
2.01 - 2.05
1.96 - 2.00
1.91 - 1.95
1.86 - 1.90
1.81 - 1.85
1.76 - 1.80
1.71 - 1.75
1.66 - 1.70
1.61 - 1.65
1.56 - 1.60
1.51 - 1.55
1.46 - 1.50
1.41 - 1.45
1.36 - 1.40
-0.55 - 1.35
0 10 20 Miles
Analysis boundary

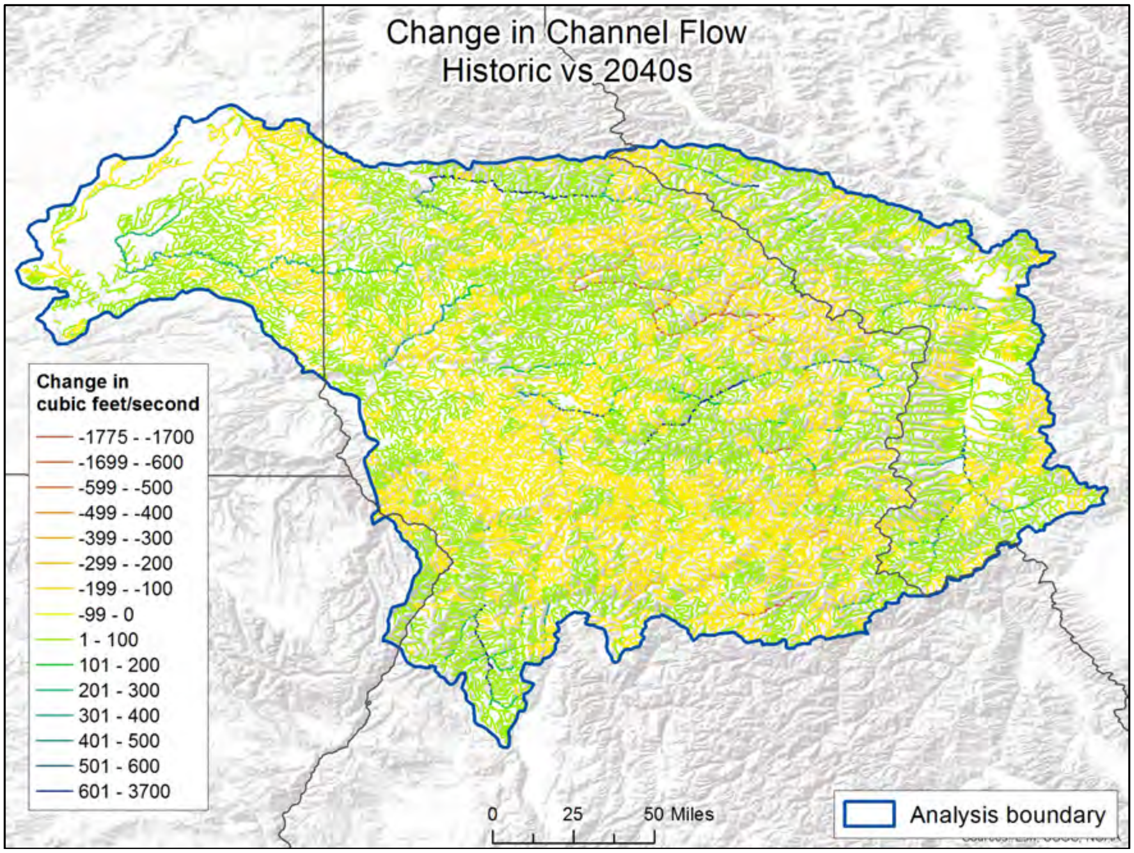
Stream Flow Metrics

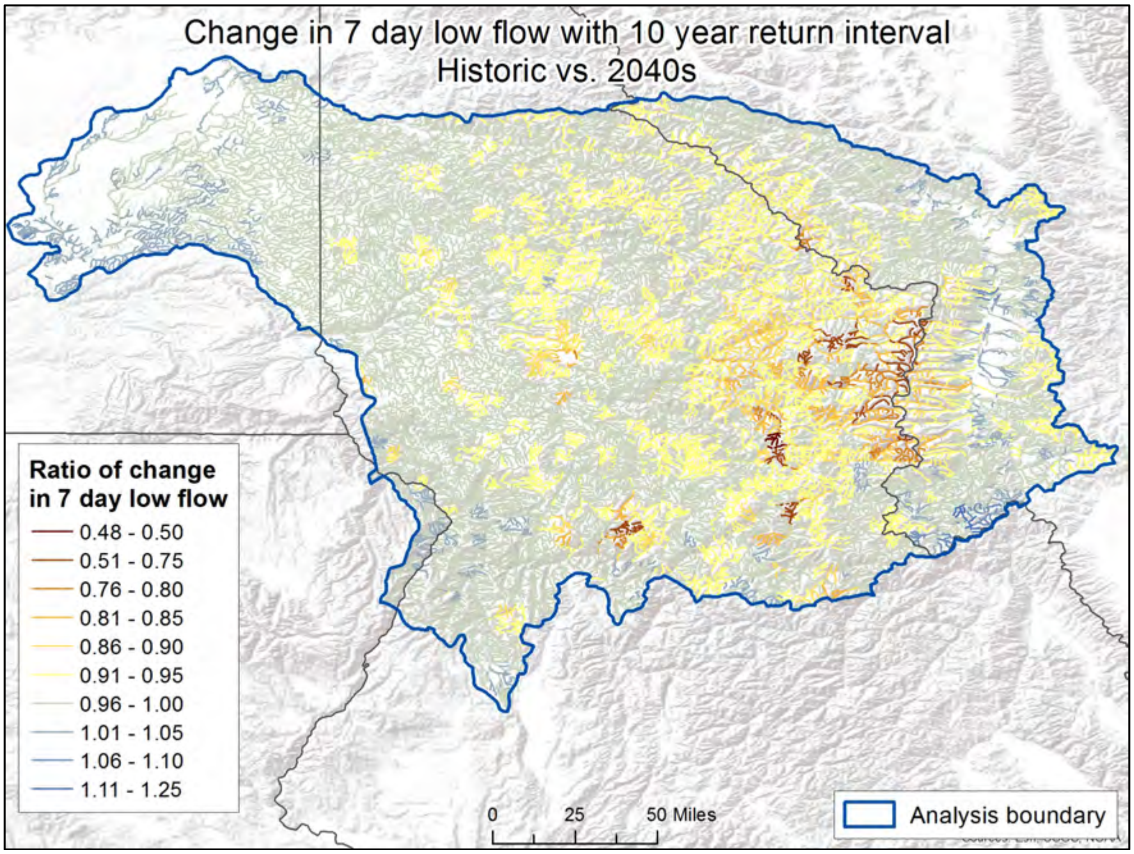
Delta fields were calculated to show the change between historic flow metrics (daily mean flow, channel-forming flow, 7-day low flow, timing of flows) and projections for the future mid-century timeslice (2040s). For all flow metrics, deltas were calculated between the future and historic timeslices (future – historic). Additionally, for daily mean flow, channel-forming flows, and the 7-day low flow with 10-year return interval, deltas were calculated as a ratio (future/historic). Delta and ratio values are symbolized with equal intervals within 2 standard deviations of the mean, and larger intervals for values outside 2 standard deviations from the mean. For delta maps, negative values are symbolized using orange or brown colors to indicate decreasing flows and positive values are in shades of green or blue to indicate increasing or stable flows. Ratio values follow the same color scheme and values lower than 1 are decreasing in flow and values above 1 are increasing.

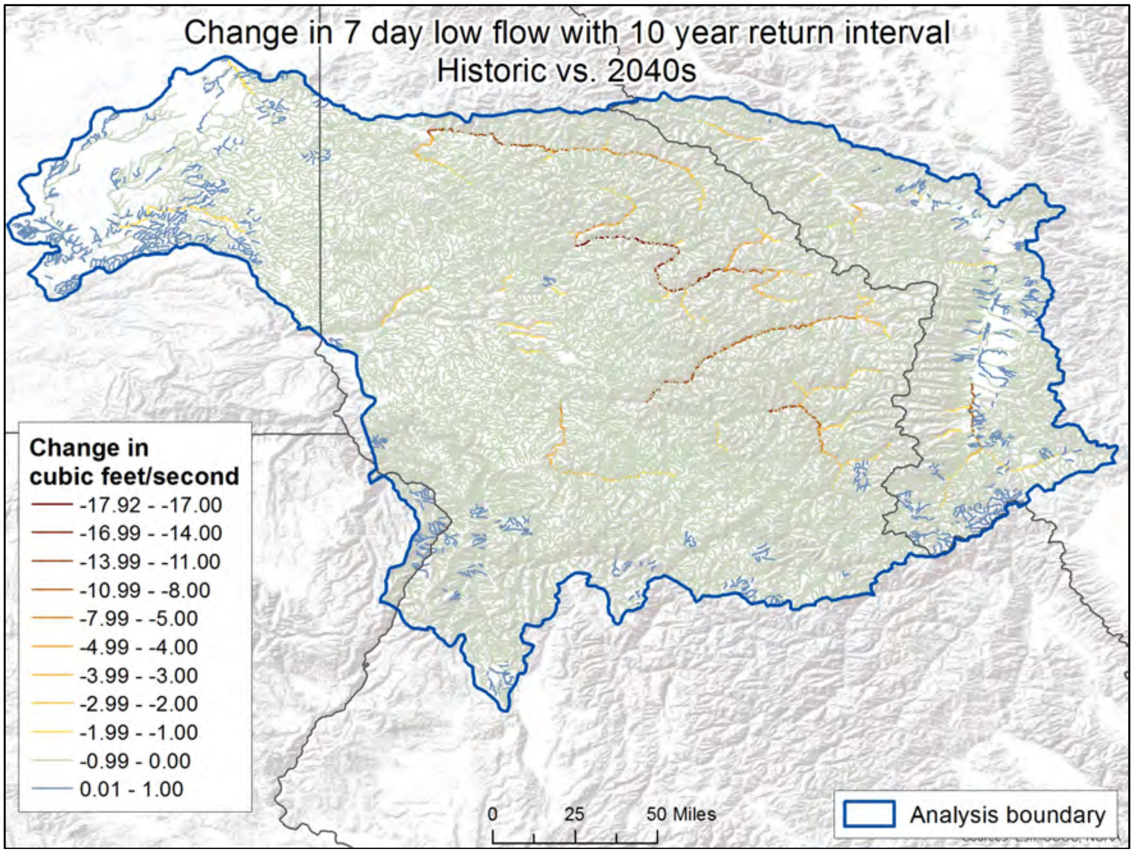


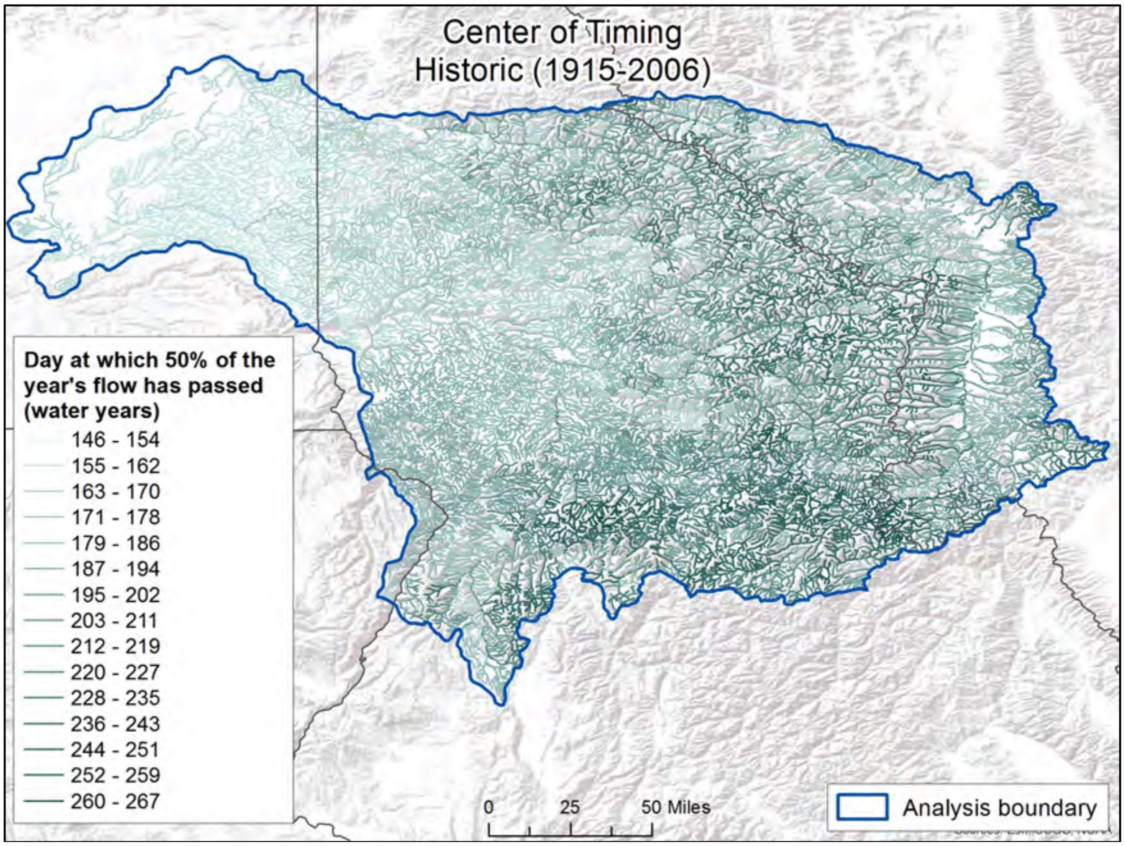

Change in Daily Mean
Historic vs 2040s
Change in cubic feet/second
-32.0 - -17.3
-17.2 - -3.5
-3.4 - 0.0
0.1 - 1.5
1.6 - 15.1
15.2 - 30.2
30.3 - 55.1
55.3 - 104.6
104.6 - 175.3
175.4 - 272.2
0
25
50 Miles
Analysis boundary
Sources: Esri, USGS, NOAA

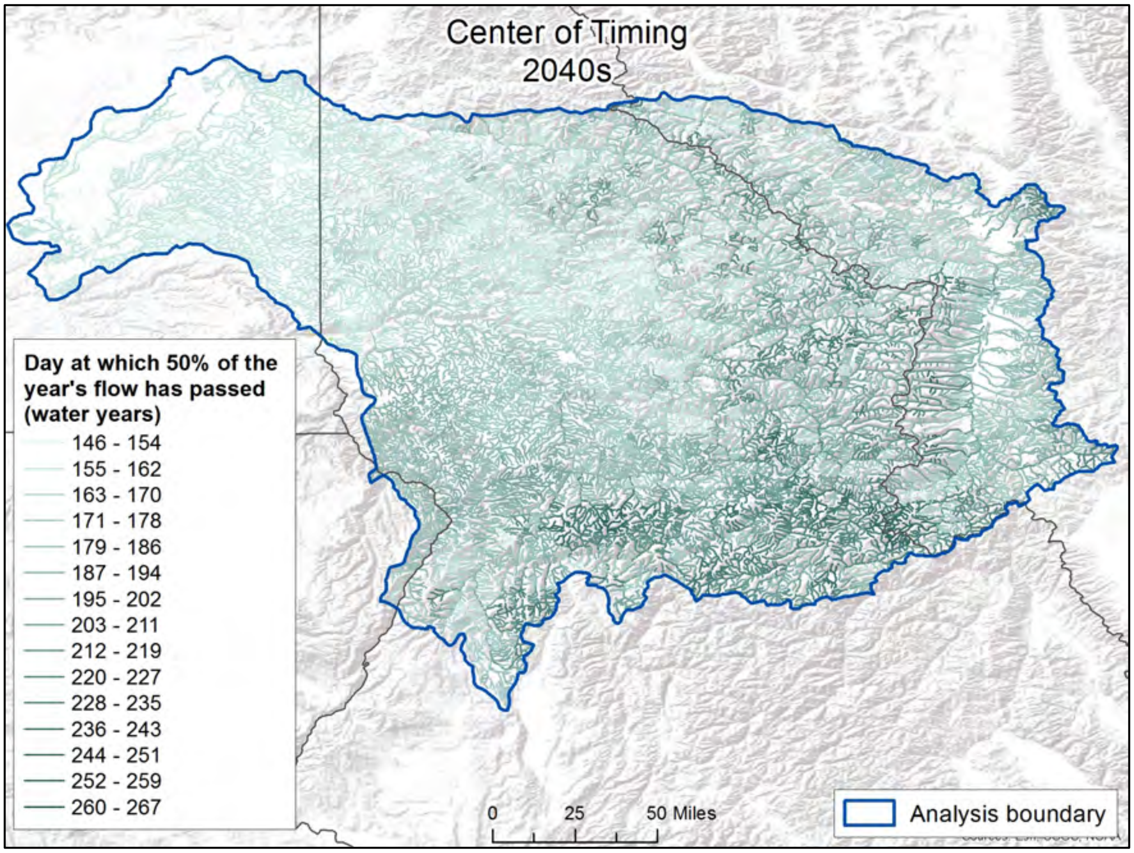

Change in Channel Flow
Historic vs. 2040s
Ratio of change in channel flow
0.00 - 0.10
0.11 - 0.50
0.51 - 0.60
0.61 - 0.70
0.71 - 0.80
0.81 - 0.90
0.91 - 1.00
1.01 - 1.10
1.11 - 1.20
1.21 - 1.30
1.31 - 1.40
1.41 - 1.50
1.51 - 2.00
0
25
50 Miles
Analysis boundary

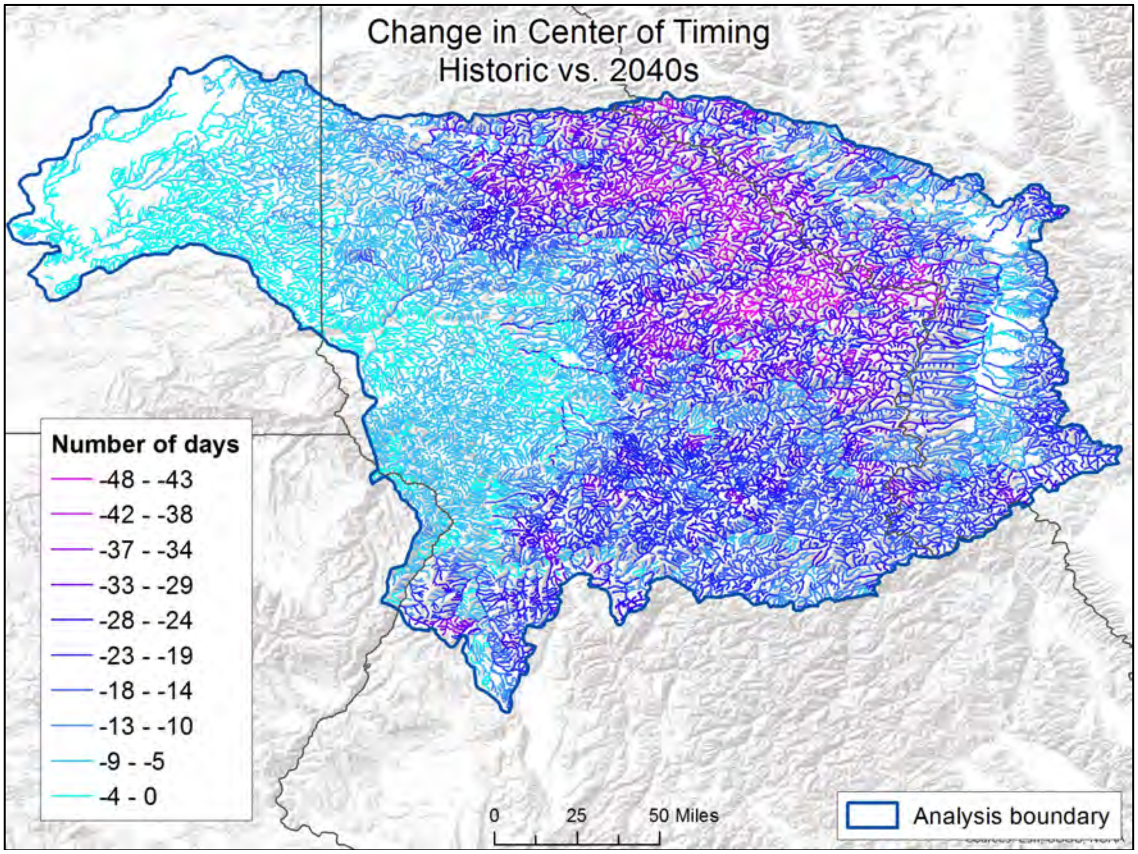

Change in Channel Flow
Historic vs 2040s
Change in cubic feet/second
-1775 - -1700
-1699 - -600
-599 - -500
-499 - -400
-399 - -300
-299 - -200
-199 - -100
-99 - 0
1 - 100
101 - 200
201 - 300
301 - 400
401 - 500
501 - 600
601 - 3700
0
25
50 Miles
Analysis boundary


Change in 7 day low flow with 10 year return interval
Historic vs. 2040s
Ratio of change in 7 day low flow
0.48 - 0.50
0.51 - 0.75
0.76 - 0.80
0.81 - 0.85
0.86 - 0.90
0.91 - 0.95
0.96 - 1.00
1.01 - 1.05
1.06 - 1.10
1.11 - 1.25
0
25
50 Miles
Analysis boundary


Change in 7 day low flow with 10 year return interval
Historic vs. 2040s
Change in cubic feet/second
-17.92 - -17.00
-16.99 - -14.00
-13.99 - -11.00
-10.99 - -8.00
-7.99 - -5.00
-4.99 - -4.00
-3.99 - -3.00
-2.99 - -2.00
-1.99 - -1.00
-0.99 - 0.00
0.01 - 1.00
0
25
50 Miles
Analysis boundary


Center of Timing
Historic (1915-2006)
Day at which 50% of the year's flow has passed (water years)
146 - 154
155 - 162
163 - 170
171 - 178
179 - 186
187 - 194
195 - 202
203 - 211
212 - 219
220 - 227
228 - 235
236 - 243
244 - 251
252 - 259
260 - 267
0
25
50 Miles
Analysis boundary


Center of Timing
2040s
Day at which 50% of the year's flow has passed (water years)
146 - 154
155 - 162
163 - 170
171 - 178
179 - 186
187 - 194
195 - 202
203 - 211
212 - 219
220 - 227
228 - 235
236 - 243
244 - 251
252 - 259
260 - 267
0
25
50 Miles
Analysis boundary


Change in Center of Timing
Historic vs. 2040s
Number of days
-48 - -43
-42 - -38
-37 - -34
-33 - -29
-28 - -24
-23 - -19
-18 - -14
-13 - -10
-9 - -5
-4 - 0
0
25
50 Miles
Analysis boundary

Summary of Stream Temperature and Flow Trends

Overall, streams in the NPCW region in 2040 are projected to become warmer, and deliver more of their flow earlier in the water year, than during the historical baseline. They are also projected to undergo noticeable, but more directionally and spatially variable changes in the daily mean flow, channel-forming flow, and 7-day low flow.

Stream temperatures in the Clearwater basin (see Stream Temperature maps) are projected to increase by as much as 2 degrees C by the 2040s. Stream temperature deltas are highly correlated with stream order: lower-order streams in the uplands are projected to have little-to-no warming, with temperature increases growing systematically larger as water moves toward larger arteries. This would suggest that exposure to warmer projected surface climate, as opposed to belowground temperature increases, is responsible for projected stream temperature increases.

Daily mean stream flow volumes by 2040 are projected to decrease by 5-10% across much of the southern portion of the NPCW ecoregion, and increase by 10-15% across much of the northern half of the basin; in extreme cases, isolated streams within these two areas are projected to see daily mean flow decrease by more than 45% and increase by more than 25%, respectively (see Stream Flow Metrics – Daily Mean Flow).

The stream flow that is responsible for creating and maintaining the size and shape of a stream is the channel forming flow. The spatial pattern of change in channel forming flows appears to be random, although ratios of change generally across the NPCW region are decreasing. Deltas of change in cubic feet per second show that larger arteries are changing significantly but in different directions (see Stream Flow Metrics – Channel Flow).

The 7-day low flow with a 10-year return interval (7Q10) is used as an indicator of low flow conditions during drought. By the 2040s, 7Q10 is projected to decrease by as much as 50%, generally in the higher elevation areas in the eastern part of the analysis boundary (see Stream Flow Metrics – 7Q10 Change Ratio). Delta values in cubic feet per second are decreasing more in the larger arteries at higher elevations (see Stream Flow Metrics – 7Q10 CFS).

Significant changes are also projected in the timing of stream flow, with the center of timing (the day when 50% of the year's water has passed) expected to occur as much as six weeks earlier in 2040 compared to the historic baseline. These projected advances in the center of timing are most pronounced in lower-order streams in the center of the watershed, while larger and more western streams are projected to see much smaller changes (see Stream Flow Metrics – Center of Timing).

4. Uncertainty and Climate Change[5]

Models used to project rapidly changing climates have a high degree of uncertainty (IPCC 2007a). While there is little debate that atmospheric CO_2 is increasing at an alarming rate and this increase will continue to cause changes in climate (IPCC 2007a), there is uncertainty about the magnitude and rate of future change (Stainforth et al. 2005; Roe and Baker 2007). This uncertainty increases as climate predictions are made at finer resolutions, for different geographic areas, and for time periods farther into the future (e.g., end of 21st century). The range of possible future climates predicted from GCMs is much greater than the variability of climate over the past two or three centuries (Stainforth et al. 2005). Futhermore, it is impossible to know how society will respond to climate change, so GCMs often model a suite of scenarios that capture a range of possible societal responses to climate change (e.g., A1b vs. A2 scenarios). Finally, it is the high variability of climate extremes, not the gradual change of average climate over time, that will drive most ecosystem responses, and these rare, extreme events are difficult to predict (Easterling et al. 2000a).

Uncertainty further compounds as we try to predict how ecosystems will respond to the effects of climate change (Araujo et al. 2005). Mechanistic ecological simulation of climate, vegetation, and disturbance dynamics across landscapes is still in its infancy (Sklar and Costanza 1991; Walker 1994; Keane and Finney 2003). Many models ignore the important interactions that disturbance, hydrology, and land use have as they project climate effects on future vegetation distribution (Notaro et al. 2007). Little is known about the interactions between climate, vegetation and disturbance, or critical plant and animal life cycle processes of reproduction, growth, and mortality (Keane et al. 2001; Gworek et al. 2007; Ibanez et al. 2007; Lambrecht et al. 2007) as they interact with climate in different ways. Climate's influences on the interactions between multiple disturbance regimes (e.g., fire and beetles) could also create novel landscape responses.

Species Distributional Models (SDMs), also called bioclimatic or species envelope models or niche models, link current climate with current distribution of a species through advanced statistical modeling. Using the statistical model, future species distribution is generated using projected climate data as inputs. SDMs relate only climate to species occurrence, resulting in predictions of potential species habitat, not distribution. In addition, numerous critical processes are not part of SDMs, including mortality, competitive interactions, phenology, tree growth, and reproduction and establishment, among others. While SDM projections can be informative, they are not prognostic. Managers and planners may want to develop adaptation strategies that are more robust to uncertainty, rather than develop strategies that rely on predictive species distribution modeling.

However, uncertainties should not be a reason for inaction. Rather, it is incumbent that uncertainty be explicitly considered in order to evaluate the complexity and variability that it

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[5] Content contributions by R. Keane, USFS Rocky Mountain Research Station.

represents. Resource managers and planners are accustomed to making decisions within the context of uncertainty. Natural systems have inherent stochasticity, while human community responses are similarly unpredictable. Yet, we have developed decision-making frameworks that allow us to reasonably accommodate uncertainty, complexity and variability in order to move forward and improve outcomes. Historically we have used past experience to help inform our future actions. Accommodating climate change-related uncertainty requires that we evaluate present and future events, posing new challenges for communities and resource managers who may have relied on evaluation of past experience (Hulme and Carter 1999, Refsgaard et al. 2013). An evolving variety of tools and approaches are being used by managers to prepare for the potential range of future conditions that are often provided by climate data. These include: Scenario Planning (Peterson et al. 2003), Adaptive Management (Hansen & Hoffman 2011), Contingency Planning/Bet Hedging (Hansen et al. 2003), and the Precautionary Principle (Hansen & Hoffman 2011), among others.

5. Vulnerability Assessment Model and Methods

Defining Terms

Exposure: A measure of how much of a change in climate or climate-driven factors a resource is likely to experience (Glick et al. 2011).

Sensitivity: A measure of whether and how a resource is likely to be affected by a given change in climate or factors driven by climate (Glick et al. 2011).

Adaptive Capacity: The ability of a resource to accommodate or cope with climate change impacts with minimal disruption (Glick et al. 2011).

Vulnerability: A function of the sensitivity of a particular resource to climate changes, its exposure to those changes, and its capacity to adapt to those changes (IPCC 2007b).

Development of Collaborative Process

This project used a collaborative, expert elicitation-based approach that involved representatives from the Nez Perce-Clearwater (NPCW) National Forest and USFS Northern Region. Expert elicitation has a long history in conservation and regulation. These approaches are effective where there is greater uncertainty about current system function or future projections but where there is a reservoir of detailed knowledge and expertise. Expert elicitation also has the benefits of being relatively rapid, encouraging ownership and buy-in, and lower cost. Further, participants in this process had extensive knowledge about the ecology, management, and threats to NPCW ecosystems and species, and also comprise many of the professionals who will use the results of the project.

Representatives from the NPCW National Forest and USFS Northern Region were invited to participate in the project through the Vulnerability Assessment Workshop.

Roles of the Vulnerability Assessment Workshop Participants

Using the vulnerability assessment model described below as a guide, workshop participants were asked to apply their knowledge and expertise about a selected resource (ecosystem, species, or ecosystem service) to evaluate its vulnerability to climate and non-climate stressors. Vulnerabilities of ecosystems and species are described in this report; vulnerabilities of ecosystem services are described in a companion report produced by Headwaters Economics.

Forest Resources

Selection Process

The initial list of forest resources (50+ resources including species, ecosystems, and ecosystem services) was developed internally by the NPCW National Forest and reviewed by the USFS Northern Region. The final list of 43 resources included 8 ecosystems, 27 species or species assemblages, and 8 ecosystem services. These were organized according to a set of agreed upon coarse filters (i.e., ecosystems) and fine filters (i.e., species or assemblages). Ecosystem services remained in a separate category altogether and are discussed in a companion report produced by Headwaters Economics.

Given the time and resources available, it was not feasible to apply the vulnerability assessment model to all 43 resources individually. Therefore, the following criteria were applied to select a subset of resources for consideration:

Ecosystems

Ecosystems, selected by the NPCW National Forest, were chosen based on those that best represented the major ecosystems of the region. The final list of eight ecosystems were evaluated during the Vulnerability Assessment Workshop; findings are for these assessments are described in Section 5 of this report.

Species

Species, also selected by the NPCW National Forest, were chosen based on those thought to be most vulnerable to climate change. However, to narrow down the list of species and assemblages to something more manageable, participants went through a quick exercise during the Vulnerability Assessment Workshop. Using the coarse filter/fine filter approach, wherein coarse filters represent ecosystems and fine filters focus on individual species not accounted for under the coarse filter, participants were asked to consider whether each species under a coarse filter was captured by the coarse filter level vulnerability assessment or whether species vulnerability needed to be assessed separately. For example, ponderosa pine, western larch, Douglas fir, and the grand fir/cedar/hemlock community were all evaluated as being captured by the coarse filter ecosystem assessment (e.g., either dry forest or mixed mesic communities) and therefore did not need to undergo a separate vulnerability assessment. However, whitebark pine, Canada lynx, and wolverine, for example, were selected for further assessment because they were identified as being more vulnerable (e.g., due to both climate and non-climate stressors) than their supporting ecosystems.

In total, twenty species were considered in the vulnerability assessment process. Vulnerability assessment summaries for these species were compiled and are described in Section 5.

Final Resource List

The final 28 ecosystems and species considered in the vulnerability assessment process included:

Ecosystems	**Species**
Aquatic	Bull trout
	Cutthroat trout
	Fall Chinook salmon
	Interior redband trout
	Spring Chinook salmon
	Steelhead
	Westslope cutthroat trout
Coastal disjunct	Red alder

Dry forest	Flammulated owl Lewis’s woodpecker Pygmy nuthatch White-headed woodpecker
Grassland/shrubland	Spalding’s catchfly
Mixed mesic	Fisher
Riparian	Coeur d’Alene salamander Giant salamander
Subalpine	Canada lynx Mountain goat Whitebark pine Wolverine
Wetlands, moist meadows, groundwater-dependent ecosystems	None

Vulnerability Assessment Model[6]

The vulnerability assessment model used in this process comprises three vulnerability components (sensitivity, adaptive capacity, and exposure), confidence evaluations for all components, and relative vulnerability and confidence for a resource (Figure 7). In this report, each component of vulnerability includes expert assigned rankings as well as narratives summarizing expert comments and information from the scientific literature. The aim of the narratives that accompany rankings is to make transparent the rationales and assumptions underlying the rankings and confidences assigned to each variable.

Sensitivity, adaptive capacity, and exposure components were broken down into specific elements better suited to assessing the vulnerability of particular resources for this assessment. For example, sensitivity comprises three main elements for ecosystems and five elements for species. Sensitivity elements for ecosystems include: sensitivity to climate (i.e., temperature and precipitation) and climate-driven changes (e.g., snowpack, soil moisture, low flows), disturbance regimes, and non-climate stressors. Elements for assessing species' sensitivity include: direct (e.g., physiology) and indirect (e.g., ecological relationships) sensitivities to climate and climate-driven changes, life history, non-climate stressors, and dependencies (e.g., dependence on sensitive habitats or specific prey). Sensitivity and adaptive capacity elements for ecosystems and species were informed by Glick et al. 2011, Manomet Center for Conservation Sciences 2012, and Lawler 2010. Exposure elements were created by EcoAdapt. Elements for each vulnerability component are described in more detail below.

Experts assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) for each component of vulnerability. Expert assigned rankings for each component were then averaged (mean) to generate an overall score. For example, rankings for each element of ecosystem sensitivity were averaged to generate an overall ecosystem sensitivity score. No scores were assigned for exposure; instead, experts were asked to rank, in order of importance, the exposure elements most important to consider for the ecosystem or species. Elements for each component of vulnerability were also assigned one of five confidence rankings (High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1). This ensured the degree of confidence assessors had in ranking each variable was explicit. Confidence rankings for each vulnerability component were averaged (mean) to generate an overall confidence score.

The user of these vulnerability results is encouraged to pay close attention to the narratives and individual rankings of sensitivity and adaptive capacity for each resource, rather than relative vulnerability rankings and summaries. Familiarity with each vulnerability component in addition to a resource's overall ranking allows one to better adapt one's understanding as exposure information varies (e.g., climate change projections vary depending on models used, greenhouse gas emissions scenarios, timeframes, etc.). This finer level of understanding better

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[6] This process was modeled after the Northeast Association of Fish & Wildlife Agencies (NEAFWA) Habitat Vulnerability Model (Manomet Center for Conservation Sciences 2012).

supports *why* a particular resource is vulnerable and *what* management actions may reduce vulnerabilities.

Further, the elements of adaptive capacity may not be independent. For example, areas with minimal human footprints are likely to be less fragmented, provide higher permeability, offer more refuge habitat, and exhibit higher levels of biotic and abiotic diversity (McKinney and Lockwood 1999). In contrast, the protected areas that tend to be more intact and diverse may also have laws that constrain management options for adaptation strategies. For example, many national parks prohibit prescribed burning, which is a common tool for increasing resilience through adaptive capacity. Managers may want to consider these tradeoffs as they develop adaptation strategies and management options for a particular resource.

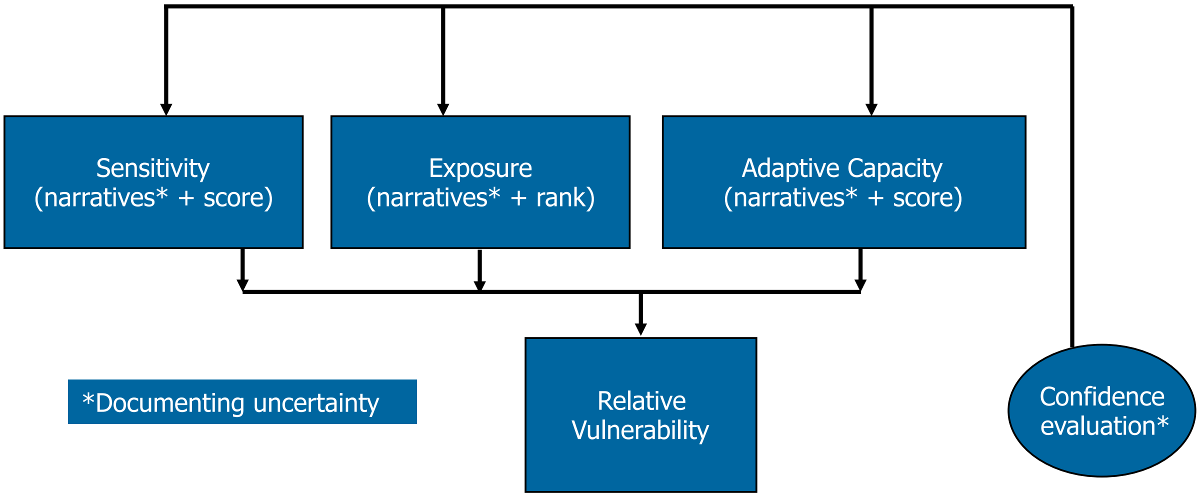

Figure 7. Structure of the vulnerability assessment model.

Model Elements – Ecosystems

This section lists the elements that were considered in the expert elicitation-based vulnerability assessment model for ecosystems. This list of elements for sensitivity and adaptive capacity were informed by Glick et al. 2011, Manomet Center for Conservation Sciences 2012, and Lawler 2010. Exposure elements were generated by EcoAdapt. The expert elicitation vulnerability assessment worksheets for ecosystems can be found on the EcoAdapt workshop support page[7].

Ecosystem Sensitivity & Exposure

1. Climate and Climate-Driven Changes. The two ways ecosystem sensitivity to climate and climate-driven changes were considered in this project included: (1) does the system inhabit a relatively narrow climatic zone, and (2) does the system experience large changes in composition or structure to small climate or climate-driven changes? Systems that inhabit a narrow climatic zone and/or experiences large changes in composition or structure in response to small changes in climate have higher sensitivity (Lawler 2010). Ecosystem benefits from climate and climate-driven changes were also considered.

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[7] http://ecoadapt.org/workshops/va-workshop-npc

2. Disturbance Regimes. Ecosystems may be sensitive to particular disturbance regimes such as wildfire, flooding, drought, insect and disease outbreak, or wind, among others. Ecosystems that experience larger changes in composition or structure due to small changes in disturbance regimes are likely more sensitive (Lawler 2010).

3. Future Climate Exposure. A number of climate and climate-driven factors may be important to consider for a system. These factors may include, but are not limited to: temperature, precipitation, climatic water deficit (i.e., reduced soil moisture), wildfire, snowpack, runoff, timing of flows, low flows, high flows, and stream temperature. Participants were asked to rank, in order of most important to least important, the climate and climate-driven factors most relevant to consider for the system and why, and document any potential areas of refugia.

4. Non-Climate Stressors. Other non-climate stressors have the potential to exacerbate the effects of climate change on ecosystems, or vice versa. Systems that have to endure multiple non-climate stressors are likely more sensitive to climate changes. Non-climate stressors can include land use conversion, agriculture and/or aquaculture, energy production and mining, transportation corridors, logging and wood harvesting, dams and water diversions, biological resource use (e.g., hunting, fishing), invasive and other problematic species, recreation, livestock grazing, fire suppression practices, or pollution and poisons, among others (Glick et al. 2011; Manomet Center for Conservation Sciences 2012). Participants were asked to identify non-climate stressors most likely to increase sensitivity of the system to climate change, assess degree stressor affects sensitivity and degree of current exposure to stressor, and evaluate confidence.

Ecosystem Adaptive Capacity

1. Extent, Integrity and Continuity. Ecosystems that are currently widespread in their geographic extent, with high integrity and continuity may be better able to withstand and persist into the future despite climate and non-climate stressors. Ecosystems that are degraded, isolated, limited in extent, or currently declining due to climate and non-climate stressors will likely have lower adaptive capacity (Manomet Center for Conservation Sciences 2012).

2. Landscape Permeability. More permeable landscapes with fewer barriers to dispersal and/or migration will likely result in greater adaptive capacity. The relative permeability of a landscape depends on natural and anthropogenic factors; for example, barriers to dispersal can include roads, land use conversion, logging and clear cuts, grazing, energy production and mining, dams and culverts, geologic features (e.g., mountains, rivers), or agriculture, among others (Lawler 2010; Glick et al. 2011).

3. System Diversity. Ecosystems with diverse physical and topographical characteristics (e.g., variety in aspects, slopes, and soil types) may be better able to persist under changing climate conditions than habitats that are less varied because they exist across widely differing conditions (Manomet Center for Conservation Sciences 2012). The level of diversity of component species and functional groups in the ecosystem may also affect the system's adaptive capacity to climate change impacts. For example, in ecosystems where each functional group is represented by multiple species, response to changes in climate varies among the species resulting in greater adaptive capacity (Glick et al. 2011). Dependency on a single keystone or foundation species can also affect the adaptive capacity of the system, contingent upon the species vulnerability to climate change.

4. Management Potential. Humans have the potential to intervene and change ecosystems in ways that reduce the impacts of climate change. For example, humans already control the flow regimes of most stream ecosystems (through dams) (Poff et al. 1997), so flow regimes could be manipulated to minimize stressful effects of climate change, such as low flows during late summer (Xu et al. 2010). The costs and benefits of management actions will vary among systems. Actions will be most feasible when resources are culturally and economically valued and the costs of implementing new management strategies are low. Further, use conflicts for the system (e.g., recreation or development pressure) may reduce the adaptive capacity of a system if management potential is low.

Model Elements – Species

This section lists the elements that were considered in the expert elicitation-based vulnerability assessment model for species. This list of elements for sensitivity and adaptive capacity were informed by Glick et al. 2011, Manomet Center for Conservation Sciences 2012, and Lawler 2010; exposure elements were generated by EcoAdapt. The expert elicitation vulnerability assessment worksheets for species can be found on the EcoAdapt workshop support page[8].

Species Sensitivity & Exposure

1. Direct Sensitivities. Physiological sensitivity is directly related to a species' physiological ability to tolerate changes in climate or climate-driven factors that are higher or lower than the range that they currently experience. Species life history may also be affected by changes in climate or climate-driven factors. Species that are able to tolerate a wide range of variables are likely less sensitive to climate change (Glick et al. 2011). Benefits to the species as a result of climate and climate-driven changes were also considered.

2. Indirect Sensitivities. Species sensitivity also likely depends on the sensitivities of ecological relationships and/or interspecific interactions. For example, the effects of climate or climate-driven changes on predator/prey relationships, foraging, habitat, pollination, dispersal, or competition, among others, are likely to influence a species' overall sensitivity to climate change. Benefits to the species' ecological relationships as a result of climate and climate-driven changes were also considered.

3. Future Climate Exposure. A number of climate and climate-driven factors may be important to consider for a species. These factors may include, but are not limited to: temperature, precipitation, climatic water deficit (i.e., reduced soil moisture), wildfire, snowpack, runoff, timing of flows, low flows, high flows, and stream temperature. Participants were asked to rank, in order of most important to least important, the climate and climate-driven factors most relevant to consider for the species and why, and document any potential areas of refugia.

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[8] http://ecoadapt.org/workshops/va-workshop-npc

4. Life History. Species reproductive strategy may influence sensitivity to climate change; for example, species with longer generation times and fewer offspring (K-selection) may be at increased extinction risk under long-term climate change. Species with a short generation time that produce many offspring (r-selection) may be better able to take advantage of climate changes (Glick et al. 2011).

5. Dependencies. Species that use multiple habitats, have multiple prey or forage species, or have multiple host plants are likely less sensitive to climate change (generalist). Conversely, species with very narrow habitat needs, single prey or forage species, or dependence on another sensitive species or habitat for life history purposes likely have greater sensitivity to climate changes (specialist). For example, species that depend on vernal pools or ephemeral wetlands, or live in alpine environments are likely to be susceptible to climate impacts such as increased temperatures or changes in precipitation regimes (Glick et al. 2011).

6. Non-Climate Stressors. Other non-climate stressors have the potential to exacerbate the effects of climate change on species, or vice versa. Species that have to endure multiple non-climate stressors are likely more sensitive to climate changes. Non-climate stressors can include land use conversion, agriculture and/or aquaculture, energy production and mining, transportation corridors, logging and wood harvesting, dams and water diversions, biological resource use (e.g., hunting, fishing), invasive and other problematic species, recreation, livestock grazing, fire suppression practices, or pollution and poisons, among others (Glick et al. 2011; Manomet Center for Conservation Sciences 2012). Participants were asked to identify non-climate stressors most likely to increase sensitivity of the species to climate change, assess degree stressor affects sensitivity and degree of current exposure to stressor, and evaluate confidence.

Species Adaptive Capacity

1. Extent, Status and Dispersal Ability. Species that are currently widespread in their geographic extent, with a robust population status, connectivity, and a high ability to disperse may be better able to withstand and persist into the future despite climate and non-climate stressors. Species that are endemic, endangered, or with isolated or fragmented populations and/or limited ability to disperse will likely have lower adaptive capacity (Manomet Center for Conservation Sciences 2012).

2. Barriers to Dispersal. In general, species that are poorer dispersers (disperse slowly and over short distances) are more susceptible to climate change and likely have less adaptive capacity (Glick et al. 2011). Similarly, the adaptive capacity of species with high innate dispersal ability may decrease if there are significant barriers to dispersal. Barriers to dispersal can include roads, land use conversion, logging and clear cuts, energy production and mining, dams and culverts, geologic features (e.g., mountains, rivers), fire suppression, grazing, or agriculture, among others (Lawler 2010).

3. Intraspecific/Life History Diversity. Species that demonstrate a diversity of life history strategies (e.g., variations in age at maturity, reproductive or nursery habitat use, or resource use) are likely to have greater adaptive capacity. Similarly, species able to express different and varying traits (e.g., phenology, behavior, physiology) in response to environmental variation have greater adaptive capacity than those that cannot modify their physiology or vary behavior to better cope with climate changes and its associated effects. Many species exhibit phenotypic plasticity in response to inter-annual variation in temperature and precipitation. Some species and/or populations will be better able to adapt evolutionarily to climate change. For example, species may have greater adaptive capacity if they exhibit characteristics such as faster generation times, genetic diversity, heritability of traits, larger population size, or multiple populations with connectivity among them to allow for gene flow.

4. Management Potential. Humans have the potential to intervene in ways that reduce the impacts of climate change on a particular species. For example, if a species is listed as threatened or endangered, it can provide opportunities for implementing specific management measures likely to help populations persist. The costs and benefits of management actions will vary among species. Actions will be most feasible when resources are culturally and economically valued and the costs of implementing new management strategies are low. Further, use conflicts for the species (e.g., recreation or development pressure) may reduce its adaptive capacity if management potential is low.

Confidence Evaluation

Each of the sensitivity, adaptive capacity, and exposure elements described above for resources were assigned a confidence rank: High, Moderate-High, Moderate, Low-Moderate, or Low. These approximate confidence levels were based on the 5-category scale developed by Moss and Schneider (2000) for the IPCC Third Assessment Report. This vulnerability assessment model not only assesses the confidence associated with the individual element rankings, but also uses these rankings to estimate the overall level of confidence for each component of vulnerability by calculating mean confidence rankings across elements.

Vulnerability Assessment Application

Model Application

EcoAdapt, in collaboration with the USFS and Headwater Economics, convened a 2-day workshop entitled *A Vulnerability Assessment Workshop for the Nez Perce-Clearwater National Forest*, held September 10-11, 2013 at the Nez Perce National Forest Supervisor's Office in Grangeville, ID. The main focus of the workshop was assessing the vulnerabilities of resources (ecosystems, species, and ecosystem services). Approximately twenty scientists and resource managers participated in this workshop from the NPCW National Forest and USFS Northern Region. Information from the workshop such as the agenda, presentations, handouts, readings, and other resources can be found on the workshop support page[9].

This workshop was structured to provide participants with a foundation of information from which they could assess the vulnerabilities of the selected resources. Participants were introduced to general vulnerability assessment theory and approaches (following the process described in Glick et al. 2011), provided with past and projected climate trends in the Nez Perce-Clearwater, and organized into several different small working group arrangements to discuss and evaluate the vulnerability of resources.

Workshop participants were directed to apply the vulnerability assessment model described above to the list of resources. As this was an expert elicitation process, participants were encouraged to make decisions based on their knowledge and expertise, and the workshop process and vulnerability assessment model were designed to be flexible to support collaborative on-the-fly modification and improvement.

Participant assessments and comments were compiled and assembled into this vulnerability assessment report. As part of this report, resource vulnerability briefings were created which synthesize participant comments and peer-review references for each resource. These vulnerability briefings are summarized in the next section of this report.

Model Application – peer review process

The draft vulnerability assessment report was sent to scientists and resource managers at the NPCW National Forest and USFS Northern Region for review. Comments and revisions from these reviewers were incorporated into the final report.

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[9] http://ecoadapt.org/workshops/va-workshop-npc

6. Vulnerability Assessment Results

Climate change is the most pressing challenge of our time, yet resource managers struggle to incorporate climate change into management decisions. Vulnerability assessments provide a foundation for understanding how and to what degree resources are threatened by climate change, and can help resource managers and conservation planners set management and planning priorities as well as enable more efficient allocation of resources. Vulnerability assessments are also the first step in developing strategies and improving management practice to better prepare for and respond to projected changes. Specifically, vulnerability assessments can be used to inform the development and implementation of adaptation strategies designed to reduce the vulnerability of resources to actual or expected climate change effects. Incorporating vulnerability and adaptation actions into management decisions will facilitate our ability to meet long-term goals for resources.

The following section presents individual climate change vulnerability assessment results for all twenty-eight Nez Perce-Clearwater ecosystems and species. The results are intended to help managers develop and prioritize adaptation strategies to conserve these resources in the face of climate change.

Aquatic Ecosystems[10]

Executive Summary

The relative vulnerability of aquatic ecosystems in this assessment is considered moderate, due to high sensitivity to climate-driven changes, moderate-high sensitivity to non-climate stressors, and moderate-high adaptive capacity. Aquatic ecosystems are sensitive to climate and climate-driven changes that affect hydrologic regimes including:

- increased stream temperatures,
- decreased snowpack,
- shifts from snow- to rain-dominant watersheds, and
- earlier snowmelt and runoff timing.

Increasing air temperatures (along with other local factors) influence stream thermal regimes, and warming stream temperatures may cause corresponding shifts in species distribution, phenology, and life histories. Changes in precipitation type, timing, and amount can lead to altered flow regimes with subsequent impacts on aquatic biota. For example, high flows resulting from earlier snowmelt and shifts from snow- to rain-dominant watersheds can affect stream complexity, alter bank morphology and stability, and increase woody debris recruitment in aquatic systems. Low stream flows resulting from decreased snowpack and earlier snowmelt may reduce connectivity of smaller streams to mainstream flows, as well as reduce habitat amount and quality.

Aquatic systems are also sensitive to non-climate stressors, including:

- transportation corridors,
- fire suppression practices,
- logging and timber harvest, and
- dams and water diversions.

Habitat homogenization (e.g., through road networks and maintenance, logging/harvest, channelization, etc.) can increase sedimentation rates and contribute to warming stream temperatures by reducing riparian vegetation. Dams impede aquatic network connectivity,

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[10] This assessment focuses on lotic ecosystems. The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

either as physical barriers or by creating new thermal zones, but could help mitigate low flows and warming temperatures driven by climate change. However, flow regulation prevents the maintenance of floodplain habitat (Poff et al. 1997, Stanford et al. 2005), and reduces the habitat heterogeneity that buffers organisms from environmental change. The adaptive capacity of aquatic systems in the Nez Perce-Clearwater (NPCW) region is moderate-high, due to its high physical and topographic diversity and fairly continuous nature, but somewhat degraded structural and functional integrity.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

The two most important physical attributes of lotic ecosystems are their flow and thermal regimes, which describe seasonal variation in water quantity and temperatures (Poff et al. 1997; Caissie 2006). These attributes are not independent; for example, the magnitude of stream flows mediates the seasonal dynamics of water temperature (Caissie 2006; Dickson et al. 2012; Armstrong and Schindler 2013). Changes in climate interact with hydrological conditions to influence these two key attributes of aquatic habitat. In particular, changes in precipitation type, timing, and amount can affect the timing and magnitude of high and low flow events. Lower flows and warming air temperatures (along with other local factors) can affect the duration and magnitude of high temperature events that exceed the physiological tolerances of coldwater species. Similarly, disturbance regimes such as wildfire can affect these two key aquatic habitat attributes. Overall, the sensitivity of aquatic systems to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[11]

It is important to note that the influence of shifts in precipitation type, timing, and amount, as well as stream temperature, will depend on location within a basin and stream network, as well as the physical context of a given stream reach. For example, just as topography can mediate the effects of climate change on terrestrial ecosystems (Dobrowski 2010; Sears et al. 2011), the diverse hydrologies found in heterogeneous watersheds can differentially filter the same regional climate signal (Armstrong and Schindler 2013).

Table 4. Hydrologic response relative to climate and climate-driven changes. Table modeled after Furniss et al. 2013.

Climate and climate-driven changes	Anticipated hydrologic response
Warmer air temperatures	• Warmer stream temperatures
Changes in precipitation (timing and amount)	• Possibility for increased landslides/erosion rates • Altered timing and volume of runoff and flows
Decreased snowpack and earlier snowmelt	• Higher winter flows • Lower summer flows
Shifts from snow- to rain-dominant watersheds	• Early winter peak flows and increased winter flood risk

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[11] Confidence associated with this evaluation was Moderate-High.

Stream water temperature varies within and among streams, and is influenced by factors such as topographic shade, upland and riparian vegetation, local air temperature and humidity, altitude, latitude, discharge, water source, and solar angle and radiation (Poole and Berman 2001; Ebersole et al. 2003). From 1980-2009, Isaak et al. (2011) found a net temperature increase in streams of the northwest (due primarily to warming air temperature), with rates of warming highest during the summer (~0.22˚C/decade). Wildfire can have long-term effects on stream temperatures by reducing shading (due to loss of streamside vegetation) or through channel widening (increases exposure to solar radiation) (Dunham et al. 2007). For example, within wildfire perimeters of a river network in central Idaho, stream temperatures were 2-3 times greater than basin averages (Isaak et al. 2010), and summer maxiumum water temperatures can remain significantly elevated for at least a decade following wildfire (Dunham et al. 2007). Similarly, summer water temperatures often exceeded 20˚C in small streams of burned watersheds compared with <15˚C in unburned watersheds (Minshall et al. 1997). Wildfires that cross shallow streams can also cause excessive temperatures resulting in fish mortality (Hitt 2003). Stream temperatures define habitable zones for almost all aquatic biota, and shifting thermal regimes may cause corresponding shifts in species distribution, phenology, and life histories (Rieman and Isaak 2010). For example, stream temperature increases over the past 20 years have resulted in an 11-20% loss of bull trout (*S. confluentus*) spawning and juvenile habitat, as cold headwater stream lengths warm beyond thermal tolerances (Isaak et al. 2010). As water temperatures rise, species may also become more susceptible to natural and introduced diseases (e.g., see McCullough 1999).[12]

Aquatic communities are highly adapted to and influenced by flow regimes, which play a key role in sustaining biodiversity and ecological integrity (Poff et al. 1997). Flow regime varies geographically in response to a number of factors including climate (precipitation and temperature), topography, geology, land cover, and position in the network. Alteration of flow magnitude, timing, frequency, duration, or rate of change of hydrologic conditions (i.e. flashiness) can induce a variety of environmental and biological responses, with subsequent impacts on biodiversity and ecosystem function (Poff et al. 1997). Variable flows create and maintain different habitat features and conditions that are essential to aquatic species.[13] Disturbances such as high flow events often reduce species abundance and suitable habitat in the short term, but maintain critical habitat features over longer time periods (Reeves et al. 1995; Stanford et al. 2005). Increased magnitude of spring discharge may have positive effects by restoring floodplain heterogeneity in dam-regulated rivers, which have lost their natural flood pulse. For example, frequent high flows transport sediment and organic resources, import large woody debris, scour floodplain soils, and shape fluvial environments (Poff et al. 1997; Yarnell et al. 2010; Luce et al. 2012), maintaining and/or enhancing productivity and diversity.

However, higher magnitude flood pulses may have negative results due to the immediate stress and mortality they can cause to aquatic organisms, particularly if high flow events exceed the

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[12] See the Aquatic Species section of this report for a summary of temperature effects on aquatic biota.

[13] See the Aquatic Species section of this report for a summary of high and low flow effects on aquatic biota.

magnitude that aquatic organisms have experienced in their evolutionary history or if anthropogenic habitat alterations have diminished the potential for organisms to find refuge during flood pulses (Waples et al. 2008a). The life histories of aquatic organisms are often synchronized to seasonal variation in flow levels (Lytle and Poff 2004). Altered timing of high flow events may reduce species abundance by causing phenological mismatch, in which organisms exhibit ontogenetic transitions at suboptimal times. While organisms may be able to adjust their phenologies through plastic or evolutionary responses, altered timing of life-history events may have cascading effects that negatively effect fitness during other portions of ontogeny (Waples et al. 2008b). Earlier spring runoff may protract periods of low flow and warm temperature, which are often stressful for culturally important organisms such as fishes. Similarly, prolonged low flows can result in the reduction or elimination of plant cover, diminished plant species diversity, or physiological stress in plants (see review in Poff et al. 1997).

Wildfire can be beneficial for aquatic systems by facilitating large woody debris recruitment, which increases aquatic habitat complexity and diversity (Reeves et al. 1995, Rieman and Isaak 2010). However, more frequent post-fire debris flows following extreme precipitation events may result in compromised aquatic habitats and/or reduced habitat connectivity (Rieman and Isaak 2010). For example, small stream channels in steep topographies may experience simplified habitats if high flows and debris flows cause scour and removal of woody debris, local alluvium, and bank soils. However, downstream channels will benefit from this same process, as more diverse habitats are created by wood and sediment deposition (Rieman and Isaak 2010).[14] Wildfire can also influence stream temperatures (see section above).

Future climate exposure

The most important climate and climate-driven factors to consider for aquatic ecosystems are those that alter hydrologic regimes and stream temperatures, including increasing air temperatures, decreased snowpack, shifts from snow- to rain-dominant watersheds, and earlier snowmelt and runoff timing. In addition, aquatic systems are also sensitive to altered disturbance regimes such as wildfire that can affect sedimentation and erosion and water temperature.

Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing, and are projected to continue to increase, with more significant increases in winter and summer (Littell et al. 2011; see Section 3). Air temperature is used as a proxy in estimating future stream temperature (Rieman et al. 2007; Wenger et al. 2011). However, recent research by Arismendi et al. (2012) indicates a less direct association between air and stream temperature trends, highlighting the importance of local, non-climatic factors in understanding future stream temperature trends. For example, the

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[14] For additional information on wildfire effects in riparian habitats, please refer to the Riparian Ecosystem section of this report.

effects of changes in air temperature on stream temperature depend on variables such as canopy cover and the residence time of water (Holtby 1988; Caissie 2006). In the summer, stream temperatures are projected to warm at rates of 0.3-0.45˚C per decade, causing a net increase of 1.2-1.8˚C by mid-century (Isaak et al. 2011). Further, stream isotherms may shift 5-143 km upstream if air temperatures rise by 2˚C (Isaak and Rieman 2013). In general, increasing air temperatures will contribute to reduced stream flows and warmer stream temperatures, which can create thermal and flow barriers and lead to shifts in species distribution and phenology (Rieman and Isaak 2010).[15] Refugia from reduced stream flow and elevated stream temperatures may include streams with significant groundwater influence and high elevation areas with intact riparian function.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Warmer air temperatures cause more precipitation to fall as rain rather than snow (Knowles et al. 2006), potentially shifting some streams from snow-dominated to transitional or rain-dominated, where the timing of flows is related to the timing of precipitation (Stewart et al. 2005; Littell et al. 2011). In the western U.S., this means more stream flow in fall and winter, and less in spring and summer (Elsner et al. 2010; Wenger et al. 2010). These changes are projected to occur soonest at mid-elevation sites (Regonda et al. 2005; Pierce et al. 2008; Nayak et al. 2010). A shift from snow to rain may also lead to large changes in hydrograph timing and magnitude (Elsner et al. 2010), and can lead to increased risk of mid-winter floods (Hamlet and Lettenmaier 2007; Luce et al. 2012). Flood magnitude is also projected to increase due to the increased occurrence of rain-on-snow events (e.g., in spring snowmelt-dominated basins; Hamlet and Lettenmaier 2007) and increased precipitation intensity (e.g., in rain-dominated basins; Easterling et al. 2000b). High flow events scour and deposit sediments, disturbing the benthos and reorganizing stream channels (Stanford et al. 2005). Increased winter flood incidence and disturbance along aquatic corridors can also affect stream complexity, alter bank morphology and stability, and increase woody debris recruitment in aquatic systems (Rieman and Isaak 2010; Yarnell et al. 2010). Changes in the magnitude and frequency of high flow events may increase mortality in aquatic organisms, or expose vulnerable life-stages (e.g., fry; see Fausch et al. 2001). However, high flow events may be critical to maintaining the habitat features that aquatic organisms depend on (Reeves et al. 1995).

About 75% of runoff in the western U.S. is currently derived from precipitation that falls as snow (Service 2004). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). As a result of snowpack declines, the spring freshet has decreased in both length and volume (Luce et al. 2012). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-

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[15] For a more complete summary of potential climate impacts on aquatic species, please refer to the Aquatic Species section of this report.

0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Warming temperatures have led to earlier runoff timing (Stewart et al. 2005) and lower summer flows (Rood et al. 2008; Luce and Holden 2009). Specifically, unregulated streams in Idaho experienced earlier peak stream flow and lower summer stream flows from 1967-2007 (Clark 2010). Similarly, in the northwestern U.S., historical changes show declines in stream flows over the last 50 years (Moore et al. 2007; Luce and Holden 2009; Clark 2010), including declines summer flows (Luce and Holden 2009; Leppi et al. 2011). However, there may be several factors that influence changes in stream flow aside from direct climate change including land use contributions, climate cycles (e.g., PDO), and evapotranspiration (Hoerling and Eischeid 2007), and it still remains a question as to whether the changes are precipitation or transpiration related (see Luce et al. 2012 for a full discussion on this). Earlier runoff timing and changes in seasonal flows (e.g., lower summer stream flows, higher winter flows) are projected to continue for the NPCW region (Wenger et al. 2010; Littell et al. 2011; see also Section 3). Changes in seasonal stream flow could be compounded during warm PDO cycles, which are correlated with reduced precipitation and may lead to overall annual declines in stream flow (Rood et al. 2005). Less snowpack that melts earlier also decreases available surface water and groundwater by limiting the duration of snowmelt-induced stream flow and groundwater recharge (Viers et al. 2013). Earlier snowmelt and decreased snowpack may also increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006). In watersheds with dense vegetation, low flow conditions may be exacerbated due to increased evapotranspiration from warmer air temperatures (Hamlet et al. 2007).

Changes in precipitation type, timing, and amount can have varying impacts on stream flow (Luce and Holden 2009). For example, high elevation headwaters may be more affected by variability in annual snowpack and higher summer temperatures whereas subalpine and montane forested riparian areas may be more affected by shifts in streamside microclimates (Luce et al. 2012). Altered stream flow (amount, timing, duration) contributes to changes in geomorphology and physical processes, and affects the ecological integrity of aquatic and riparian systems (Poff et al. 1997; Furniss et al. 2010). Changes in stream flow timing can disrupt cues for fish, modify aquatic food web structure, or affect riparian plant recruitment (see review in Poff et al. 1997). Low stream flows could reduce connectivity of smaller streams to mainstream flows, habitat amount and quality (e.g., by reducing the volume of pools or velocity of water), or delivery of food from upstream sources (Harvey et al. 2006; Furniss et al. 2010; Yarnell et al. 2010). Reduced stream flows may also cause some sections of stream to dry and become impassible to fish (Rieman and McIntyre 1996). Further, the magnitude of stream flows mediates the seasonal dynamics of water temperature (Caissie 2006; Dickson et al. 2012; Armstrong and Schindler 2013), and low flows during summer generate warmer stream temperatures (Caissie 2006). Changes in stream flow and patterns of groundwater recharge may also result in range contraction and/or local loss of species (Luce et al. 2012). Groundwater-dominated systems may buffer short-term variations in stream flow, although long-term drought (e.g., lasting several years) may increase sensitivity of these systems (Lall and Mann 1995; Shun and Duffy 1999). Further, in years with smaller snowpack, groundwater inputs may be diminished and headwater stream temperatures warmer (Isaak et al. 2011).

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Fire can induce hydrologic changes, including changes to snowmelt, runoff, peak flows, and low flows (Luce et al. 2012). Increased solar radiation post-fire has been linked to advances in snowmelt timing (1-2 weeks), with variable changes in peak flows (e.g., severely burned small basins experienced significant changes in peak flows but at the basin scale, no increase was observed) (Troendle et al. 2010; Luce et al. 2012). Changes in runoff following fire have been primarily attributed to changes in soil properties (i.e., surface sealing, water repellency) (Luce et al. 2012). Water repellency seems to be associated with certain vegetation communities (e.g., chaparral, subalpine fir), and is more likely where fires burn severely; further, dry, hot summers of the western U.S. may be ideal conditions for water repellent behavior (Luce et al. 2012). Peak flows in streams can be significantly higher post-fire than pre-fire, though it is more frequently documented that post-fire peak flow is less than pre-fire peak flow (Shakesby and Doerr 2006). Altered wildfire regimes can also affect fire-related sedimentation regimes; larger-scale fires may increase the number of riparian areas exposed to mudslides and overland flow, leading to increased channel disturbance from post-fire debris flows (Isaak et al. 2010; Rieman and Isaak 2010) and possible extirpations of local fish populations (Brown et al. 2001). Increasing wildfire frequency or severity can also increase the aerial extent of streams exposed to wildfire-induced high temperatures (e.g., by removing vegetation shading) (Dunham et al. 2007).

Again, it is important to note that the impacts of climate change will depend on location within a basin and stream network, as well as the physical context of a given stream reach.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of aquatic systems, including transportation corridors, fire suppression practices, logging and wood harvesting, dams and water diversions, invasive species, mining, recreation, and grazing.[16] Human land uses such as dams or water diversions, logging, mining, livestock grazing, channelization, transportation corridors, and development of floodplain habitat have reduced the diversity of habitat available for aquatic organisms as well as disrupted water quality.

Transportation corridors and road management can have negative impacts on the biotic integrity of aquatic ecosystems. Road networks affect the riparian environment by altering sediment delivery mechanisms, increasing fine sediment yields, increasing incidence of invasive species, modifying vegetation community structure and function, altering large woody debris recruitment, simplifying stream function, and providing travel routes for grazing animals (Williams 1954; Young et al. 1967; Roath and Krueger 1982; Trombulak and Frissell 2000). Channelization also reduces sinuosity and the habitat heterogeneity provided by an intact floodplain (Stanford et al. 2005). Generally, as the density of roads in a watershed increases, aquatic habitat quality decreases. Road construction causes severe disturbance to soils on slopes (Rieman and Clayton 1997) and, in a scientific literature review considering the effects of roads on aquatic systems, Trombulak and Frissell (2000) underscored the importance of avoiding construction of new roads and removing or restoring existing problematic roads. Roads built decades ago are often located in valley bottoms next to streams and are difficult to relocate (Swift and Burns 1999), and current recreation use and a lack of sufficient maintenance can result in increased sediment delivery (Grace and Clinton 2007). Timber harvest and road building can accelerate the frequency and volume of debris slides and hillslope sediment loss (Naiman et al. 2005). Road infrastructure, such as culverts, may also alter local hydrographs or create barriers to aquatic species dispersal, further reducing the connectivity of stream networks that may already be experiencing fragmentation due to thermal limits (Fausch et al. 2002, Rieman and Isaak 2010). Roads produce runoff and sediment in almost every precipitation event, which may have significant impacts on aquatic biota as they occur more frequently (Luce et al. 2012). Road maintenance or construction can also exacerbate climate-driven warming stream temperatures by removing riparian vegetation and reducing shaded stream portions (Isaak et al. 2010).

Fire suppression practices of the past century affect aquatic systems in a variety of ways. Without fire, aquatic systems are deprived of important nutrients and woody debris delivered by post-fire debris flows, which can lead to nutrient deficits and reduced habitat complexity and diversity. However, extensive wildfires can lead to increases in summer water temperatures, particularly in first- and second-order streams, decreases in habitat

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[16] The collective degree these stressors increase sensitivity of aquatic ecosystems was considered Moderate-High. Participant confidence associated with this evaluation was Moderate-High. Current exposure to these non-climate stressors in the NPCW region was judged to be Moderate by workshop participants (Confidence: Moderate-High).

heterogeneity, channel alterations, and significant restructuring and movement of large woody debris (Minshall et al. 1997). Fire suppression practices may also exacerbate shifting wildfire regimes, as they create the potential for more frequent and/or intense fires, and may increase the overall stream area exposed to temperature increases within burned areas (Isaak et al. 2010). Alternatively, fire suppression may be a necessary future tool to prevent immediate stream temperature rise in specified areas where critical habitat or small populations of sensitive fish species occur; however, immediate impacts of fire suppression should be weighed against longer-term consequences (Isaak et al. 2010).

Logging along riparian corridors adjacent to aquatic systems can both increase sedimentation rates and remove large tree species that provide crucial shade (Holtby 1988). These landscape alterations may exacerbate warming stream temperatures by increasing solar radiation, especially if changing climate conditions prevent reestablishment of pre-fire riparian vegetation types (Caissie 2006, Isaak et al. 2010).

Dams can provide both a threat and benefit to riparian ecosystems. Historically, dams have impeded aquatic network connectivity, either as physical barriers or by creating new thermal zones, but many have been modified to facilitate fish passage (Isaak et al. 2010). Climate changes may exacerbate fragmentation caused by dams by further thermally isolating local populations. Through flow regulation, dams can reduce peak flows for increased water storage and increase or decrease low base flows (Poff et al. 2011). In the Pacific Northwest, regulated streams have experienced an average flow decrease of 2.8% over the past thirty years, compared to an average flow decrease of 2.1% in unregulated streams (Littell et al. 2011). Changes in flow regime can affect riparian vegetation and aquatic biotia, as these organisms are adapted to their historic flow and thermal regimes and floodplain habitat, which dams and water diversions can significantly alter (Lytle and Poff 2004, Poff et al. 2007). For example, flow regulation can lead to declines in riparian vegetation richness and abundance (Ecovista et al. 2003; Beauchamp et al. 2006), increase invasive species establishment (Beauchamp and Stromberg 2007), or shift riparian plant composition (Hadley and Emmett 1998). However, dams that create temperature-stratified reservoirs may be beneficial to aquatic biota, as they periodically release cold water to downstream habitats and could help mitigate low flows and warming temperatures driven by climate change (e.g., see Null et al. 2011). Further, depending on where dams are located, they may be able to elevate low flows (Poff et al. 2006). However, release of this water in winter can affect thermal regimes in salmonid spawning habitat (e.g., see ODFW 2000), potentially affecting incubation rates and altering salmonid life-cycle phenology. Water diversions for human use can further exacerbate low flows and increasing stream temperatures.

Aquatic ecosystems were also considered sensitive to energy production and mining, invasive species, land use conversions, and livestock grazing by workshop participants, but current exposure to these stressors is considered low within the NPCW region.[17]

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW aquatic ecosystems was considered moderate-high by workshop participants and reviewers.[18] Aquatic ecosystems are transcontinental, featuring high physical and topographic diversity. Anadromous fish (i.e., Chinook and steelhead) were historically keystone species in the region, providing marine-derived nutrients for both aquatic and terrestrial systems and species. However, overall productivity in stream corridors now appears reduced compared to historic levels as salmon and steelhead numbers have declined over the past several decades. Aquatic systems are generally continuous in the NPCW region, but feature varied disruptions due to a variety of human-related activities (e.g., dams, habitat alteration on valley bottoms). Transportation corridors (i.e. road networks and maintenance) are the main drivers behind stream degradation and connectivity issues; logging and clear cuts, energy production and mining, and grazing also play a role, but occur on a smaller scale in the region. Land use conversions, agriculture, and dams and water diversions also impede aquatic system continuity and integrity, but occur only in very small parts of the NPCW region.

Management potential

Aquatic ecosystems were judged by workshop particpants to be very highly valued systems by the public. Workshop participants identified road networks and access, grazing, timber fuels management (particularly near riparian areas), and mining as potential use conflicts for the NPCW region. Management strategies from the peer-reviewed literature are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. Also, please refer to Rieman and Isaak (2010) for additional management actions that could be considered. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature-generated Proposed Management Strategies[19]:

- Focus on critical roads and relocate or restore them. Select critical roads by (1) identifying the highest ecological priority areas, (2) within those, identifying the most damaging roads, and (3) within those, identifying the roads that can be effectively decommissioned or mitigated (Luce et al. 2001).

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[17] For a summary of invasive species impacts on aquatic species, please see the Aquatic Species section of this report. For a summary of grazing and invasive species impacts on riparian habitats, please see the Riparian Ecosystems section of this report.

[18] Confidence associated with this evaluation was Moderate-High.

[19] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. See Luce et al. (2012) and Rieman and Isaak (2010) for additional management considerations.

  - Potential benefits: Minimize the harmful effects of roads on the aquatic environment, and limit the impacts of relocation/restoration.
  - Potential challenges: May require increased institutional capacity or funding.
- Focus thinning and other activities related to fuels reduction in areas with existing road systems, and use minimal impact harvest techniques (Brown et al. 2004b).
  - Potential benefits: Likely improve aquatic function.
- Limit intensity, range, and timing of grazing to protect the most sensitive riparian habitats, particularly those with low channel confinement, low gradient stream banks, and those with high exposure to future climate impacts (Ecovista et al. 2003).
  - Potential benefits: Reduce sediment loads, decrease incidence of invasive species, and improve system resilience.

Coastal Disjunct Ecosystem[20]

Executive Summary

The relative vulnerability of the coastal disjunct ecosystem in this assessment is considered moderate, due to its moderate-high sensitivity to climate-driven changes and non-climate stressors, and moderate adaptive capacity. Coastal disjunct communities are sensitive to climate-driven changes such as:

- reduced soil moisture,
- drought,
- extreme temperature events, and
- wildfire.

Drier and hotter conditions, which lead to seasonal soil moisture deficits, could cause higher root disease mortality and/or habitat conversion. Altered fire regimes that result in lethal crown fires or stand-replacing fires may restrict the regeneration capacities of the coastal disjunct system.

Coastal disjunct systems are also sensitive to several non-climate stressors including:

- timber harvest,
- fire suppression practices,
- grazing, and
- recreation.

These non-climate stressors can exacerbate system sensitivity by amplifying the effects of climate-driven changes. For example, denser forests resulting from fire suppression practices may experience higher soil moisture stress due to increased competition and, during times of system-wide drought, could contribute to higher tree mortality rates and/or facilitate the spread of root disease. The adaptive capacity of the coastal disjunct ecosystem is considered moderate, due to low geographic extent and integrity but high component species and functional group diversity as well as high societal value.

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[20] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

Coastal disjunct ecosystems typically occur in areas with moderate air temperatures and abundant, well-balanced soil moisture and precipitation. Coastal disjunct species (e.g., western red cedar, *Thuja plicata*, and red alder, *Alnus rubra*) are often scattered amongst other mixed mesic species and occur along riparian areas or wetlands with abundant sub-surface moisture or in upslope sites with perched water tables (Scott et al. 2013). The coastal disjunct system exhibits sensitivity to drought and reduced soil moisture, extreme temperatures, and wildfire. Overall, the sensitivity of the coastal disjunct ecosystem to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[21]

Table 5. Potential coastal disjunct response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated coastal disjunct response
Drought and reduced soil moisture	• Upland edges dry out and/or convert to shrub or dry-adapted species • Increased vulnerability to root pathogens
Extreme temperature (hot or cold) events	• Range contraction • Reduced seedling survival
More frequent or severe wildfire	• Reduced regeneration potential of component species, potential habitat conversion to more fire-tolerant, disturbance-adapted species

Drought and reduced soil moisture pose the greatest climate-driven threat to the coastal disjunct system. For example, if overall moisture availability decreases, the drier, upland edges of the coastal disjunct system may dry out and be converted to drier shrub or forest types, especially in sites with southern exposures where soil moisture deficits will likely increase (Bollenbacher et al. 2013; Scott et al. 2013). Additionally, heightened moisture stress can also increase the vulnerability of coastal disjunct species to root pathogens (Scott et al. 2013). Extreme temperature (hot or cold) events may contract the range of this ecosystem and limit survivorship of seedlings (Bollenbacher et al. 2013; Scott et al. 2013).

Altered wildfire regimes also present a potential threat to the coastal disjunct ecosystem. Coastal disjunct ecosystems feature historically low fire frequency and mixed fire severity, and can go over 250 years without a stand-replacing fire (Scott et al. 2013). If wildfires become more frequent or more severe, coastal disjunct ecosystems will struggle to naturally regenerate due to reduced seed source and degradation of their required moist microclimate. Also, due to fire suppression practices of the past century, coastal disjunct ecosystems now often have higher tree densities, increasing the likelihood of more intense stand-replacing fires during dry summers. Increasing wildfire frequency and/or severity might result in drier margins of the

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[21] Confidence associated with this evaluation was Moderate.

coastal disjunct ecosystem being replaced by other more fire-tolerant and disturbance-tolerant species (Bollenbacher et al. 2013).

Future climate exposure

Climate and climate-driven factors most relevant to consider for the coastal disjunct ecosystem include increased drought, reduced soil moisture, high temperatures, and increased wildfire frequency and severity. On average, precipitation across the Nez Perce-Clearwater (NPCW) region has not been notably different during recent years (i.e. last 30 years) compared to 1901-1980, with increased precipitation in spring and summer, and decreased precipitation in winter (see Section 3). Future precipitation trends are difficult to predict, though many models indicate no overall annual precipitation changes for the NPCW region through 2040, with precipitation increases occurring in spring and winter and decreases occurring in summer (Hamlet et al. 2005; Morgan et al. 2008; Littell et al 2011; see also Section 3).

Warming temperatures and precipitation decreases (e.g., due to decreased snowpack and earlier snowmelt) may increase the length of the summer drought season (Stewart et al. 2009) and may lead to earlier desiccation of soils (Hamlet et al. 2007). July 1 soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see also Section 3) and, in general, drought frequency and severity could increase throughout the western U.S. (IPCC 2007a). Increased drought and reduced soil moisture may increase water stress in coastal disjunct vegetation, potentially constricting the coastal disjunct system to the more mesic expanses of its current range (Bollenbacher et al. 2013; Scott et al. 2013) and/or enhancing vulnerability of component species to root pathogens (Scott et al. 2013).

Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Extreme heat days are projected to both increase in frequency and last 3-6 days longer by the end of the 21st century (Diffenbaugh et al. 2005). Warmer temperatures, extreme heat events, and decreased summer precipitation could compound soil moisture declines due to increased evapotranspiration and earlier soil moisture recharge, particularly on southern aspects. This could cause a contraction of the coastal disjunct ecosystem range away from warmer and increasingly xeric areas and contribute to higher root disease mortality (Bollenbacher et al. 2013; Scott et al. 2013). Potential areas of refugia from low soil moisture and extreme temperature include north-facing slopes and areas with abundant groundwater. However, dispersal to these refugia zones may be difficult for coastal disjunct species, particularly in a fragmented landscape (Gavin 2009). Spatial patterns indicate that several coastal disjunct species (e.g., western red cedar) have not reached their potential distribution despite the presence of favorable refugia, which may indicate limited dispersal ability (Gavin 2009).

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes (Holden and Jolly 2011). The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). Altered fire regimes and more stand-replacing fires could lead to regeneration issues for the coastal disjunct ecosystem due to reduced seed source (Bollenbacher et al. 2013; Scott et al. 2013). Refugia zones may include north-facing slopes and continued colonization of riparian areas.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of coastal disjunct systems, including timber harvest, fire suppression practices, recreation, livestock grazing, energy production and mining, transportation corridors, and dams and water diversions.[22] The coastal disjunct ecosystem occupies small patches across the NPCW region, occurring in riparian areas, wetlands, and in upslope sites with saturated soils (Scott et al. 2013). The upland portions of this ecosystem may be more sensitive than those in wetlands or along riparian areas due to higher current exposure to timber harvest practices. Large-scale disturbances, such as clear cutting, may accelerate upland ecosystem conversion to drier shrub habitats as it allows for fast-colonizing shrub species to invade and outcompete seral tree species (Scott et al. 2013). Further, the drier edges of the upland coastal disjunct system are already facing conversion to drier shrub and/or forest systems due to reduced soil moisture availability. The combination of harvest-induced disturbance and reduced soil moisture could accelerate and/or increase the percentage of upland habitat facing conversion to shrublands (Bollenbacher et al. 2013; Scott et al. 2013).

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[22] The collective degree these stressors increase sensitivity of coastal disjunct ecosystems was considered Moderate-High. Participant confidence associated with this evaluation was Moderate. Current exposure to these non-climate stressors in the NPCW region was judged to be Low-Moderate by workshop participants (Confidence: Low-Moderate).

The coastal disjunct system is a historically dense system with long fire return intervals, but fire suppression practices have resulted in stand densities higher than historically present. This new ecosystem structure could cause both more severe wildfires, especially dangerous crown fires, as well as exacerbate system sensitivity to other climate stressors. For example, denser forests resulting from fire suppression practices may experience higher soil moisture stress due to increased competition and, during times of system-wide drought, could contribute to higher tree mortality rates and/or facilitate the spread of root disease (Scott et al. 2013). Additionally, denser forests and lack of fire can alter succession patterns, excluding shade-intolerant and early successional species (Bollenbacher et al. 2013).

The coastal disjunct ecosystem is also sensitive to recreation and livestock grazing practices, which disturb and alter the landscape. Large-scale disturbance could increase invasion rates of shade-tolerant weeds, exotic plant species, and upland dry shrubs, which could increase overall competition for soil moisture and/or facilitate ecosystem conversion (NPCW Forest Plan Assessment 2012c; Bollenbacher et al. 2013; Scott et al. 2013). Similar to the effects of harvest-induced disturbance, grazing or recreation-induced disturbance could compound climate-related soil moisture deficits by accelerating and/or increasing the percentage of the coastal disjunct system facing habitat conversion (Bollenbacher et al. 2013; Scott et al. 2013).

Energy production and mining, transportation corridors, and dams and water diversions were also identified by workshop participants as potential non-climate stressors to the coastal disjunct ecosystem; however, the current degree of exposure to these stressors is thought to be low within the NPCW region.

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW coastal disjunct ecosystems was considered moderate by workshop participants and reviewers.[23] The coastal disjunct ecosystem is somewhat isolated, and exhibits low structural/functional integrity but high component species and functional group diversity. Though coastal disjunct ecosystems occur across the West, with a geographic extent including Washington, Oregon and California, the Idaho portion is isolated from these other regions. The coastal disjunct ecosystem exists only in limited, “patchy” areas due to its moist microclimate requirements, limited dispersal ability of component vascular plant species (Gavin 2009), and current barriers to system continuity including timber harvest, land use conversion, and agricultural practices in Palouse (Bollenbacher et al. 2013). Given potential dispersal limitations of component species, the coastal disjunct ecosystem could struggle to adapt to climate change on fragmented landscapes (Gavin 2009). Though it has low topographical diversity, the coastal disjunct ecosystem has very high component species diversity and functional group diversity.

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[23] Confidence associated with this evaluation was Moderate.

Management potential

Coastal disjunct ecosystems were judged by workshop particpants to be a highly valued system by the public. Workshop participants identified timber harvest, grazing, recreational OHV use, and mining as potential use conflicts for the NPCW region. Workshop participants proposed that either eliminating timber harvest or using selective thinning practices could reduce disturbance and protect the coastal disjunct system both now and in the future. In addition, peer-reviewed literature suggests that thinning dense stands and thinning in robust areas (e.g., areas with more soil moisture) could also be used to maintain coastal disjunct system resilience and protect more vulnerable areas (e.g., dry, upland sites) (Scott et al. 2013). Workshop participants also recommended evaluation of coastal disjunct systems for grazing suitability/unsuitability, and that the system be closed to OHV use. These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Workshop-generated Proposed Management Strategies[24]:

- Eliminate harvest and/or use selective thinning in coastal disjunct areas.
  - Potential benefits: Reduce disturbance and disturbance-related habitat conversion, increase overall system resilience.
  - Potential challenges: Possible conflicts with timber harvest interests.
- Conduct grazing suitability assessments for coastal disjunct stands.
  - Potential benefits: May identify most vulnerable coastal disjunct stands that can be prioritized for grazing exclusion.
  - Potential challenges: May require additional institutional capacity.
- Prevent OHV use in coastal disjunct stands.
  - Potential benefits: Limit disturbance events that could compound climate-driven changes.
  - Potential challenges: Difficult to regulate and enforce, may have little impact.

Literature-generated Proposed Management Strategy[25]:

- Thinning in resilient coastal disjunct areas (e.g., areas with high relative soil moisture), and overly dense stands.
  - Potential benefits: Protect driest (most vulnerable) sites by reducing disturbance-induced habitat conversion (Scott et al. 2013), increase local moisture levels, increase stand resilience, reduce stand/fuel density and crown fire risk, reduce root pathogen risk (Bollenbacher et al. 2013; Scott et al. 2013).
  - Potential challenges: Possible conflicts with timber harvest interests.

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[24] Workshop-generated management strategies were developed by participants at the Nez Perce-Clearwater Vulnerability Assessment Workshop held in September 2013.

[25] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

Dry Forest Ecosystems[26]

Executive Summary

The relative vulnerability of the dry forest ecosystem in this assessment is considered high, due to its high sensitivity to climate-driven changes and non-climate stressors, and low-moderate adaptive capacity. Dry forest communities are sensitive to climate and climate-driven changes including:

- drought,
- soil moisture, and
- wildfire.

Drought and soil moisture changes could alter the range and species composition of dry forests. For example, dry forest species may be able to expand into drier mixed mesic zones, but soil moisture deficits could also increase mortality of old growth ponderosa pines. Low severity, high frequency fires may increase dry forest area, but in areas where fire suppression practices have created denser, structurally homogeneous dry forest communities, lethal crown fires can occur leading to seed source and nutrient loss and regeneration issues for dry forest species.

Dry forest systems are also sensitive to non-climate stressors, including:

- fire suppression practices, and
- insect and disease outbreaks.

These non-climate stressors may exacerbate climate-driven changes. For example, fire suppression practices have increased forest density and altered forest structure, increasing the likelihood of more frequent and severe forest fires. Climate change is projected to cause significant growth of bark beetle populations, increasing the risk of widespread beetle-related mortality. Within the Nez Perce-Clearwater (NPCW) region, the dry forest ecosystem exhibits low-moderate adaptive capacity due to its degraded condition, and moderate physical and topographical diversity.

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[26] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

Dry forest ecosystems, which include Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) species, generally occur in warm, low elevation areas where moisture is limited, exhibiting high sensitivity to soil moisture and drought, and high sensitivity to wildfire. Dry forest ecosystems may demonstrate some sensitivity to warmer temperatures and precipitation changes, however these changes may benefit the system. Overall, the sensitivity of dry forest ecosystems to climate and climate-driven changes was evaluated as high by workshop participants.[27]

Table 6. Potential dry forest response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated dry forest response
Warmer temperatures	• Shifts to higher elevations (Douglas fir) • Reduced regeneration at lower elevations
Precipitation changes	• Increased habitat area • Increased colonization opportunities
Reduced soil moisture	• Increased old growth ponderosa pine mortality due to competition with dense, young stands • Limited establishment during summer • May expand into mixed mesic habitat
Drought	• Limited establishment during summer • May expand into mixed mesic habitat
Wildfire	• Low severity, high frequency fires: Increased dry forest extent and return to open forest structure • Crown re-burns: Reduced seed source and regeneration problems post-fire

Douglas fir is limited by cold temperatures at elevation (K. Hazelbaker, pers. comm., 2014) and by moisture in the southern parts of its range (Scott et al. 2013). Ponderosa pine is primarily limited by moisture availability, although young seedlings are susceptible to cold nighttime temperatures (Scott et al. 2013). Precipitation and drought influence ponderosa pine and Douglas fir establishment, as both species have limited establishment and growth opportunities during the dry summer months of July and August (Scott et al. 2013). Soil moisture deficits may also increase old growth ponderosa pine mortality due to heightened competition with dense stands of young trees (Mehl and Haufler 2001). Alternatively, changes in precipitation (either more or less) may create more areas for dry forest types. For example, ponderosa pine can colonize hot, dry conditions of a disturbed site and/or expand into drier mixed mesic areas (K.

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[27] Confidence associated with this evaluation was Moderate-High.

Hazelbaker, pers. comm., 2014). Similarly, during cool, wet climates Douglas fir or denser ponderosa pine can also become established (Bollenbacher et al. 2013).

Wildfire appears to be the most significant climate-driven change affecting dry forest ecosystems. Low severity, high frequency fires - representative of historic fire regimes- may actually increase dry forest extent. For example, low severity fires result in thick barked ponderosa pine surviving and thin barked Douglas fir or ponderosa pine seedlings dying, allowing ponderosa pine forest to develop into large patches of open, old growth structure able to persist for long periods of time (Bollenbacher et al. 2013). Similarly, large crown fires that occur in denser Douglas fir or ponderosa pine stands may help re-establish open forest structure and bring areas back to the initial stand establishment phase. However, wildfires that re-burn large, crown-fire burn areas may limit forest establishment due to loss of seed source, limited soil moisture, and high surface soil temperature (Bollenbacher et al. 2013). In addition, current dry forests feature altered forest structure and composition (e.g., increased forest density and increased occurrence of small ponderosa pine, Douglas fir, and grand fir (*Abies grandis*) compared to historic patterns of old growth ponderosa pine), which increases the risk of extensive crown fires and resultant nutrient and microbe loss, which can limit dry forest regeneration (Jain and Graham 2005).

Future climate exposure
Climate and climate-driven factors most relevant to consider for the dry forest ecosystem include wildfire, soil moisture and drought, and temperature. The dry forest ecosystem historically featured frequent, low-severity fires and only occasional stand-replacement fires (Scott et al. 2013). Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes (Holden and Jolly 2011). The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). Dry forests are adapted to

fire in open forest conditions, but uncharacteristic fire behavior combined with denser forests could limit dry forest regeneration by increasing crown fire frequency, reducing seed sources, and reducing nutrient availability (Jain and Graham 2005; Turner et al. 2012; Bollenbacher et al. 2013).

Although dry forest ecosystems currently occur in areas where moisture is limited, warming temperatures and precipitation decreases (e.g., due to decreased snowpack and earlier snowmelt) may increase the length of the summer drought season (Stewart et al. 2009) and may lead to earlier desiccation of soils (Hamlet et al. 2007). July 1 soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see also Section 3). Reduced soil moisture may result in the forest edge moving up slope, particularly on southerly aspects where moisture deficits are greater (Bollenbacher et al. 2013). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Ponderosa pine is the most heat- and drought-adapted conifer species in the NPCW region, and may actually expand its range into Douglas fir and/or mixed mesic habitat if soil moisture deficits increase (Scott et al. 2013; K. Hazelbaker, pers. comm., 2014). However, old growth ponderosa pine may face heightened mortality during times of moisture stress due to competition from high-density stands of small ponderosa pine individuals (Mehl and Haufler 2001). Refugia areas from moisture deficits may include more mesic zones, which may be currently occupied by other systems and species.

Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Warmer temperatures may provide opportunities for Douglas fir expansion into higher elevation areas where they are currently temperature-limited (K. Hazelbaker, pers. comm., 2014) while simultaneously reducing tree regeneration at lower elevations (Turner et al. 2012).

Sensitivity and current exposure to non-climate stressors

Workshop participants and peer-reviewers identified several non-climate stressors that affect the sensitivity of dry forest systems, including fire suppression practices and insect and disease outbreaks.[28] The key non-climate stressors impacting dry forest ecosystems are fire suppression practices and insect and disease outbreaks. Fire suppression allows for Douglas fir, grand fir or denser ponderosa pine to become established relatively quickly (decades rather than centuries), resulting in a high density, stressed and structurally homogeneous forest (Bollenbacher et al. 2013). This new structure and increased biomass can enhance the risk of lethal crown fires, which can reduce regeneration success of dry forest stands following disturbance (Turner et al. 2012), and may accelerate shifting fire regimes. However,

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[28] The collective degree these stressors increase sensitivity of dry forest ecosystems was considered High. Participant confidence associated with this evaluation was High. Current exposure to these non-climate stressors in the NPCW region was judged to be High by workshop participants (Confidence: Moderate-High).

management techniques such as thinning can help return dry forests to a density more typical of a low severity fire regime (Mehl and Haufler 2001; Bollenbacher et al. 2013) and help re-establish spatial heterogeneity within dry forest stands.

Both Douglas fir and ponderosa pine are sensitive to bark beetle infestations, which may become more common under projected climate change scenarios (Fettig et al. 2007; Bollenbacher et al. 2013). Bark beetles and other herbivorous insects have played important historical roles in forest ecosystems by acting as major mortality agents, reducing tree competition and restarting successional stages. However, climate change impacts are projected to cause significant growth of bark beetle populations and disrupt these historical, co-evolved normative relationships (Logan et al. 2003). Increasing temperatures may reduce cold mortality and increase the reproductive cycle of these bark beetles, while precipitation changes and moisture stress will make tree species more susceptible to infestation, increasing the risk of widespread beetle-related mortality (Fettig et al. 2007). Increased bark beetle herbivory can lead to more woody fuel, which can enhance the likelihood of large, stand-replacing fires (Logan et al. 2003; Bollenbacher et al. 2013). However, current research indicates that recent bark beetle outbreaks (4-13 years prefire) may not affect subsequent fire severity (Harvey et al. 2013). Increased bark beetle mortality can also facilitate vegetation shifts, which could alter species composition and function within the dry forest ecosystem (Fettig et al. 2007). Douglas fir is also highly sensitive to a variety of root pathogens, especially during times of heightened moisture stress, which may facilitate conversion to ponderosa pine-dominated landscapes in the future (Scott et al. 2013).

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW dry forest ecosystems was considered low-moderate by workshop participants and reviewers.[29] The dry forest system can be found widely across the western U.S., and while the system occurs fairly continuously within the NPCW region, historic dry forest structure and composition occurs only in isolated patches (Mehl and Haufler 2001). Dry forest systems are considered to be very degraded in the NPCW region (Mehl and Haufler 2001), mainly due to shifts in forest structure, composition, and density. For example, 85-98% of old growth, low density ponderosa pine stands have been lost and replaced with high density stands of small to medium sized grand fir, Douglas fir, and ponderosa pine, which are more vulnerable to climate change impacts such as increased fire severity and soil moisture stress (Mehl and Haufler 2001). Diversity of the system – both in terms of physical and topographical diversity as well as component species diversity – is considered moderate. Keystone species for the ecosystem include ponderosa pine and Douglas fir.

Management potential

Dry forest ecosystems were judged by workshop particpants to be a highly valued system by the public. Workshop participants identified wildland-urban interface (WUI) fire management

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[29] Confidence associated with this evaluation was Moderate-High.

as a potential use conflict. Increasing public education and outreach around fire-resistant homes and landscapes, especially for residents living in/near the WUI, could help reduce fire risk in the dry forest ecosystem both now and in the future. In addition, peer-reviewed literature suggests using managed disturbance (e.g., thinning and prescribed fire) to restore old growth ponderosa pine stands to their historic open structure (Mehl and Haufler 2001; Bollebacher et al. 2013). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Workshop-generated Proposed Management Strategy[30]:

- Increase public education and outreach around fire-resistant homes and landscapes, especially for residents living in/near WUI.
    - Potential benefits: Reduce fire risk.
    - Potential challenges: Funding, staff time, institutional capacity, public engagement.

Literature-generated Proposed Management Strategy[31]:

- Use targeted thinning of smaller individuals followed by re-introduction of low severity, high frequency fire regimes to restore open, old growth ponderosa pine stands (Mehl and Haufler 2001; Bollenbacher et al. 2013).
    - Potential benefits: Re-establish open forest structure, increase system resilience, reduce fuel density and crown fire risk, increase soil moisture, reduce insect and disease vulnerability (Fettig et al. 2007; Bollenbacher et al. 2013).
    - Potential challenges: May increase rates of savannification (Scott et al. 2013).

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[30] Workshop-generated management strategies were developed by participants at the Nez Perce-Clearwater Vulnerability Assessment Workshop held in September 2013.

[31] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

Grassland Ecosystems[32]

Executive Summary

The relative vulnerability of the grassland ecosystem in this assessment is considered moderate, due to its low sensitivity to climate changes, moderate sensitivity to non-climate stressors, and moderate adaptive capacity. Grassland ecosystems are generally only slightly sensitive to climate and climate-driven changes, including:

- altered wildfire regimes,
- precipitation changes, and
- drought.

Altered fire regimes could increase invasion rates and colonization of several weed species (e.g., cheatgrass), but could also facilitate grassland expansion into burned conifer systems. Precipitation changes have varying impacts as well. For example, increased precipitation could increase soil moisture and allow for colonization of other moisture-dependent species (e.g., conifers) while decreased precipitation combined with warmer temperatures could allow grassland range expansion by stressing adjacent conifer systems.

Grassland systems appear much more sensitive to non-climate stressors including:

- invasive species,
- fire suppression practices, and
- grazing.

Invasive plants can outcompete native species, alter ecological processes, and alter habitat suitability. Grazing-related disturbances, particularly overgrazing, can also lead to an increase in bare ground, which facilitates germination and higher densities of invasive plants. These non-climate stressors may exacerbate system sensitivity to climate changes. For example, livestock grazing causes soil compaction and increased bare ground, which can increase runoff and erosion following rain events and reduce the amount of productive topsoil needed to sustain plant communities. However, targeted, light grazing may help reduce non-native plant abundance. The adaptive capacity of grassland ecosystems is moderate, due to moderate-high

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[32] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

physical, topographical, and component species diversity but altered structural and functional integrity.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

The Palouse Prairie grassland ecosystem in the Nez Perce-Clearwater (NPCW) region exhibits sensitivity to wildfire, precipitation changes, and drought. Overall, the sensitivity of grassland ecosystems to climate and climate-driven changes was evaluated as moderate by workshop participants.[33]

Table 7. Potential grassland response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated grassland response
Wildfire	• Increased frequency or severity: Limited native species regeneration, increased invasive species establishment • Reintroduction of natural fire regimes: Promote grassland re-establishment by removing and excluding conifers
Precipitation increases	• Habitat conversion (e.g., conifer encroachment)
Drought	• Degradation of grassland system and component species

Altered fire regimes, particularly increased fire severity, may limit regeneration of native species such as Idaho fescue (*Festuca idahoensis*) and bluebunch wheatgrass (*Pseudoroegneria spicata*) (Ecovista et al. 2003). More frequent or severe wildfires could also increase invasion rates and dominance of non-native annual grasses and weed species including yellow starthistle (*Centaurea solstitialis*) and cheatgrass (*Bromus tectorum*) (Ecovista et al. 2003; Bradley et al. 2009; NPCW Forest Plan Assessment 2012b; Bollenbacher et al. 2013).

Precipitation changes, especially increased precipitation, could cause habitat conversion of grassland ecosystems. Contemporary grasslands are accustomed to dry summers and limited precipitation (~10-30 inches per year) distributed evenly over fall, winter, or spring. Increased precipitation could increase soil moisture and allow for colonization of other moisture-dependent species, such as conifers (NPCW Forest Plan Assessment 2012a; Bollenbacher et al. 2013). While grassland ecosystems are adapted to dry summer conditions, they still rely on ample growing season precipitation. Prolonged summer drought or drought occurring during the growing season could lead to grassland degradation, particularly for bluebunch wheatgrass on flat landscapes (Bollenbacher et al. 2013).

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[33] Confidence associated with this evaluation was Moderate.

Grassland ecosystems could experience a range expansion following higher temperatures and/or decreased precipitation, as these changes can stress adjacent conifer systems leading to habitat conversion to grasslands.

Future climate exposure

Wildfire is the most significant climate-driven factor to consider for grassland ecosystems. Frequent, low severity wildfires are a historical component of Palouse Prairie grasslands (Bowker et al. 2004), and have traditionally played an important role in enhancing the cover of certain native perennial species such as bluebunch wheatgrass (Ecovista et al. 2003; Bowker et al. 2004). However, in the NPCW region wildfires are projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. Altered fire regimes may cause increased invasive annual weed encroachment within the grasslands ecosystem while simultaneously inhibiting the regeneration of native forbs and grasses (Ecovista et al. 2003; Bradley et al. 2009; Bollenbacher et al. 2013). Alternatively, altered fire regimes that resemble historic fire regimes in areas where fire suppression practices have occurred may benefit grassland systems by removing and excluding conifers and allowing for grassland reestablishment (Heyerdahl et al. 2006).

On average, precipitation across the Nez Perce-Clearwater (NPCW) region has not been notably different during recent years (i.e. last 30 years) compared to 1901-1980, with increased precipitation in spring and summer, and decreased precipitation in winter (see Section 3). Future precipitation trends are difficult to predict, though many models indicate no overall annual precipitation changes for the NPCW region through 2040, with precipitation increases occurring in spring and winter and decreases occurring in summer (Hamlet et al. 2005; Morgan et al. 2008; Littell et al 2011; see also Section 3). Warming temperatures and precipitation decreases (e.g., due to decreased snowpack and earlier snowmelt) may increase the length of the summer drought season (Stewart et al. 2009) and may lead to earlier desiccation of soils (Hamlet et al. 2007). These climate and climate-driven changes may or may not affect grassland systems.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of grassland ecosystems, including invasive weeds, fire suppression practices, and livestock grazing.[34] Invasive weeds may pose the largest non-climate threat to grassland ecosystems, as

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[34] The collective degree these stressors increase sensitivity of grassland ecosystems was considered Moderate. Participant confidence associated with this evaluation was Moderate-High. Current exposure to these non-climate

exotic annual grasses, noxious weeds, and exotic forbs are already well-established across the grassland landscape (Ecovista et al. 2003; Bollenbacher et al. 2013). Invasive plants can outcompete native species, alter ecological processes, and alter habitat suitability. For example, cheatgrass is less nutritious for herbivorous wildlife species than native grasses, which could negatively impact obligate big game species that rely on grassland ecosystems for forage during late summer, winter, and drought periods (Ecovista et al. 2003). As an annual grass, cheatgrass has a much shallower and weaker root system than native perennial grasses (especially bunchgrasses). As a result, very little organic matter from the roots is contributed to the soil, reducing the amount of soil carbon and limiting development of soil structure, which in turn reduces infiltration and percolation. Recent research indicates that changes in precipitation and soil moisture affect the phenology of some exotic species (Wolkovich et al. 2013), which may influence invasion success. Invasive plants may exacerbate climate-driven changes; for example, elevated cheatgrass abundance may contribute to more frequent and intense fires, as it has a naturally shorter fire-return interval than native species and its fine-textured leaves create a larger fuel source (Ecovista et al. 2003).

Fire suppression practices in the NPCW region have allowed for conifer encroachment onto historic grasslands (Heyerdahl et al. 2006). Fire suppression practices may also lead to shifts from grass/forb-dominated communities to shrub-dominated communities, which can reduce overall species richness and diversity (Hanson et al. 2008). Fire suppression activities have also increased fuel density within grasslands and adjacent ecosystems, increasing the potential for more frequent and intense wildfires. Paired with climate shifts that promote altered fire regimes, fire suppression practices may contribute to fire regime changes that could restrict the natural regeneration capacities of native grassland species while facilitating noxious weed invasion (Ecovista et al. 2003).

Grazing, depending on both timing and intensity, has been shown to affect the composition of successional plant communities and allow for increased noxious weed invasion within grassland ecosystems. For example, overgrazing native grasslands typically facilitates the invasion of yellow starthistle, which may crowd out native grasses and form large monocultures with no grazing potential (Ecovista et al. 2003). Grazing may also exacerbate climate-driven changes. For example, livestock grazing increases bare ground (Olson et al. 1997) and soil compaction, which can increase runoff and erosion following rain events and lead to a reduction in the amount of rich topsoil needed to sustain plant communities. Grazing-related disturbances (e.g., increased bare ground) can also lead to higher densities of invasive plants (e.g., cheatgrass), which can contribute to altered fire regimes due to higher fuel densities and shorter fire return intervals (Ecovista et al. 2003). Alternatively, light grazing could reduce densities of some non-native species in grassland systems, especially if it is timed to coincide with non-native species peak growth periods and native species low growth or dormancy periods (e.g., during summer) (Hendersen et al. 2012). For example, Olson et al. (1997) showed that sheep grazing reduced

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stressors in the NPCW region was judged to be Moderate to Moderate-High by workshop participants (Confidence: Moderate).

cover of the non-native perennial forb spotted knapweed (*Centaurea manculosa*) while having minimal impact on the native Idaho fescue grass community (Olson et al. 1997).

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW grassland ecosystems was considered moderate by workshop participants and reviewers.[35] Within the NPCW region, the grassland ecosystem exhibits moderate-high topographic and component species diversity, altered structural and functional integrity, and occurrence in isolated patches. Palouse Prairie grassland ecosystems are endemic to the local Idaho and Montana region, occupying foothills and canyon breaklands, but due to large geographic features (e.g., mountain ranges) and past land use conversion, are spread out in patches throughout the landscape (Ecovista et al. 2003). The Palouse Prairie encompasses the largest portion of grassland ecosystem in the NPCW region, but only 1% of the original prairie acreage still remains after a century of agricultural conversion for hay, pasture, and grain harvest (Ecovista et al. 2003; NPCW Forest Plan Assessment 2012a). Bluebunch wheatgrass and Idaho fescue are the two most common native grass species (Ecovista et al. 2003; NPCW Forest Plan Assessment 2012a). However, invasive weeds have drastically altered the remaining grassland ecosystems, as they have become well established on low elevation, steep, south-facing, solar-loaded slopes since introduction in the 1800s (Ecovista et al. 2003).

Management potential

Grassland ecosystems were judged by workshop particpants to be a highly valued system by the public. Workshop participants identified fire suppression practices and improper livestock grazing as potential use conflicts. Peer-reviewed literature suggests that reintroducing prescribed fire, especially in areas where fire exclusion has been practiced, could re-establish and protect grassland systems both now and in the future (Ecovista et al. 2003; Heyerdahl et al. 2006). In addition, peer-reviewed literature suggests preventing overgrazing, restricting grazing on sensitive or degraded grassland sites, and controlling the timing of grazing to avoid peak native grass growth periods in order to better protect grassland systems (Ecovista et al. 2003; Heyerdahl et al. 2006; Hendersen et al. 2012). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature-generated Proposed Management Strategies[36]:

- Use prescribed fires (modeled after natural fire regimes) in areas where fire exclusion has been practiced.

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[35] Confidence associated with this evaluation was Moderate.

[36] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

  - Potential benefits: Promote re-establishment of grasslands, reduce conifer encroachment, reduce fuel build-up and natural fire risk (Ecovista et al. 2003; Heyerdahl et al. 2006).
  - Potential challenges: May facilitate invasive plant encroachment, can temporarily impair grasslands, may require increased institutional capacity to study proper burn intervals and impacts on native vegetation to achieve maximum benefit.
- Prevent overgrazing and institute climate- and species-informed grazing restrictions (e.g., protect climate-sensitive areas and avoid grazing during peak native grassland species growth periods).
  - Potential benefits: Increase resilience of grassland systems, promote native species recruitment and growth, reduce invasive weed abundance and/or establishment (Hendersen et al. 2012), increase forage for wildlife (Ecovista et al. 2003; Heyerdahl et al. 2006).
  - Potential challenges: Public buy-in, may require increased institutional capacity.

Mixed Mesic Ecosystems[37]

Executive Summary

The relative vulnerability of the mixed mesic ecosystem in this assessment is considered moderate, due to its moderate-high sensitivity climate-driven changes, high sensitivity to non-climate stressors, and moderate adaptive capacity. Mixed mesic communities are sensitive to climate-driven changes such as:

- reduced soil moisture,
- drought, and
- wildfire.

Drier and hotter conditions, which lead to seasonal soil moisture deficits, could lead to habitat conversion, species composition shifts, and/or higher disease or insect mortality of mixed mesic species. Changing fire regimes could have variable impacts; moderate severity fires may allow for historic, fire-tolerant species such as western white pine and western larch to recolonize portions of the mixed mesic range, but high fire severity could be detrimental to all species within the system.

Mixed mesic systems are also sensitive to non-climate stressors including:

- disease and insect outbreaks, and
- fire suppression practices.

These non-climate stressors may exacerbate system sensitivity to climate and climate-driven changes. For example, white pine blister rust, bark beetle infestations, and fire suppression practices may facilitate changing wildfire regimes by causing shifts in ecosystem composition and structure towards less fire-resilient species. The adaptive capacity of the mixed mesic ecosystem is considered moderate as it has low structural and functional integrity, but high physical, topographical, and component species diversity.

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[37] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

The mixed mesic ecosystem typically occurs in warm, moist areas and includes species such as western white pine (*Pinus monticola*), western larch (*Larix occidentalis*), grand fir (*Abies grandis*), western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and moist Douglas fir (*Pseudotsuga menziesii*). Historically, western white pine was a dominant species in the mixed mesic system (Harvey et al. 2008); however, since the beginning of the 20th century, more than 95% of the western white pine population has been eliminated by white pine blister rust, mountain pine beetle infestations, and salvage harvest (Harvey et al. 2008; Bollenbacher et al. 2013). In areas with the highest risk of white pine blister rust, only 1% of the original western white pine population may remain (Hann et al. 1997 *in* Harvey et al. 2008). The loss of dominant western white pine has led to major increases in late successional, shade-tolerant, and severely drought- and fire-intolerant species, such as grand fir, hemlock, and cedar. Consequently, the mixed mesic ecosystem exhibits sensitivity to decreased precipitation in the form of reduced soil moisture and drought, as well as altered wildfire regimes. Overall, the sensitivity of the mixed mesic ecosystem to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[38]

Table 8. Potential mixed mesic response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated mixed mesic response
Reduced soil moisture and drought	• Range contraction away from drier edges, southern aspects, and spur ridges • Species composition changes, positive feedback cycle toward more shade-tolerant, drought-intolerant species • Increased susceptibility to disease or insect attack
Wildfire	• Moderate severity fires: Reduced grand fir, cedar, and hemlock regeneration due to reduced seed source, potential recolonization opportunities for western white pine and western larch • Increased fire severity: Reduced seed source and regeneration opportunities for all species

While the mixed mesic system is characterized by periodic drought, reduced soil moisture or prolonged drought could cause ecosystem range contraction, changes in species composition, and/or increased disease and insect mortality. The timing of soil moisture deficits and drought periods also influences species distribution. For example, drought periods in the summer

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[38] Confidence associated with this evaluation was Moderate-High.

severely restricted the distribution of both western red cedar and western hemlock, while reduced soil moisture in spring appears to be most important for determining the range of western white pine (Mathys et al. 2014). Reduced soil moisture could result in water-stressed trees that are less able to resist insect or pathogen attack, leading to increased mortality of grand fir (Steele et al. 1995), cedar, hemlock, and remaining western white pines (Mathys et al. 2014). These impacts may also cause elevated positive feedback cycles leading toward accelerated succession of these shade-tolerant, drought-intolerant species (Bollenbacher et al. 2013). Additionally, moisture stress could cause a retraction from the drier portions and southern aspects of the current mixed mesic ecosystem range. Western white pine and larch, in particular, could be forced to retreat to northern aspects with deep soils to escape soil moisture deficits, allowing dry-adapted ponderosa pine (*Pinus ponderosa*) to encroach on drier mixed mesic edges such as spur ridges (Bollenbacher et al. 2013). Warming regional temperatures may intensify soil moisture deficits, drought episodes, and the associated impacts on the mixed mesic ecosystem (Bollenbacher et al. 2013). Potential areas of refugia from moisture and temperature-related effects include riparian areas, northern aspects, higher elevations, and the northern range of the Nez Perce-Clearwater (NPCW) region.

Altered wildfire regimes also present a potential threat to mixed mesic ecosystems. Historically, western white pine-dominated mixed mesic ecosystems were relatively fire-tolerant and had short fire return intervals. Modern mixed mesic ecosystems, with a greater abundance of dense, fire-intolerant cedar and grand fir, favor longer fire return intervals and mixed fire severity. Combined with fire suppression practices of the past century, the mixed mesic ecosystem now features less fire-resilient trees at higher densities, increasing the potential for stand-replacing fires during dry periods (Bollenbacher et al. 2013). Increased frequency and/or severity of fire is unlikely to favor grand fir, cedar, and hemlock regeneration due to reduced seed source, which may present a potential opportunity for other more fire-tolerant species, such as western white pine and western larch, to recolonize burned portions of the mixed mesic system. These more fire-tolerant species can withstand moderate severity fires, but increased fire severity will likely be detrimental to all species in the mixed mesic system (Bollenbacher et al. 2013; Scott et al. 2013). Potential refugia areas include north-facing slopes and riparian areas.

Future climate exposure

Climate and climate-driven factors most relevant to consider for the mixed mesic ecosystem include altered wildfire regimes, drought, reduced soil moisture, and increased temperatures. Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes (Holden and Jolly 2011). The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). Altered fire regimes and more stand-replacing fires may lead to reduced seed sources and cause regeneration issues for the mixed mesic ecosystem. Western white pines and western larch are better adapted to moderate severity fires than other component species, but will also be vulnerable to more severe fires caused by increased forest density (Bollenbacher et al. 2013).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005) and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Snowpack declines and earlier snowmelt may lead to earlier soil moisture recharge, which may lead to earlier desiccation of soils (Hamlet et al. 2007). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3) and, in general, drought frequency and severity may increase throughout the western U.S. (IPCC 2007a). Drought or soil moisture declines could lead to shifts in mixed mesic species distribution and composition (Mathys et al. 2014) and/or heighten mixed mesic ecosystem vulnerability to insect and pathogen attack (Steele et al. 1995; Mathys et al. 2014), potentially increasing overall system disturbance (Bollenbacher et al. 2013).

Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Extreme heat days are expected to both increase in frequency and last 3-6 days longer by the end of the 21st century (Diffenbaugh et al. 2005). Warmer temperatures, extreme heat events, and declines in summer precipitation could compound soil moisture deficits due to increased evapotranspiration, particularly on southern aspects. These changes may contribute to higher disease or insect mortality, and/or cause range contraction of the mixed mesic system away from its drier edges (Bollenbacher et al. 2013). Elevated temperatures could also limit the regeneration of mixed mesic species, mainly by exacerbating soil moisture deficits (Bollenbacher et al. 2013; Scott et al. 2013).

Sensitivity and current exposure to non-climate stressors
Workshop participants identified two non-climate stressors that affect the sensitivity of mixed mesic ecosystems, including fire suppression practices and insect and disease outbreaks.[39] White pine blister rust (caused by the pathogen *Cronartium ribicola*) has caused significant (>95%) mortality of western white pines throughout the mixed mesic ecosystem (Harvey et al. 2008), leading to a large-scale shift in species composition to severely fire- and drought-intolerant species, such as cedar, grand fir, and hemlock (Bollenbacher et al. 2013). These drought-intolerant species are particularly susceptible to root disease, which can cause tree mortality and/or accelerate succession toward more shade-tolerant, drought-intolerant species. This creates a positive feedback cycle that may lead to further disease vulnerability and disturbance (Bollenbacher et al. 2013). Climate-driven changes, such as increased and/or prolonged drought, may enhance disease-related disturbance in the mixed mesic ecosystem by increasing water stress among species.

Insect outbreaks also act as significant non-climate stressors for certain species in the mixed mesic ecosystem, and infestations may become more common under projected climate change scenarios (Fettig et al. 2007; Bollenbacher et al. 2013). For example, native mountain pine beetles (*Dendroctonus ponderosae*) historically acted as major mortality agents for western white pine, reducing tree competition and restarting successional stages (Harvey et al. 2008). However, climate change impacts are projected to cause significant growth of mountain pine beetle populations (e.g., by reducing cold mortality and increasing the reproductive cycle; Logan et al. 2003), and heightened moisture stress could make western white pine individuals more susceptible to infestation (Fettig et al. 2007; Mathys et al. 2014). Increased infestation rates could potentially reduce or eliminate the remnant western white pine population in the NPCW region and perpetuate species shifts within the mixed mesic system. Grand fir individuals, though not sensitive to mountain pine beetle infestation, are sensitive to *Scolytus subscaber* attack (Furniss and Kegley 2011), and may be more vulnerable to infestation during periods of moderate to severe moisture stress (Steele et al. 1995). *S. subscaber* infests the live branches of mature grand fir trees, though it rarely causes tree mortality or economic damage (Furniss and Kegley 2011). Thinning of dense grand fir stands may increase water availability and resin production (Fettig et al. 2007) in grand fir individuals, which could reduce *S. subscaber* invasion success (Steele et al. 1995).

Fire suppression practices of the past decade have increased mixed mesic forest density, increasing available fuel, elevating fire risk, and reducing local water availability and forest resilience. Fire suppression practices can also exacerbate system sensitivity to other climate stressors (Bollenbacher et al. 2013). For example, denser forests may experience higher soil moisture stress due to increased competition. During times of system-wide drought, when soil moisture stress reduces tree vigor, the mixed mesic system could experience higher tree mortality rates, root disease and/or insect outbreaks (Logan et al. 2003; Scott et al. 2013).

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[39] The collective degree these stressors increase sensitivity of mixed mesic ecosystems was considered High. Participant confidence associated with this evaluation was Moderate-High. Current exposure to these non-climate stressors in the NPCW region was judged to be Moderate-High by workshop participants (Confidence: High).

Within the NPCW region, fire suppression practices are currently employed along transportation corridors, while wilderness and roadless areas see considerably less fire suppression activity.

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW mixed mesic ecosystems was considered moderate by workshop participants and reviewers.[40] Within the NPCW region, the mixed mesic ecosystem has high system continuity and diversity, but degraded structural and functional integrity. The mixed mesic ecosystem can be found almost continuously at mid-elevations in northwestern Montana and north of the Salmon River in Idaho with minor landscape disruptions caused by timber harvest, mining, transportation corridors, grazing lands, and impassable geologic features. The mixed mesic system has high component species and topographic diversity, but due to changes in species composition and abundance, is largely degraded in terms of system integrity. Western white pine and western larch, fire-tolerant species that used to be dominant in this ecosystem, now occur only sporadically (Harvey et al. 2008), contributing to reduced fire, drought and overall system resilience (Harvey et al. 2008; Bollenbacher et al. 2013). Species composition of this system changes from north to south within the NPCW region, with western white pine acting as a foundation species in the northern range, ponderosa pine as a foundation species in the southern range, and western larch as a foundation species system-wide. The mixed mesic ecosystem is also home to coastal disjunct species (e.g., western red cedar, *Thuja plicata,* and red alder, *Alnus rubra*).

Management potential

Mixed mesic ecosystems were judged by workshop particpants to be a moderate to highly valued system by the public. Workshop participants identified resistance to stand replacement restoration as a potential use conflict. Peer-reviewed literature and reviewers suggest replacing late successional stands of grand fir, western red cedar, and western hemlock with early seral species (e.g., western white pine, western larch, and ponderosa pine) to return the mixed mesic system to its historic composition, structure, and resilience (Bollenbacher et al. 2013; Nez Perce and Clearwater National Forests Habitat Type Groups 2013; Scott et al. 2013; K. Hazelbaker, pers. comm., 2014). Peer-reviewed literature also suggests selective thinning as a method to reduce mixed mesic forest density and vulnerability to moisture stress (Fettig et al. 2007; Hagle 2008; Harvey et al. 2008; Bollenbacher et al. 2013; Scott et al. 2013). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

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[40] Confidence associated with this evaluation was Moderate-High.

Literature- and Reviewer-generated Proposed Management Strategies[41]:

- Replace late successional grand fir, western red cedar, and western hemlock stands with early seral species such as western white pine, western larch, and ponderosa pine.
    - Potential benefits: Reduce system vulnerability to drought, disease, and wind throw, increase fire resilience (Bollenbacher et al. 2013; Nez Perce and Clearwater National Forests Habitat Type Groups 2013; Scott et al. 2013), increase harvest profits (western white pine is a commercially valuable species; Harvey et al. 2008).
    - Potential challenges: Public buy-in and support, climate change impacts could limit planting opportunities (Scott et al. 2013), western white pine restoration requires white pine blister rust resistant individuals to be effective in the long-term.
- Continue selective thinning practices, targeting late seral, disease-susceptible individuals and species.
    - Potential benefits: Increase soil moisture, reduce disease and insect vulnerability, increase system resilience, integrity, and function both now and in the future (Fettig et al. 2007; Hagle 2008; Harvey et al. 2008; Bollenbacher et al. 2013; Scott et al. 2013).
    - Potential challenges: Possible conflicts with timber harvest interests.

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[41] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. Reviewer-generated strategies were gathered from regional experts in the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014.

Riparian Ecosystems[42]

Executive Summary

The relative vulnerability of riparian ecosystems in this assessment is considered moderate to moderate-high, due to its high sensitivity to climate-driven changes and non-climate stressors, and moderate-high adaptive capacity. Riparian communities are sensitive to climate and climate-driven changes including:

- decreased snowpack,
- earlier snowmelt and runoff timing,
- shifts from snowfall to rainfall, and
- wildfire.

Changes in precipitation type, timing, and amount (i.e., decreased snowpack, earlier snowmelt and runoff) can affect stream flow patterns and volume, altering sediment loading and debris and riparian vegetation composition, among others. Altered wildfire regimes can also affect riparian systems, as fire influences the species composition, structure, and environmental condition of riparian communities plant composition and habitat extent.

Riparian systems are also highly sensitive to non-climate stressors including:

- invasive species,
- grazing,
- transportation corridors (i.e. road networks), and
- dams and water diversions.

These non-climate stressors may exacerbate climate-driven changes. For example, grazing has the potential to exacerbate climate-driven changes in soil moisture and water availability on riparian vegetation by further compacting soil, damaging soil structure, and reducing infiltration, all of which lead to degraded conditions for germination. Adaptive capacity of riparian ecosystems is considered moderate-high as they feature high physical, topographical, component species, and functional group diversity and can be found across the Nez Perce-Clearwater region with generally high continuity.

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[42] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

Riparian ecosystems are systems with a high water table (due to proximity to aquatic ecosystems or subsurface water), and have distinct vegetation and soil characteristics, high species diversity and density, and allow for continuous interactions to occur between riparian, aquatic, and terrestrial ecosystems. Riparian areas provide critical ecological functions, including maintaining water quality and quantity, providing habitat for aquatic and terrestrial biota, sediment retention, building and maintaining stream banks, and provisioning of ecosystem services (Brinson et al. 2002; Naiman et al. 2005). Riparian plant communities maintain aquatic habitat for native fishes by providing shade (i.e., moderates stream temperature), contributing large woody debris and organic matter inputs, and providing streamside habitat and bank stabilization (Luce et al. 2012). Riparian areas are characterized by high natural disturbance rates (e.g., flooding, debris flow) and high soil moisture, and exhibit high sensitivity to changes in water supply. In particular, changes in precipitation type, timing, and amount can affect stream flow patterns and volume, altering sediment loading and debris and riparian vegetation composition (Huxman and Scott 2007; Chambers and Pellant 2008). Riparian systems are also moderately sensitive to wildfire, as it can help maintain ecological diversity. Overall, the sensitivity of riparian ecosystems to climate and climate-driven changes was evaluated as high by workshop participants.[43]

Table 9. Potential riparian ecosystem response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated riparian ecosystem response
Decreased snowpack and earlier snowmelt	• Lower summer flows, which could reduce connectivity and may lead to losses of riparian habitats along smaller watercourses • Declines in available surface water and groundwater, which could result in range contraction or local loss of species • Altered riparian recruitment or potential vegetation shifts to more dry-adapted species
Shifts from snow to rain	• Increased winter flood risk, which can increase scour events, alter bank morphology and stability, and increase sedimentation rates and woody debris recruitment
Wildfire	• Reduced speed and ability of riparian plant recolonization and regeneration • Larger-scale fires may increase the number of riparian areas exposed to mudslides and overland flow

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[43] Confidence associated with this evaluation was Moderate-High.

Riparian systems are highly adapted to and influenced by flow regimes, which play a key role in determining abiotic and biotic conditions and sub-basin scales (Poff et al. 1997). Moderate disturbance enhances species diversity (Merritt and Cooper 2000), resets riparian succession through scour (Naiman et al. 2008), and gradually receding flows provide varied, moist microclimates important for seedling recruitment (Shafroth et al. 1998). In particular, four flood characteristics are important to riparian environments: magnitude, frequency, timing, and duration (Luce et al. 2012). Flooding affects both geomorphological and hydrological features of riparian areas, including sediment, large wood, and organic material transport; saturation; and shaping of fluvial environments (Yarnell et al. 2010; Luce et al. 2012). For example, flood flows are more important than total annual water yields in terms of sediment transport (Buffington and Montgomery 1997). Flooding can also facilitate the establishment of vegetation stands or reset successional processes (e.g., see Rood et al. 1998). Riparian vegetation can help to constrain flood flows, trap sediment and debris, and limit stream bank erosion. Further, riparian plant species are able to recover rapidly from or take advantage of disturbance due to a range of adaptations (i.e., morphological, physiological, reproductive) (Dwire and Kauffman 2003). For example, the seed dispersal of cottonwoods and willows is tightly correlated with seasonal hydrology (Rood et al. 1998).

The influence of shifts in precipitation type, timing and amount on riparian systems will depend on location within a basin and stream network, as well as the physical context of a given stream reach. For example, high elevation headwaters may be more affected by variability in annual snowpack and higher summer temperatures whereas subalpine and montane-forested riparian areas may be more affected by shifts in streamside microclimates (Luce et al. 2012). Increases in the rate, magnitude, and timing (i.e., earlier in spring) of spring discharge may lower the availability and diversity of stream habitats, lead to higher sediment transport and redistribution (ultimately increasing habitat variability), and result in longer duration of warm water conditions (associated with earlier timing of snowmelt). Conversely, decreases in the rate, magnitude, and timing (i.e. later in spring) of spring discharge may provide larger regions of shallow stream habitat and improve connectivity, limit stream channel heterogeneity and species productivity, and help sustain lower water temperatures longer into the summer (associated with later timing of snowmelt) (see Yarnell et al. 2010 for a full discussion and literature review). In general, shifts in the magnitude of flow may affect abiotic channel conditions, shifts in timing may affect biotic conditions, and shifts in the rate of change may affect both abiotic and biotic conditions (Yarnell et al. 2010). Drought and limited moisture stresses riparian vegetation by reducing the ability of plants to photosynthesize and/or germinate, and eliminates seasonal flooding required by some species (e.g., cottonwood; Obedzinski et al. 2001) (Poff et al. 2011). Further, drought and reduced water tables can also allow exotic species better adapted to drier conditions to outcompete native riparian species (e.g, see Stromberg et al. 2007 *in* Poff et al. 2011).

Wildfire is another important disturbance regime with both benefits and detriments to riparian ecosystems. Wildfire helps maintain the ecological diversity of riparian ecosystems, both through resetting the successional clock directly and by inducing additional sedimentation processes (e.g., overland flow, mudslides) that contribute to successional riparian dynamics

(Dwire and Kauffman 2003). Fire influences the species composition, strucutre, and environmental conditions of riparian communities (Reeves et al. 2006; Pettit and Naiman 2007; Stromberg and Rychener 2010). Research indicates that most riparian areas burn either similar to or less frequently/more moderately than adjacent uplands (Everett et al. 2003; Luce et al. 2012). Riparian areas can also act as a buffer against fire, although in some cases riparian areas may burn more frequently and/or more severely than adjacent uplands or serve as corridors for fire movement (Pettit and Naiman 2007; Countryman 1971 *in* Luce et al. 2012; for a full discussion please see Luce et al. 2012). Fire effects occurring upstream can influence downstream conditions (Wipfli et al. 2007), as well as future fire behavior (Pettit and Naiman 2007). Riparian vegetation typically recovers quickly following fire, although this is dependent on vegetation type. High riparian soil moisture often prevents the combustion of soil organic materials and below-ground plant tissues, and riparian species have evolved adaptations that promote both survival on-site during fire (e.g., sprouting and thick bark) and after fire through recolonization of burned sites (e.g., via wind and water seed dispersal) (Dwire and Kauffman 2003). During dry periods and extreme fire weather, human-induced alterations to the landscape (e.g., timber harvest, fire suppression) may heighten the risk of severe fire, reducing the regeneration capacities of riparian vegetation by burning below-ground plant tissues (Dwire and Kauffman 2003). Lost vegetative protection and altered soil properties following fire can lead to surface erosion and mass wasting (e.g., debris flows), potentially extirpating local populations and simplifying habitats (Luce et al. 2012). However, these large events also provide important coarse material (e.g., cobbles, logs) that eventually contribute to habitat complexity and quality (Reeves et al. 1995).

Future climate exposure

The most important climate and climate-driven factors to consider for riparian ecosystems are those that alter hydrologic regimes and local soil moisture, including decreased snowpack, shifts from snow- to rain-dominant watersheds, and earlier snowmelt and runoff timing. In addition, riparian systems are also sensitive to altered wildfire regimes.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Warmer air temperatures cause more precipitation to fall as rain rather than snow (Knowles et al. 2006), potentially shifting some streams from snow-dominated to transitional or rain-dominated, where the timing of flows is related to the timing of precipitation (Stewart et al. 2005; Littell et al. 2011). In the western U.S., this means more stream flow in fall and winter, and less in spring and summer (Elsner et al. 2010; Wenger et al. 2010). These changes are projected to occur soonest at mid-elevation sites (Regonda et al. 2005; Pierce et al. 2008; Nayak et al. 2010). A shift from snow to rain may also lead to large changes in hydrograph timing and magnitude (Elsner et al. 2010), and can lead to increased risk of mid-winter floods (Hamlet and Lettenmaier 2007; Luce et al. 2012). Increased winter flood incidence can increase scour events, alter bank morphology and stability, and increase sedimentation rates and woody debris recruitment, affecting riparian extent and vegetation recruitment and success (Rieman and Isaak 2010; Yarnell et al. 2010).

About 75% of runoff in the western U.S. is currently derived from precipitation that falls as snow (Service 2004). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). As a result of snowpack declines, the spring freshet has decreased in both length and volume (Luce et al. 2012). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack predictions are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), but it appears that SWE (a measure of April 1st snowpack) is projected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Warming temperatures have led to earlier runoff timing (Stewart et al. 2005) and lower summer flows (Rood et al. 2008; Luce and Holden 2009). Specifically, unregulated streams in Idaho experienced earlier peak stream flow and lower summer stream flows from 1967-2007 (Clark 2010). Similarly, in the northwestern U.S., historical changes show declines in stream flows over the last 50 years (Moore et al. 2007; Luce and Holden 2009; Clark 2010), including declines summer flows (Luce and Holden 2009; Leppi et al. 2011). However, there may be several factors that influence changes in stream flow aside from direct climate change including land use contributions, climate cycles (e.g., PDO), and evapotranspiration (Hoerling and Eischeid 2007), and it still remains a question as to whether the changes are precipitation or transpiration related (see Luce et al. 2012 for a full discussion on this). Earlier runoff timing and changes in seasonal flows (e.g., lower summer stream flows, higher winter flows) are projected to continue for the NPCW region (Wenger et al. 2010; Littell et al. 2011; see also Section 3). Changes in seasonal stream flow could be compounded during warm PDO cycles, which are correlated with reduced precipitation and may lead to overall annual declines in stream flow (Rood et al. 2005). Less snowpack that melts earlier also decreases available surface water and groundwater by limiting the duration of snowmelt-induced stream flow and groundwater recharge (Viers et al. 2013). Earlier snowmelt and decreased snowpack may also increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006).

Changes in precipitation type, timing, and amount can have varying impacts on stream flow (Luce and Holden 2009). Altered stream flow (amount, timing, duration) contributes to changes in geomorphology and physical processes, and affects the ecological integrity of aquatic and riparian systems (Poff et al. 1997; Furniss et al. 2010). Changes in stream flow timing and volume can affect species reproductive success or seedling recruitment due to lack of favorable microclimates and/or competition with other species (Rood et al. 2005; Yarnell et al. 2010). Low stream flows could reduce connectivity of smaller streams to mainstream flows, potentially leading to losses of riparian habitats along smaller watercourses (Furniss et al. 2010; Yarnell et al. 2010). Reduced stream flows may also contribute to declines in riparian arthropod abundance and specialization (Yarnell et al. 2010). Changes in stream flow and patterns of groundwater recharge may also result in range contraction and/or local loss of species (Luce et al. 2012). Groundwater-dominated systems may buffer short-term variations in stream flow, although long-term drought (e.g., lasting several years) may increase sensitivity of these systems (Lall and Mann 1995; Shun and Duffy 1999). Further, in years with smaller snowpack, groundwater inputs may be diminished and headwater stream temperatures warmer (Isaak et al. 2011). Reduced flows, decreases in groundwater discharge, and/or soil moisture reductions may heighten water stress in riparian areas, and could lead to impaired riparian recruitment and/or potential vegetation shifts to more dry-adapted species (Bollenbacher et al. 2013). Potential refugia areas from climate impacts may include north facing slopes, areas with seeps and springs, groundwater-dependent areas, and/or riparian corridors in all but the most extreme weather conditions.

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes (Holden and Jolly 2011). The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Increasing temperatures may result in warming riparian microclimates, resulting in shifts in streamside vegetation composition (i.e., may become more similar to upland vegetation). Consequently, these riparian areas may burn more like surrounding uplands (Luce et al. 2012). More intense and/or severe fires in riparian ecosystems, especially if below-ground organic material and plant tissue is incinerated, may reduce the speed and ability of riparian plant recolonization and regeneration, which could allow for increased noxious weed invasion and reduced riparian system functionality (Dwire and Kauffman 2003). Fire can also induce hydrologic changes, including changes to snowmelt, runoff, peak flows, and low flows (Luce et al. 2012). Increased solar radiation post-fire has been linked to advances in snowmelt timing (1-2 weeks), with variable changes in peak flows (e.g., severely burned small basins experienced significant changes in peak flows but at the basin scale, no increase was observed) (Troendle et al. 2010; Luce et al. 2012). Changes in runoff following fire have been primarily attributed to changes in soil properties (i.e., surface sealing, water repellency) (Luce et al. 2012). Water repellency seems to be associated with certain vegetation communities (e.g., chaparral, subalpine fir), and is more likely where fires burn severely; further, dry, hot summers of the western U.S. may be ideal conditions for water repellent behavior (Luce et al. 2012). Peak flows in streams can be significantly higher post-fire than pre-fire, though it is more frequently documented that post-fire peak flow is less than pre-fire peak flow (Shakesby and Doerr 2006). Altered wildfire regimes can also affect fire-related sedimentation regimes; larger-scale fires may increase the number of riparian areas exposed to mudslides and overland flow, which could lead to altered riparian successional patterns across larger areas (Dwire and Kauffman 2003). Potential areas of refugia may include broad valley bottoms and headwalls (Camp et al. 1996; Everett et al. 2003).

It is well known that terrestrial, aquatic, and riparian ecosystems do not function independently of one another (e.g., see Nakano and Murakami 2001). Changes in water temperature will likely have the strongest direct effects on freshwater ecosystems, but many of these effects could be transmitted to riparian and terrestrial ecosystems due to the strong linkages among these habitats. For example, changes in temperature may have direct effects on the phenology and community composition of stream invertebrates, which in turn could modulate the timing, magnitude, and compositions of reciprocal subsidies between riparian and freshwater habitats. The direct and indirect effects of stream temperature on riparian ecosystems are not well understood and future changes will be difficult to predict. For a more complete discussion regarding the effects of changing stream temperature on aquatic biota, please refer to the Aquatic Ecosystems and Aquatic Species summaries in this report.

Again, it is important to note that the impacts of climate change will depend on location within a basin and stream network, as well as the physical context of a given stream reach.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of riparian ecosystems, including invasive and problematic species, grazing, transportation corridors, dams and water diversions, recreation, and fire suppression practices.[44] For example, both orange hawkweed and meadow hawkweed have become well-established in riparian areas within the NPCW region, forming monoculture mats that exclude all other vegetation (Ecovista et al. 2003). By outcompeting native vegetation for shade and moisture, invasive plants may alter important riparian vegetation functions such as regulating stream temperature through shade (Ecovista et al. 2003) or affect the quality, types, and abundances of substrates for invertebrates (Obedzinski et al. 2001). Altered hydrologic and disturbance regimes could facilitate invasive species colonization, further affecting the ecological structure and function of riparian communities (Furniss et al. 2010). Recent research also indicates that some exotic species exhibit higher tracking of interannual variation in temperature, shifting their flowering

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[44] The collective degree these stressors increase sensitivity of riparian ecosystems was considered High. Participant confidence associated with this evaluation was Moderate. Current exposure to these non-climate stressors in the NPCW region was judged to be Moderate by workshop participants (Confidence: Moderate).

with climate change while native species do not (Wolkovich et al. 2013), potentially facilitating exotic invasions.

Domestic and wild grazing (e.g., cattle, sheep, elk), particularly overgrazing, may alter community structure and composition and impair riparian ecosystem function. In particular, there are four areas of ecological impact: (1) soil compaction, (2) herbage removal, (3) physical damage to plants, and (4) changes in fluvial processes that affect germination sites (Kauffman 1988 *in* Poff et al. 2011). Livestock graze young riparian woody species (e.g., cottonwood and willow seedlings) along with herbaceous species (e.g., sedges, mannagrass, bluejoint reedgrass). Overgrazing can compact soil, limiting plant cover and regeneration capacity of native species such as black cottonwood, while simultaneously allowing for further spread of invasive species into the riparian zone (Ecovista et al. 2003; Poff et al. 2011). Additionally, through the cumulative effect of herbivory and vegetation alteration (e.g., by trampling), grazing disrupts aggradation processes in riparian ecosystems, leading to increased bank erosion, channel incisions, loss of crucial topsoil needed for plant regeneration, lowered water tables, and shifts in vegetation composition (Ecovista et al. 2003; Poff et al. 2011). Grazing has the potential to exacerbate climate-driven changes in soil moisture and water availability on riparian vegetation by further compacting soil, damaging soil structure, and reducing infiltration, all of which lead to degraded conditions for germination.

Transportation corridors and road management can have negative impacts on riparian ecosystems. Road networks affect the riparian environment by altering sediment delivery mechanisms, increasing fine sediment yields, increasing incidence of invasive species, modifying vegetation community structure and function, altering large woody debris recruitment, simplifying stream function, and providing travel routes for grazing animals (Williams 1954; Young et al. 1967; Roath and Krueger 1982; Trombulak and Frissell 2000). Generally, as the density of roads in a watershed increases, aquatic habitat quality decreases. Road construction causes severe disturbance to soils on slopes (Rieman and Clayton 1997) and, in a scientific literature review considering the effects of roads on aquatic systems, Trombulak and Frissell (2000) underscored the importance of avoiding construction of new roads and removing or restoring existing problematic roads. Roads built decades ago are often located in valley bottoms next to streams and are difficult to relocate (Swift and Burns 1999), and current recreation use and a lack of sufficient maintenance can result in increased sediment delivery (Grace and Clinton 2007). Timber harvest and road building can accelerate the frequency and volume of debris slides and hillslope sediment loss (Naiman et al. 2005). Roads produce runoff and sediment in almost every precipitation event, which may have significant impacts on aquatic biota as they occur more frequently (Luce et al. 2012).

Dams can provide both a threat and benefit to riparian ecosystems. Through flow regulation, they can reduce peak flows for increased water storage and increase or decrease low base flows (Poff et al. 2011). In the Pacific Northwest, regulated streams have experienced an average flow decrease of 2.8% over the past thirty years, compared to an average flow decrease of 2.1% in unregulated streams (Littell et al. 2011). Changes in flow regime can affect riparian vegetation and aquatic biotia. For example, flow regulation can lead to declines in riparian vegetation richness and abundance (Ecovista et al. 2003; Beauchamp et al. 2006), increase invasive species establishment (Beauchamp and Stromberg 2007), or shift riparian plant composition (Hadley and Emmett 1998). However, dams that create temperature-stratified reservoirs may be beneficial to aquatic biota, as they periodically release cold water to downstream habitats and could help mitigate low flows and warming temperatures driven by climate change (e.g., see Null et al. 2011). Further, depending on where dams are located, they may be able to elevate low flows (Poff et al. 2006).

Workshop participants also considered riparian and wetland ecosystems to be highly sensitive to fire suppression practices and OHV and ATV recreation but the current degree of exposure to these activities is thought to be low within the NPCW region.

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW riparian ecosystems was considered moderate-high by workshop participants and reviewers.[45] Riparian ecosystems feature high topographic, component species, and functional group diversity, and are continuous by nature. Within the NPCW region, however, riparian areas feature somewhat degraded structural and functional integrity due to a variety of activities including logging, mining, road networks, dams and water diversions, and grazing, all of which alter landscapes, impede natural processes, and/or restrict riparian continuity. Agriculture, land use conversion, and geologic features also may impede riparian landscape permeability and functionality, but occur in very small areas relative to the expanse of the aforementioned activities.

Management potential

Riparian ecosystems were judged by workshop participants to be highly valued systems by the public. Workshop participants identified dams and water diversions, invasive and other problematic species, grazing, and road networks as potential use conflicts for the NPCW region. Management strategies from the peer-reviewed literature are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature-generated Proposed Management Strategies[46]:

- Focus on critical roads and relocate or restore them. Select critical roads by (1) identifying the highest ecological priority areas, (2) within those, identifying the most damaging roads, and (3) within those, identifying the roads that can be effectively decommissioned or mitigated (Luce et al. 2001).

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[45] Confidence associated with this evaluation was High.

[46] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. See Luce et al. (2012) for additional management considerations.

  - Potential benefits: Minimize the harmful effects of roads on the aquatic environment, and limit the impacts of relocation/restoration.
  - Potential challenges: May require increased institutional capacity or funding.
- Focus thinning and other activities related to fuels reduction in areas with existing road systems, and use minimal impact harvest techniques (Brown et al. 2004b).
  - Potential benefits: Likely improve aquatic function.
- Restore low elevation, mixed severity ponderosa pine forests along riparian corridors (Rieman et al. 2000). In riparian areas that have departed from their expected range of conditions, restoration may want to consider their position in the landscape relative to elevation, location within the stream network, and climate regime (Pettit and Naiman 2007).
  - Potential benefits: Treatments (e.g., road decommission and relocation, culvert replacement, thinning to restore old forest structure) could create more suitable habitat in the long term. However, land managers will need to consider a variety of spatial and temporal scales and treatment design to improve scientific understanding (Rieman et al. 2000).
  - Potential challenges: In the short-term, sediment loads and the risk of landslides and debris flows from steep-facing drainages could increase.
- Limit intensity, range, and timing of grazing to protect the most sensitive riparian habitats, particularly those with low channel confinement, low gradient stream banks, and those with high exposure to future climate impacts (Ecovista et al. 2003).
  - Potential benefits: Reduce sediment loads, decrease incidence of invasive species, and improve system resilience.

Subalpine Ecosystems[47]

* This vulnerability assessment focuses on the upper subalpine zone (above 6000 ft) within the Nez Perce-Clearwater region, focusing mainly on subalpine areas that could support whitebark pine.

Executive Summary

The relative vulnerability of the subalpine ecosystem in this assessment is considered moderate-high to high, due to its high sensitivity to a number of climate and non-climate stressors, and moderate adaptive capacity. Subalpine communities are sensitive to climate and climate-driven changes such as:

- temperature increases,
- reduced soil moisture, and
- wildfire.

Warmer temperatures and precipitation changes may result in summer soil moisture deficits and drought conditions, potentially shifting species composition. Changing fire regimes could have variable impacts; increases in moderate-severity fire may favor whitebark pine recolonization, but increasing incidence of large, stand-replacing fires is likely to be detrimental to all species within the system.

The key non-climate stressor affecting subalpine systems of the Nez Perce-Clearwater (NPCW) region is insect and disease outbreaks. Bark beetles and other herbivorous insects have played important historical roles in forest ecosystems by acting as major mortality agents, reducing tree competition and restarting successional stages. Beetle outbreaks were historically limited in higher subalpine systems due to cold temperatures (Logan and Powell 2001), but warmer temperatures during the 20th century have caused elevated and widespread beetle-induced mortality in subalpine tree species. Increasing temperatures may reduce cold mortality and increase the reproductive cycle of these bark beetles, while precipitation changes and moisture stress could make tree species more susceptible to infestation, further increasing the risk of widespread beetle-related mortality. Fire suppression activities in lower elevation zones (i.e., non-wilderness subalpine) act as an additional non-climate stressor, and may increase the risk

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[47] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

of more frequent and/or intense wildfires. The adaptive capacity of subalpine ecosystems in the region is considered moderate, due to its high elevation location with limited room to vertically migrate, high component species diversity, and moderately degraded ecosystem integrity.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

Current subalpine ecosystems feature a mix of whitebark pine (*Pinus albicaulis*) and other high elevation associates including subalpine fir (*Abies lasiocarpa*), Englemann spruce (*Picea engelmannii*), mountain hemlock (*Tsuga mertensiana*), and subalpine larch (*Larix lyallii*). Whitebark pine, a drought- and fire-tolerant but shade-intolerant species typically colonizes the highest and harshest sites where it can exist in relatively pure stands as a climax species (Bollenbacher et al. 2013). At lower elevations, whitebark pine acts as an early seral species and is successionally replaced by more shade-tolerant and water-dependent subalpine associates, such as subalpine fir and Englemann spruce (Callaway and Walker 1997; Bollenbacher et al. 2013). Throughout the subalpine zone, whitebark pine acts as a “nurse” tree for other conifers on the most stressful, exposed sites, creating more moderate environmental conditions that allow them to establish (Callaway and Walker 1997). Over the past century, however, the subalpine system has seen a significant shift in species composition. Through a combination of beetle and white pine blister rust mortality and lower-elevation fire occurrence, whitebark pine has been largely replaced by subalpine fir, and early successional whitebark pine communities now exist in only a small portion of their previous range (Sala et al. 2001; USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013). Subalpine fir and spruce provide critical habitat for the threatened Canada lynx (*Lynx canadensis*), but increased subalpine fir abundance may increase the vulnerability of the entire subalpine ecosystem, as subalpine fir is more sensitive to increased temperatures, moisture stress, and wildfire (Bollenbacher et al. 2013; Scott et al. 2013).

The subalpine ecosystem occurs at high elevations with deep snow and cold temperatures, and exhibits overall sensitivity to temperature, drought, soil moisture, and wildfire. Overall, the sensitivity of subalpine ecosystems to climate and climate-driven changes was evaluated as high by workshop participants.[48]

Table 10. Potential subalpine response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated subalpine response
Warmer air temperatures	• Subalpine fir may retreat to cooler aspects and experience reduced growth • Increased whitebark pine krummholz growth • Increased invasion of whitebark pine onto former snowfields

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[48] Confidence associated with this evaluation was Moderate-High.

	• Increased subalpine forest density resulting from longer growing season
Drought and reduced soil moisture	• Subalpine fir, spruce, subalpine larch, and mountain hemlock may retreat to more mesic aspects, potentially increasing whitebark pine establishment and dominance on southerly and dry aspects • Reduced subalpine fir growth • Increased vulnerability to insect or disease outbreak
Wildfire	• Low-moderate severity fires: Increased colonization/regeneration opportunities for whitebark pine • Severe, stand-replacing fires: High tree mortality, reduced seed source and regeneration opportunities for all species

Warming temperatures and precipitation shifts that lead to reduced soil moisture or increased drought may reduce tree growth, cause species composition shifts and/or allow drought-tolerant whitebark pine to become more dominant relative to other component species, potentially restoring historic landscape conditions (Peterson et al. 2002; Bollenbacher et al. 2013). For example, subalpine fir typically features reduced growth during times of low soil moisture (Peterson et al. 2002), and may be forced to retreat to cooler, more mesic sites within the subalpine system if moisture stress increases (Peterson et al. 2002; Bollenbacher et al. 2013; Scott et al. 2013). High species stress from low moisture conditions can also increase species vulnerability to insect and pathogen attack. For example, whitebark pine is vulnerable to both introduced white pine blister rust (caused by the pathogen, *Cronartium ribicola)* and native mountain pine beetles (*Dendroctonus ponderosae)*, and is less able to fend off attack during times of stress from low moisture conditions (Bollebacher et al. 2013). Moisture stress may be compounded by the higher abundance of subalpine fir across the subalpine landscape. For example, in a previous study, mature subalpine firs used more water than mature whitebark pines, and stand-level transpiration increased substantially following succession of whitebark pines and the co-dominance of subalpine fir (Sala et al. 2001). However, these trends were studied during periods of high relative soil moisture (i.e., spring); it is unknown if the same trends occur during periods of low moisture (e.g., summer) (Sala et al. 2001).

Whitebark pine is fire-tolerant and is known for colonizing fire-disturbed areas, while spruce and subalpine fir are less fire-tolerant and act as late successional species (Bollenbacher et al. 2013). Subalpine forests historically featured fire-return intervals of 50 to 300 years, and fire was historically the key successional agent in the subalpine system, removing subalpine fir and maintaining open stands of fire-tolerant whitebark pine (Sala et al. 2001). However, increased mortality of whitebark pine from blister rust and mountain pine beetles and the exclusion of fire in lower elevation areas (i.e., non-wilderness), have led to increased abundance and density of subalpine fir and other subalpine associates, all relatively fire-intolerant species (Sala et al. 2001; USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013). Although whitebark pine is more tolerant of fire, all species in the subalpine system are likely to suffer if large, stand-replacing fires become more frequent (Bollenbacher et al. 2013).

Future climate exposure

Climate and climate-driven factors most relevant to consider for the subalpine ecosystem include warming temperatures, increased drought and reduced soil moisture, and changing wildfire regimes. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Warming minimum temperatures may increase subalpine ecosystem extent. For example, increased minimum temperatures have been shown to stimulate growth of whitebark pine krummholz forms and allow for increased whitebark pine invasion into former snowfields, as warmer temperatures effectively lengthen the growing season by causing earlier snow- and soil-melt and reducing the severity and frequency of nighttime freezing and wind desiccation events (Millar et al. 2004). Decreased snowpack, earlier snowmelt, and a longer growing season also favor higher subalpine forest density (Klasner and Fagre 2002; Peterson et al. 2002; Korner 1998 *in* Millar et al. 2004), although species composition may be influenced by moisture availability.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005) and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Snowpack declines and earlier snowmelt may lead to earlier soil moisture recharge, which may lead to earlier desiccation of soils (Hamlet et al. 2007). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3). Warming temperatures and decreasing moisture availability can limit tree growth, especially for less drought-adapted species such as subalpine fir (Peterson et al. 2002). Temperature increases and moisture declines may also drive subalpine fir, spruce, subalpine larch, and mountain hemlock to more mesic sites and cooler, northern slopes, allowing whitebark pine and lodgepole pine to dominate southerly aspects (Peterson et al. 2002; Bollenbacher et al. 2013). However, soil moisture is unlikely to limit whitebark pine establishment in its upper elevation zones (Weaver et al. 2001 *in* Millar et al. 2004).

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Although whitepark pine is fire-tolerant, featuring the ability to both withstand and recolonize following fire, more intense or frequent stand-replacing fires may reduce overall seed source. This is problematic as its seed sources may already be declining due to non-climate stressors including mountain pine beetle infestations and white pine blister rust mortality (Keane and Parsons 2010; USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013). In addition, more frequent and/or intense fires may eliminate whitebark pine individuals that are resistant to white pine blister rust, reducing the genetic resilience of whitebark pine in the NPCW region (Keane and Parsons 2010). Other, less fire-tolerant subalpine associates (e.g., subalpine fir, Englemann spruce) could suffer from more frequent or intense fires, as they are not adapted to either withstand or regenerate after severe fire (Bollenbacher et al. 2013).

Sensitivity and current exposure to non-climate stressors

Workshop participants identified fire suppression activity in lower elevation, non-wilderness areas as a non-climate stressor that affects the sensitivity of subalpine ecosystems.[49] Scientific literature also identified insect and disease outbreaks as a potential non-climate stressor. The subalpine system is sensitive to two different bark beetle species: the mountain pine beetle is a native pest that attacks whitebark pine, while spruce beetles (*Dendroctonus rufipennis* Kirby) attack Englemann spruce (Bollenbacher et al. 2013). Spruce beetles typically only attack large trees (16"+ diameter-at-breast-height (dbh) class) and spruce stands are generally susceptible when their basal area is greater than 150 square feet per acre and spruce makes up 65%+ of the canopy (Holsten et al. 1999). These conditions rarely occur across vast areas, thus spruce beetle infestations usually result in high local mortality but only over small acreages (Bollenbacher et al. 2013). However, if forest heterogeneity is not reintroduced into

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[49] The collective degree these stressors increase sensitivity of subalpine ecosystems was considered High. Participant confidence associated with this evaluation was High. Current exposure to these non-climate stressors in the NPCW region was judged to be High by workshop participants (Confidence: High).

contemporary subalpine systems, spruce beetle impacts could become problematic in the future (Bollenbacher et al. 2013).

Comparatively, mountain pine beetles (MPB) have caused widespread mortality of whitebark pine (Logan and Powell 2001; Bentz et al. 2010; Jewett et al. 2011) throughout the subalpine range, and infestations could become more common under projected climate change scenarios (Fettig et al. 2007; Bollenbacher et al. 2013). Bark beetles and other herbivorous insects have played important historical roles in forest ecosystems by acting as major mortality agents, reducing tree competition and restarting successional stages. MPB outbreaks were historically limited in higher subalpine systems due to cold temperatures (Logan and Powell 2001), but warmer temperatures during the 20th century have caused elevated and widespread MPB-induced mortality of whitebark pine in the NPCW region, especially among cone-bearing trees with a dbh of more than 20 cm (Keane and Parsons 2010). Warmer temperatures associated with climate change may reduce MPB cold mortality, accelerate life cycles, cause significant population growth, and result in more successful invasions by increasing emergence and invasion synchronicity (Logan and Powell 2001; Bentz et al. 2010; Jewett et al. 2011). Further, precipitation changes and moisture stress could enhance whitebark pine susceptibility to infestation, increasing the risk of widespread beetle-related mortality (Fettig et al. 2007). Increased bark beetle herbivory can also lead to more woody fuel, which could enhance the likelihood of large, stand-replacing fires, especially during dry summer periods (Logan et al. 2003; Bollenbacher et al. 2013). However, current research indicates that recent bark beetle outbreaks (4-13 years prefire) may not affect subsequent fire severity (Harvey et al. 2013).

In addition, whitebark pine is also sensitive to exotic white pine blister rust, and roughly 80% of white pines in the northern Rocky Mountains are infected (Tomback 2007). Although it is not known how climate change will affect white pine blister rust itself, heightened moisture stress and associated declines in tree vigor may increase white pine blister rust infestations and associated mortalities (USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013), which could further alter species composition in the subalpine system.

Fire suppression activities in lower elevations can affect subalpine forests. For example, in the absence of fire, subalpine fir and spruce have replaced whitebark pine at lower elevations due to natural succession (USFWS 2011; Keane et al. 2012), creating homogeneous forests in both species composition and age structure (Bollenbacher et al. 2013). These factors heighten the risk of more frequent and/or intense fires, particularly stand-replacing fires that would limit the regeneration of all subalpine component species (Bollenbacher et al. 2013). However, managed wildfires in upper subalpine zones (i.e., wilderness) have benefitted the Nez Perce-Clearwater National Forest subalpine system (K. Hazelbaker, pers. comm., 2014).

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW subalpine ecosystems was considered moderate by workshop participants and reviewers,[50] due to its high elevation location with limited room to vertically migrate, high component species diversity, and moderately degraded ecosystem integrity. Although it is an ecosystem found across the West, upper subalpine zones are found in discrete "patches" isolated at high elevations with limited room to migrate vertically. Non-climate stressors (e.g., insects, disease) moving upwards from lower elevations and into subalpine ecosystems may further "squeeze" this habitat. The subalpine ecosystem has high component species diversity, but low physical, topographic, and functional group diversity. Further, it has moderately degraded structural and functional integrity within the NPCW region due to the loss of whitebark pine. Whitebark pine is considered a keystone species in the subalpine ecosystem (Bollenbacher et al. 2013), especially at higher elevations, but has experienced mass mortality from mountain pine beetles and white pine blister rust (Keane et al. 2012), leading to shifts in forest structure.

Management potential

Subalpine ecosystems were judged by workshop participants to be very highly valued systems by the public. Workshop participants identified whitebark pine restoration in lynx spruce-fir habitat as a potential use conflict. However, reviewers and peer-reviewed literature identified whitebark pine restoration as a key strategy for maintaining resilience and integrity of subalpine ecosystems. Other recommended management strategies included maintaining natural fire in subalpine systems to protect whitebark pine populations and to prevent fuel buildup (USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature- and Reviewer-generated Proposed Management Strategies[51]:

- Continue to manage natural fires in the upper subalpine system (i.e., wilderness areas).
    - Potential benefits: Maintain or increase forest heterogeneity, maintain whitebark pine habitat, reduce fuel loads and risk of stand-replacing fires (USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013; R. Keane, pers. comm., 2013; K. Hazelbaker, pers. comm., 2014).
- Implement whitebark pine restoration (e.g., planting rust-resistant seedlings, using prescribed burning or thinning in non-wilderness areas, planting seeds and seedlings to colder environments, etc.; Keane et al. 2012).

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[50] Confidence associated with this evaluation was High.

[51] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. Reviewer-generated strategies were gathered from regional experts in the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014.

- Potential benefits: Improve ecosystem resilience by restoring keystone species (Keane et al. 2012).
- Potential challenges: May required additional institutional capacity, research, and funding.

Wetlands, Moist Meadows, and Groundwater-Dependent Ecosystems[52]

Executive Summary

The relative vulnerability of wetlands, moist meadows and groundwater-dependent ecosystems (GDEs) in this assessment is considered low-moderate to moderate, due to their moderate sensitivity to climate changes, low sensitivity to non-climate stressors, and moderate adaptive capacity. GDEs, wetlands, and moist meadows are sensitive to climate and climate-driven changes that affect water supply, including:

- decreased precipitation,
- decreased snowpack and earlier snowmelt,
- shifts from snow to rain, and
- reduced soil moisture.

GDEs, wetlands, and moist meadows are dependent on their individualized hydrologic regimes (i.e. surface water vs. groundwater) for water supply. Consequently, surface water wetlands are highly sensitive to precipitation changes whereas GDEs are more sensitive to decreased snowpack and the subsequent impacts to groundwater recharge. Changes in precipitation type, timing, and amount may have significant impacts on these systems including habitat conversion, decreased available surface water and groundwater, erosion and channel incision, or decreased habitat for aquatic biota, among others. Although not mentioned above, these systems are also sensitive to altered wildfire regimes, which can increase system vulnerability to flash floods and elevated sediment and debris loading.

GDEs, wetlands, and moist meadows are also sensitive to several non-climate stressors including:

- timber harvest,
- grazing,
- invasive species, and
- land use conversion for agriculture and development.

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[52] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Although current exposure to such stressors is considered low in the Nez Perce-Clearwater (NPCW) region, they could exacerbate climate-driven changes. For example, grazing has the potential to exacerbate climate-driven changes in soil moisture and water availability for riparian vegetation by further compacting soil, damaging soil structure, and reducing infiltration, which could facilitate faster system degradation or conversion. The adaptive capacity of these systems is considered moderate due to isolation and fragmentation, although many of these habitats do exhibit high structural and functional integrity.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Wetlands, moist meadows, and groundwater-dependent ecosystems (GDEs) feature generally high soil moisture and varying chemical soil compositions that are controlled by localized groundwater and surface water hydrologic regimes. Wetlands and moist meadows provide a number of ecosystem services including filtering water; attenuating floods; acting as sources of recharge to, sinks for, or discharge from groundwater; helping stabilize stream banks in adjacent riparian ecosystems; and providing highly productive habitat (Hammersmark et al. 2008; Loheide et al. 2009; Lowry et al. 2011). Wetlands, moist meadows, and GDEs also provide habitat for rare species and/or migratory birds.

Surface water wetlands are highly sensitive to precipitation changes, particularly those that affect surface flows and soil moisture such as drought or decreased precipitation, decreased snowpack and earlier snowmelt, and/or shifts from snowfall to rainfall. Wetlands located in the uplands of mountainous landscapes, plateaus, and high plains are dependent on precipitation for their water source, whereas wetlands in the lowlands of mountainous landscapes, plateaus, and high plains receive their water from groundwater discharge and/or streams (Winter 2001). Wetlands in riverine landscapes (characterized by broad lowlands) receiving water from regional groundwater flow systems may be better able to buffer variations in climate due to sustained water input (Winter 2001).

In mid- to high-elevations, moist meadows typically receive consistent water supply from snowmelt, which provides surface water, recharges groundwater, and elevates soil moisture and baseflow during summer (Viers et al. 2013). A high groundwater table is necessary to sustain moist meadow plants (Elmore et al. 2006; Loheide and Gorelick 2007), while periodic flooding is important for maintaining meadow functioning (Hammersmark et al. 2008). Moist meadows are sensitive to climate-driven changes in both surface water and groundwater, including changes in snowmelt, precipitation and groundwater, and changes in the amplitude, duration, and timing of surface and subsurface flows (Viers et al. 2013).

GDEs (which can include wetlands and peatlands, wet forests, springs, rivers and lakes) are reliant on groundwater supply to maintain system composition, structure, and function. They link groundwater and surface water, as well as groundwater and terrestrial ecosystems (Boulton 2005; Barquin and Scarsbrook 2008), and contribute to the regional biodiversity of freshwater and wetland ecosystems (Klove et al. 2011). GDEs may be more resilient to climate and climate-driven changes, but also exhibit sensitivity to prolonged drought, as well as decreased snowpack and the subsequent impacts to groundwater recharge. Similar to wetlands, GDEs located in headwater systems may be more vulnerable than lowland ecosystems, although headwater systems may be more adapted to natural variability (Klove et al. 2011).

Wetlands, moist meadows, and GDEs also exhibit some sensitivity to wildfire, which can influence the species composition, structure, and environmental conditions of these communities. However, these ecosystems have generally higher fuels and soil moisture content than in surrounding uplands, and may serve as fire breaks (Dwire and Kauffman 2003).[53] Overall, the sensitivity of wetlands, moist meadows, and GDEs to climate and climate-driven changes was evaluated as low-moderate by workshop participants.[54]

Table 11. Potential wetland, moist meadow, and GDE response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated wetland, moist meadow, and GDE response
Reduced precipitation, drought, reduced soil moisture	• Range contraction and/or habitat conversion, especially of surface water systems • Increased invasion of dry-adapted species and/or invasive species • Increased tree encroachment
Precipitation changes (shifts from snow- to rain-dominant or transient watersheds)	• Increased winter flood risk, leading to erosion of moist peat and topsoil, channel incision, loss of bank stability, reduced opportunities for water storage and recharge, drying of meadows, or drier, isolated wetlands with more terrestrial, dry-adapted species
Decreased snowpack and earlier snowmelt	• Decreased available surface water and groundwater • Stressed hydric/mesic vegetation, promoting more xeric conditions • Reduced flows potentially leading to shifts in vegetation composition or habitat conversion, increased stream temperatures, and/or decreased quality, quantity, and duration of habitat available for aquatic biota

Wetlands, moist meadows, and groundwater-dependent systems could benefit from certain climate and climate-driven changes, including increased precipitation, soil moisture, and

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[53] For additional discussion regarding fire, please see the Riparian Ecosystem section of this report.
[54] Confidence associated with this evaluation was Moderate.

snowpack. Increases in these factors may enhance relative water availability, provide more suitable habitat, or increase system extent via paludification, where higher water tables cause adjacent conifer mortality (Chadde et al. 1998).

Future climate exposure[55]

The most relevant climate and climate-driven factors to consider for wetlands, moist meadows, and GDEs are those that alter water supply and flow regimes including precipitation declines, decreased snowpack and earlier snowmelt, shifts from snow to rain, and reduced soil moisture and drought. In addition, altered wildfire regimes could increase system vulnerability to flash flood events and increase sediment and debris loading.

On average, precipitation across the Nez Perce-Clearwater (NPCW) region has not been notably different during recent years (i.e. last 30 years) compared to 1901-1980, with increased precipitation in spring and summer, and decreased precipitation in winter (see Section 3). Future precipitation trends are difficult to predict, though many models indicate no overall annual precipitation changes for the NPCW region through 2040, with precipitation increases occurring in spring and winter and decreases occurring in summer (Hamlet et al. 2005; Morgan et al. 2008; Littell et al 2011; see Section 3). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Drought and declines in summer precipitation could stress wetlands or moist meadow systems that rely mainly on surface water, possibly resulting in range contractions and/or vegetation conversions to more dry-adapted species (Poff et al. 2002). Possible refugia from precipitation changes and increased drought include seeps/springs and windward slopes.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Warmer air temperatures cause more precipitation to fall as rain rather than snow (Knowles et al. 2006), potentially shifting some streams from snow-dominated to transitional or rain-dominated, where the timing of flows is related to the timing of precipitation (Stewart et al. 2005; Littell et al. 2011). In the western U.S., this means more stream flow in fall and winter, and less in spring and summer (Elsner et al. 2010; Wenger et al. 2010). These changes are projected to occur soonest at mid-elevation sites (Regonda et al. 2005; Pierce et al. 2008; Nayak et al. 2010). A shift from snow to rain may also lead to large changes in hydrograph timing and magnitude (Elsner et al. 2010), and can lead to increased risk of mid-winter floods (Hamlet and Lettenmaier 2007; Luce et al. 2012). Increased flooding could lead to erosion of moist peat and topsoil (Weixelman et al. 2011), channel incision (Viers et al. 2013), and reduce opportunities for water storage and recharge.

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[55] Additional information on future climate exposure and potential impacts can be found in the Riparian Ecosystem section of this report.

Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). About 75% of runoff in the western U.S. is currently derived from precipitation that falls as snow (Service 2004), and as a result of snowpack declines, the spring freshet has decreased in both length and volume (Luce et al. 2012). Warming temperatures have also led to earlier runoff timing (Stewart et al. 2005). Less snowpack that melts earlier decreases available surface water and groundwater by limiting the duration of snowmelt-induced stream flow and groundwater recharge (Viers et al. 2013). Earlier snowmelt and decreased snowpack has also increased the length of the summer drought season (Stewart et al. 2009) and contributed to increased wildfire activity (Westerling et al. 2006). These climate-driven changes may lead to earlier timing of peak evapotranspiration and reduced soil moisture (Hamlet et al. 2005). For example, July 1 soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see also Section 3). Warming temperatures and changes in precipitation may lead to increased evapotranspiration rates, potentially contributing to a further reduction in soil moisture.

Warming temperatures have also resulted in lower summer flows (Rood et al. 2008; Luce and Holden 2009). For example, unregulated streams in Idaho experienced earlier peak stream flow and lower summer stream flows from 1967-2007 (Clark 2010). Earlier runoff timing and changes in seasonal flows (e.g., lower summer stream flows, higher winter flows) are projected to continue for the NPCW region (Wenger et al. 2010; Littell et al. 2011; see also Section 3). Changes in seasonal stream flow could be compounded during warm PDO cycles, which are correlated with reduced precipitation and may lead to overall annual declines in stream flow (Rood et al. 2005).

Changes in precipitation type, timing, and amount may have significant impacts on wetlands, moist meadows, and groundwater-dependent ecosystems. For example, shifts from snow to transitional watersheds and earlier peak runoff can lead to flashier runoff events that could increase sediment loads and erosion, exacerbate stream incision, affect channel and bank stability, reduce opportunities for water storage and recharge, and lead to drying of meadows (Viers et al. 2013). Declines in snowpack and earlier snowmelt could lead to a decrease in the water table (e.g., by affecting groundwater recharge), which may stress more hydric and mesic vegetation, promoting more xeric conditions (Viers et al. 2013). Soil moisture declines may result in species range contraction or changes to species composition (Poff et al. 2002), tree encroachment (Millar et al. 2004), or affect weed species colonization rates (Galatowitsch et al. 1999). Increased intermittent flows can alter channel morphology and stream bank stability, decreasing the quality, quantity, and duration of habitat available for aquatic biota (Shafroth et al. 2002; Lowry et al. 2011; Wenger et al. 2011). Earlier timing and longer low flows could also affect aquatic biota by increasing stream temperatures, potentially favoring warm water species and leading to declines in cold water species (Wenger et al. 2011). Reduced flows could also lead to shifts in vegetation composition and potential habitat conversion of GDEs, wetlands, or moist meadows.

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). More frequent or intense fires, especially those that occur near or upslope of wetlands, moist meadows, or GDEs, could increase the vulnerability of these systems to flash flood events and their associated impacts (e.g., increased sediment and debris loading, degraded bank stability, and increased erosion and incision) (Viers et al. 2013).

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of wetlands, moist meadows, and GDEs, including timber harvest, invasive species, and grazing.[56] Scientific literature also identified land use conversion for agricultural or urban development as a potential non-climate stressor. The degree of current exposure to these stressors is thought to be low in the NPCW region.

Timber harvest and other off-site human activities can alter the hydrologic and nutrient regimes that control the vigor and species composition of GDEs, wetlands, and moist meadows (Chadde et al. 1998; Poff et al. 2002). The hydrologic conditions of these systems, particularly surface

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[56] The collective degree these stressors increase sensitivity of wetlands, moist meadows, and GDEs was considered Low. Participant confidence associated with this evaluation was High. Current exposure to these non-climate stressors in the NPCW region was judged to be Low by workshop participants (Confidence: Moderate-High).

flow-through systems, are tightly linked with upland drainage integrity and condition, thus harvest activities in upland areas can have implications beyond their individual site (Chadde et al. 1998). Although GDEs, wetlands, and moist meadow systems currently experience a low degree of exposure to these activities in the NPCW region, timber harvest in upland areas and other offsite actions may contribute to or compound the changing hydrologic conditions that these systems face due to climate and climate-driven changes.

Invasive species have become well established in the NPCW region, but are not yet present in high numbers in wetlands, moist meadows, or GDEs. However, climate and climate-driven changes and/or increased anthropogenic activity may facilitate heightened invasion of noxious weeds (Chadde et al. 1998; Poff et al. 2002). For example, declines in spring runoff may create drier wetland edges where dry-adapted invaders can become established and warmer air temperatures may facilitate growth and reproduction rates of invasive species (Poff et al. 2002). Further, OHV and ATV use and/or altered fire regimes may increase colonization opportunities for Canada thistle (C*irsium arvense*), a common wetland invader that follows disturbance (Chadde et al. 1998).

Grazing can directly decrease native and endemic wetland, moist meadow, and GDE vegetation by removal and trampling, which can simultaneously allow for noxious weed encroachment (Chadde et al. 1998). Grazing, particularly overgrazing, may also increase soil compaction, reducing the water storage capacity of wetland soils and altering local hydrologic conditions (Chadde et al. 1998; Ecovista et al. 2003). Grazing has the potential to exacerbate climate-driven changes in soil moisture and water availability on riparian vegetation by further compacting soil, damaging soil structure, and reducing infiltration, all of which may facilitate faster wetland, moist meadow, and GDE degradation and vegetation conversion.

Some estimates suggest that 56% of Idaho's wetlands have been lost since 1860, mainly due to conversion for agriculture and/or urban purposes (Ecovista et al. 2003). However, current exposure to land use conversion is considered low within the NPCW region. As crucial habitat and refugia for focal species such as the Coeur d'Alene salamander (*Plethodon idahoensis*) and the western toad (*Anaxyrus boreas*), further wetland, moist meadow, and GDE decline and degradation could threaten the future persistence of these species (Ecovista et al. 2003). GDEs, wetlands, and moist meadows already face shifting hydrologic conditions due to climate and climate-driven changes; continuing to keep land use conversion activities to a minimum will help prevent synergistic impacts that could reduce habitat integrity and extent.

Adaptive Capacity

System extent, integrity, continuity, and diversity

The adaptive capacity of NPCW wetland, moist meadow, and groundwater-dependent ecosystems was considered moderate by workshop participants and reviewers.[57] This evaluation was due to high structural and functional integrity and moderate species diversity,

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[57] Confidence associated with this evaluation was High.

but low continuity and physical/topographical diversity. Wetlands are found throughout many western states but are inherently discrete features; in the NPCW region, they tend to be very small and scattered throughout the landscape (Ecovista et al. 2003). Some wetland fragmentation occurs on USFS lands, mainly stemming from past land use conversion (Ecovista et al. 2003). Despite past losses of overall wetland abundance (Ecovista et al. 2003), remaining wetlands still exhibit high structural and functional integrity. Wetlands, moist meadows, and GDEs in the NPCW region have low to moderate component species diversity, and very low physical, topographical, and functional group diversity, especially among wetland-obligate species.

Management potential

GDEs, wetlands, and moist meadows were judged by workshop participants to be of high ecological value and low to moderate value to the public. Workshop participants identified timber harvest, invasive species, and grazing as potential use conflicts for wetlands, moist meadows, and GDEs in the NPCW region. Recreation (e.g., ATV/OHV use) may also represent a potential use conflict for these systems. Peer-reviewed literature suggests that using and/or enhancing special area designations could help protect these systems from various human activities (e.g., timber harvest, road construction, grazing) and their associated impacts (e.g., heightened sedimentation and/or invasive species establishment) both now and in the future (Chadde et al. 1998; Ecovista et al. 2003). Peer-reviewed literature also suggests that preventing timber harvest and grazing in these watersheds, evaluating wetland, moist meadow, and GDE sites for harvest or grazing suitability, and/or monitoring areas with grazing activities or areas adjacent to timber harvest activities could help protect them from unsuitable or unsustainable disturbance (Chadde et al. 1998; Ecovista et al. 2003). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this system[58]. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature-generated Proposed Management Strategies[59]:

- Implement additional and/or enhance special area designations for wetlands, moist meadows, and GDEs.
    - Potential benefits: Protect systems from various human disturbance (e.g., timber harvest, road construction, grazing) and/or reduce disturbance-related impacts (e.g., sedimentation and invasive species risk) (Chadde et al. 1998; Ecovista et al. 2013).
    - Potential conflicts: Administrative or social barriers.
- Prevent timber harvest or grazing in wetland, moist meadow, or GDE watersheds or upland areas.

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[58] Additional management strategies that may be applicable can be found in the Riparian Ecosystems section of this report.

[59] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

  - Potential benefits: Prevent system disturbance and disturbance-related impacts (e.g., sedimentation and invasive species risk) (Chadde et al. 1998; Ecovista et al. 2013).
  - Potential conflicts: Administrative or social barriers.
- Evaluate wetland, moist meadow, and GDE sites for harvest or grazing suitability prior to starting those activities.
  - Potential benefits: Identify wetlands, moist meadows, or GDEs that may be the most vulnerable to potential impacts (Chadde et al. 1998; Ecovista et al. 2013).
  - Potential conflicts: May require additional institutional capacity and funding.
- Monitor wetlands, moist meadows, or GDEs adjacent to harvest activities or with current grazing activities. Consider monitoring vegetation, sediment loading, invasive species, and water chemistry, among others.
  - Potential benefits: Identify impacts with potential to modify management/activity levels accordingly.
  - Potential conflicts: May required additional institutional capacity or funding.

Aquatic Species[60]

Summary includes: Fall and Spring Chinook salmon, steelhead, cutthroat trout, westslope cutthroat trout, bull trout, and interior redband trout

Executive Summary

The relative vulnerability of aquatic species in this assessment is considered moderate, due to moderate-high sensitivity to climate-driven changes and non-climate stressors, and moderate-high adaptive capacity. Aquatic species are sensitive to climate and climate-driven changes that affect water temperatures and hydrologic regimes such as:

- increased air temperatures,
- decreased snowpack,
- earlier snowmelt and runoff timing, and
- shifts from snowfall to rainfall.

Warming temperatures and changes in precipitation significantly affect the hydrology of watersheds. For example, increasing air temperatures can influence stream thermal regimes, and warming stream temperatures may cause corresponding shifts in species distribution, energy budgets, phenology, and life histories. Decreased snowpack, earlier snowmelt, and shifts from snow to rain can lead to altered timing of flows and runoff, higher winter flows and increased flood risk, and lower summer flows, which can affect critical habitat features that species depend on, increase mortality of different species life stages, and/or exacerbate warming stream temperatures.

Aquatic species are also sensitive to non-climate stressors including:

- invasive species, and
- habitat homogenization.

Habitat homogenization (e.g., through transportation corridors, dams and water diversions, grazing, etc.) can reduce the habitat available to aquatic species, affect water quality, and exacerbate impacts of climate-driven changes in stream temperature and flow regimes. The adaptive capacity of aquatic species in the Nez Perce-Clearwater (NPCW) region is considered

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[60] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

moderate-high, due their moderate-high geographic extent and connectivity, high dispersal ability, and high intraspecific/life diversity.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

A challenge in understanding the effects of climate change on aquatic ecosystems is that aquatic organisms do not experience climate directly. Instead, changes in climate interact with hydrological conditions to influence key attributes of aquatic habitat, such as flow and thermal regimes. Further, just as topography can mediate the effects of climate change on terrestrial ecosystems (Dobrowski 2010; Sears et al. 2011), the diverse hydrologies found in heterogeneous watersheds can differentially filter the same regional climate signal (Armstrong and Schindler 2013). For example, the effects of changes in air temperature on stream temperature depend on variables such as canopy cover and the residence time of water (Holtby 1988; Caissie 2006). Further, the biological effects of changes in aquatic habitat conditions depend on the attributes of the organism and the level to which new habitat conditions deviate from optimal conditions. Changes in habitat may affect the same species differently, by having different effects at different life stages, or by differentially affecting individuals or populations with different phenotypes. While some of this complexity leads to irreducible uncertainties, some of it can be unraveled to generate predictions regarding the sensitivities of aquatic species to climate change. Below, the direct and indirect sensitivities of focal aquatic species are summarized.

The two most important attributes of lotic ecosystems are their flow and thermal regimes, which describe seasonal variation in water quantity and temperatures (Poff et al. 1997; Caissie 2006). These attributes are not independent; for example, the magnitude of stream flows mediates the seasonal dynamics of water temperature (Caissie 2006; Dickson et al. 2012; Armstrong and Schindler 2013). However, flow and temperature have distinctly different effects on organisms, so it is useful to consider them separately when assessing the sensitivity of organisms to climate-driven changes. Overall, the sensitivity of aquatic species to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[61]

Table 12. Potential aquatic species response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated aquatic species response
Warmer stream temperatures	• Higher metabolism requiring more food to survive, declines in growth rate, and may sexually mature earlier • Increased winter and spring temperatures: Eggs incubate more rapidly and young fish emerge earlier • Loss of habitat suitability (e.g., in stream reaches

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[61] Confidence associated with this evaluation was Moderate-High.

	that become too warm) • Displacement of cold-adapted fish by warm-adapted fish
Altered flow regimes	• High flow events: Increase scour events and sedimentation, direct mortality • Low flow events: Declines in habitat area, potential increased vulnerability to predators, reduced food, warmer stream temperatures, decreased growth, decreased survival

Sensitivity to temperature

Physiology and energy budgets: Temperature affects biological rates, particularly in the poikilotherms that dominate aquatic communities. Most attributes of physiological performance show a dome-like relationship to temperature; increases in temperature increase performance initially, but past the optimal temperature, performance drops off sharply due to thermal stress (Brett 1971; Hanson et al. 1997; Martin and Huey 2008). Temperature has direct effects on the every day function of organisms, mediating maximum rates of performance, such as swimming speed and digestive capacity. By affecting rates of respiration, temperature also affects metabolic costs. In general there is a non-linear increase in metabolic costs with increasing temperature. Initially, physiological performance increases with temperature faster than metabolic costs do, so the scope for growth and aerobic activity increases with temperature (Brett 1971; Hanson et al. 1997). However, at some temperature, known as the *pejus* temperature, performance begins to decrease (Portner and Knust 2007). Since temperature mediates the *maximum* capacities of organisms, the realized effects depend on ecological context. For example, the effects of temperature on fish growth depend on food availability (Brett 1971). If there is little food available and temperatures increases, a fish's energy budget responds to increases in metabolic costs, but not to increases in digestive capacity (i.e. it is not digestively constrained), so growth rates decline (Brett 1971; Crozier et al. 2010).

In addition to affecting the energy budgets of aquatic organisms, temperature affects their developmental rates and thus the timing of key life-cycle transitions (i.e. phenology). In theory, animals have evolved to synchronize their phenologies with seasonal variation in environmental conditions. There is concern that environmental change can cause the cues that trigger animal phenology to become unreliable, leading to mismatch between animal phenologies and optimal environmental conditions (Edwards and Richardson 2004; Winder and Schindler 2004).

Species-specific considerations: temperature

Energy budgets (salmonids): Salmonids are cold-water fishes, and generally exhibit maximum levels of physiological performance at approximately 15˚C (Brett 1971). Even bull trout, which are thought to require colder water temperatures than other salmonids, exhibited maximum growth rates at 16˚C when fed unlimited rations (Mesa et al. 2013). Since fishes usually feed at a fraction of their maximum ration (Armstrong and Schindler 2011), optimal temperatures for growth typically range from 8-15˚C (Brett 1971). Interestingly, the thermal physiology of salmonids may vary as much within a species as it does among species. For example, sub-populations of rainbow trout that rear in warm interior regions exhibit maximum growth rates at temperatures of 20˚C or higher, suggesting they are locally adapted to high temperatures (Rodnick et al. 2004).

Salmonids show seasonal variation in energy budgets, losing weight when food abundance and temperature combine sub-optimally, and rapidly growing and storing energy when food is abundant and temperatures are warm enough to provide sufficient digestive capacity (but cool enough to prevent thermal stress) (Scheuerell et al. 2007; Armstrong and Bond 2013). Seasonal variation in water temperature typically encompasses temperatures that hinder growth from both ends of the spectrum. Cool temperatures may reduce foraging ability and digestive capacity during winter, whereas high temperatures lead to excessive metabolic costs during late summer, especially if insect hatches have subsided and food abundance is low. For example; The John Day River of interior Oregon exhibits high summer temperatures that cause salmonids to seek thermal refuge (Torgersen et al. 1999a) or even die, in large-scale mortality events. However, steelhead were recently shown to spawn in warmer regions of the John Day watershed, likely because cool temperatures during winter and spring constrain growth opportunities for juveniles (Falke et al 2013). Thus, the net effect of changes in temperature on growth will be context dependent, but should generally depend on whether reduced thermal constraints on growth during fall, winter, and spring can mitigate increased thermal stress during summer. The critical context is that of food abundance—if there is insufficient food, warming temperatures during winter may actually exacerbate energy loss (Connolly and Petersen 2003), whereas if there is sufficient food, fish can likely mitigate negative effects of warmer summers, as long as thermal stress is not sufficient to cause mortality or long-term physiological harm. Fish are capable of rapidly storing energy, as maximum rates of energy intake are typically 3-9 times that of basal energetic costs under ideal thermal conditions (Hanson et al. 1997; Armstrong and Schindler 2011). Further, recent evidence shows that fish can manipulate their physiology to reduce energetic costs during periods of low foraging opportunity (Armstrong and Bond 2013).

In a warmer future, cold-blooded animals such as fish will incur higher metabolic costs and be more likely to endure physiological stress during the summer. However, warmer temperatures will generally increase physiological performance and growth potential during fall, winter, and spring. Maintaining foraging opportunities and thermal refugia will increase the probability that increased growth during the cooler months can mitigate decreased performance during warmer months.

Chinook salmon: Stream-rearing Pacific salmon are vulnerable to high temperatures during the summer, especially during periods with low flows. The capacity for streams to rear juvenile salmon is thought to mediate the population productivity of Chinook salmon (Quinn 2005). Recent evidence suggests that the negative density-dependence of survival is stronger for Chinook salmon in years with warmer temperatures (Crozier et al. 2010). This suggests warming temperatures could decrease the carrying capacity of streams. The likely mechanism underlying this observation is that the optimal temperature for fish growth decreases with decreasing food abundance. Thus as the number of fish increases, and there is less food to go around, fish growth will decline unless there is an associated increase in food abundance.

The effects of warming temperatures are especially concerning for adult salmon because they cease feeding during their spawning migrations. Since warming temperatures increase metabolic costs, they in turn decrease the time that a salmon has to migrate before it burns through its energy supplies. This implication is evident in the migratory behavior of salmon. When salmon encounter warm temperatures during their spawning migrations, they actively seek cooler water, which slows their metabolism (Torgersen et al. 1999; Newell and Quinn 2005; Goniea et al. 2006). Chinook salmon may be especially vulnerable to warmer summer temperatures, because the Spring-run life-history (i.e. stream-type), which is prevalent in the interior states, has an especially long freshwater residence time; these fish migrate during the spring and then reside in freshwater over the summer before spawning in the fall (Quinn 2005). Thus these fish may spend upwards of seven months in freshwater, relying on stored energy the entire time. Fall-run (ocean-type) fish spend less time in freshwater as adults, but their spawning migration encounters the warmest part of the watershed (the downstream portion) during the warmer part of the year (later summer and early fall). This suggests fall-fun Chinook salmon are more vulnerable to acute stress from extreme temperatures (e.g., >20˚C), whereas spring-run fish are more vulnerable to energy loss from prolonged exposure to sub-optimal temperatures.

Bull trout: Bull trout are of conservation concern across their range in the Western U.S. (Rieman et al. 2007). In the interior regions, warming temperatures have coincided with restricted spatial distributions of bull trout (Eby, unpublished data). Bull trout appear to have a colder thermal-niche than other salmonids (Dunham et al. 2003) and exhibit high mortality at temperatures exceeding 20˚C (Selong et al. 2001).

Cutthroat trout: Like bull trout, cutthroat trout are often found in cooler streams than other salmonids. While this is partly due to the thermal physiology of cutthroat trout (Bear et al. 2007), interactions with invasive trout species (e.g., hybridization with rainbow trout) may also constrain the range of cutthroat trout to cooler headwater streams (Novinger and Rahel 2003). The suitability of headwater streams for cutthroat trout could actually improve with warming, as cold temperatures currently constrain the distribution and density of cutthroat trout in high elevation headwater streams (Coleman and Fausch 2007). However, warming could also increase the potential for rainbow trout to invade current cutthroat strongholds.

Phenology (salmonids): Changes in thermal regimes are known to affect the phenology of salmonids. The incubation rates of salmonid embryos are well-studied and embryos require a certain amount of cumulate temperature (i.e. degree-days) prior to hatching (Quinn 2005). Altered thermal regimes will change the amount of time required for incubation. These biological responses can already be seen in rivers where large flood-control dams have altered thermal regimes in salmonid spawning habitat (ODFW 2000). A major concern with altered incubation rates is that they could cause early life-cycle events in salmonids to occur at sub-optimal times. For example, if spawning dates remain constant and incubation rates increase, embryos will hatch earlier, and fry will have to begin exogoneous feeding earlier. If optimal foraging conditions do not exhibit a synchronized response, the resulting trophic mismatch could lead to decreased growth rates for recently emerged age-0 salmonids. This sort of trophic mismatch has been documented in a variety of systems, including plankton communities in lakes (Winder and Schindler 2004) and ungulates in terrestrial ecosystems (Post and Forchhammer 2008).

Many species of salmonids exhibit ontogenetic shifts in habitat. These shifts can be subtle, such as age-0 fish moving from the margins of the stream toward the thalweg or extreme, such as steelhead that migrate from small mountain tributaries to pelagic habitats of the Pacific Ocean (Quinn 2005). Changes in temperature have the potential to alter the timing of these habitat shifts, or to change the optimal time for transitioning between habitats. For example, warmer temperatures could alter the cues for migration and cause fish to migrate at different times. Alternatively, fish could migrate at the same time, but the seasonal dynamics of conditions in destination habitats could change. Mismatch between the timing of migration and the timing of optimal habitat conditions has the potential to reduce the survival of individuals and the productivity of populations. For example, warming may shift the timing of peak food availability in the estuary, as well as the timing of cues for smoltification, and water velocities along smolt travel routes. If these climate-induced changes lead to trophic mismatch and reduce the foraging opportunities of smolts as they reach the ocean, there could be a reduction in early-marine survival of salmon smolts, which is thought to limit population productivity (Quinn 2005). Recent studies have documented shifts in the migration phenology of salmon (Kovach et al. 2012), but fewer studies have been able to explore whether these shifts are leading to mismatch events (Scheuerell et al. 2009). Maintaining population-level diversity in migration phenology will likely buffer salmonids from mismatch and reduce inter-annual variation in survival (Schindler et al. 2010).

Warming may also change the propensity of populations to migrate. In many salmonids multiple migratory strategies persist in the same population (Quinn 2005). In theory these multiple strategies co-exist because they generate comparable levels of fitness over time (Gross 1985). Climate change will likely change the costs and benefits associated with migration, in turn changing the proportions of migrants in populations. This has the potential to reduce the resiliency of populations, by diminishing the population-diversity that confers stability through the portfolio effect (Schindler et al. 2010). Further, certain migratory phenotypes may have particular cultural and economic significance, so the societal value of a species can change even if its abundance or biomass remains constant. For example, warming conditions in the Columbia River could reduce the survival of anadromous rainbow trout (steelhead), leading to an increase in the stream-resident phenotype, which is less prized by anglers.

For many fishes, reproductive phenology is temperature-dependent; i.e., spawning is induced by specific changes in temperature. Thus changes in thermal regimes could alter the spawn-timing of fishes. The timing of spawning is difficult to monitor effectively, but recent data show that some populations of Pacific salmon are migrating to spawning grounds at earlier dates

(Kovach et al. 2013). The combination of early spawning migrations and increased water temperatures during incubation could cause initial life-cycle transitions in salmonids to occur substantially earlier in year.

In a warmer future, the optimal timing of salmon life-cycle events will likely change. Salmon phenology is already changing in association with recent warming. Salmon evolved in climates that exhibited high inter-annual variation, so they should exhibit tactics that buffer them from mismatch with the environment. It remains to be seen whether these coping mechanisms will continue to work as temperatures reach levels that are outside historic levels of variation.

Sensitivity to Flow Regimes

Flow has several direct and indirect effects on aquatic organisms. High flow events scour and deposit sediments, disturbing the benthos and reorganizing stream channels (Stanford et al. 2005). The effects of high flow events are scale-dependent. Over the small spatial and temporal extents at which disturbances occur, high flow events cause mortality. However at broader scales, high flow events may be critical to maintaining the habitat features that aquatic organisms depend on (Reeves et al. 1995). Species have evolved their life-cycles to both capitalize on –and take refuge from – high flow events (Lytle and Poff 2004). Changes in the timing of high flow events may have deleterious effects by exposing flood-vulnerable life-stages to flooding or by not exposing flood-reliant life-stages to flooding. Changes in the magnitude and frequency of high flow events may increase mortality in aquatic organisms by exposing them to levels of physical stress that exceed what these organisms have evolved to cope with.

Low flow events can also cause stress for organisms for several reasons. As flow decreases, so does habitat area, which causes organisms to compete for less space, and may make aquatic organisms more vulnerable to terrestrial and avian predators. Decreased flows are accompanied by decreased stream velocities, which result in less food being delivered in the stream drift. Lastly, low flows during summer generate warmer stream temperatures (Caissie 2006). The combination of reduced food abundance and increased water temperatures are likely to result in decreased growth potential for aquatic organisms. Indeed low flows have been shown to decrease the growth and survival of trout (Xu et al. 2010).

Species-specific considerations: flow regimes

Pacific salmon: Pacific salmon spawn in the fall and offspring emerge in the late winter or spring. Low flows during the summer and fall may be stressful for migrating adults. Mass mortality events in both fall and spring-run Chinook salmon have been linked to high temperatures due to low flows (Bartholow and Henriksen 2006; Salmon Recovery, http://www.salmonrecovery.gov/Habitat/JohnDaywatertemps.aspx). Some salmon populations may also depend on high flows to allow passage to upstream spawning areas. For example, spring-run (stream-type) Chinook often migrate to spawning grounds during the high flows that occur from late-winter through early-summer (Quinn 2005). However, high flow events during the fall and winter can scour the gravels where embryos incubate, reducing egg-to-fry survival.

Increased severity of winter floods has been linked to decreased egg-to-fry survival in Washington (Greene et al. 2005).

Snowmelt and the resulting runoff in spring may be important for aiding the seaward migration of salmon smolts. Reduced flows during the spring have both direct and indirect effects on smolt migrations. The reduced stream velocities increase the travel time required for smolts to reach the ocean – this in turn increases the time of exposure to predators. Low flows may also make smolts more vulnerable to predators per unit of time exposed. Increased temperatures associated with low flows may increase the maximum swimming speed of salmon, but they also increase the swimming speeds and digestive capacities of fish preying on salmon (Petersen and Kitchell 2011). Thermal constraints on smallmouth bass prevent them from being effective predators on Chinook salmon smolts in watersheds such as the John Day River (Lawrence et al. 2012). With warming, species such as smallmouth bass, walleye, and Northern Pikeminnow will almost certainly become more effective predators on salmon smolts (Petersen and Kitchell 2011). Spring-run Chinook are particularly vulnerable to predation because they originate higher in river networks and have longer migrations to sea. However, although fall-run Chinook have shorter seaward migrations, many populations emigrate as age-0 fry (Quinn 2005), which makes them vulnerable to broader size-spectra of predators, likely increasing their predation risk per unit time of migration.

Brook trout, brown trout, and bull trout: Brook trout, bull trout, and brown trout spawn in the fall. Like Pacific salmon, these species may be vulnerable to increased frequency and timing of high flow events during the embryo-incubation period of winter (Wenger et al. 2011). The effects of decreased egg-to-fry survival on population productivity depends on whether subsequent life stages are limiting for survival (i.e. population bottlenecks). Salmonids exhibit strong density-dependence in juvenile survival (Quinn 2005) so reduced embryo survival is likely to increase the survival rates of juveniles.

Cutthroat and rainbow trout: Cutthroat and rainbow trout spawn in the late winter and spring, and emerge later in the spring or in the summer. High winter flows that are detrimental to fall-spawning salmon do not negatively affect embryo survival of cutthroat and rainbow trout, and may even be beneficial (Wenger et al. 2011). Cutthroat trout are negatively impacted by interactions with invasive trout species including brown trout and brook trout. Increased severity of winter flooding could benefit cutthroat trout by reducing competitive interactions with non-native fall-spawning salmonids (Wenger et al. 2011).

Lower flows may harm salmonids by increasing mortality during migration or reducing the carrying capacity of juvenile rearing habitat. Higher peak flows could decrease egg-to-fry survival in fall-spawning salmonids.

Future climate exposure

Important climate and climate-driven changes to consider for aquatic species include those that alter hydrologic regimes and stream temperatures, such as increasing air temperatures, decreased snowpack, earlier snowmelt and runoff timing, and shifts from snowfall to rainfall.

Aquatic species are also sensitive to increased wildfire as it can impact stream habitat and water quality in both the short- and long-term.

Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing, and are projected to continue to increase, with more significant increases in winter and summer (Littell et al. 2011; see Section 3). In the summer, stream temperatures may warm at rates of 0.3-0.45˚C per decade, causing a net increase of 1.2-1.8˚C by mid-century (Isaak et al. 2011). Further, stream isotherms may shift 5-143 km upstream if air temperatures rise by 2˚C (Isaak and Rieman 2013). In general, increasing air temperatures may contribute to reduced stream flows and warmer stream temperatures, which can create thermal and flow barriers and lead to shifts in species distribution and phenology (Rieman and Isaak 2010). However, recent research by Arismendi et al. (2012) indicates a less direct association between air and stream temperature trends, highlighting the importance of local, non-climatic factors (e.g., shading from riparian vegetation). Refugia from reduced stream flow and elevated stream temperatures may include streams with significant groundwater influence and high elevation areas with intact riparian function. Please refer to the *Sensitivity* section above for a summary of aquatic biota interactions with stream temperature.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Warmer air temperatures cause more precipitation to fall as rain rather than snow (Knowles et al. 2006), potentially shifting some streams from snow-dominated to transitional or rain-dominated, where the timing of flows is related to the timing of precipitation (Stewart et al. 2005; Littell et al. 2011). In the western U.S., this means more stream flow in fall and winter, and less in spring and summer (Elsner et al. 2010; Wenger et al. 2010). These changes are projected to occur soonest at mid-elevation sites (Regonda et al. 2005; Pierce et al. 2008; Nayak et al. 2010). A shift from snow to rain may also lead to large changes in hydrograph timing and magnitude (Elsner et al. 2010), and can lead to increased risk of mid-winter floods (Hamlet and Lettenmaier 2007; Luce et al. 2012). Increased winter flood incidence can increase scour events, alter bank morphology and stability, and increase sedimentation rates and woody debris recruitment in aquatic systems (Rieman and Isaak 2010; Yarnell et al. 2010).

About 75% of runoff in the western U.S. is currently derived from precipitation that falls as snow (Service 2004). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). As a result of snowpack declines, the spring freshet has decreased in both length and volume (Luce et al. 2012). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Warming temperatures have led to earlier runoff timing (Stewart et al. 2005) and lower summer flows (Rood et al. 2008; Luce and Holden 2009). Specifically, unregulated streams in Idaho experienced earlier peak stream flow and lower summer stream flows from 1967-2007 (Clark 2010). Similarly, in the northwestern U.S., historical changes show declines in stream flows over the last 50 years (Moore et al. 2007; Luce and Holden 2009; Clark 2010), including declines summer flows (Luce and Holden 2009; Leppi et al. 2011). However, there may be several factors that influence changes in stream flow aside from direct climate change including land use contributions, climate cycles (e.g., PDO), and evapotranspiration (Hoerling and Eischeid 2007), and it still remains a question as to whether the changes are precipitation or transpiration related (see Luce et al. 2012 for a full discussion on this).

Earlier runoff timing and changes in seasonal flows (e.g., lower summer stream flows, higher winter flows) are projected to continue for the NPCW region (Wenger et al. 2010; Littell et al. 2011; see also Section 3). Changes in seasonal stream flow could be compounded during warm PDO cycles, which are correlated with reduced precipitation and may lead to overall annual declines in stream flow (Rood et al. 2005). Less snowpack that melts earlier also decreases available surface water and groundwater by limiting the duration of snowmelt-induced stream flow and groundwater recharge (Viers et al. 2013). Earlier snowmelt and decreased snowpack may also increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006).

Changes in precipitation type, timing, and amount can have varying impacts on stream flow. Altered stream flow (amount, timing, duration) contributes to changes in geomorphology and physical processes, and affects the ecological integrity of aquatic and riparian systems (Poff et al. 1997; Furniss et al. 2010). Low stream flows could reduce connectivity of smaller streams to mainstream flows, and reduce habitat amount and quality (Furniss et al. 2010; Yarnell et al. 2010; Luce et al. 2012). Changes in stream flow and patterns of groundwater recharge may also result in range contraction and/or local loss of species (Luce et al. 2012). Groundwater-dominated systems may buffer short-term variations in stream flow, although long-term drought (e.g., lasting several years) may increase sensitivity of these systems (Lall and Mann 1995; Shun and Duffy 1999). Further, in years with smaller snowpack, groundwater inputs may be diminished and headwater stream temperatures warmer (Isaak et al. 2011). Please refer to the *Sensitivity* section above for a summary of aquatic biota interactions with stream flow regimes.

Aside from being sensitive to hydrologic changes, aquatic species are also sensitive to altered disturbance regimes such as wildfire. Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Increased wildfire frequency or severity could increase the aerial extent of streams exposed to wildfire-induced high temperatures, remove streamside vegetation, and lead to increased channel disturbance from post-fire debris flows (Dunham et al. 2007; Isaak et al. 2010; Rieman and Isaak 2010; Luce et al. 2012). Fire can also induce hydrologic changes, including changes to snowmelt, runoff, peak flows, and low flows (Luce et al. 2012). Increased solar radiation post-fire has been linked to advances in snowmelt timing (1-2 weeks), with variable changes in peak flows (e.g., severely burned small basins experienced significant changes in peak flows but at the basin scale, no increase was observed) (Troendle et al. 2010; Luce et al. 2012). Changes in runoff following fire have been primarily attributed to changes in soil properties (i.e., surface sealing, water repellency) (Luce et al. 2012). Water repellency seems to be associated with certain vegetation communities (e.g., chaparral, subalpine fir), and is more likely where fires burn severely; further, dry, hot summers of the western U.S. may be ideal conditions for water repellent behavior (Luce et al. 2012). Peak flows in streams can be significantly higher post-fire than pre-fire, though it is more frequently documented that post-fire peak flow is less than pre-fire peak flow (Shakesby and Doerr 2006). The impacts of wildfire on fish species are varied, and include local extirpations (that may or may not be permanent; Rinne 2003; Howell 2006), altered growth and maturation (Dunham et al. 2007), renewal of spawning gravels (Reeves et al. 1995), or a temporary productivity boost due to nutrient inputs (Malison and Baxter 2010).

It is important to note that the impacts of climate change will depend on the specific species under consideration, as well as the location within a basin and stream network and the physical context of a given stream reach.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of aquatic species, including invasive species and habitat homogenization (e.g., road networks and maintenance, dams and water diversions, mining, logging and wood harvesting, grazing, fire suppression practices).[62]

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[62] The collective degree these stressors increase sensitivity of aquatic species was considered Moderate-High. Participant confidence associated with this evaluation was Moderate-High. Current exposure to these non-climate stressors in the NPCW region was judged to be Moderate by workshop participants (Confidence: Not stated).

Invasive species

Invasive species in freshwater ecosystems range from micro-organisms to large fish and include "semi-invasive" species, which may be native to some watersheds in a region, but have been introduced beyond their historical range (e.g., rainbow trout). Native salmonids are under threat due to three types of interactions with invasive species: predation, competition, and hybridization. One of the biggest threats to native salmonids has been the introduction of closely related species, often congeners. Introduced brook trout and rainbow trout hybridize with closely related bull trout and cutthroat trout, respectively. Hybrid swarms can have devastating effects, resulting in widespread declines or extirpation (Rieman et al. 1997; Kanda et al. 2002). The effect of climate on hybridization is complex and hard to predict. Culling invasive salmonids may be a valuable conservation strategy, but is controversial because rainbow trout and brook trout support substantial recreational fisheries.

Invasive species that prey on native salmonids include lake trout, brown trout, rainbow trout, walleye, smallmouth bass, crayfish, and others. A frequently observed result in ecology is that habitat features can mediate predator-prey interactions as much, or even more than the abundance of predator and prey (Kauffman et al. 2007; Armstrong et al. 2010). Warming temperatures can increase predation rates by increasing the foraging efficiency and digestive capacity of predators. This may be especially problematic with invasive warm-water fishes, such as smallmouth bass. As temperatures exceed 15˚C, the physiological performance of native salmonids decreases, while that of their warm-water predators increases (Hanson et al. 1997). Further, the population sizes and spatial extent of warm-water invasives are likely to increase with warming temperatures, leading to higher levels of predation (Lawrence et al. 2012). Flow regimes likely mediate predator-prey interactions, but empirical studies on this topic are scarce. Lower flows are likely to make native salmonids more vulnerable to predation by invasive species, but a lack of empirical data has limited the ability to model predator-prey dynamics under altered flow regimes.

It is difficult to predict how climate change will affect competitions between native salmonids and invasive species. Warmer temperatures will increase metabolic costs, which could exacerbate competition if trophic resources do not also increase (Brett 1971; Crozier et al. 2010). Further, warm water tolerant species like brown trout may increase in abundance as the frequency of thermal stress events increases.

Warmer temperatures will likely exacerbate the negative effects of introduced species on native salmonids. Increased severity of winter flooding may provide spring-spawning native salmonids some relief from invasive fall-spawning salmonids.

Habitat homogenization

Habitat homogenization is one of the most widespread impacts of human development. For example, floodplain habitat is among the most endangered habitat types on the planet. Rivers are inherently heterogeneous, exhibiting high levels of spatial variation in habitat variables such as water velocity and temperature (Stanford et al. 2005). Human land uses such as dams or water diversions, logging, mining, livestock grazing, channelization, transportation corridors, and development of floodplain habitat have reduced the diversity of habitat available for aquatic organisms as well as disrupted water quality. For example, livestock grazing can affect both water quality and aquatic habitat by increasing bank erosion and channel incision, compacting soil, limiting plant cover and regeneration capacity of native species, and lowering the water table (Ecovista et al. 2003; Poff et al. 2011).

Road networks and road management can affect the aquatic environment by altering sediment delivery mechanisms, increasing fine sediment yields, increasing incidence of invasive species, modifying vegetation community structure and function, altering large woody debris recruitment, simplifying stream function, and providing travel routes for grazing animals (Williams 1954; Young et al. 1967; Roath and Krueger 1982; Trombulak and Frissell 2000). Generally, as the density of roads in a watershed increases, aquatic habitat quality decreases. Road construction causes severe disturbance to soils on slopes (Rieman and Clayton 1997) and, in a scientific literature review considering the effects of roads on aquatic systems, Trombulak and Frissell (2000) underscored the importance of avoiding construction of new roads and removing or restoring existing problematic roads. Roads built decades ago are often located in valley bottoms next to streams and are difficult to relocate (Swift and Burns 1999), and current recreation use and a lack of sufficient maintenance can result in increased sediment delivery (Grace and Clinton 2007). Timber harvest and road building can accelerate the frequency and volume of debris slides and hillslope sediment loss (Naiman et al. 2005). Roads produce runoff and sediment in almost every precipitation event, which may have significant impacts on aquatic biota as they occur more frequently (Luce et al. 2012).

Dams can provide both a threat and benefit to riparian ecosystems. Dams and water diversions have eliminated habitat and impeded connectivity for aquatic species, and can contribute to lower stream flows in summer; however, dams that create temperature-stratified reservoirs may be beneficial to aquatic species, as they periodically release cold water to downstream habitats and could help mitigate low flows and warming temperatures driven by climate change (e.g., see Null et al. 2011). However, release of this water in winter can affect thermal regimes in salmonid spawning habitat (e.g., see ODFW 2000), potentially affecting incubation rates and altering salmonid life-cycle phenology. Through flow regulation, dams can reduce peak flows for increased water storage and increase or decrease low base flows (Poff et al. 2011). For example, depending on where dams are located, they may be able to elevate low flows (Poff et al. 2006).

Habitat heterogeneity increases the ability of aquatic organisms to endure climate-related stress, particularly when different patches of habitat respond differentially to the same regional climate signal (Torgersen et al. 1999). For example, myriad aquatic organisms exploit the stable temperatures of groundwater-dominated habitat during cold- or hot-spells that render other habitats unsuitable (Torgersen et al. 2012). Likewise, off-channel habitats may be important refuge from high flow events, whereas main-channel habitats are critical when low flows isolate and desiccate off-channel habitats (Rypel et al. 2012). Maintaining and restoring habitat heterogeneity is an adaptation strategy that will likely be more robust to uncertainty than alternative strategies such as those that rely on predictive species distribution modeling.

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of NPCW aquatic species was considered moderate-high by workshop participants and reviewers.[63] This evaluation was due to high connectivity, high dispersal ability but significant barriers to dispersal (e.g., transportation corridors, logging, energy production and mining, dams and water diversions), and high intraspecific/life history diversity.

Intraspecific/life history diversity

Today's phenotypes may perform poorly in a warmer future, but species can change their phenotypes over time through two primary mechanisms: evolution and phenotypic plasticity.

The raw material for evolution is genetic variation. If genetic variation is high, there is more potential for natural selection to craft phenotypes that perform better in a warmer future. A simulation model of evolutionary responses to warming in Fraser River salmon showed that the initial level of genetic variation had a strong influence on whether populations will be viable under multiple warming scenarios (Reed et al. 2011). Salmonids have lost much of their historical genetic diversity at the southern extent of their range. For example, the abundance of Pacific salmon species are at ~1-10% of their historical abundance in the lower-48 states, and though all five species may be extant, roughly 30% of populations have been extirpated (Gustafson et al. 2007). Further, many salmon runs are now dominated by hatchery stocks, which have low genetic diversity and can cause outbreeding depression in wild stocks (Araki et al. 2007; Araki et al. 2008).

As temperatures increase in streams, salmonids will more frequently experience temperatures that are sub-optimal for growth (>15˚C) and harmful (>20˚C). The key question is whether salmonids can evolve in ways that increase their thermal tolerance. Research suggests that *Tcrit*, the temperature at which fish become incapacitated, is similar among populations and species of salmonids (Myrick and Cech 2000; Rodnick et al. 2004). This suggests there are evolutionary constraints on thermal tolerance, and salmonids will never be able to survive temperatures above ~30˚C. However, there is evidence that salmonids can evolve better performance in temperatures ranging from 15-30˚C (i.e., temperatures in between the theoretical optimum temperature and Tcrit). Rainbow trout from warm desert streams (the redband ecotype) exhibit the same Tcrit as other salmonids, but they have a higher aerobic scope, maximum rate of respiration, and metabolic power (Rodnick et al. 2004). This means that desert redband trout have an elevated capacity to function at high temperatures. Similar thermal adaptation has been documented among populations of Fraser River sockeye (Eliason et al. 2011) that experience different thermal regimes during spawning migrations. These studies confirm that salmonids are capable of evolving traits that perform better in warmer water temperatures. The remaining questions are whether populations threatened by warming have sufficient genetic variation to evolve, and whether warm-performance phenotypes have

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[63] Confidence associated with this evaluation was High.

high enough heritability to evolve quickly (Reed et al. 2011). Populations of Chinook salmon rearing in cool regions do show heritability for heat tolerance phenotypes (Beacham and Withler 1991). Salmonids have been a model organism for documenting rapid evolution in the wild (Quinn 2005), and can show measurable changes in phenotypes over periods of a century or less (Quinn 2005). However, there is little known regarding the pace at which populations can evolve physiological traits that confer high performance in warm water. There are trade-offs associated with warm-performing phenotypes. For example, populations from warm streams in southeast Oregon had reduced swimming performance at 12˚C compared to a population from a cooler stream (Rodnick et al. 2004). Thus the ability to survive hot summers may come at the expense of performance during other seasons. Further, large trout have lower thermal tolerance than small trout (Rodnick et al. 2004; Breau et al. 2011), so a warmer future may include smaller, less fecund individuals.

Salmonids exhibit extreme genetic diversity, both within and among populations. Preserving this genetic diversity will increase adaptive capacity.

Phenotype plasticity occurs when a single genotype can produce different phenotypes. This creates the potential for animals to exhibit traits that perform better in a warmer future, without having to evolve new genes (Crozier et al. 2008). Examples of plastic responses include altered migration timing (McCullough 1999; Hyatt et al. 2003; Crozier et al. 2008), behavioral thermoregulation (Torgersen et al. 1999; Armstrong et al. 2013), altered age-at-maturity (Crozier et al. 2008), habitat shifts (Crozier and Zabel 2006), and flexibility in organ size (Armstrong and Bond 2013). Land use and management actions may affect the capacity for salmonids to exhibit plastic responses to climate change. For example, behavioral thermoregulation may be critical to the persistence and health of salmonids across their geographical range (Torgersen et al. 1999; Goniea et al. 2006; Breau et al. 2011; Armstrong et al. 2013). However, the capacity for animals to thermoregulate depends on the persistence of spatial variation in water temperature. Loss of floodplain habitat and depleted water tables can reduce thermal heterogeneity and eliminate the thermal refugia that enable salmonids to find optimal temperatures during hot and cold spells (Torgersen et al. 2012).

Behavioral thermoregulation may be critical to the persistence of salmonids, yet habitat homogenization threatens the expression of this adaptive behavior.

Pacific salmon: Pacific salmon integrate across freshwater and marine habitats. This makes them both resilient and vulnerable to climate change. Spending 1-6 years at sea buffers salmon from climate-related disturbances in freshwater. For example, if there is a fire or drought and the disturbance lasts for only a year, then individuals at sea will not be impacted. The longer lived a species, and the more life-history diversity and overlapping generations, the more the species is expected to be buffered from ephemeral disturbances in freshwater (Quinn 2005).

Though Pacific salmon spread risk by performing their life-cycle across a large geographic extent, it also exposes them to more potential climate stressors. For example, a Chinook salmon returning to Idaho is subject to myriad climate-related threats, such as ocean acidification, altered phenology in marine food webs, increased risk from invasive warm-water predators in migratory routes, and increased energetic costs or physiological stress during migration.

Management Potential

Aquatic species were judged by workshop participants to be very highly valued species by the public. Workshop participants identified transportation corridors (i.e., road networks and maintenance), fuels/vegetation management, grazing, dams, and mining as potential use conflicts for the NPCW region. Management strategies from the peer-reviewed literature are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for these species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Literature-generated Proposed Management Strategies[64]:

- Focus on critical roads and/or road infrastructure (e.g., culverts) and relocate or restore them (Luce et al. 2001; Rieman and Isaak 2010). Select critical roads by (1) identifying the highest ecological priority areas, (2) within those, identifying the most damaging roads, and (3) within those, identifying the roads that can be effectively decommissioned or mitigated (Luce et al. 2001).
    - Potential benefits: Minimize the harmful effects of roads on the aquatic environment, limit the impacts of relocation/restoration, improve stream network connectivity, buffer effects of peak flow events (reduce erosion and sedimentation).
    - Potential challenges: May require increased institutional capacity, research, or funding.
- Focus fuels/vegetation management in areas with existing road systems, and use minimal impact harvest techniques (Brown et al. 2004b).
    - Potential benefits: Likely improve aquatic function, improve aquatic habitat diversity and complexity by increasing post-fire woody debris flows (Isaak et al. 2010), potentially increase terrestrial subsidies for aquatic species by changing the riparian tree community from conifers to red alder, which seem to host more invertebrates (Piccolo and Wipfli 2002).
    - Potential challenges: Adminstrative or social barriers.
- Limit intensity, range, and timing of grazing to protect the most sensitive aquatic habitats, particularly those with low channel confinement, low gradient stream banks, and those with high exposure to future climate impacts (Ecovista et al. 2003).
    - Potential benefits: Reduce sediment loads, decrease incidence of invasive species, and improve system resilience.
    - Potential challenges: Adminstrative or social barriers.

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[64] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. See Luce et al. (2012) for additional management considerations.

Species management: Foraging opportunities and food webs

Temperatures increase metabolic costs for fishes, but whether this effect is negative depends on the feeding rate of a fish (Brett 1971). In the ranges of ~10-18˚C, the effects of warming are highly contingent on food abundance. This suggests that managers have the potential to increase the thermal tolerance of fishes by maintaining and restoring the productivity of food-webs. For example, the decline of bull trout has been largely blamed on the loss of cold-water habitats, but may also be due to the widespread decline of Pacific salmon, which were historically a primary food source for many populations of bull trout (Lowry 2009). The thermal niche of bull trout would likely be wider if opportunities to forage on salmon were restored. Salmonids rearing in streams will be increasingly susceptible to low flow periods during the late summer, when metabolic costs are high and total habitat volume is reduced. In general, aquatic insect emergence subsides by late summer, and fish rely more heavily on terrestrially-derived food sources, such as spiders, grasshoppers and other invertebrates (Wipfli and Baxter 2011; Nakano and Murikami 2001). The magnitude of these terrestrial subsidies may be influenced by land-use practices and forest type. For example overgrazing can reduce riparian vegetation and diminish terrestrial subsidies (Edwards and Huryn 1996), whereas logging can actually increase terrestrial subsidies by changing the riparian tree community from conifers to red alder, which seem to host more invertebrates (Piccolo and Wipfli 2002). While the effects of logging on stream subsidies is complex and likely context-dependent, it is clear that restoring riparian vegetation can buffer streams from climate change by not only decreasing stream temperatures (Holtby 1987), but also by increasing the magnitude of terrestrial subsidies (Edwards and Huryn 1996; Wipfli and Baxter 2011), thereby increasing food abundance and thermal tolerance.

Canada Lynx and Wolverine[65]

Executive Summary

The relative vulnerability of Canada lynx and wolverine in this assessment is considered high, due to high sensitivity to climate and non-climate stressors, and low-moderate adaptive capacity. Lynx and wolverine are sensitive to climate and climate-driven changes including:

- increased temperatures (wolverine),
- decreased snowpack depth (wolverine and lynx), and
- wildfire (lynx).

Increased temperature may reduce availability of wolverine summer habitat and/or force wolverines to move to higher elevations. Decreased snowpack may increase lynx competition with other predators, and may reduce available wolverine den sites, cub survival, or dispersal. Altered wildfire regimes may benefit lynx by providing and/or maintaining early seral coniferous stands.

Canada lynx and wolverine are also sensitive to non-climate stressors including:

- logging and timber harvest (lynx),
- recreation (lynx), and
- loss of core habitat and connectivity due to land use changes (lynx and wolverine).

A number of activities including fire suppression practices, trapping, and livestock grazing may also affect lynx. Lynx and wolverines are particularly vulnerable to habitat fragmentation due to land use changes (e.g., roads, urban development). Land use changes combined with climate-driven declines in snowpack are likely to further limit core habitat and connectivity for both species. The adaptive capacity of lynx and wolverine is considered low-moderate, due to their small, isolated populations and lack of connectivity (e.g., due to dispersal barriers).

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[65] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity & Exposure

Sensitivity to climate and climate-driven changes

Canada lynx, *Lynx canadensis*, and wolverine, *Gulo gulo*, are found in high elevation, alpine/subalpine areas (1500 m to >2600m) with deep snowpack. Canada lynx exhibits indirect sensitivities to climate and climate-driven changes such as decreased snowpack and altered wildfire regimes. Wolverine exhibits both direct and indirect sensitivities to climate and climate-driven changes including increasing temperature, decreased snowpack, and earlier snowmelt. Overall, the sensitivity of Canada lynx and wolverine to climate and climate-driven changes was evaluated as high by workshop participants.[66]

Canada lynx occur in boreal and montane forests, and select habitats where snowshoe hares, their main prey source, are most abundant (Mowat et al. 1999). These habitats generally include dense, early seral coniferous stands, although mature stands with dense understory are also used for hunting (Mowat et al. 1999), denning, and movement corridors (Koehler 1990; Aubry et al. 1999). Deciduous-dominated stands with dense understories can also support hares (Koehler 1990) and lynx. In particular, dense understory cover appears to be an important habitat feature for both hares (Hodges 1999a; Hodges 1999b) and lynx (Mowat and Slough 2003), and stand structure appears to be more important than forest cover type (Mowat et al. 1999). Lynx and snowshoe hare may utilize a wider range of habitats in summer, including areas with less dense understory (Hodges 1999a; Hodges 1999b; Mowat et al. 1999). It is important to note that southern populations of lynx (i.e., below the Canadian border) may prey on a wider diversity of species (e.g., red squirrel, grouse, flying squirrel, ground squirrel, porcupine, beaver, mice, voles, and shrews, among others) due to lower average hare densities and differences in small mammal communities (Ruediger et al. 2000).

In central Idaho, Douglas fir on moist sites at higher elevations may serve as primary lynx habitat; Douglas fir, grand fir, western larch, and aspen forests interspersed within subalpine forests may also contribute to lynx habitat (Ruediger et al. 2000). Subsequent to disturbance, lodgepole pine communities often provide high quality foraging habitat for lynx (Ruediger et al. 2000; Mowat and Slough 2003). Natural disturbances (e.g., insect and disease outbreaks, wildfire) could benefit lynx by creating additional foraging habitat.

In addition to stand structure, snow conditions are also an important factor to consider in lynx habitat. Lynx have large feet and long legs, making them specialized hunters in deep, soft snow. Deep snowpack currently limits the potential for competition with other predators, such as coyotes, mountain lions, or bobcats (Ruediger et al. 2000). However, crusting or compaction of snow (e.g., as a result of more freeze-thaw events), or loss of snowpack, may reduce the competitive advange of lynx (Buskirk et al. 1999), or contribute to lynx starvation and reduced recruitment (Ruediger et al. 2000).

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[66] Confidence associated with this evaluation was Moderate-High.

Table 13. Potential Canada lynx or wolverine response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated Canada lynx and wolverine response
Warming temperatures	Wolverine • Reduce availability of summer habitat and/or force migration to higher elevations • Reduce caching/refrigeration success of carrion prey, potentially affecting wolverine population dynamics
Decreased snowpack	Lynx • Increase competition (interference and exploitation) with other predators (e.g., coyotes) Wolverine • Reduce cub survival • Reduce dispersal • Reduce caching/refrigeration success of carrion prey, potentially affecting wolverine population dynamics • Reduce potential den sites
Altered wildfire regimes	Lynx • Provide and/or maintain early seral coniferous stands

Wolverines also select for high elevation habitats with deep snow in winter (Inman 2013). Wolverines were once considered to be a habitat generalist, however recent findings indicate that all wolverine reproductive dens are located in areas with deep, persistent spring snow cover (Magoun and Copeland 1998; Aubry et al. 2007; Schwartz et al. 2007; Copeland et al. 2010). Spring snow cover is thought to aid survival of young (i.e., by providing a thermal advantage; Pullianinen 1968) and provide refuge from predators (Pullianinen 1968; Persson et al. 2003). Schwartz et al. (2009) also suggest that successful dispersal of wolverines is largely limited to dispersal paths defined by the presence of persistent spring snow cover. Wolverines rely on carrion buried in snow as a winter food source, digging 10 feet into snowpack to access cached carcasses. As they require snowpack for both breeding and food storage, spring snow cover generally indicates areas of year-round habitat use (Copeland et al. 2010).

Wolverines may also exhibit some sensitivity to temperature. Wolverines move up in elevation during the summer, suggesting avoidance of summer heat. However, recent research indicates that wolverine preference for lower summer temperatures may be due to elevational or habitat preferences (e.g., persistent spring snow) rather than a response to high summer temperatures (Copeland et al. 2010). Regardless, temperature may play an important role in habitat selection at finer spatial scales (Copeland et al. 2010). Elevational movement may also reflect seasonal variation in prey availability (Copeland et al. 2007; Krebs et al. 2007).

Life History

The Canada lynx is iteroparous, typically has one reproductive event per year, and reaches sexual maturity around 2 years of age. Breeding occurs between March and April, kittens are born between May and June (Quinn and Parker 1987, Slough and Mowat 1996 *in* Ruediger et al. 2000), and lynx have been reported to have between 1 and 4 kittens per litter (Ruediger et al. 2000). Den sites typically occur in downed logs or root wads in or adjacent to foraging habitat, which provides thermal cover and predator protection for kittens, and an abundance of prey for mothers (Ruediger et al. 2000). Lynx may move den sites several times while raising kittens, following prey density (Ruediger et al. 2000). Lynx are solitary and territorial, occupying subalpine habitats between 1500 to 2000 m (Aubry et al. 1999); in Montana, lynx have large home ranges (e.g., 238 km^2 for males and 115 km^2 for females) with low prey densities (Squires and Laurion 1999).

Wolverines are iteroparous with low fecundity (Inman 2013). They can have one reproductive event each year and produce up to 4 young under ideal conditions, although young mortality rates may be high (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/gulo-gulo), especially if winter food sources are scarce (Inman 2013). Reproductive denning begins in early February to mid-March, and dens may be located under boulders or downed trees buried beneath snow (Magoun and Copeland 1998). Lactation occurs from February through April, and juvenile growth occurs during summer; both processes are dependent on abundant food (Inman 2013). Male wolverines typically reach sexual maturity at or after 2 years old, while females reach sexual maturity at or after 1 year of age. Females tend to produce biannually or less frequently (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/gulo-gulo). Female reproductive activity is likely linked with prey availability (Inman 2013), while litter production and litter size is related to female age (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/gulo-gulo). In the Greater Yellowstone Area, wovlerines typically select for elevations above 2600 m, and have large home ranges (e.g., 797 km^2 for males and 303 km^2 for females) that rarely or never overlap with other individuals (Inman 2013).

Future climate exposure

Important climate and climate-driven changes to consider for lynx and wolverine include warming temperatures (wolverine), decreased snowpack (wolverine, lynx), earlier spring snowmelt (wolverine), and altered wildfire regimes (lynx). Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Increasing temperatures could affect wolverines in several ways. For example, increasing summer temeprtures may reduce the availability of wolverine summer habitat and/or force wolverines to higher elevations (Copeland et al. 2010). Increasing temperatures and/or declines in spring snow cover may reduce the caching/refrigeration success of carrion prey, potentially affecting wolverine population dynamics (Inman 2013).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005) and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). McKelvey et al. (2011) also project significant snowpack declines in central Idaho by 2045, although some locations maintain spring snow cover and connectivity to other snow-covered areas (e.g., along the Idaho-Montana border). Warming temperatures and declines in spring snow cover will likely reduce the size of wolverine habitat patches and connectivity, which could lead to many small and isolated wolverine populations (Copeland et al. 2010; McKelvey et al. 2011). The potential costs associated with movement between these shrinking snow islands are likely to increase (e.g., wolverines have farther to travel), and could lead to fewer successful wolverine dispersals (McKelvey et al. 2011). Decreased snowpack may reduce potential den sites, and earlier spring snowmelt may reduce the warmth provided by deep, persistent snow surrounding reproductive dens, which could affect cub survival (Copeland et al. 2010; McKelvey et al. 2011). Decreased snowpack or earlier snowmelt may increase competition (exploitation and interference) between lynx and other predators (e.g., coyotes). For example, coyotes and lynx have historically occupied different winter habitats due to the inability of coyotes to travel and effectively hunt in deep snow (Crete and Lariviere 2003). Decreased snowpack or earlier snowmelt resulting in crusting or snow compaction may reduce the competitive advange of lynx (Buskirk et al. 1999), leading to increased competition for snowshoe hares or shifting lynx out of areas occupied by coyotes, thus limiting lynx population size (e.g., see O'Donoghue et al. 1997).

Earlier snowmelt and decreased snowpack may increase the length of the summer drought season (Stewart et al. 2009) and may lead to earlier desiccation of soils (Hamlet et al. 2007). July 1 soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see also Section 3), and warming temperatures and an increasing number of high temperature events, both projected for the region (e.g., see Section 3; Diffenbaugh et al. 2005), may exacerbate soil moisture deficits (Bollenbacher et al. 2013). Warmer temperatures and declines in soil moisture could have negative impacts on subalpine forest species such as spruce and subalpine fir. Changes in subalpine vegetation communities may affect forage and cover for snowshoe hares, with subsequent impacts on lynx.

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Wildfire and other disturbance regimes (e.g., insects and disease) historically played an important role in maintaining diverse forest successional stages. In the Rocky Mountains, wildfire regimes included frequent (35-100 years) stand-replacing or mixed-severity fire, and infrequent (200+ years) stand-replacing fires (Hardy et al. 1998 *in* Ruediger et al. 2000). Major fire events created diverse, early successional forests that provided habitats preferred by snowshoe hares (although not in the short-term), and therefore foraging habitat for lynx (Koehler 1990). Less intense, more frequent fires also created more structural diversity at a smaller scale (Ruediger et al. 2000). Increasing fire frequency may benefit lynx by providing and/or maintaining early seral coniferous stands, although more frequent, severe wildfires could negatively impact lynx.

Sensitivity and current exposure to non-climate stressors

Lynx may be sensitive to non-climate stressors such as logging and timber harvest, fire suppression practices, trapping, livestock grazing, recreational activities, and loss of core and connectivity habitat due to land use changes. Wolverines are also sensitive to loss of core and connectivity habitat due to land use change.

Logging and timber harvest may have varying impacts on lynx. For example, clearcuts may restrict lynx movement and use patterns (Koehler 1990). Further, snowshoe hares appear to avoid clearcuts and, as a result, lynx do not appear to spend much time hunting in these areas (Mowat et al. 1999). Salvage logging following wildfire or other disturbances can negatively affect lynx if large-diameter trees are removed, as large dead trees are important for providing foraging cover and/or denning habitat (Ruediger et al. 2000). Conversely, timber harvest or disturbance events (e.g., insect and disease outbreaks) in mature forests may benefit lynx by creating regenerating environments (Mowat and Slough 2003). However, compared to regenerating burns, regenerating clearcuts are unlikely to support lynx densities due to lack of sufficient understory density (Mowat and Slough 2003).

A number of activities including fire suppression practices, trapping, livestock grazing, and recreation may also affect lynx. Fire suppression practices, which have led to a reduction in large forest fires, have reduced the amount of available habitat for lynx and may have contributed to population declines (Koehler 1990). In some cases, trapping may also have contributed to declines (Poole 2001 *in* Mowat and Slough 2003). Livestock grazing has the potential to affect lynx indirectly by impacting snowshoe hare habitat and populations (e.g., by changing the structure and/or composition of native plant communities that support snowshoe hare; Ruediger et al. 2000). Increased winter recreation (i.e. snowmobiling) has provided coyotes with access to deep snow areas, potentially increasing competition between coyotes and lynx (e.g., see O'Donoghue et al. 1997), although this relationship is inconclusive (e.g., see Dowd and Gese 2012). Anecdotal evidence suggests that recreational activities in wolverine habitat can cause wolverines to abandon their dens, although there are other examples of no apparent effect (Heinemeyer et al. 2010; Heinemeyer and Squires 2012).

Lynx and wolverines are particularly vulnerable to habitat fragmentation due to land use changes (e.g., roads, urban development). Wolverine occurrence has been correlated with remoteness from human development (Rowland et al. 2003; May et al. 2006; May et al. 2008) and, while there is no indication that wolverine dispersal is currently limited by human development, increasing road and housing densities may deter dispersal (Inman 2013). Exurban and suburban development, as well as highways, may also impede lynx movement (Ruediger et al. 2000). In general, human development limits core habitat and connectivity for both lynx and wolverine.

Adaptive Capacity

Species extent, integrity, continuity, and diversity

The adaptive capacity of the lynx and wolverine is considered low-moderate due to their small, isolated populations and lack of connectivity (e.g., due to a number of dispersal barriers). Although they are wide ranging and can be found throughout most of northern North America, populations in the U.S. have been significantly reduced due to habitat fragmentation and urbanization. Northern Idaho and Montana feature isolated populations of both species with low population densities. The Canada lynx is listed as a threatened species, while the wolverine is a candidate species for listing as of 2010.

Lynx can disperse large annual distances (100-1000 km) (Slough and Mowat 1996 *in* Climate Change Sensitivity Database http://climatechangesensitivity.org/node/67; Mowat et al. 1999; Ruggiero et al. 1999b). Various human activities – including land use conversion, agriculture, logging, dams and water diversions, urban development, and transportation corridors – can prevent lynx dispersal, either by creating impassable barriers or by creating large open spaces that lynx behaviorally avoid. Wolverines also have high dispersal ability; both sexes can disperse more than 100 km and movements of more than 150 km are not uncommon. Some dispersal barriers do exist (e.g., roads, agricultural, industrial, and urban development), but with varying impacts on wolverine dispersal. For example, wolverines have been documented to cross major highways, but road mortalities may play a large factor in declines of low-density populations. Further, it is unknown how these barriers will affect wolverine dispersal in response to climate change, as previous studies have only documented their impacts in relation to annual movement cycles (Packila et al. 2007 *in* Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/gulo-gulo; Schwartz et al. 2009).

Lynx and wolverine typically exhibit low life history and genetic diversity, but may exhibit moderate behavioral diversity. For example, in the absence of snowshoe hare, lynx will prey on a variety of other species, including red squirrels, grouse, and voles (Ruediger et al. 2000). In addition, lynx can occupy a variety of forest cover types. Wolverines have evolved to time breeding, lactation, and juvenile growth (all energetically-demanding processes) to occur earlier that other carnivores, an adaptation that allows them to exploit cached food during winter and relatively high food sources during summer (Inman 2013). Wolverines in the Northern Rockies feature only a few haplotypes, indicating low genetic diversity (Schwartz et al. 2009). The lynx features relative genetic homogeneity across all populations, which may indicate genetic exchange across isolated populations (Koehler and Aubry 1994 *in* Ruediger et al. 2000). However, changing climate patterns may result in an increase in genetic differentiation, potentially leading to isolated east-west populations of lynx in North America (Row et al. 2014)

Management potential

Lynx and wolverine were judged by workshop participants to be very highly valued species by the public. Workshop participants identified winter recreation and timber harvest as potential use conflicts for lynx, and recommended that lynx core habitat or important connectivity routes could be closed to winter recreation. Peer-reviewed literature identified loss of core habitat and connectivity due to land use changes as an additional use conflict. Maintaining and/or enhancing large areas of contiguous habitats and corridors to facilitate connectivity for both species were recommended as a potential management strategy (Ruggiero et al. 1999a; Ruediger et al. 2000; Inman 2013). Please refer to Ruediger et al. (2000) for a list of potential management actions for lynx, and Inman 2013 for a list of potential management actions for wolverine. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Coeur D'Alene and Idaho Giant Salamanders[67]

Executive Summary

The relative vulnerability of the Coeur D'Alene Salamander (CDS) and the Idaho Giant Salamander (IGS) in this assessment is considered high, due to high sensitivity to climate and climate-driven changes, moderate-high sensitivity to non-climate stressors, and low adaptive capacity. Salamanders are highly dependent on moist and shaded habitats, and are sensitive to climate and climate-driven changes including:

- altered precipitation,
- reduced soil moisture or drought, and
- wildfire.

Changes in precipitation that reduce soil moisture, lower stream flows, or increase drought incidence can affect salamander habitat availability and quality and reduce foraging and breeding opportunities. Wildfires during late summer and fall may cause further habitat fragmentation and reduce foraging and breeding success for salamanders

Salamanders are also sensitive to non-climate stressors including:

- human disturbances (e.g., dams and water diversions, road construction, and timber harvest),
- pollution, and
- introduced species.

These non-climate stressors can further alter and fragment suitable habitat for salamanders, as well as affect salamander fitness. Salamanders have low adaptive capacity due to their small, isolated and declining populations, and low dispersal ability.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Coeur D'Alene salamanders (CDS), *Plethodon idahoensis,* and Idaho giant salamanders (IGS), *Dicamptodon aterrimus*, are found in mountainous, coniferous forests, and are closely tied to

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[67] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

water for reproduction and thermal and hydric stability, often inhabiting talus areas or fractured rock formations near cool, wet environments, such as along stream edges, seepages, and waterfall spray zones. During wet weather, they can also be found in talus areas further from water and under leaf litter and bark (Cassirer et al. 1993; Cassirer et al. 1994). Both species need areas of high moisture for survival during their non-dormant periods (April-September), and exhibit sensitivity to precipitation changes, drought, reduced soil moisture, and wildfire. Overall, the sensitivity of salamanders to climate and climate-driven changes was evaluated as high by workshop participants.[68]

Table 14. Potential CDS and IDS response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated CDS and IDS response
Reduced soil moisture	• More frequent CDS retreat underground or to rocky, mesic areas to avoid desiccation • Reduced foraging, breeding, and dispersal opportunities
Drought (late summer and fall)	• Loss of habitat (e.g., seepages and low flow streams could dry out) • Reduced foraging, breeding, and dispersal opportunities
Precipitation changes (e.g., more rain-on-snow events)	• Higher exposure to desiccation if retreat routes are blocked by sedimentation
Wildfire	• Reduced IGS larval success if fire reduces canopy cover, increases humidity, and compounds soil moisture reductions • More frequent CDS retreat underground to avoid desiccation • Reduced foraging, breeding, and dispersal opportunities from loss of habitat • Higher exposure to desiccation if retreat routes are blocked by post-fire sedimentation events

Reduced moisture can impact salamander fitness, foraging activity, and retreat habitat (Ohanjanian 2004). For example, precipitation changes that lead to increased summer drought and/or reduced soil moisture could force CDS to retreat underground more often to avoid desiccation (Ohanjanian 2004). Reduced moisture can also limit critical forest habitat and/or canopy cover between watercourses, affecting salamander dispersal ability and causing habitat fragmentation (Ohanjanian 2004). Prolonged drought in late summer and fall may also reduce foraging and breeding success for CDS (Cassirer et al. 1994). CDS that occupy seepages and low-volume creeks may be most sensitive to moisture changes, as these sites have low flows during much of the year (Ohanjanian 2004) and may be more vulnerable to moisture reductions

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[68] Confidence associated with this evaluation was Moderate-High.

resulting from climate change. Altered precipitation patterns such as increased rain-on-snow events are likely to increase sedimentation rates and landslides (Harp 1997 *in* Raymond et al. 2013), which can block underground CDS retreat routes (Ohanjanian 2004). Conversely, IGS appear tolerant of a wide range of local conditions within streams, although habitat could become lost or unsuitable as a result of drought or increased temperatures (Nature Serve 2013, http://www.natureserve.org/explorer).

Similar to drought, wildfires during late summer and fall may cause further habitat fragmentation and reduce foraging and breeding success for salamanders in the Nez Perce-Clearwater (NPCW) region. CDS and IGS live and have evolved in areas where natural fires occur (Nature Serve 2013, http://www.natureserve.org/explorer), but climate-driven changes in fire frequency and intensity may negatively impact these species. Fires reduce canopy cover and may decrease local soil moisture and relative humidity, which can reduce larval success for the IGS (Idaho Department of Fish and Game (IDFG) 2005f) and force underground retreat of CDS. In addition, burned areas increase the risk of mudslides and altered sedimentation regimes (e.g. see Riparian Ecosystems summary), which can bury CDS retreat routes and/or leave them exposed to desiccation.

Life history

CDS are nocturnal, lungless terrestrial salamanders that feed on invertebrates (Wilson and Larsen 1988) and may spend up to 7 months per year underground (Cassirer et al. 1994). CDS are only active above-ground between April-May and mid-September-early November, typically retreating to moist underground areas from June to mid-September when weather is warmer and hibernating during winter. CDS are opportunistic feeders that prey on aquatic and semi-aquatic invertebrates (Wilson and Larsen 1988), typically foraging at night around moist habitats or in adjacent wet forests, but never far from their retreat cover (Cassirer et al. 1994). CDS are highly dependent on wet habitats, requiring moisture for respiration and hydration, and thus are closely associated with fissured bedrock or wet talus along streams and springs that protects them from desiccation in the summer and freezing in the winter (Cassirer et al. 1994). CDS are iteroparous, K-selectionists. Mating typically occurs above-ground in late summer and fall, and occasionally in the spring (Lynch 1984 *in* Cassirer et al. 1994). CDS can have 2 reproductive events per year, before and after hibernation. CDS eggs are laid in water, and young emerge in September (Ohanjanian 2004; IDFG 2005e). Males reach sexual maturity in 3.5 years and breed annually, while females reach maturity after 4.5 years and breed biannually (Lynch 1984 *in* Cassirer et al. 1994).

IGS are iteroparous and typically have one reproductive event per year, but can produce over 40 young under optimal conditions (Lohman and Bury 2005). IGS reach sexual maturity at 3 to 4 years of age, but may only breed biannually or less frequently (Lohman and Bury 2005). IGS eggs are laid in water, and larval development is positively correlated with canopy cover (IDFG 2005f).

Future climate exposure

Important climate and climate-driven factors to consider for salamanders in the NPCW region include changes in precipitation leading to reduced soil moisture and drought, decreased stream flow, and altered wildfire regimes.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Warmer air temperatures cause more precipitation to fall as rain rather than snow (Knowles et al. 2006), potentially shifting some streams from snow-dominated to transitional or rain-dominated, where the timing of flows is related to the timing of precipitation (Stewart et al. 2005; Littell et al. 2011). In the western U.S., this means more stream flow in fall and winter, and less in spring and summer (Elsner et al. 2010; Wenger et al. 2010). A shift from snow to rain may also lead to large changes in hydrograph timing and magnitude (Elsner et al. 2010), and can lead to increased risk of mid-winter floods (Hamlet and Lettenmaier 2007; Luce et al. 2012). Flooding can affect both geomorphological and hydrological features of salamander habitat, including sediment transport, saturation, and scour.

Warming temperatures have led to earlier runoff timing (Stewart et al. 2005) and lower summer flows (Rood et al. 2008; Luce and Holden 2009). Specifically, unregulated streams in Idaho experienced earlier peak stream flow and lower summer stream flows from 1967-2007 (Clark 2010). Earlier runoff timing and changes in seasonal flows (e.g., lower summer stream flows, higher winter flows) are projected to continue for the NPCW region (Wenger et al. 2010; Littell et al. 2011; see also Section 3). Changes in seasonal stream flow could be compounded during warm PDO cycles, which are correlated with reduced precipitation and may lead to overall annual declines in stream flow (Rood et al. 2005). Less snowpack that melts earlier also decreases available surface water and groundwater by limiting the duration of snowmelt-induced stream flow and groundwater recharge (Viers et al. 2013).

Earlier snowmelt and decreased snowpack may also increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006), and may lead to earlier desiccation of soils (Hamlet et al. 2007). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3). Reduced soil moisture could cause increased habitat fragmentation and altered breeding and foraging success for the CDS and IGS, especially if the extent of wet forests is reduced. As moisture levels decrease, salamanders may become more dependent on areas of rocky retreat and streams near mature forests to reduce temperature stress and rates of evaporation. Potential refugia include shaded, moist areas on north-facing slopes and areas near streams.

Wildfire is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). Altered fire regimes could contribute to salamander habitat loss, degradation, or fragmentation. Increased wildfire activity could also increase the risk of mudslides and sedimentation (e.g., see Riparian Ecosystems summary), contributing to blocked CDS underground escape routes.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of salamanders, including human disturbances, pollution, and introduced species.[69] For example, dams and water diversions, mining, blasting of rock for road construction, pollution, timber harvest, and managed forest fires can all impact salamander habitat and fitness. Damming and other water diversion methods may flood habitat or dry downstream areas, impacting water quality and availability and/or reducing salamander habitat extent (Cassirer et al. 1994; IDFG 2005e). Road construction, trail construction, and recreation activities may further fragment habitat, cause sedimentation fill near streams and seeps, increase runoff pollution, and/or result in salamander mortality (IDFG 2005e). Managed forest fires, controlled burns, and timber harvest can also impact salamander habitat and reduce reproductive potential by reducing overhead forest canopy, increasing sedimentation, altering water quality and flow, and reducing soil moisture and relative humidity (IDFG 2005e). Current IGS abundance appears to be higher in unfragmented headwater drainages with few to no roads (Sepulveda and Lowe 2009).

Salamanders are highly sensitive to pollution that enters their habitat through overland or stream flows. Pollutants can come from a variety of sources, including pesticides (IDFG 2005f), herbicides used for weed control, and chemical applications commonly used for fire suppression, mining, or road maintenance (IDFG 2005e). Pollutants can affect salamanders in numerous ways (e.g., by disrupting embryo development, causing mortality; see Smithsonian Environmental Research Center, http://www.serc.si.edu/education/resources/watershed/stories/salamanders.aspx), and climate-driven changes, such as increasing temperatures and low flows, could magnify the impacts of pollutants on CDS and IGS (e.g., by increasing relative concentration levels).

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[69] The collective degree these stressors increase sensitivity of salamanders was considered Moderate-High. Participant confidence associated with this evaluation was High. Current degree of exposure to these stressors was evaluated as Low (Confidence: Moderate).

Salamanders are also sensitive to the introduction of non-native species (e.g., fish and frogs). For example, introduced bullfrogs or non-native game fish, whose ranges may expand under changing climate conditions, can increase competition or predation risk for the CDS (IDFG 2005e).

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of the CDS and IGS is considered low due their small, isolated and declining populations, low dispersal ability, and limited genetic exchange.[70] CDS occur in isolated populations in northern Idaho, western Montana, and southeastern British Columbia, Canada, while IGS occur only in Idaho and Montana. CDS can be found in drainages of the St. Joe River and North Fork of the Clearwater River (Wilson et al. 1997), as well as in the Selway, Kootenai, and Moyie drainages (IDFG 2005e), while IGS can be found in parts of the Coeur d'Alene, Clearwater, Lochsa, and Salmon River drainages (IDFG 2005f). CDS are listed as an S2 Imperiled species in Idaho and as a Sensitive species in the USFS Northern Region, and IGS are listed as an S3 Vulnerable species in Idaho. There are usually less than five CDS individuals in a given habitat site (IDFG 2005e), and 95% of known populations in Idaho and Montana have been verified extant since 1987 (Cassirer et al. 1994). Although not all location occurrences have been identified, it appears that the southern, eastern, and western edges of CDS distribution are limited by lack of moisture, geological features, and high temperatures (Wilson and Larsen 1998). IGS are thought to be declining in the Clearwater and South Fork Salmon River drainages (IDFG 2005f). Small salamander populations may be experiencing genetic pressures typical of small populations (e.g., inbreeding depressions) (Ralls et al. 1986 *in* Cassirer et al. 1994).

Salamander populations experience high isolation and limited dispersal ability due to habitat fragmentation caused by natural features (e.g., dry areas) and human disturbance (e.g., roads, timber harvest) (Wilson et al. 1989 *in* Cassirer et al. 1994; Cassirer et al. 1994; Wilson et al. 1997). Habitat fragmentation interferes with natural metapopulation dynamics (Sepulveda and Lowe 2009; Mullen et al. 2010). For example, there appears to be little genetic exchange between widely separated (>100 km) CDS populations (Howard et al. 1993), and genetic exchange between adjacent populations may be controlled by the presence of suitably moist dispersal corridors (Cassirer et al. 1994). IGS usually disperse no more than 1 km (Nature Serve 2013, http://www.natureserve.org/explorer), and IGS in the Salmon River drainage may be experiencing little genetic exchange with other populations, as they are separated by almost 40 km (IDFG 2005f). However, IGS readily recolonizes disturbed areas as they are recovered or restored, with recolonization likely occurring within catchments (Mullen et al. 2010). IGS may also demonstrate some adaptive capacity to a high frequency of natural disturbances (e.g., landslides and sedimentation) (Sepulveda and Lowe 2009).

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[70] This evaluation is based on information in the scientific literature, as workshop participants were unable to evaluate adaptive capacity in the time allotted.

Management potential

Workshop participants did not identify any use conflicts or management options for CDS or IGS in the NPCW region, but peer-reviewed literature suggests that CDS and IGS populations would benefit from management strategies that maintain essential hydrological characteristics of habitat, protect mature forest canopy to provide shade and reduce evaporation rates, and maintain or maximize connectivity between riparian forested habitats (IDFG 2005e; IDFG 2005f; Sepulveda and Lowe 2009). The scientific literature identified road and trail construction and timber harvest activities as anthropogenic, non-climate stressors that may affect salamander habitat. Potential management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

<u>Literature-generated Proposed Management Strategies</u>[71]:

- Prevent loss, degradation, and fragmentation of salamander habitat (e.g., maintain road condition, protect roadless areas, restore connectivity, include stream-side protection zones to buffer streams from anthropogenic activities, maintain 60% canopy cover, leave understory vegetation, avoid activity within 25 ft of seep sites and 100 ft of stream sites).
    - <u>Potential benefits:</u> Maintain salamander habitat connectivity, maintain canopy cover, increase habitat resiliency by preventing habitat alterations (e.g., increased sedimentation, altered hydrology, loss of important rock structures used for shelter).
    - <u>Potential challenges:</u> Possible conflicts with recreational access.
- Install culverts and overland flow barriers near roadside seeps.
    - <u>Potential benefits:</u> Facilitate salamander migration and reduce habitat alteration via sedimentation.
- Time nearby timber harvest to occur during periods when salamanders are dormant below-ground (e.g., November-March, July-August) to minimize disturbance to habitat, movements, and breeding activities.
    - <u>Potential benefits:</u> Reduce salamander vulnerability to direct and indirect timber harvest impacts (e.g., mortality and increased sedimentation).
    - <u>Potential challenges:</u> Possible conflicts with timber harvest practices (e.g., winter weather conditions).
- Survey and monitor salamander populations in the NPCW region.
    - <u>Potential benefits</u>: Improve understanding of population status and trends, habitat condition, and potential impacts of climate and non-climate stressors.
    - <u>Potential challenges</u>: May require increased institutional capacity.

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[71] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region. Strategies generated from Cassirer et al. 1994; IDFG 2005e; IDFG 2005f; Sepulveda and Lowe 2009.

Dry Forest Birds[72]

Executive Summary

The relative vulnerability of dry forest birds in this assessment is considered moderate to moderate-high, due to moderate sensitivity to climate and climate-driven changes, high sensitivity to non-climate stressors, and moderate-high adaptive capacity. Dry forest birds are sensitive to climate and climate-driven changes including:

- increasing temperature,
- precipitation changes, and
- wildfire.

Warming temperatures (direct sensitivity) as a result of climate change may lead to an overall upward elevational shift of some dry forest bird species. Precipitation changes (indirect sensitivity) may increase or decrease dry forest bird habitat extent or quality. For example, precipitation changes could allow ponderosa pine to expand into new habitat areas, but soil moisture reductions could also lead to heightened mortality of old growth ponderosa pine. Low severity, high frequency fire may expand dry forest habitat and help maintain open, old growth forest structure that dry forest birds require for nesting, but wildfires that re-burn large, crown-fire burned areas may limit forest establishment and translate to decreased dry forest habitat.

Dry forest birds are also sensitive to non-climate stressors including:

- fire suppression practices,
- timber harvest, and
- livestock grazing.

These non-climate stressors may exacerbate climate-driven changes and/or alter dry forest bird habitat. For example, increased stand densities as a result of fire suppression practices may reduce dry forest bird occupancy, result in unsuitable breeding habitat, or reduce shrub and grass understories causing declines in associated insect prey. Further, increased fuel loads resulting from fire suppression activities can increase the likelihood of more frequent and severe forest fires. The adaptive capacity of dry forest birds is considered moderate-high, as

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[72] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

most species have small populations with patchy distribution across the Nez Perce-Clearwater landscape, but have high dispersal ability.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Dry forest birds in the Nez Perce-Clearwater (NPCW) region – including the white-headed woodpecker (*Picoides albolarvatus*), flammulated owl (*Otus flammeolus*), Lewis's woodpecker (*Melanerpes lewis*), and pygmy nuthatch (*Sitta pygmaea*) – occupy open, old growth dry forest ecosystems consisting of multi-storied, low density, and moderate to open canopied ponderosa pine (*Pinus ponderosa*) and Douglas fir (*Pseudotsuga menziesii*). Dry forest bird species exhibit both direct and indirect sensitivity to climate and climate-driven changes in temperature, precipitation, and wildfire. Overall, the sensitivity of dry forest bird species to climate and climate-driven changes was evaluated as high by workshop participants.[73]

Table 15. Potential dry forest bird responses relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated dry forest bird response
Warmer temperatures	• Upward elevational shift of flammulated owls • Higher nesting and incubation success of white-headed woodpeckers • Increased forage opportunities • Altered habitat range
Precipitation changes and reduced soil moisture	• Altered habitat range, and potential decline of old growth ponderosa pine habitat
Wildfire	• Low severity, high frequency fires: Increased habitat extent, quality, and complexity, reduced predator abundance • Crown re-burns: Reduced habitat availability and complexity

Some dry forest bird species demonstrate direct sensitivity to changes in temperature. For example, flammulated owls may currently be limited at low elevations due to high diurnal temperatures and limited at high elevations by cold nighttime temperatures that affect their ability to thermoregulate (Waterbury et al. 2009). Warming temperatures as a result of climate change may result in an overall upward elevational shift of this species (McCallum 1994 *in* Waterbury et al. 2009). Further, warmer temperatures are linked with higher nesting and incubation success of white-headed woodpeckers (Hollenbeck et al. 2011).

Warming temperatures and precipitation changes also indirectly affect dry forest birds by influencing prey availability and habitat extent. For example, warmer temperatures are linked with higher surface-bark insect abundance, possibly increasing the forage opportunities for

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[73] Confidence associated with this evaluation was Moderate.

white-headed woodpeckers and other dry forest birds (Hollenbeck et al. 2011). Similarly, local precipitation and temperature patterns influencing insect abundance are thought to control the timing of Lewis's woodpecker breeding more than local photoperiods (Dudley and Saab 2003). Changes in precipitation and temperature increases may alter habitat availability for dry forest bird species. For example, decreased precipitation may create more areas for dry forest type, as ponderosa pine can colonize hot, dry conditions of disturbed sites (Bollenbacher et al. 2013) and/or expand into drier mixed mesic zones (K. Hazelbaker, pers. comm., 2014). Warming temperatures may also expand the range of Douglas fir into higher elevations (K. Hazelbaker, pers. comm., 2014) and increase survivorship of ponderosa pine seedlings across the NPCW region (Scott et al. 2013), which could provide more habitat for dry forest bird species. However, it is important to note that dry forest tree species have limited establishment and growth opportunities during the dry summer months of July and August and during drought periods (Scott et al. 2013). Further, soil moisture deficits could increase old growth ponderosa pine mortality due to heightened competition with dense stands of young trees (Mehl and Haufler 2001), possibly reducing preferred, old growth habitat for dry forest birds. During cool, wet climates, Douglas fir or denser ponderosa pine can also become established (Bollenbacher et al. 2013).

Wildfire appears to be the most significant climate-driven change indirectly impacting dry forest birds, as low severity, high frequency understory fires maintain the open, dry forest habitat required by these species. For example, thick-barked ponderosa pine typically survives such fires while thin-barked Douglas fir or ponderosa pine seedlings do not, allowing ponderosa pine forest to develop into large patches of open, old growth structure able to persist for centuries (given appropriate moisture and temperature regimes) (Bollenbacher et al. 2013). Low severity-high frequency fires, representative of historic fire regimes, may also increase dry forest habitat extent. Large crown-fires that occur in Douglas fir or denser ponderosa pine forest may help bring areas back to the initial stand establishment phase (Bollenbacher et al. 2013), potentially returning it to open-grown forest preferred by these bird species. For example, burned ponderosa pine forest created by stand-replacing fires may be important source habitat for Lewis's woodpecker (Saab et al. 2004). Depending on pre-fire forest structure, burn severity and size, geographic area, and post-fire age, wildfire may also benefit dry forest birds by reducing predator abundance (Abele et al. 2004), enhancing the quality of breeding habitats (Saab et al. 2004), or providing other important habitat features. For example, the abundance of several Lewis's woodpecker predators (e.g., tree squirrels, magpies, raccoons) were lower in burned forests than unburned forests (Abele et al. 2004). Additionally, burned conifer forests typically feature more snags (Abele et al. 2004), which a variety of dry forest birds use for nesting sites. Although wildfire is largely beneficial to dry forest birds, wildfires that re-burn large, crown-fire burned areas may limit forest establishment and translate to decreased dry forest habitat (Bollenbacher et al. 2013). For example, re-burned dry forest areas that feature altered forest structure or non-forested condition cannot typically support flammulated owls (Bollenbacher et al. 2013). Current dry forests feature altered forest structure and composition (e.g., increased forest density and increased occurrence of young stands compared to historic patterns of old growth ponderosa pine), which increases the risk of extensive crown fires and nutrient and microbe loss. This can limit dry forest regeneration (Jain and Graham 2005) and reduce habitat availability for dry forest birds.[74]

Life history

White-headed woodpeckers are monogamous, cavity-nesting birds that rely on large, open grown ponderosa pine for nesting cavities, and more closed canopy ponderosa pine stands for forage. They are iteroparous, and typically produce one clutch per year beginning at the end of May – each clutch has 4 to 5 eggs. Typically 1 to 3 young make it to fledgling status, and associate with parents through autumn. White-headed woodpeckers are also non-migratory, and occupy the same home range and breeding site year after year, though they may wander outside of their home range to pursue food (Garret et al. 1996 *in* Mellen-McLean et al. 2013).

Flammulated owls migrate to the neotropics during the winter, typically return to Idaho in early May, begin egg laying in mid-June, hatch in early July, fledge young by the end of July, and migrate south from August through October (McCallum 1994 *in* Waterbury et al. 2009). Flammulated owls are obligate cavity nesters, and commonly nest in open, old growth ponderosa pine, Douglas fir, and/or co-dominant stands, and in old woodpecker holes in large trees and snags (Idaho Department of Fish and Game (IDFG) 2005a). Breeding pairs are genetically monogamous, and male owls share active parenting duties with females (Arsenault et al. 2002).

Lewis's woodpeckers are monogamous migrants, forming long-term or multiple season mate pairs (Bock 1970 *in* Abele et al. 2004). The majority of the Lewis's woodpecker population migrates to the southern U.S. during the winter (Bock 1970, Hadow 1973 *in* Abele et al. 2004) as food sources decline in the NPCW region. Migration occurs between mid-August and late October, and returning individuals come back to breed between April and May (Bock 1970 *in* Abele et al. 2004). Lewis's woodpeckers have one clutch per year, averaging 6-7 eggs per clutch that hatch in late June (Bock 1970 *in* Abele et al. 2004). Lewis's woodpeckers typically have 1 to 2 successful fledglings (Koenig 1986 in Abele et al. 2004) that emerge from the nesting cavity 28-34 days after hatching (Dudley and Saab 2003). Female Lewis's woodpeckers breed one year after hatching, while males may delay first breeding for more than one year (Bock 1970 *in* Abele et al. 2004).

Pygmy nuthatches are cooperatively breeding passerines, and breeding pairs typically feature several male helpers that are relatives (Ghalambor and Dobbs 2006). Pygmy nuthatches typically have only one brood per year, but have high reproductive success, as more than 86% of nests successfully produce at least one offspring (Martin 1995). Nesting begins in mid-April and lasts through July; females lay one egg per day, have clutches of roughly 7 eggs, and incubate anywhere from 12-17 days (Ghalambor and Dobbs 2006). From April through September, fledglings emerge from the nest (i.e., when they are 14 to 22 days old) (Kingery and Ghalambor 2001), but remain somewhat dependent on adults 23-28 days post-fledging (Norris

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[74] Please see the Dry Forest Ecosystem section of this report for a more complete summary of climate impacts on dry forest habitats and species.

1958 *in* Ghalambor and Dobbs 2006). Pygmy nuthatches are highly social, and may form post-breeding flocks that stay within the natal territory (Ghalambor and Dobbs 2006). Females typically breed in their second year and breed annually (Norris 1958, Kingery and Ghalamber 2001), while males may wait longer as they assist their parents instead of breeding (Ghalambor and Dobbs 2006).

Future climate exposure
Important climate and climate-driven factors to consider for dry forest birds include increasing temperatures, precipitation changes that affect soil moisture and drought, and wildfire. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing (Littell et al. 2011; see Section 3), and are projected to continue to increase, with more significant increases in winter and summer (see Section 3). Warmer temperatures may impact dry forest bird species by increasing local insect forage opportunities, improving nesting success, or causing elevational shifts in response to temperature limits (McCallum 1994 *in* Waterbury et al. 2009; Hollenbeck et al. 2011).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Earlier snowmelt and decreased snowpack may increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006), and may lead to earlier soil moisture recharge, which could cause earlier desiccation of soils (Hamlet et al. 2007). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3).

Increasing temperatures, soil moisture deficits, and drought may affect dry forest bird habitat extent. For example, warmer temperatures may provide opportunities for Douglas fir expansion into higher elevational areas where they are currently temperature-limited (K Hazelbaker, pers. comm., 2014). Similarly, reduced soil moisture may result in the forest edge moving upslope, particularly on southerly aspects where moisture deficits are greater, or expansion of dry forest habitat into mixed mesic zones that may be limited by moisture availability (Bollenbacher et al. 2013; K. Hazelbaker, pers. comm., 2014). Ponderosa pine, a heat- and drought-adapted conifer species in the NPCW region, may actually expand its range into Douglas fir and/or mixed mesic habitat if soil moisture deficits increase (Scott et al. 2013; K. Hazelbaker pers. comm., 2014). Concomitantly, old growth stands may experience higher mortality due to altered forest structure (Mehl and Haufler 2001), which could reduce dry forest bird nesting habitat. Expanded dry forest extent could benefit dry forest birds in the long-term by increasing habitat and food availability, but declines in old growth ponderosa pine forests would likely reduce nesting and foraging opportunities in the short-term.

Altered wildfire regimes also have the potential to affect dry forest bird habitats. The dry forest ecosystem historically featured frequent, low-severity fires and only occasional stand-replacement fires (Scott et al. 2013). Wildfire is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013). Dry forests are adapted to fire in open forest conditions, but denser forests resulting from fire suppression practices and uncharacteristic fire behavior could limit dry forest regeneration by reducing seed sources, increasing crown fire frequency, and reducing nutrient availability (Jain and Graham 2005; Bollenbacher et al. 2013). Aside from impacting habitat extent, reduced seed source could also affect foraging success of dry forest birds. For example, the white-headed woodpecker feeds primarily on ponderosa pine seeds from late summer through winter (Blair and Servheen 1995 *in* Mellen-McLean et al. 2013), and more intense fires could lead to food shortages for this species (Mellen-McLean et al. 2013).

Sensitivity and current exposure to non-climate stressors

Workshop participants identified several non-climate stressors that affect the sensitivity of dry forest bird species, including fire suppression practices, timber harvest, and livestock grazing.[75] These non-climate stressors can alter or degrade dry forest structure, indirectly affect dry forest birds. Fire suppression activities prevent normal wildfire disturbance that maintains mature, open, and moderate canopy cover dry forest structure. As a result of fire suppression practices, 85-98% of old growth, low density ponderosa pine stands have been lost and replaced with high density stands of small to medium sized grand fir, Douglas fir, and ponderosa pine (Mehl and Haulfer 2001). Altered vegetation structure (e.g., increased true fir understory) and increased fuel loads resulting from fire suppression activities increase the chance of more frequent and/or intense stand-replacing fires, which can reduce seed source and dry forest

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[75] The collective degree these stressors increase sensitivity of dry forest bird species was considered High. Participant confidence associated with this evaluation was High. Current exposure to these non-climate stressors was evaluated as High (Confidence: High).

extent, impacting habitat and food availability for dry forest birds (Mellen-McLean et al. 2013). In addition, increased forest stand density (e.g., through fire suppression activities) can reduce dry forest bird occupancy, result in unsuitable breeding habitat (Wisdom et al. 2000), or reduce shrub and grass understories causing declines in associated insect prey (Morgan 1994). For example, the Lewis's woodpecker requires low canopy cover in order to adequately maneuver and catch aerial insects (Abele et al. 2004), and is found in lower abundance in dense forest habitat. Reduced fire frequency can also lead to increased predator abundance. For example, increased downed wood and shrub cover in the absence of fire allows for increased small mammal occupancy and predation of white-headed woodpecker nests (Hollenbeck et al. 2011).

Timber harvest in dry forests, particularly harvest activities aimed at removing old growth ponderosa pine or snags, have led to widespread loss of dry forest bird nesting and foraging habitat (Sallabanks et al. 2001 *in* IDFG 2005c). However, depending on the type of timber harvest, habitat quality may be positively or negatively affected. For example, the highest densities of flammulated owls were found on sites with no or limited selective cutting (Moore and Frederick 1991), and populations have been seen to decline following timber harvest (Granzreb and Ohmart 1978, Howie and Ritcey 1987 *in* Waterbury et al. 2009). Similarly, logged forests hold significantly fewer pygmy nuthatches than unlogged forests (Sydeman et al. 1988 *in* IDFG 2005c). Timber harvest practices such as clear-cutting can lead to habitat loss and fragmentation (Mellen-McLean et al. 2013), and may have a non-linear effect on dry forest bird populations. For example, flammulated owls are typically found in aggregated population clusters (Arsenault et al. 2002) rather than random distribution. Large-scale timber harvest in a populated tree cluster could eliminate a large portion of the local flammulated owl population and reduce dispersal ability (Moore and Frederick 1991). Conversely, in northern Idaho, Lewis's woodpeckers will nest in clear-cuts (depending on other habitat features) (Abele et al. 2004). Furthermore, post-fire salvage logging can enhance nesting habitat for the Lewis's woodpecker if clumped distributions of large diameter snags are maintained (Saab et al. 2002). The loss of large snags could negatively impact all dry forest bird species, as snags are their primary nesting sites. For example, Lewis's woodpecker uses snags as both a nesting site and for winter food caches (IDFG 2005b).

Livestock grazing (intensity, duration, and timing) can also alter dry forest bird habitat, although subsequent impacts on species may be variable. For example, grazing in ponderosa pine forests of the Southwest U.S. has been documented to reduce herbaceous and shrub understory volume, change understory plant composition, and decrease the overall number of plant species present in dry forest zones, which may affect local insect abundance and foraging success of dry forest birds (Finch et al. 1997). Livestock grazing in the Southwest has also been documented to increase tree density by reducing competition between pine seedlings and other understory vegetation (Doescher et al. 1987, Karl and Doescher 1993, Covington and Moore 1994 *in* Finch et al. 1997), which could affect dry forest bird breeding habitat and/or prey abundance. Conversely, short-term (<5 years) grazing in riparian habitats may have little impact on cavity-nesting species, including Lewis's woodpeckers (Saab et al. 1995). The impact of grazing on dry forest birds in the NPCW region is currently unknown.

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of NPCW dry forest bird species was considered moderate-high by workshop participants and reviewers,[76] as most of the associated species have small populations with patchy distribution across the NPCW landscape and limited reproductive events, but have a high ability to disperse.

The white-headed woodpecker is a Sensitive species in the USFS Northern Region, a USFWS Species of Special Concern, and an S2 Imperiled species in Idaho. Idaho's population is estimated at roughly 320 individuals, although overall population size may be declining (Sauer et al. 2014a). There are locally distributed populations in western Idaho and a vagrant population in the Idaho Panhandle (IDFG 2005c; Blair and Servheen 1995 *in* Mellen-McLean et al. 2013). Although white-headed woodpeckers can be found from British Columbia (B.C.) to California, densities are higher in the southern range and California features a different subspecies than the northern populations (Mellen-McLean et al. 2013). White-headed woodpeckers are non-migratory, and rarely leave their home range (Mellen-McLean et al. 2013). They prefer old growth ponderosa pine, but will feed and nest in other mature, open-canopied tree species (e.g., Douglas fir) when necessary (Frederick and Moore 1991). White-headed woodpeckers have low reproductive potential, reproducing only once per year and typically only having 1-3 young (Garrett et al. 1996 *in* Mellen-McLean et al. 2013).

The flammulated owl is also a Sensitive species in the USFS Northern Region and a Species of Special Concern with S3B Vulnerable Breeding/Low Breeding populations in Idaho (IDFG 2005a). The flammulated owl can be found from B.C. to southern Mexico, and is a neotropical migrant in the winter, occupying winter habitats from central Mexico to Guatemala and El Salvador (Moore and Frederick 1991; IDFG 2005a). Flammulated owls are widely distributed in Idaho but decline in abundance from west to east and at elevations outside of 1915 to 2800 m (Waterbury et al. 2009; McCallum 1994 *in* Waterbury et al. 2009). These owls typically occur in clustered, localized populations with large distances between each cluster (Groves et al. 1997; McCallum 1994 *in* Waterbury et al. 2009). Flammulated owls prefer low density, mature ponderosa pine stands with moderate canopy cover for nesting, but will occupy pure Douglas fir stands and forest types that feature similar stand structure to ponderosa pine if necessary (Moore and Frederick 1991; Waterbury et al. 2009). In addition, they require grassland edges for foraging (Goggans 1986, Reynolds and Linkhart 1987 *in* IDFG 2005a). Flammulated owls are genetically monogamous during each breeding season and have low reproductive potential, reproducing only once per year and usually having no more than 2 fledglings (McCallum 1994 *in* Arsenault et al. 2002; IDFG 2005a).

Lewis's woodpeckers are an S3B Vulnerable Breeding species in Idaho (IDFG 2005b). They can be found from B.C. to California and throughout Idaho, though they do not breed in the southeastern portion of the state and migrate to the southern U.S. during winter (Tobalske

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[76] Confidence associated with this evaluation was Moderate.

1997 *in* IDFG 2005b). Population size of this species is hard to quantify, as they are largely nomadic and occur sporadically (Bock 1970, Tobalske 1997 *in* IDFG 2005b), but overall populations are thought to be declining both regionally and across the U.S. (Sauer et al. 2014b). Lewis's woodpeckers may be more resilient than other dry forest bird species, as they can breed and nest in a variety of open forest types, including ponderosa pine forests, riparian forests, aspen groves, and oak woodlands (Abele et al. 2004; Tobalske 1997 *in* IDFG 2005b). Further, they are more nomadic in general, with presence and migration paths being controlled more by food source (e.g., insects and arthropods) than forest type (Abele et al. 2004).

The pygmy nuthatch population is declining, and is considered an S1 Critically Imperiled species in Idaho and classified as a Sensitive species in the USFS Northern Region (IDFG 2005c). It can be found from B.C. to northern Mexico (Kingery and Ghalambor 2001) and is non-migratory, usually staying within its territorial range (Ghalambor and Dobbs 2006). There are approximately 5300 resident pygmy nuthatches in Idaho (IDFG 2005c), but they are patchily distributed and have their highest density in the northern portions of the state (Groves et al. 1997; Kingery and Ghalambor 2001). Although nation-wide pygmy nuthatch population trends are relatively stable (Sauer et al. 2014c), Idaho pygmy nuthatch populations declined about 48.3% per year from 1980-2004 (IDFG 2005c), likely due to altered habitat structure and loss of open, old growth ponderosa pine forests through fire suppression, logging, and grazing (Sallabanks et al. 2001 *in* IDFG 2005c). However, recent trends in the Northern Rockies and Montana indicate that pygmy nuthatch populations may be increasing (Sauer et al. 2014c). Pygmy nuthatches roost and nest in snags or dead portions of live trees (IDFG 2005c) and typically associate with old growth ponderosa pine, although they will inhabit other dry forest types if necessary (Kingery and Ghalambor 2001). The pygmy nuthatch is limited to southern slopes at elevations between 607 to 1067 m (Burleigh 1972 *in* IDFG 2005c).

Management potential

Dry forest birds were judged by workshop participants to be moderately valued species by the public. Workshop participants and reviewers identified timber harvest - specifically removal of old growth ponderosa pine - as a potential use conflict for dry forest birds, and recommended that selective harvest (thinning) in combination with surface fuels treatments be used to protect dry forest bird habitat both now and in the future. Peer-reviewed literature corroborates that the protection of cone bearing, old growth ponderosa pine and associated old growth forest features (e.g., snags and open forest structure) is particularly important for dry forest bird species (IDFG 2005a; IDFG 2005b; Mellen-McLean et al. 2013). This management strategy is further outlined below, but please note that it represents only a general, preliminary idea of how to manage use conflicts for dry forest bird species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Proposed Management Strategy[77]:

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[77] This proposed management strategy was identified by workshop participants and reviewers, and supported by the peer-review literature. Workshop-generated management strategies were developed by participants at the Nez Perce-Clearwater Vulnerability Assessment Workshop held in September 2013. Reviewer-generated strategies

- Implement selective harvest/thinning in combination with surface fuels treatments in dry forest habitat, particularly in dense stands. Avoid removal of cone bearing, old growth ponderosa pine and old growth forest features (e.g., snags).
    - Potential benefits: Maintain and/or enhance dry forest bird habitat, create open forest structure, and reduce fire vulnerability.
    - Potential challenges: Adminsistrative or social barriers.

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were gathered from regional experts in the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014. Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

Fisher[78]

Executive Summary

The relative vulnerability of the fisher in this assessment is considered moderate, due to moderate-high sensitivity to climate and climate-driven changes, moderate sensitivity to non-climate stressors, and moderate-high adaptive capacity. Fishers are sensitive to climate and climate-driven changes such as:

- increased temperatures,
- changes in precipitation that result in reduced soil moisture,
- snowpack depth, and
- altered wildfire regimes.

Fishers exhibit some sensitivity to temperature extremes, relying on canopy cover and riparian areas to ameliorate high temperatures. Changes in precipitation can affect soil moisture and vegetation composition, potentially altering habitat suitability for fishers. Changes in snowpack may influence fisher mobility and/or cause shifts in competition dynamics. Altered fire regimes that result in lethal crown fires or stand-replacing fires may eliminate important fisher habitat characteristics such as snags, mature stands, ground cover, and canopy cover.

Fishers are also sensitive to non-climate stressors including:

- timber harvest,
- trapping, and
- disease.

These non-climate stressors can also reduce fisher population numbers by exacerbating climate-driven changes and reducing fisher habitat integrity. For example, timber harvest can reduce the structural diversity of fisher habitat, limiting denning, resting, and foraging opportunities for fishers. The fisher exhibits moderate-high adaptive capacity due to its isolated and small regional population, moderate ability to disperse, and moderate genetic, behavioral, and life history diversities.

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[78] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Fishers (*Martes pennanti*) may exhibit some direct sensitivity to increasing temperatures and changing snowpack characteristics, but the fisher's sensitivity to climate will largely be driven by climate-related changes, such as changes in temperature, precipitation, and wildfire, that affect important habitat elements. Overall, the sensitivity of fisher to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[79]

Table 16. Potential fisher response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated fisher response
Warmer temperatures	• Reduced foraging activity and increased dependency on specific habitat for thermoregulation
Precipitation changes (e.g., leading to reduced soil moisture)	• Habitat alteration that could affect availability of prey, den, and rest sites
Decreased snowpack	• Increased juvenile dispersal and/or increased mobility • Increased competitive advantage over marten
Wildfire	• Historic fire regimes: Increased habitat complexity • More frequent stand-replacing fires: Increased directly mortality, elimination of important habitat characteristics

Fishers exhibit some sensitivity to temperature extremes, avoiding extreme daily high temperatures by seeking dense canopies, canyon bottoms, riparian areas, and shaded structures, and foraging during cooler periods of the day (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). Fisher distributions appear to be influenced by snowpack; deep snow limits fisher movement, particularly juvenile dispersal (Olsen et al. 2014), and mitigates competitive interactions between fisher and marten (*Martes americana*) (Krohn et al. 1997). Reductions in snowpack could increase juvenile dispersal success during winter (Olsen et al. 2014) or alter competitive relationships between fisher and marten. For example, decreased snowpack or changes to snowpack characteristics (e.g., freeze/thaw/compaction) and may provide fishers a competitive advantage (Krohn et al. 1997).

In the western U.S. , fishers are considered a habitat specialist. In the Nez Perce-Clearwater (NPCW) region, fishers are found mainly in mesic, canopied forests and riparian corridors with abundant vertical and horizontal structure at both the stand and landscape scale (Buskirk and Powell 1994; Lofroth et al. 2011; Schwartz et al. 2013), including large trees, a variety of tree

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[79] Confidence associated with this evaluation was Moderate-High.

class sizes, snags, and downed logs (Schwartz et al. 2013). Resting and denning structures (e.g., large, decaying trees) are thought to be the most important habitat elements required for maintenance of fisher populations (Lofroth et al. 2011; Weir et al. 2012). Temperature and precipitation changes that impact the extent and vegetative composition of mixed mesic coniferous sites could affect the availability of fisher prey, den, and rest sites (Schwartz et al. 2013). For example, precipitation changes that result in reduced soil moisture or drought may reduce canopy cover, kill larger trees, or alter vegetation composition (e.g., induce shifts from western red cedar, *Thuja plicata*, to ponderosa pine, *Pinus ponderosa*), removing habitat value within the NPCW region (Schwartz et al. 2013) and potentially affecting prey dynamics.

Fishers are also sensitive to disturbance regimes such as wildfires, windthrow, and insect and disease outbreaks that influence habitat extent and structural complexity. Fires and windthrow events typically increase horizontal and vertical structure within fisher habitat, providing downed logs, snags, and damaged live trees for fisher occupancy (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). However, large-scale or uncharacteristically severe wildfire may increase direct fisher mortality or degrade habitat quality and connectivity, further isolating small fisher populations by (USFWS 2011). For example, more intense fires may eliminate important fisher habitat characteristics such as snags, mature stands, ground cover, and canopy cover (Powell and Zielinski 1994), impacting fisher abundance and distribution in the long term (Safford 2006). Insect and disease outbreaks may be both beneficial and detrimental to fisher depending on the outbreak scale. Bark beetles can contribute to conifer mortality, especially during times of moisture stress, and facilitate shifts in species composition and structure (Fettig et al. 2007) that can reduce available habitat for fishers. Conversely, fishers can use perches and cavities created by diseases and woodpeckers (Purcell et al. 2012; Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). However, major mortality events resulting from insect and disease outbreaks may reduce fisher habitat.

Life history

The fisher, a member of the Mustelidae family, is highly dependent on mature upland and lowland mixed conifer forests with abundant physical ground structure (e.g., large trees, logs, and snags), high canopy cover, and small mammal prey. Fishers in the NPCW region shift cover type seasonally, using mature forests in summer and younger forests in winter, although both cover types typically feature high snag and log density, large trees, and high cover (Lofroth et al. 2011). Habitat shifts are likely related to prey use and availability (Lofroth et al. 2011). Fishers are considered prey generalists (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti) and opportunistic predators, preying on a variety of small animals such as snowshoe hares, voles, red squirrels, mice, and small birds (Jones 1991; Powell 1993).

The fisher is a habitat specialist, preferring mixed mesic hardwood forest stands along riparian corridors containing western red cedar, subalpine fir, Pacific yew, and Englemann spruce, and avoiding ponderosa pine and lodgepole pine stands or other xeric areas (Lofroth et al. 2011; Schwartz et al. 2013). Fishers are believed to use habitat at four different levels: (1) resting site,

(2) stand, (3) home range, and (4) landscape (Zielinksi 2013). Fishers use deformed or decaying large mature trees, snags, and downed logs as resting and denning sites (Zielinski 2013), and require canopy cover for hunting and predator avoidance (Lofroth et al. 2011). Fishers are solitary and territorial, and have relatively large home ranges in the NPCW region (average of 82.6 km^2 for males, 40.8 km^2 for females), likely due to lower prey availability (Lofroth et al. 2011). Female fishers require large trees at both the stand and landscape level, and a diversity of tree sizes is also favorable, likely because they provide cover to aid hunting and predator escape (Schwartz et al. 2013). Fishers do not generally occur in open areas (<40% crown cover) or areas that have recently been logged (Lofroth et al. 2011), especially logged areas that leave no large trees behind (Schwartz et al. 2013).

Fishers can live up to 10 years (Powell et al. 2003; Arthur et al. 1992 *in* USFWS 2011). Although fishers reach sexual maturity after 1 year, they typically begin breeding when they are 2 years old (Powell et al. 2003) and have long reproductive periods, with young being born almost 1 year after copulation (Powell et al. 2003; Wright and Coulter 1967, Frost et al. 1997 *in* USFWS 2011). Fishers are iteroparous, with one breeding event per year usually in late February through April (Jones 1991; Weckwerth and Wright 1968, Roy 1991 in USFWS 2011). Fisher litters typically include 2-3 kits, and reproductive rates vary depending on prey availability (Jones 1991; Powell and Zielinski 1994; Powell et al. 2003; Weckwerth and Wright 1968, Roy 1991 *in* USFWS 2011).

Future climate exposure

Climate and climate-driven factors important to consider for fishers include increasing temperatures, changes in precipitation (including snowpack and soil moisture), and altered wildfire regimes. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing, and are projected to continue to increase, with more significant increases in winter and summer (Littell et al. 2011; see Section 3). Warmer temperatures could reduce foraging times and increase dependency on specific habitat features (e.g., large canopy cover and riparian areas) to ameliorate thermal extremes.

Precipitation changes (e.g., snowpack declines, earlier snowmelt) may affect fisher ecological relationships, habitat extent, and vulnerability. Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Decreased snowpack could allow fisher to utilize new areas, aid juvenile fisher dispersal during winter, and/or provide fisher a competitive advantage for prey (Krohn et al. 1997; Olsen et al. 2014).

Earlier snowmelt and decreased snowpack may increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006), and may lead to earlier soil moisture recharge, which could lead to earlier desiccation of soils (Hamlet et al. 2007). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Soil moisture is also projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3). These climate-driven changes could affect the vulnerability of fisher habitat by altering water availability, limiting plant growth or regeneration success, or shifting vegetation composition. For example, reduced soil moisture could allow ponderosa pine encroachment onto drier mixed mesic sites (K. Hazelbaker, pers. comm., 2014), potentially reducing fisher habitat, as fishers typically avoid dry sites and sites colonized by ponderosa or lodgepole pine (Schwartz et al. 2013). Fisher distributions in the northern Rocky Mountains are projected to shift to the north and east by 2090 in response to climatic shifts (e.g., warmer and wetter winters) and changing habitat suitability. The largest habitat losses are projected to occur near the center of the current fisher range, in northern Idaho south of Moscow (Olsen et al. 2014). North-facing slopes and riparian corridors may represent viable fisher refugia areas in the NPCW region, and more northerly and easterly sites in Montana (e.g., Glacier National Park and south of Kalispell, MT) could transition into suitable fisher habitat in the future (Olsen et al. 2014).

Historic fire regimes have played an important role in creating habitat structure for fishers. However, wildfire is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Increased fire intensity and frequency may reduce the structural diversity and connectivity of mature, mixed coniferous forests (Morgan et al. 2008; Bollenbacher et al. 2013), potentially degrading fisher habitats and resulting in further isolation of small NPCW fisher populations (USFWS 2011). For example, increased incidence of stand-replacing fires can create open, patchy, and uninhabitable forest swaths; if these unforested areas are greater than 1 km in length it is unlikely that fishers will travel across them to reach forest cover (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). Further, stand-replacing fires can cause fisher habitat to be unusable for many decades (Safford 2006), and mature seral forest characteristics can take hundreds of years to develop (USFWS 2011). Refugia areas from fire may include accessible habitat types with lower fire frequency, suitable structural diversity, and adequate prey abundance.

Sensitivity and current exposure to non-climate stressors

Workshop participants identified timber harvest as a non-climate stressor that affects the sensitivity of fisher.[80] Scientific literature also identified trapping and disease as potential non-climate stressors. Timber harvest activities, if not planned to accommodate fisher needs, have the potential to degrade mature seral forest characteristics and impact fisher fitness and foraging potential. For example, large-scale clear-cuts or severely reduced canopy cover can create open forest areas that fishers are unlikely to occupy or pass through (Lofroth et al. 2011; Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). However, climate- and species-informed timber harvest practices can be used to protect and/or create critical fisher habitat, and recent USFS management strategies are based on meeting fisher needs (Nez Perce and Clearwater National Forests Habitat Type Groups 2013; B. Bollenbacher, pers. comm., 2014). For example, maintaining large trees and snags at both stand and landscape scales for fisher resting and denning sites, as well as maintaining a diversity of tree sizes and high relative canopy cover will help allow for fisher thermal regulation, hunting cover, and predator escape (Schwartz et al. 2013). In addition, silviculture treatments and harvest can be used to mimic historic disturbances and create a landscape mosaic of young to late-seral trees more suitable for fisher (USFWS 2011; B. Bollenbacher, pers. comm., 2014), and may also reduce the risk of catastrophic fire.

Although not a targeted species, fishers are vulnerable to incidental trapping in Idaho, as they co-occur with other fur-bearing species targeted for trapping such as marten and bobcats (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti). With low reproductive rates and a small population size, reductions in the NPCW fisher population due to incidental trapping could negatively impact fisher population growth rates (Powell and Zielinski 1994).

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[80] The collective degree this stressor increases sensitivity of fisher was considered Moderate. Participant confidence associated with this evaluation was Moderate. Current degree of exposure to this stressor was evaluated as Moderate (Confidence: Moderate).

Fishers are also vulnerable to a variety of diseases and parasites, although there are no indicated correlations between fisher infection rates, habitat management techniques, and climate change impacts in the NPCW region (USFWS 2011). Four main pathogens – canine distemper virus, toxoplasmosis, parvoviruses, and rabies – can cause local fisher mortalities (Gabriel et al. 2012) and are carried by sympatric host species in the Northern Rockies (Zielinski 2013). Fishers in California are also vulnerable to parasitic infection, including intestinal invertebrates and bacterial, protozoan, and arthropod disease agents (Brown et al. 2008); however, there is little known about disease incidence and impacts on fishers in the NPCW region (USFWS 2011).

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of fisher was considered moderate-high by workshop participants and reviewers.[81] This evaluation was due to its isolated and small regional population, moderate ability to disperse, and moderate genetic, behavioral, and life history diversities. Fishers are considered a transcontinental species found throughout northern North America, but the Northern Rocky Mountains (NRM) fisher population, found in northern Montana and Idaho, is listed as a Distinct Population Segment by the USFWS due to its genetic differentiation and physical separation from other fisher populations (USFWS 2011). The NRM fisher population is currently proposed for listing as a Threatened or Endangered species (Fisher Petition 2013), although listing has not been found to be warranted in the recent past (USFWS 2010; USFWS 2011).

Fishers have moderate dispersal ability, with average dispersal distances of 50 to 75 km, but they are capable of migrating distances of up to 100 km annually (Powell and Zielinksi 1994; Lewis and Happe 2009). Fisher dispersal ability depends on the suitability of and distance to available habitat and the presence of hospitable migration corridors in between (Buskirk and Powell 1994). For example, areas lacking large trees and canopy cover will hinder fisher migration. Although total suitable fisher habitat area may increase over time in the northern Rocky Mountains, these habitat areas may not be contiguous (Olsen et al. 2014). Currently within the NPCW region, fisher habitat connectivity is considered high due to large amounts of intact habitat and the remote character of the area, but given future climate projections and habitat shifts, large portions of contiguous habitat in the NPCW region may be lost by 2090, potentially resulting in habitat islands (Olsen et al. 2014).

Barriers to fisher dispersal include habitat fragmentation caused by both natural and human disturbances (USFWS 2011; Olsen et al. 2014). Human barriers to migration include timber harvest areas, clear-cuts, energy production and mining activities, roadways and development, and land use conversions. Natural barriers (e.g., mountain ranges and semi-arid montane valleys with little cover) also prevent fisher dispersal and, combined with human development, are predicted to be a major barrier for fisher migration in the future, possibly preventing fishers

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[81] Confidence associated with this evaluation was High.

from colonizing new areas of suitable habitat to the north and east (Olsen et al. 2014). Migration barriers, both natural and human-caused, could decrease the adaptive capacity of fishers in the NPCW region by decreasing dispersal, reducing genetic exchange, and increasing the potential for inbreeding and accumulation of deleterious genetic mutations as fishers are forced into smaller habitat islands.

Fishers in the NPCW region exhibit moderate life history and behavioral diversity. They display moderate prey and habitat flexibility, typically featuring larger home ranges to make up for less suitable habitat or prey availability and occupying different forest types in the absence of typical hardwoods (Lofroth et al. 2011). Fishers also display some behavioral diversity – they avoid foraging during high temperatures and seek out habitat zones that ameliorate thermal extremes (Climate Change Sensitivity Database, http://climatechangesensitivity.org/species/martes-pennanti).

Management potential

Fishers were judged by workshop participants to be of moderate value to the public. Workshop participants identified timber harvest as a potential use conflict for fisher, and workshop participants and reviewers recommended that current riparian harvest exclusions be continued to protect critical fisher habitat both now and in the future. In addition, workshop participants, reviewers, and peer-reviewed literature suggest that selective harvest can be used in sites adjacent to riparian corridors to preserve fisher denning, resting, and foraging habitat characteristics (e.g., large trees, structural diversity, snags, downed logs, and high canopy cover) (USFWS 2011; Schwartz et al. 2013). This management strategy is further outlined below, but please note that it represents only a general, preliminary idea of how to manage use conflicts for this species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Proposed Management Strategy[82]:

- Continue to exclude timber harvest from riparian areas and use selective timber harvest/thinning in adjacent areas to manage for fisher-preferred mature forest characteristics (e.g., large trees, structural diversity, snags, downed logs, and high canopy cover).
    - Potential benefits: Maintain fisher habitat complexity, integrity, and connectivity.

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[82] This proposed management strategy was identified by workshop participants and reviewers, and supported by the peer-review literature. Workshop-generated management strategies were developed by participants at the Nez Perce-Clearwater Vulnerability Assessment Workshop held in September 2013. Reviewer-generated strategies were gathered from regional experts at the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014. Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

Mountain Goat[83]

Executive Summary

The relative vulnerability of the mountain goat in this assessment is considered moderate, due to moderate sensitivity to climate-driven changes and non-climate stressors, and moderate adaptive capacity. Mountain goats are sensitive to climate and climate-driven changes such as:

- increased temperatures, and
- decreased snowpack depth and earlier snowmelt.

Warming summer temperatures could increase summer heat stress, while decreased snowpack depth and earlier snowmelt could increase forage opportunities and kid survival.

Mountain goats are also sensitive to non-climate stressors including:

- hunting and
- anthropogenic disturbances (e.g., snowmobiling, helicopters, roads).

These non-climate stressors may cause additive population declines or abandonment of current ranges, which could have long-term consequences for mountain goat populations when combined with impacts from climate and climate-driven changes. The adaptive capacity of mountain goats in the Nez Perce-Clearwater (NPCW) region is considered moderate, due their small and isolated populations, moderately high dispersal ability, and moderate behavioral diversity.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

The mountain goat, *Oreamnos americanus missoulae*, occupies subalpine and alpine habitats with rocky, steep terrain, and exhibits direct and indirect sensitivities to changes in temperature, snowpack depth, and snowmelt timing.[84]

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[83] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

[84] Sensitivity to specific climate changes was not assessed by workshop participants. Overall sensitivity estimates presented here represent the best available information from the scientific literature.

Table 17. Potential mountain goat response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated mountain goat response
Warmer temperatures	• Increased summer heat stress may reinforce existing thermoregulatory behavior (e.g., move to cooler slopes/aspects to avoid high temperatures) • Increased forage opportunities if grasses move upwards in elevation
Decreased snowpack and earlier snowmelt	• Altered habitat: Increased conifer encroachment, which could disrupt migration or increase predation; increased grass abundance could increase forage opportunities • Reduced kid mortality in late winter and spring

Mountain goats exhibit both direct and indirect sensitivity to warming temperatures. For example, mountain goat activity is often reduced during hot midsummer days (Cote and Festa-Bianchet 2003; Chadwick and Douglas 1973 *in* Innes 2011), and they will often select north-facing slopes, snowfields, and windy sites to thermoregulate (Fox 1983; Stevens 1979, Welch 1991 *in* Innes 2011). Mountain goats shift habitat use in response to changes in food availability (e.g., due to snowpack), and will move from low elevations (spring) to high elevations (summer) to obtain the most nutritious forage (Innes 2011). Warming temperatures may provide enhanced foraging opportunities if there is an increase in the upper limits of plant growth.

Deep snow can limit mountain goat movement and use of forage habitats in winter, potentially restricting them to windswept ridges where they consume grasses or sedges (Innes 2011). Snow depth can also affect the timing of snowmelt, green-up and availability of vegetation. Late snowmelt may increase mountain goat mortality, particularly juveniles and aging individuals (Innes 2011), whereas mountain goats may benefit from early springs/snowmelt due to earlier access to high-quality forage (Hamel et al. 2009). However, early springs may allow predators (e.g., mountain lions, gray wolves, brown bears) better access to mountain goat habitats, when offspring are most vulnerable to predation (Hamel et al. 2010; Klein 1953 *in* Innes 2011).

Life history

Mountain goats feature a thick winter pelage that they grow before early December and shed between May and August (Brandborg 1955, Holroyd 1967, Smith 1988 *in* Cote and Festa-Bianchet 2003). Summer coats are short, and are grown between June and autumn (Holroyd 1967, Smith 1988 *in* Cote and Festa-Bianchet 2003). Mountain goats typically reach sexual maturity around 2 years old, begin breeding around age 3, and can breed every year up to 10 years old, after which breeding productivity declines (Cote and Festa-Bianchet 2003; Idaho Department of Fish and Game (IDFG) 2005d). Mountain goats have low reproductive potential; they have one reproductive event per year, usually giving birth to only one kid, although litter sizes may be influenced by resource availability (Cote and Festa-Bianchet 2003). Further, mountain goats do not reproduce every year; only 50-80% of females may breed during a given year (Festa-Bianchet et al. 1994). Breeding occurs from late October through early December, and kids are born from mid-May to early June (Cote and Festa-Bianchet 2003). Mountain goats feature birth synchrony; 80% of kids are born within a 2-week period, which may maximize their fitness and foraging opportunities (Cote and Festa-Bianchet 2001). Kids begin eating vegetation after one week, and eat solely vegetation after 4-5 weeks, although they depend on adults to help them forage in snow. Mountain goats typically form large nursery groups, and males leave these nursery groups when they are 2 to 4 years old (Romeo et al. 1997; Chadwick 1977 *in* Cote and Festa-Bianchet 2003).

Future climate exposure

Important climate and climate-driven factors to consider for mountain goats include warming temperatures, decreased snowpack depth and earlier snowmelt. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing (Littell et al. 2011), and are projected to continue to increase, with more significant increases in winter and summer (see Section 3). Increasing summer temperatures may heighten heat stress in mountain goats, reinforcing existing thermoregulatory behavior (e.g., grazing on northerly and easterly slopes and among melting snowfields) (Fox 1983; Stevens 1979, Welch 1991 *in* Innes 2011).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005) and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Decreased snowpack and earlier snowmelt may have varying impacts on mountain goats. For example, lower snowpack and shorter duration of snow cover may increase food availability for mountain goats during winter and spring (Martin 2000; Toweill 2004), which could improve kid survival rates (Cote and Festa-Bianchet 2003; Innes 2011). Conversely, food availability may decrease due to increased tree encroachment into subalpine and alpine habitats (Rochefort 1994; Bjork and Molau 2007; Pettorelli et al. 2007). Tree encroachment may also increase predation risk (as predators use the structure for cover) (Cote and Festa-Bianchet 2003; Festa-Bianchet and Cote 2007). Similarly, earlier timing of snowmelt may increase predator access to mountain goat habitat, potentially increasing offspring mortality (Hamel et al. 2010).

Sensitivity and current exposure to non-climate stressors

The scientific literature identified two non-climate stressors that may affect the sensitivity of mountain goats, including legal harvest through hunting and human disturbance.[85] Mountain goats can be legally hunted as big game in both Idaho and Montana, but native populations are very sensitive to overharvest. For example, recruitment and productivity declined following increased legal harvest of mountain goats in Idaho (Kuck 1977, Herbert 1978 *in* Cote and Festa-Bianchet 2003). In addition, harvesting of females may have larger population impacts than harvesting of males (Cote and Festa-Bianchet 2003). Hunting may exacerbate climate impacts on native mountain goat populations, leading to additive mortality and unsustainable population declines (Cote and Festa-Bianchet 2003).

Mountain goats are also sensitive to human disturbance activities such as snowmobiling, helicopters, and road use. Repeated disturbance events can cause displacement from habitat, kid separations, nursery group dissolution, and/or injury if goats flee (Chadwick 1983 *in* IDFG 2005d).

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of mountain goats was considered moderate by workshop participants,[86] due to their small and isolated populations, moderately high dispersal ability, and moderate behavioral diversity. Mountain goats can be found in subalpine and alpine habitats throughout western North America. Idaho and Montana feature native mountain goat populations (subspecies *O.americanus missoulae*) (Cote and Festa-Bianchet 2003; IDFG 2005d). Mountain goats are an S2 Imperiled species in Idaho (IDFG 2005d), and population sizes are currently in the mid- to high 2000s, though they fluctuate annually. Idaho population declines were recorded between 2000 and 2004, though the reason for these declines is unknown (IDFG 2005d). These goats primarily occupy isolated subalpine habitats ranging from 7,000 to 10,000 feet in elevation (Toweill 2004; IDFG 2005d), with high habitat connectivity in summer and low habitat connectivity in winter. Mountain goats typically move to lower elevations in winter and follow receding snowpack back to summer ranges. They have moderately high dispersal ability, especially during summer, where they can travel 2 to 5 km per day or more (Cote and Festa-Bianchet 2003); seasonal migration distances vary widely (8 to 30 km) depending on herd, sex, and geographic location (Innes 2011). Mountain goat movement is restricted to areas with nearby escape terrain, such as rocky slopes or cliffs, as dispersal through wooded areas can be dangerous (i.e., tree cover increases predation risk).

Mountain goats exhibit low to moderate behavioral diversity. For example, they will avoid foraging during the warmest parts of the day to avoid heat stress (Cote and Festa-Bianchet 2003), which may become more important as temperatures warm. As herbivorous generalists,

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[85] Workshop participants were unable to address this evaluation in the time allotted.

[86] Confidence associated with this evaluation was Moderate.

they will eat whatever vegetation they can find, though they prefer grasses and shrubs when available (Cote and Festa-Bianchet 2003).

Management potential
Mountain goats were judged by workshop participants to be a highly valued species by the public. Workshop participants did not know of any potential use conflicts for mountain goats in the NPCW region, but regional literature suggests that hunting could impact this species (IDFG 2005d). Peer-reviewed literature suggests preventing harvest of female mountain goats and setting conservative and adjustable herd-specific harvest limits (e.g., harvest no more than 2-3% of the population, close or reduce season following overharvest of females) to protect native NPCW mountain goat populations both now and in the future (IDFG 2005d). This management strategy is not discussed here (as it falls under the purview of IDFG), however more specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Red Alder[87]

Executive Summary

The relative vulnerability of the red alder in this assessment is considered low, due to moderate sensitivity to climate and climate-driven changes, low sensitivity to non-climate stressors, and moderate-high adaptive capacity. Red alders are sensitive to climate and climate-driven changes such as:

- decreased snowpack and earlier snowmelt,
- reduced soil moisture and drought, and
- altered wildfire regimes.

Precipitation changes that reduce soil moisture or cause drought can reduce red alder survival, germination, and growth. More frequent fire could increase the colonization opportunities for this fast growing pioneer species, but more intense or stand-replacing fires could damage alder stands. Red alder may benefit from increasing temperatures, as they are currently limited by cold temperatures, freezing, and frost events. Warming temperatures could increase red alder growth rates, reduce freeze damage, and allow them to expand to higher elevations.

Red alders are also sensitive to non-climate stressors including:

- fire suppression activities, and
- disturbance events (e.g., timber harvest and transportation corridors).

Fire suppression activities reduce natural wildfire disturbance that red alders depend on for new colonization opportunities, and can lead to conifer succession. Fire suppression activities can also exacerbate future changing fire regimes by increasing fuel density. Although red alders in the Nez Perce-Clearwater (NPCW) region are fairly isolated, their adaptive capacity is considered moderate-high due to high seed dispersal ability and high behavioral, phenotypic, and life history diversity.

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[87] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Red alder (*Alnus rubra*) is an early successional species that prefers moist, well-drained soils (Deal and Harrington 2006; Cortini et al. 2012). Red alders are highly sensitive to cold temperatures, precipitation changes leading to drought and reduced soil moisture, and wildfire. Conversely, warming temperatures, increased precipitation, and changes in snowpack depth and snowmelt timing may benefit red alder. Overall, the sensitivity of red alder to climate and climate-driven changes was evaluated as low-moderate by workshop participants.[88]

Table 18. Potential red alder response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated red alder response
Warmer temperatures	• Reduced cold mortality • Enhanced seed production • Reduced dormancy length and enhanced growth • Range expansion to higher elevations by reducing frost/freezing events
Decreased snowpack and earlier snowmelt	• Range expansion to higher elevations • Reduced top breakage • Potential range contraction, habitat conversion, and/or reduced fitness if available soil moisture declines
Precipitation changes leading to reduced soil moisture and/or drought	• Replacement by dry-adapted conifers • Limited spring soil moisture: Reduced germination, growth, and survival • Drought: Increased seedling mortality on open sites
Wildfire	• Increased frequency of low-moderate severity fires: Increased colonization opportunities • Increased fire intensity: Reduced growth and survival

Red alders are currently limited by cold temperatures, freezing, and frost events (Deal and Harrington 2006; Cortini et al. 2012), and tree mortality or reduced seed production may occur following severe winter freezes (Worthington 1957 *in* Deal and Harrington 2006). Further, red alder does not typically grow at high elevations due to snow and ice breakage (Deal and Harrington 2006). Red alder is adapted to humid climates, and warmer temperatures could increase red alder growth rates by breaking their cold dormancy earlier in the season (Deal and

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[88] Confidence associated with this evaluation was Moderate.

Harrington 2006; Spittlehouse 2008). Warmer annual temperatures and warmer minimum temperatures may also allow red alder to move higher in elevation.

Precipitation changes (more or less) may have variable impacts on red alder. For example, altered precipitation patterns that reduce soil moisture or increase drought incidence may have negative impacts on red alder. Reduced soil moisture, especially in spring, may limit red alder germination, growth and survival (Haeussler 1988 *in* Deal and Harrington 2006; Cortini et al. 2012), and could lead to replacement of red alder stands by more drought-tolerant conifers (Deal and Harrington 2006). Red alder rarely grows on droughty soils, particularly in the NPCW region (Deal and Harrington 2006), and drought and heat injury are the main mortality agents for red alder seedlings in disturbed, open sites (Haeussler 1988 *in* Deal and Harrington 2006). Increased precipitation, particularly during the growing season, may benefit red alder by increasing areas with enough soil moisture to support this species.

Low to moderate severity wildfires typically create expansion and colonization opportunities for red alder. As an early successional and shade-intolerant species, red alder is a common pioneer species following wildfire disturbance (Deal and Harrington 2006; Cortini et al. 2012). Red alders can typically survive light surface fires because they have slightly fire-resistant bark and occur in moist sites with very little flammable debris (Deal and Harrington 2006; Worthington 1957 *in* Deal and Harrington 2006), and red alder stands have actually been used as fire breaks to protect more flammable conifer stands (Deal and Harrington 2006). While increased fire frequency could enhance colonization areas for red alder, more intense fires could negatively impact red alder growth and survival in burned stands (Deal and Harrington 2006).

Life history

Red alder is a fast-growing, polycyclic, prolific pioneer species that excels at colonizing disturbed areas (Deal and Harrington 2006). Individual red alder trees reach sexual maturity between 3 and 4 years, and in pure red alder stands dominant trees reach maturity between 6 and 8 years (Stettler 1978 *in* Deal and Harrington 2006). Red alders are generally monoecious (Hitchcock et al. 1964 *in* Deal and Harrington 2006), and seeds are typically a result of outcrossing (Stettler 1978 *in* Deal and Harrington 2006). Red alders are prolific seed producers, but seed quantity and quality can vary annually and between stands (Worthington 1957, Brown 1985, McGee 1988 *in* Deal and Harrington 2006). Seed crop failure is rare, but reduced seed production may occur following severe winter freezes (Worthington 1957 *in* Deal and Harrington 2006). Seed dispersal is thought to be controlled by weather; dry weather opens catkins and releases seeds, while wet weather keeps catkins closed and/or can terminate a dispersal event by closing previously opened catkins (Deal and Harrington 2006). Seed dispersal occurs primarily by wind, but can also occur via water and animal transport (Deal and Harrington 2006), and typically starts in September and continues through February (McGee 1988 *in* Deal and Harrington 2006). Seed germination generally occurs earlier in open environments compared to forested environments (beginning in late February and March respectively), and germination success is higher in disturbed seedbeds with exposed mineral soil (Deal and Harrington 2006). Red alder enters a period of winter dormancy, which may allow it to avoid frost and freezing damage and reduce incidence of snow breakage (Deal and

Harrington 2006). Red alder also excels at fixing atmospheric nitrogen, which can make disturbed sites more hospitable for coniferous species (Deal and Harrington 2006; Cortini et al. 2012).

Future climate exposure

Important climate and climate-driven factors to consider for red alder include increasing temperature, precipitation changes (including drought and reduced soil moisture), decreased snowpack and earlier snowmelt, and wildfire. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures are also projected to increase, with more significant increases in winter and summer (see Section 3). Warming annual average temperatures and increasing minimum temperatures could reduce the length of red alder cold dormancy and increase seasonal growth rates (Deal and Harrington 2006; Spittlehouse 2008). In addition, warming temperatures may allow red alder range expansions to higher elevations by reducing incidence of frost and freezing events (Cortini et al. 2012).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008); and earlier runoff timing (Stewart et al. 2005). Over the last 60 years, there has been less snow in the mountains of the western U.S. (Barnett et al. 2008). However, from 1916-2003, models suggest that snow water equivalent (SWE; a measure of April 1st snowpack) has increased 0-0.5% per year in the NPCW region, mainly due to increased regional precipitation at high elevations (Hamlet et al. 2005). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Earlier snowmelt and decreased snowpack may also increase the length of the summer drought season (Stewart et al. 2009) and contribute to increased wildfire activity (Westerling et al. 2006), and may lead to earlier soil moisture recharge, which could lead to earlier desiccation of soils (Hamlet et al. 2007). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3).

Changes in precipitation timing and amount may have varying impacts on red alder. For example, decreased snowpack may reduce the incidence of snow breakage in higher elevation red alder stands, while earlier snowmelt may allow for increased red alder colonization at higher elevations (Deal and Harrington 2006). Reduced warm season precipitation, heightened drought incidence, and reduced soil moisture could negatively affect red alder survival, growth, and reproduction, as red alder relies on moist microclimates (Cortini et al. 2012). Further, large shifts in relative moisture could cause red alder stands to be replaced by more drought-tolerant species, such as Douglas fir (*Pseudotsuga menziesii*) or other conifers (Deal and Harrington 2006). North-facing slopes represent potential refugia areas from decreased precipitation and moisture deficits (Deal and Harrington 2006).

Wildfire, though historically present in this region, is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Red alder could benefit from increased fire incidence, as it is a pioneer species that excels at colonizing disturbed landscapes (Deal and Harrington 2006; Cortini et al. 2012). In addition, red alder can survive low to moderate surface fires due to slightly fire-resistant bark and a moist microclimate with low fuel flammability (Deal and Harrington 2006). However, more intense or stand-replacing fires resulting from drier conditions may affect growth and survivorship in red alder stands (Deal and Harrington 2006; Cortini et al. 2012).

Sensitivity and current exposure to non-climate stressors

Workshop participants identified two non-climate stressors that affect the sensitivity of red alder, including fire suppression practices and disturbance events.[89] Red alder is favored by low to moderate severity fire, as it survives these events and can successfully colonize and establish in burned areas (Deal and Harrington 2006). Fire suppression practices reduce natural wildfire disturbance and colonization opportunities for red alder, and facilitate natural succession patterns that can cause red alder to be replaced by conifers. Fire suppression practices also increase fuel density, which can lead to more frequent and intense fires. More frequent fires fueled by climate change and higher fuel loads may increase colonization opportunities for red alder, however, more intense fires could have negative impacts on burned alder stands (Deal and Harrington 2006; Cortini et al. 2012).

Disturbance events (e.g., timber harvest and creation/maintenance of transportation corridors) can create more red alder colonization and growth opportunities. Red alder is a pioneer species that can colonize disturbed mineral soils (Deal and Harrington 2006), and anthropogenic disturbance activity may contribute to increased red alder abundance throughout the NPCW

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[89] The collective degree these stressors increase sensitivity of red alder was considered Low. Participant confidence associated with this evaluation was Moderate. Current degree of exposure to these stressors was not assessed by participants.

region. Under optimal conditions, red alder can actually act as a nuisance species by outcompeting conifer seedlings (Deal and Harrington 2006) and preventing natural succession patterns following disturbance.

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of red alders was considered moderate-high by workshop participants.[90] Although red alders in the NPCW region are fairly isolated, they have high seed dispersal ability and high behavioral, life history, and phenotypic diversity. Red alders feature transcontinental and robust populations, existing below 1100 m along the coast from southeast Alaska to southern California (Deal and Harrington 2006). The NPCW region hosts an isolated disjunct population of red alders that occur in patchy distribution, typically along streams and lakeshores. Red alders have high seed dispersal ability due to high seed counts, multiple seed transport methods (e.g., wind, water, animals), and relatively long dispersal distances (e.g., seeds are typically dispersed up to 1 km away from parent stands) (Deal and Harrington 2006).

Red alders also feature high behavioral, phenotypic, and life history diversity. Red alder can produce three different types of branches and self-prune in response to light availability, and can exist across a variety of habitat types if enough soil moisture is present. Red alders also shift carbon allocation to root biomass and stop short-term growth under moisture stress (Chan 1990 *in* Deal and Harrington 2006), which may allow them to survive drought periods. Further, red alder typically enters a period of dormancy to survive cold winter temperatures. Phenotypic variation is high among individual trees, and genetic differentiation is thought to exist among different geographic populations of red alder as they feature differing growth rates and sensitivity to frost and other factors (Deal and Harrington 2006). Red alder also demonstrates two reproductive strategies: seeds (which can be outcrossed or self-pollinated) and vegetative coppice regeneration (Deal and Harrington 2006). These different reproduction strategies allow for both the establishment of new trees through seed germination and the regeneration of damaged trees.

Management potential

Red alders were judged by workshop participants to be moderately valued by the public. Workshop participants thought that there are none/few use conflicts for this species, as it is a pioneer species and generally benefits from human activities that cause disturbance across NPCW landscapes. Peer-reviewed literature suggests that disturbance that exposes bare mineral soil, including timber harvest (e.g., thinning, large-group selection and/or clear-cutting) or fire, can be used to regenerate red alder stands (Deal and Harrington 2006). This management strategy is further outlined below, but please note that more specific management strategies for this species that address climate and non-climate stressors will be developed and presented in future documents.

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[90] Confidence associated with this evaluation was Moderate.

Literature-generated Proposed Management Strategy[91]:

- Use thinning or prescribed burns to encourage regeneration of red alder stands.
    - Potential benefits: Increase red alder recruitment, maintain alder benefits to ecosystem (e.g., nitrogen fixation) (Deal and Harrington 2006).
    - Potential challenges: Adminsistrative or social barriers.

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[91] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

Spalding's Catchfly[92]

Executive Summary

The relative vulnerability of Spalding's catchfly in this assessment is considered high, due to moderate-high sensitivity to climate and climate-driven changes, moderate sensitivity to non-climate stressors, and low adaptive capacity. Spalding's catchfly is sensitive to climate and climate-driven changes including:

- precipitation changes,
- reduced soil moisture or prolonged drought, and
- wildfire.

Prolonged drought or reduced soil moisture could limit catchfly growth and reproduction by inducing dormancy and/or degrade the integrity of the fescue grasslands on which Spalding's catchfly depends. Alternatively, precipitation increases in the fall and winter could cause catchfly individuals to leave dormancy and reproduce during the following growing season. Wildfire may have mixed impacts. For example, fire in spring and fall could facilitate recruitment by reducing grass cover and litter accumulation, but summer fires could remove reproducing catchfly individuals, increase invasive weed encroachment and establishment, and/or reduce pollinator abundance.

Spalding's catchfly is also sensitive to several non-climate stressors including:

- livestock grazing (particularly overgrazing),
- invasive species,
- land use conversion, and
- herbicide drift.

These non-climate stressors can prevent catchfly reproduction, reduce recruitment success, alter grassland plant composition and habitat condition, and may exacerbate climate-driven changes. For example, invasive weeds can reduce Spalding's catchfly pollinator visitation and lead to reduced fitness and inbreeding depressions, and may also increase the risk of more frequent or intense fires. Although overgrazing or grazing at inappropriate times (e.g., when catchfly flowers) may be detrimental to this species, light grazing practices may help stimulate

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[92] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

catchfly recruitment by reducing competition and litter depth. The adaptive capacity of Spalding's catchfly is considered low, due to their small, fragmented population and low genetic diversity, although their ability to become dormant under sub-optimal conditions may buffer them in the short-term from climate impacts.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Spalding's catchfly (*Silene spaldingii*) is typically found in grasslands, shrub habitats, and open ponderosa pine stands that feature moderate precipitation, warm summers with drought periods, and deep soils that retain decent soil moisture. Spalding's catchfly requires mesic conditions, and is usually associated with areas that retain soil moisture throughout the dry summer season, including swales, northerly aspects, low-lying areas, and talus or basalt outcrops (Hill and Gray 2004). Accordingly, Spalding's catchfly is sensitive to precipitation changes, prolonged drought, and reduced soil moisture, as well as wildfire. Overall, the sensitivity of Spalding's catchfly to climate and climate-driven changes was evaluated as moderate-high by workshop participants.[93]

Table 19. Potential Spalding's catchfly response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated Spalding's catchfly response
Precipitation changes	• Facilitate conversion of grassland systems • Increased fall or winter precipitation: Leave dormancy and reproduce during following growing period
Prolonged drought and/or reduced soil moisture	• Fall or winter: Enter dormancy, potentially limiting catchfly growth and reproduction in short term (1-2 years) • Degrade grassland systems
Wildfire	• Spring or fall fire (5-10 year return intervals): Higher recruitment • Summer fire: Population decline due to direct mortality

Prolonged summer drought or drought during the growing season may degrade or compromise the integrity of the fescue grasslands on which Spalding's catchfly depends (Hill and Gray 2004; Bollenbacher et al. 2013). Precipitation changes that lead to drought or reduced soil moisture, particularly in the fall and winter, could limit catchfly growth and reproduction in the short term by inducing prolonged plant dormancy (Lesica and Crone 2007). Dormant periods typically last 1-2 years, though some individuals can remain dormant for up to 6 years (Lesica and Crone 2007). However, dormant individuals are more likely to flower in successive years than their vegetative counterparts (Lesica and Crone 2007). Wet summers are also associated with

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[93] Confidence associated with this evaluation was Moderate-High.

prolonged dormancy in Spalding's catchfly, potentially due to concomitant resource depletion following elevated seed production (Lesica and Crone 2007). Alternatively, precipitation increases in the fall and winter could cause catchfly individuals to leave dormancy and reproduce during the following growing period (Lesica and Crone 2007). Precipitation increases could also allow increased abundance of more woody and moisture-dependent species (e.g. ponderosa pine), which could reduce the geographic extent and reproductive success of Spalding's catchfly and other prairie species (Hill and Gray 2004).

Wildfire may have positive impacts on Spalding's catchfly, depending on seasonality. Spring and fall fires can stimulate seedling recruitment by temporarily reducing competition and litter accumulation (Lesica 1999; Hill and Gray 2004), creating safe recruitment sites and leading to concomitant increases in population size (Lesica 1999). Recruitment is typically higher in burned areas versus unburned areas (Lesica 1999). Although spring and fall fires benefit Spalding's catchfly, summer fires may have more unfavorable effects on catchfly population size by removing growing and flowering individuals during the late-summer fire season (Lesica 1999). The full effects of fire on dormant individuals are still unknown (Lesica 1999). Fire may also increase establishment and spread of competitive and disturbance-adapted weed species, which can outcompete Spalding's catchfly and alter ecological conditions of the grassland system (Hill and Gray 2004). In addition, fire may cause heightened mortality for the ground-nesting bumblebee, *Bombus fervidus*, which is the main pollinator for Spalding's catchfly (Lesica 1993; Lichthardt and Gray 2002).

Life history

Spalding's catchfly is an iteroparous, long-lived, herbaceous perennial geophyte (Hill and Gray 2004) that has four life history stages: seedling (rosette), dormant, vegetative (non-reproductive), and flowering (reproductive) (Lesica and Crone 2007). After a dormant winter period, seeds germinate and plants emerge in mid- to late May (Lesica and Crone 2007). Peak blooms occur late, starting in mid-July, with seed setting occurring through August and occasionally into October (Hill and Gray 2004; Lesica and Crone 2007). This blooming period allows Spalding's catchfly to avoid competition with other grassland species, which typically enter a period of aestivation (summer dormancy) during these months; however, this blooming period also coincides with fire season, with more frequent and/or intense fire potentially killing reproducing catchfly individuals (Lesica 1999; Hill and Weddell 2003 *in* Hill and Gray 2004). Spalding's catchfly reproduces only by seed (Lesica 1999), and experiences significantly higher fitness through pollinator outcrossing compared to self-pollination (Lesica 1993). Bumblebees serve as the main pollinator for Spalding's catchfly (Lesica and Heidel 1996 *in* USFWS 2005). Spalding's catchfly is partially self-compatible between different flowers of the same plant (USFWS 2005), but exclusion of pollinators may result in a greater than 99% reduction in fitness due to inbreeding depression (Lesica 1993). Catchfly dormancy, which follows reproduction and lasts typically 1-2 years, is believed to be both an important adaptation to semi-arid climates like the grasslands of the Nez Perce-Clearwater (NPCW) region (Lesica 1997; Lesica and Crone 2007) and an important life history trait for this species, as dormant individuals have a higher probability of flowering the following season than vegetative individuals (Lesica and Crone 2007). Spalding's catchfly has a long lifespan under natural circumstances, and populations can exist many years without recruitment due to prolonged dormancy (Hill and Gray 2004), providing a potential buffer from short-term climate-driven events.

Future climate exposure

Important climate and climate-driven factors to consider for Spalding's catchfly include precipitation changes, reduced soil moisture, drought, and wildfire. On average, precipitation across the Nez Perce-Clearwater (NPCW) region has not been notably different during recent years (i.e. last 30 years) compared to 1901-1980, with increased precipitation in spring and summer, and decreased precipitation in winter (see Section 3). Future precipitation trends are difficult to predict, though many models indicate no overall annual precipitation changes for the NPCW region through 2040, with precipitation increases occurring in spring and winter and decreases occurring in summer (Hamlet et al. 2005; Morgan et al. 2008; Littell et al 2011; see also Section 3). Precipitation increases in the fall and winter could cause catchfly individuals to leave dormancy and reproduce during the following growing period (Lesica and Crone 2007). Precipitation increases could also facilitate conversion of Spalding's catchfly habitat and reduce catchfly reproductive success by allowing more woody (e.g. ponderosa pine) and moisture-dependent species to invade grassland systems (Hill and Gray 2004).

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including more precipitation falling as rain compared to snow (Knowles et al. 2006); reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005); and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Earlier snowmelt and decreased snowpack may increase the length of the summer drought season (Stewart et al. 2009) and may lead to earlier soil moisture recharge, which could lead to earlier desiccation of soils (Hamlet et al. 2007). July 1 soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see also Section 3), and warming temperatures and an increasing number of high temperature events, both projected for the region (e.g., see Section 3; Diffenbaugh et al. 2005), may exacerbate soil moisture deficits (Bollenbacher et al. 2013). Although specific information on future drought does not currently exist for the NPCW region, in general, drought frequency and severity is projected to increase throughout the western U.S. (IPCC 2007a). Drought or reduced soil moisture could degrade the fescue grasslands on which Spalding's catchfly depends (Hill and Gray 2004; Bollenbacher 2013). Drought and reduced soil moisture occurring in the fall or winter could also induce catchfly dormancy, which may be energetically favorable, as low soil moisture can cause early senescence of vegetative individuals during the growing season (Lesica and Crone 2007). Refugia areas for Spalding's catchfly are uncertain, as they already occur in higher elevation sites within the NPCW region.

Wildfire has played a historic role in shaping NPCW grassland communities, and is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier summers are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004). Fire intensity (Brown et al. 2004a) and area burned (Spracklen et al. 2009; Littell et al. 2010) are also projected to increase. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Spring and fall fires with roughly 5 to 10 year fire return intervals are thought to allow optimum growth rates and enhance seedling recruitment for Spalding's catchfly. Shorter fire return intervals fail to allow for fuel accumulation and would likely provide little benefit to catchfly recruitment rates (Lesica 1999). In addition, more frequent summer fires could kill growing and reproducing catchfly individuals (Lesica 1999) and/or allow increased establishment and spread of competitive and disturbance-adapted weed species (Hill and Weddell 2003 *in* Hill and Gray 2004). These species can outcompete Spalding's catchfly and irreversibly alter ecological conditions of the grassland system (Hill and Gray 2004). Further, increased fire frequency and/or severity could also negatively affect ground-dwelling bumblebees, potentially reducing pollination rates and fitness within Spalding's catchfly populations (Lesica 1993; Lichthardt and Gray 2002).

Sensitivity and current exposure to non-climate stressors

Workshop participants identified two non-climate stressors that affect the sensitivity of Spalding's catchfly, including livestock grazing and invasive species.[94] Scientific literature also identified land use changes and herbicide drift as potential non-climate stressors. Livestock grazing timing and intensity can impact Spalding's catchfly in variable ways. For example, grazing can directly prevent catchfly reproduction by removing the flower and seed capsules on the upper part of the stem, cause mortality of vegetative individuals, and/or degrade the grassland community on which Spalding's catchfly depends by altering plant composition and facilitating weed invasion (Hill and Gray 2004). Heavy or prolonged grazing, grazing in late summer during catchfly reproduction, and/or grazing on mesic sites appear to have the most negative effects on Spalding's catchfly reproduction and survival (Hill and Gray 2004). In addition, *B. fervidus* nests very close to the ground surface, and livestock grazing may crush their nesting cavities, potentially affecting catchfly pollination rates and fitness (Lesica 1993; Hill and Gray 2004). Alternatively, light grazing may benefit Spalding's catchfly recruitment by reducing cover of dominant grasses and associated litter depth (Lesica 1997; Lesica 1999). Spalding's catchfly currently experiences high exposure to light grazing practices, especially in canyon grassland areas (Hill and Gray 2004), but low exposure to overgrazing in the NPCW region. Current light grazing practices have not resulted in any known catchfly extirpations in the NPCW region (S. Shelly, pers. comm., 2014).

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[94] The collective degree these stressors increase sensitivity of catchfly was considered Moderate. Participant confidence associated with this evaluation was Moderate-High. Current degree of exposure to these stressors was evaluated as Low (Confidence: Moderate-High).

Spalding's catchfly is also highly sensitive to invasive and non-native plants, which can cause irreversible changes in grassland habitat physical condition and disrupt normal ecological processes. Much of the canyon grasslands have suffered degradation by invasive weeds, particularly yellow starthistle (*Centaurea solstitialis*) and cheatgrass (*Bromus tectorum*) (Hill and Gray 2004). Elevated invasive cheatgrass abundance may contribute to more frequent and intense fires, which can affect Spalding's catchfly recruitment, as cheatgrass has a naturally short fire-return interval and its fine-textured leaves create a large fuel source (Ecovista et al. 2003). Direct impacts of invasive species on catchfly vigor are undetermined; a 2-year study by Menke (2003) did not find any direct differences in catchfly vigor between invaded and non-invaded sites at Garden Creek Ranch in Idaho (Menke 2003). However, studies in Washington suggest that higher invasive plant cover translates to lower catchfly vigor (Caplow 2002a *in* USFWS 2005). Flowering invasive species can reduce pollination success for Spalding's catchfly. For example, the invasive weed, St. Johnswort (*Hypericum perforatum*), flowers at the same time as Spalding's catchfly and can reduce bumblebee visitation and associated fitness of adjacent catchfly individuals (Lichthardt and Gray 2002; Lesica and Heidel 1996 *in* USFWS 2005). Invasive plants can also exacerbate climate-driven changes. For example, invasive weed presence could compound soil moisture shortages, which could induce catchfly dormancy (Hill and Gray 2004). Invasive weeds are already established in many NPCW grassland systems, for example, within the Clearwater and lower Snake River basins (Tisdale 1986 *in* Hill and Gray 2004; Ecovista et al. 2013), but catchfly-occupied grasslands within the Nez Perce National Forest are currently in good-to-excellent condition (S. Shelly, pers. comm., 2014) and represent high priority areas for weed monitoring and management.

Land use changes, such as conversion to agriculture or development areas, also affect Spalding's catchfly. Much of the historic Palouse Grasslands have been converted to cropland or pastures (Lesica 1999) due to their deep, fertile soils, while the canyon grasslands remain unconverted due to their steep and rugged topography (Hill and Gray 2004). As a result of past land use conversion, Spalding's catchfly now exists in isolated patches along the fringes of its former range (Lesica 1999). On the Palouse Grasslands, any remnant catchfly individuals and native grassland habitat are surrounded by large expanses of cultivated fields or other human development (Hill and Gray 2004), which can reduce gene flow among isolated catchfly patches as pollinator-assisted pollen transfer may be very rare at distances over one mile (USFWS 2005). Cropland conversions can also lead to erosion of fertile topsoil and loss of carbon, making grassland restoration difficult (Hill and Gray 2004). Additionally, pesticide application and livestock grazing may cause bumblebee population declines, ultimately affecting catchfly pollination opportunities and fitness (Lesica 1993). Although current exposure to land use alteration is considered low, preventing future conversions, especially conversions for recreational purposes along river canyons, can help prevent synergistic impacts that could lead to further declines in Spalding's catchfly persistence in the NPCW region.

Spalding's catchfly may also be sensitive to aerial herbicide sprays. For example, herbicide drift as a result of herbicides sprayed on crops may affect Spalding's catchfly populations in the Palouse Prairie. Further, aerial sprays aimed at reducing weeds in canyon grasslands are also problematic, because unpredictable wind currents and high temperatures can volatize herbicides and distribute them to unintended locations. Herbicide drift will likely continue, as aerial spraying is the most cost effective application method (Hill and Gray 2004); however, its impacts could become more problematic for Spalding's catchfly in the future as warming regional temperatures may increase herbicide volatilization. Herbicides can eliminate or reduce populations of Spalding's catchfly, potentially compounding the issues catchfly may face under changing climate conditions (Hill and Gray 2004). The current exposure to and effects of herbicide sprays on Spalding's catchfly populations in the NPCW region are currently unknown.

Adaptive Capacity

Species extent, status, and diversity

The adaptive capacity of Spalding's catchfly was considered low by workshop participants and reviewers,[95] due to a small, fragmented population size and low genetic diversity. Spalding's catchfly is an endemic species to the Palouse Prairie region of Washington, Oregon, and Idaho, and can be found in disjunct populations in northwest Montana and British Columbia, Canada. Spalding's catchfly is listed as an USFWS Threatened species and an S1 Critically Imperiled species in both Idaho and Montana. Current catchfly populations are found in small patches (typically less than 10 acres) and in distinct physiogeographic areas. In the NPCW region these include the Palouse Grasslands in Idaho, canyon grasslands along river systems including the Snake, Clearwater, and Salmon Rivers in Idaho, and in intermontane valleys of northwestern Montana (Hill and Gray 2004). Approximately 98% of the catchfly population is in canyon grasslands (Hill and Gray 2004). Populations are highly fragmented, often separated from other populations by several hundred feet or more (Hill and Gray 2004). These smaller habitat islands may not be large enough to support *B. fervidus* pollinator populations (Lesica 1993), which can increase rates of selfing, reduce recruitment, and lead to the decline or extirpation of small populations (Lesica 1993). Most fragmentation is a result of geologic features, past land use, and agricultural conversion (Lesica 1999). Genetic diversity of Spalding's catchfly in the NPCW region is low, as this species may carry a large load of deleterious recessive genes (Lesica 1993) and the small, remnant populations of Spalding's catchfly are vulnerable to inbreeding depressions, edge effects, and other genetic limitations that arise from small population sizes (Lesica 1993; Hill and Gray 2004). In addition, Spalding's catchfly has a low seed dispersal range, usually no more than 1 kilometer. There are roughly 3300 catchfly individuals in the NPCW National Forest (M. Hays, pers. comm., 2014).

Management potential

Spalding's catchfly was judged by workshop participants to be of low value to the public. Workshop participants identified livestock grazing as a potential use conflict in the region, and recommended that overgrazing be prevented to protect Spalding's catchfly populations now and in the future. In addition, preventing grazing on mesic sites, during catchfly reproduction periods (i.e. late summer), and limiting length of grazing time may improve recruitment and protect catchfly habitat (Hill and Gray 2004). Prescribed burning in the spring or fall, particularly in areas where natural fire has been suppressed, may also benefit Spalding's catchfly

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[95] Confidence associated with this evaluation was High.

recruitment (Lesica 1999). These management strategies are further outlined below, but please note that they represent only general, preliminary ideas of how to manage use conflicts for this species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Workshop-generated Proposed Management Strategy[96]:

- Continue to prevent overgrazing on catchfly sites.
    - Potential benefits: Reduce catchfly mortality, prevent degradation of grassland community, reduce soil compaction, reduce incidence of bare ground and invasive species encroachment.
    - Potential conflicts: May require public buy-in.

Literature-generated Proposed Management Strategies[97]:

- Prevent grazing in late summer, grazing on mesic catchfly sites, and limit length of grazing time.
    - Potential benefits: Reduce catchfly mortality, protect catchfly reproduction, seed crop, and recruitment (Hill and Gray 2004), reduce invasive weed encroachment, protect below-ground bumblebee nests, increase overall resilience of grassland systems and Spalding's catchfly.
    - Potential challenges: May require additional institutional capacity and enhanced public outreach and cooperation.
- In the absence of natural fire, implement prescribed burns in spring and fall in catchfly habitat.
    - Potential benefits: Enhance recruitment, reduce competition, reduce litter depth, prevent conifer encroachment (Lesica 1999).
    - Potential challenges: May increase invasive species presence, effects on dormant individuals not fully understood.

Reviewer-generated Proposed Management Strategy[98]:

- Prioritize catchfly-occupied grasslands within the NPCW region for weed monitoring and management.
    - Potential benefits: Prevent invasive species establishment and competition with Spalding's catchfly, maintain grassland integrity.
    - Potential challenges: May require increased research, funding, and/or institutional capacity.

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[96] Workshop-generated management strategies were developed by participants at the Nez Perce-Clearwater Vulnerability Assessment Workshop held in September 2013.

[97] Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

[98] Reviewer-generated management strategies were developed by regional experts from the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014.

Whitebark Pine[99]

Executive Summary

The relative vulnerability of whitebark pine in this assessment is considered moderate to moderate-high to high, as it demonstrates moderate-high sensitivity to climate and climate-driven changes, high sensitivity to non-climate stressors, and moderate-high adaptive capacity. Whitebark pine is sensitive to climate and climate-driven changes such as:

- temperature increases,
- precipitation changes (e.g., snowpack declines) resulting in reduced soil moisture, and
- wildfire.

Climate and climate-driven changes may have variable impacts on whitebark pine. For example, warmer temperatures may "push" whitebark pine off the mountain, or could allow increased growth and encroachment into previously unoccupied areas. Moisture deficits could allow whitebark pine to outcompete its drought-intolerant competitors, but could also reduce whitebark pine seedling germination and survival. Increased low-to-moderate severity wildfire may benefit whitebark pine by increasing opportunities for colonization and regeneration.

Whitebark pine is also highly sensitive to non-climate stressors including:

- mountain pine beetle outbreaks,
- white pine blister rust, and
- fire suppression practices (in lower subalpine areas).

These non-climate stressors may interact synergistically and/or exacerbate climate-driven changes. For example, prolonged moisture stress may enhance susceptibility of whitebark pine to infestation, increasing the risk of widespread beetle-related mortality. Whitebark pine exhibits moderate-high adaptive capacity due to its large geographic extent and ability to disperse fairly long distances, as well as its relatively high genetic diversity and demonstrated potential for life history diversity and behavioral/phenotypic plasticity.

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[99] The following summary represents vulnerability assessment results for a given ecosystem or species, which is comprised of evaluations and comments from an expert-elicitation workshop, peer-review comments following the workshop, and relevant references from the literature. During the workshop, participants assigned one of seven rankings (Very High-7, High-6, Moderate-High-5, Moderate-4, Low-Moderate-3, Low-2, or Very Low-1) to each finer resolution element of sensitivity and adaptive capacity, and provided a corresponding confidence score (e.g., High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) to the ranking. These individual rankings and confidence scores were then averaged (mean) to generate rankings and confidence scores for sensitivity and adaptive capacity. Exposure to climate and climate-driven changes was ranked in order of importance, and also included a confidence evaluation. Results presented in a range (e.g. from moderate to high) reflect variability assessed by participants. The aim of this summary is to expand understanding of resource vulnerability to changing climate conditions, and to provide a basis for developing appropriate adaptation responses. The resulting document is an initial evaluation of vulnerability based on existing information and expert input. It is intended to be a living document that can be revised and expanded upon as new information becomes available.

Sensitivity and Exposure

Sensitivity to climate and climate-driven changes

Whitebark pine (*Pinus albicaulis* Engelm.) is associated with harsh, cold, high elevations with deep winter snow. Whitebark pine is shade-intolerant, and can be found as a climax species at alpine treeline and on ridgelines where other species are excluded due to moisture, temperature, and other environmental constraints (e.g., wind) (Tomback 2007; USFWS 2011). Whitebark pine occurs at the highest extent of subalpine vegetative zones with limited opportunity for continued vertical migration (Keane et al. 2012), and physical factors (e.g., wind, snow) can prevent tree establishment at higher elevations (Malanson et al. 2007). In addition, whitebark pine exists as an early or mid-successional species below treeline in subalpine habitats (USFWS 2011).[100] Whitebark pine exhibits sensitivity to warming temperatures, water deficits, and wildfire. Overall, the sensitivity of whitebark pine to climate and climate-driven changes is considered high.[101]

Table 20. Potential whitebark pine response relative to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated whitebark pine response
Warmer air temperatures	• "Pushed" off the mountain by moving species' lower elevational limits upward • Expansion into prior snowfields • Increased krummholz growth • Longer growing season
Reduced soil moisture or drought (i.e., resulting from decreased snowpack and earlier snowmelt)	• Reduced seed germination • Reduced seedling survival • Increased susceptibility to insect or disease outbreaks
Wildfire	• Low-moderate severity fire: Increased colonization/regeneration opportunities • High-intensity: Loss of mature, cone-bearing individuals and reduced regeneration

Whitebark pine forests appear to have complex responses to warmer temperatures and moisture deficits. For example, historical analogs of warmer climates indicate whitebark pine was maintained and even increased in some areas under past warmer, drier climates (Tausch et al. 1993). Similarly, warming throughout the 20th century in the southeastern Sierra Nevada was positively correlated with whitebark pine growth (Millar et al. 2004). Continued warming could effectively lengthen the growing season by causing earlier snow- and soil-melt, and reducing the severity and frequency of nighttime freezing and wind desiccation events, allowing for increased krummholz growth and/or whitebark pine encroachment onto previous snow fields (Millar et al. 2004). Whitebark pine may become more competitive on moisture-limited sites

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[100] Please refer to the Subalpine Ecosystem section of this report for additional information.
[101] This evaluation is based on information in the scientific literature and from a topic expert.

(Bollenbacher et al. 2013). However, significant whitebark pine mortality occurred at lower subalpine elevations in the Sierra Nevada that were experiencing warmer, drier conditions relative to regional species distribution (Millar et al. 2012).

Reductions in snowpack and associated declines in soil moisture may decrease site suitability, increase mortality, or lead to regeneration problems for whitebark pine. Whitebark pine seedlings are vulnerable to increased runoff that removes valuable topsoil and seeds, as well as denser snowpack that can rip shallow-rooted seedlings out of the ground as it shifts and slides downhill (Keane and Parsons 2010). Whitebark pine seedlings are also vulnerable to moisture declines in seedbeds, which limit germination and seedling survival (McCaughey 1990). Warming temperatures and moisture stress can also increase species' susceptibility to insect and pathogen attack (USFWS 2011; Bollenbacher et al. 2013).

Wildfire also plays a key role in whitebark pine succession and success. Whitebark pine is a fire-adapted species that readily regenerates in large burned areas (Arno and Hoff 1989; Tomback et al. 2001b *in* Keane et al. 2012), and has historically experienced all fire regimes (non-lethal, stand-replacing, and mixed-severity; Keane and Parsons 2010). It demonstrates many adaptations that allow it to survive low to moderate severity fire including thick bark, high and thin crowns, and deep roots (USFWS 2011; Keane et al. 2012). Non-lethal surface fires have historically helped whitebark pine to maintain dominance in the overstory, as it survives and continues cone production following fire while other species (e.g., Englemann spruce and subalpine fir) are eliminated (Keane 2001; Keane and Parsons 2010; Keane et al. 2012). Additionally, whitebark pine is better able to colonize burned and/or disturbed areas than competitors, as disturbance (i.e., fire) creates caching habitat for nutcrackers and removes competitors, creating optimal growing conditions for whitebark pine (Tomback et al. 1993 *in* Keane et al. 2012). Whitebark pine seeds are primarily dispersed by the Clark's nutcracker (*Nucifraga columbiana*), which inhabit patchy forest openings resulting from moderate to low severity fire and can transport whitebark pine seeds over fairly long distances (>32 km) to burned/disturbed areas (Tomback 2007; Lorenz et al. 2011). In general, whitebark pine benefits from wildfire and/or the cessation of fire suppression practices, as it is better able to both survive fire and regenerate in burned areas than its competitors (e.g., Englemann spuce and subalpine fir) (USFWS 2011; Keane et al. 2012). However, moderate- to high-intensity burns (> 60% are burned) can lead to mortality of mature trees and reduced regeneration capacity (e.g., see Keane and Parsons 2010).

Life history

Whitebark pine can be found in both treeline and upper subalpine communities. At harsh sites at or above treeline, whitebark pine occurs in relatively pure climax stands and often displays a krummholz form (Tomback 2007; USFWS 2011). Below treeline and within subalpine mixed conifer communities, whitebark pine can occur as an early successional or seral co-dominant with subalpine fir and other species (Tomback 2007; Keane and Parsons 2010; USFWS 2011). Whitebark pines can live between 500 and 1000 years (Arno and Hoff 1989) if not replaced by more shade-tolerant subalpine fir, mountain hemlock, and Englemann spruce (Keane and Parsons 2010). Whitebark pines typically start producing cones between 30 and 60 years of age, but cone production rates are highly influenced by canopy volume, and the highest cone production rates typically begin when whitebark pines are between 125-250 years old (Arno and Hoff 1990, Krugman and Jenkinson 1974 *in* Keane et al. 2012). Whitebark pines are monoecious (Tilley et al. 2011; USFWS 2011), and cones are typically produced in the upper branches and crowns of mature trees (Larson 2009). Whitebark pine seeds are heavy, wingless, and fixed to host cones, which prevents wind dissemination (USFWS 2011). Although over 110 species feed on whitebark pine cones and seeds, whitebark pine relies on the Clark's nutcracker for seed dispersal (Tilley et al. 2011; Keane et al. 2012), and seeds are typically harvested by the nutcracker during late summer and early fall. The nutcracker can transport up to 100 seeds at a time (Keane and Parsons 2010) and over a distance of 32.6 km (Lorenz et al. 2011), typically caching seeds in open, recently disturbed areas where whitebark pine can establish with limited competition (Keane and Parsons 2010). Seed germination, stimulated by snowmelt, spring rain, or summer showers (Keane et al. 2012), typically occurs more than two years after caching by the nutcracker (Tilley et al. 2011), and germinated seeds survive best in well-drained, mesic to moderately dry soils (Izlar 2007) with ectomycorrhizal fungi (Mohatt et al. 2008). Currently, active growing season for whitebark pine begins in late May and continues through early September (Izlar 2007). Low to moderate severity wildfire is the main disturbance regime that creates preferred zones for nutcracker caching (Keane and Parsons 2010); fire helps maintain old growth whitebark pine stands (with higher cone production) by removing subalpine fir and other fire-intolerant competitors and by eliminating small, thin-barked seedlings (Keane 2001; USFWS 2011; Keane et al. 2012).

Whitebark pine and Clark's nutcracker have a mutualistic relationship, and population declines in one species may lead a positive feedback cycle of population decline in both species. Nutcracker populations have been declining since 1997, and abundance surveys indicate that nutcrackers are decreasing by more than 1.5% per year across much of the NPCW region (Lorenz and Aubry 2011). Although exact causes for this decline are unknown (Lorenz and Aubry 2011), it may be correlated with lower cone production in declining whitebark pine stands (McKinney et al. 2009), as nutcrackers have historically been observed to emigrate regionally (Bock and Lepthien 1976, Davis and Williams 1957 *in* Keane et al. 2012). Declining or emigrating nutcracker populations, combined with climate and non-climate impacts to whitebark pine, could negatively affect the extent of NPCW whitebark pine populations, potentially exacerbating future population declines in these mutualistic species.

Future climate exposure

Important climate and climate-driven factors to consider for whitebark pine include temperature increases; reduced soil moisture and drought (i.e., from decreased snowpack and earlier snowmelt); and wildfire. Average annual air temperature in the NPCW region has been increasing by 0.13˚C per decade, and is projected to warm 2.1-2.5˚C by 2040 (Isaak et al. 2011; Littell et al. 2011; see also Section 3), with warmer seasonal temperatures generally occurring in summer. Minimum and maximum temperatures have also been increasing, and are projected to continue increasing, with more significant warming in winter and summer (Littell et al. 2011; see Section 3). Warming temperatures could have varying impacts on whitebark pine. For example, increased minimum temperatures could extend the growing season and allow whitebark pine to invade prior snowfields and experience increased krummholz growth (Millar et al. 2004). Alternatively, warming temperatures may “push” whitebark pine off the mountain by moving its lower elevational limits upward (Warwell et al. 2007; Schrag et al. 2007). Warmer winter temperatures could also accelerate the expansion and establishment of beetle populations (Logan et al. 2003) as well as blister rust (Koteen 1999 *in* Keane et al. 2012) in high-elevation whitebark pine areas, and may result in increased high-elevation whitebark pine mortality (e.g., see Jewett 2009). Temperature increases may also exacerbate soil moisture deficits.

Climate change (in particular, warming temperatures) in the Pacific Northwest is projected to lead to changes in snowpack (Pierce et al. 2008), including reduced April 1st snowpack (Mote et al. 2005; Regonda et al. 2005) and earlier melting of snowpack (Mote et al. 2005; Stewart et al. 2005; Westerling et al. 2006; Barnett et al. 2008). Annual snowpack projections are difficult to accurately model, since they are dependent on both temperature trends (better understood) and precipitation trends (poorly understood), however SWE (a measure of April 1st snowpack) is expected to decline in the NPCW region by ~21% by 2040 (Littell et al. 2011). Snowpack declines and earlier snowmelt may lead to earlier soil moisture recharge, which may lead to earlier desiccation of soils (Hamlet et al. 2007). Soil moisture is projected to decline in the NPCW region by 2040 (Littell et al. 2011; see Section 3). Projected declines in snowpack volume and earlier snowmelt and associated soil moisture deficits could affect whitebark pine site suitability, increase mortality, or alter susceptibility to insect and pathogen attack. Models of future species distributions also suggests that under projected drought-like conditions, high elevations in the Greater Yellowstone Ecosystem become dominated by spruce-fir forests, with a decrease in area occupied by whitebark pine (Schrag et al. 2007).

Whitebark pine historically experienced mixed severity fires every 60-300+ years and stand-replacing fires every 250+ years. In the past, whitebark pine has benefitted from mixed severity wildfire by surviving and colonizing burned areas, leading to a complex landscape mosaic of different successional stages (Keane and Parsons 2010; Keane et al. 2012). Wildfire is projected to increase in extent in response to increased drought and decreased precipitation combined with warmer temperatures (Littell et al. 2009; Littell et al. 2010). In the western U.S., large and widespread fires are more likely when early, warm springs are followed by warm, dry summers (Westerling et al. 2006; Morgan et al. 2008; Littell et al. 2009). Due to a combination of climate and fuel conditions, an increase in fire season length (~78 days longer; Westerling et al. 2006), as well as significant increases in area burned and the number of large fires (Running 2006) have also been observed in the western U.S. On average, warmer, drier conditions are projected to lead to more frequent, severe fires in forest ecosystems (McKenzie et al. 2004), increases in annual number and area burned (Flannigan et al. 2009; Krawchuk et al. 2009; Marlon et al. 2009), as well as an increase in crown fires due to increased fuel density and decreased fuel moisture (Bollenbacher et al. 2013). Fire intensity is also projected to increase (Brown et al. 2004a). In the northwestern and southwestern U.S., topography strongly influences patterns of burn severity (Holden et al. 2009; Dillon et al. 2011). Holden and Jolly (2011) further demonstrate that fire danger varies by topography and time of season. For example, in early season (July), fire danger on north-facing slopes is less than south-facing slopes at upper elevations, whereas at peak season (mid- to late September), fire danger is similar across terrain, and in late season (after September 29th), valley bottoms recover fuel moisture but dry fuel moisture and high fire danger persist on low elevation, south-facing slopes. The effects of wildfire will likely vary from place to place, as the relative importance of fuels, weather, topography, vegetation, climate, and land use (including fire exclusion) also vary by location (e.g., see Abatzoglou et al. 2013).

Increased frequency or size of wildland fires could facilitate expansion of whitebark pine populations, as nutcrackers disperse seeds into large burn areas more effectively than wind-dispersed seeds of whitebark pine competitors (provided sufficient seed sources are available) (Lorenz and Sullivan 2009; Tomback et al. 1990 *in* Keane et al. 2013). Similarly, some modeling suggests climate-mediated changes in disturbance regimes (i.e., fire) will help maintain whitebark pine within its current range (Loehman et al. 2011). However, increased frequency of large-scale, stand-replacing fires could be detrimental to whitebark pine. For example, more stand-replacing fires could eliminate the "patchy" habitat the Clark's nutcracker prefers, as well as eliminate whitebark pine individuals that are resistant to disease and insects (Keane and Parsons 2010). Furthermore, though whitebark pine can establish following fire, there may be a significant time lag (up to 40 years) before successful establishment can be seen due to the extreme nature and high disturbance of these sites (Keane and Parsons 2010). Refugia for whitebark pine may include some high-elevation microsites with increased fire activity that allow for increased whitebark pine regeneration and growth under climate change (Bunn et al. 2003).

Sensitivity and current exposure to non-climate stressors

Information provided by a topic expert and supported by the scientific literature identified several non-climate stressors that affect the sensitivity of whitebark pine, including mountain pine beetles, white pine blister rust, and fire suppression practices in lower elevations. The most significant ongoing mortality episode of subalpine forests in western North America is in whitebark pine forests (Millar et al. 2012). The primary reason for biome-wide mortality of whitebark pine is attributed to outbreaks of white pine blister rust (*Cronartium ribicola*) and mountain pine beetle (*Dendroctonus ponderosae*) (Logan and Powell 2001; Tomback and Achuff 2010), both of which are linked to changes in climate. In the NPCW region, mountain pine beetles and white pine blister rust have caused widespread decline of mature whitebark pine (Keane and Parsons 2010; Keane et al. 2012), and caused shifts in ecosystem composition (USFWS 2011; Keane et al. 2012). Fire suppression practices can accelerate successional replacement of whitebark pine and increase their vulnerability to disease or beetle attack (USFWS 2011; Keane et al. 2012). Fire suppression has not played as large of a role as mountain pine beetles and blister rust in whitebark pine decline in the NPCW region, as a majority of whitebark pine is located in wilderness areas where fires are allowed to burn naturally (K. Hazelbaker, pers. comm., 2014).

Bark beetles and other herbivorous insects have played important historical roles in forest ecosystems by acting as major mortality agents, reducing tree competition, and restarting successional stages. Large-scale, native mountain pine beetle mortality events were historically rare in whitebark pine forests, as cold temperatures limited mountain pine beetle life cycles and invasion success (Logan and Powell 2001). The life cycle of the mountain pine beetle is temperature-dependent; warmer temperatures associated with climate change can increase tree phloem temperature (Logan and Powell 2001), accelerate beetle life cycles, enhance beetle population growth, and cause more successful whitebark pine invasions by increasing emergence and invasion synchronicity (Logan and Powell 2001; Bentz et al. 2010; Jewett et al. 2011). For example, Logan and Powell (2001) found that more than a 2˚C increase in phloem temperature caused shifts to synchronous, univoltine beetle life cycles; shorter life cycles and synchronous invasions typically result in higher tree mortality (Logan and Powell 2001; Jewett et al. 2011). Warmer regional temperatures have caused elevated and widespread mountain pine beetle-induced mortality of whitebark pine since the early 20th century, especially among cone-bearing trees over 20 cm in diameter at breast (dbh) height (Keane and Parsons 2010), and this trend is likely to continue (Hicke and Logan 2009). In addition, infestation zones may move to higher elevations as regional air temperature warms (Logan and Powell 2001; Powell and Logan 2005). Prolonged moisture stress may also enhance susceptibility of whitebark pine to infestation, increasing the risk of widespread beetle-related mortality (Fettig et al. 2007; Tomback 2007). Increased beetle herbivory leads to more woody fuel, which could enhance the likelihood of large, stand-replacing fires, especially during dry summer periods (Logan et al. 2003; Bollenbacher et al. 2013). However, current research indicates that recent bark beetle outbreaks (4-13 years prefire) may not affect subsequent fire severity (Harvey et al. 2013). Additionally, mountain pine beetles may decrease population-wide whitebark pine resilience to white pine blister rust by killing disease-resistant individuals (Tomback 2007).

Whitebark pine is also sensitive to exotic white pine blister rust (Tomback and Achuff 2010), and roughly 80% of white pines in the northern Rocky Mountains are infected (Tomback 2007). White pine blister rust typically inhibits the trees' ability to produce seeds (Arno and Hoff 1989) and prevents cone production years before the host tree dies; declining numbers of cones may cause nutcracker population declines or emigration (McKinney et al. 2009; Keane et al. 2012), reducing seed dispersal opportunities for whitebark pines (Tomback 2007). White pine blister rust can also increase whitebark pine susceptibility to beetle-related mortality (Tomback and Achuff 2010). In a study conducted by Mahalovich et al. (2006), roughly 48% (n=108) of inland Northwest trees, including individuals from NPCW whitebark pine populations, showed some resistance to white pine blister rust (Mahalovich et al. 2006; Mahalovich and Hipkins 2011). This genetic resistance is moderately heritable (Mahalovich et al. 2006), and seeds from resistant individuals could become an important base for future restoration efforts (Mahalovich et al. 2006; Tomback 2007; Mahalovich and Hipkins 2011). Although it is not precisely known how climate change will affect white pine blister rust itself, a warmer climate may accelerate the spread of blister rust (Koteen 1999 *in* Keane et al. 2012), and heightened moisture stress and associated declines in tree vigor may increase white pine blister rust infestations and associated mortalities.

Fire suppression practices in lower subalpine elevations have led to the successional replacement of whitebark pine with subalpine fir in some parts of its range (Barrett 2008, Keane 2001b *in* Keane et al. 2012). The absence of fire allows spruce and subalpine fir to outcompete and replace shade-intolerant whitebark pine; spruce and fir now appear in much larger areas than historically present, while whitebark pine presence has decreased (Tomback 2007; USFWS 2011; Keane et al. 2012; Bollenbacher et al. 2013). The exclusion of fire reduces disturbance, leading to homogenous, dense forests that are less resilient to fire, insects, and disease. High-density forest conditions can also exacerbate moisture stress for individual whitebark pines, enhancing mortality or susceptibility to white pine blister rust infection or mountain pine beetle infestation (USFWS 2011). Further, in the absence of mixed severity fires, the amount of open, “patchy” habitat preferred by the Clark’s nutcracker is reduced, thus limiting the available open areas for whitebark pine seed dispersal and colonization (Tomback 2007; Keane and Parsons 2010). In the NPCW region, it appears that fire suppression practices have had less influence on whitebark pine mortality than beetles or blister rust (K. Hazelbaker, pers. comm., 2014), mainly due to the location of whitebark pine in wilderness or roadless areas.

Adaptive Capacity

Species extent, status, and diversity

Whitebark pine exhibits moderate-high adaptive capacity due to its geographic extent and ability to disperse fairly long distances, as well as demonstrated genetic diversity and the potential for life history diversity and behavioral/phenotypic plasticity.[102] Whitebark pine occurs across 5 million acres in the northern Rocky Mountains and appears in varying forms across elevations ranging from 900 to 3200 m (Tomback 2007), but has a small and declining population. It is now a Candidate species for ESA listing (USFWS 2011) and is considered a Sensitive species within the USFS Northern Region. Whitebark pine is considered a keystone and foundational species, and can be found as a climax species on high ridges and mountaintops and as early or mid-seral species in lower elevation mixed conifer stands (USFWS 2011; Keane et al. 2012). It has the ability to disperse fairly long distances (>32 km; Lorenz et al. 2011) and across large elevational gradients due to its main disperser, the Clark’s nutcracker, and thus can exploit a variety of habitat openings at varying elevations (Tomback 2007; Keane et al. 2012).

Whitebark pine demonstrates high levels of genetic diversity within inland Northwest seed source zones (Mahalovich and Hipkins 2011). For example, compared to whitebark pine in Oregon and Washington, inland Northwest whitebark pine individuals had higher mean expected heterozygosity and number of alleles per locus (Mahalovich and Hipkins 2011). In addition, 100% of tested individuals (n=108) had polymorphic loci (Mahalovich and Hipkins 2011). Whitebark pine individuals also feature heritable genetic resistance to white pine blister rust (Mahalovich et al. 2006). Despite past hypotheses of inbreeding depressions in this species due to their discontinuous populations, current analyses indicate little inbreeding in inland Northwest populations (Mahalovich and Hipkins 2011), though this could change if climate and climate-driven changes reduce wind pollination or seed caching at higher elevations, which

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[102] This evaluation is based on information in the scientific literature and from a topic expert. No confidence was assessed as part of this evaluation.

currently contribute to high within-population genetic variation in whitebark pine (Mahalovich and Hipkins 2011). For example, nutcracker caching results in clumped whitebark pine population patterns, with tree clusters featuring genetically distinct but related stems that are not genetically related to other nearby clusters (Tomback 2007). In addition, two genotypic groups of white pines, differentiated by their responses to water stress and high temperatures, appear to occur in the Sierra Nevada (Millar et al. 2012). One group appears to be adapted to warmer and drier conditions of the current century, while the other group may be more adapted to the cool, wet conditions characteristic of the Little Ice Age, the period when whitebark pine first established (Millar et al. 2012). However, it is unknown if these genotypic groups also occur in the NPCW region.

Whitebark pine also exhibits potential for life history and behavioral diversity as well as some phenotypic plasticity. For example, whitebark pine is capable of extremely slow growth under adverse conditions (McCune 1988 *in* Tomback 2007), which may increase its ability to survive short-term climatic events such as drought. Due to its long life span (can be >600 years), whitebark pine may also be able to take advantage of climatic conditions favorable for regeneration, despite unfavorable climate changes in the short-term (Loehman et al. 2011). In areas with long-term snow cover, whitebark pine can reproduce by asexual layering – flexible branches are bent into the ground by snow and create a krummholz stand (Tilley et al. 2011). Whitebark pine seedlings are also believed to show phenotypic differences in morphology and performance depending on local site conditions (Izlar 2007), such as displaying the krummholz growth form (Keane et al. 2012). In addition, whitebark pine can also utilize the “candelabra” or lyrate growth form, which features a flat top canopy, potentially increasing cone production and increasing the visibility of cones for Clark’s nutcrackers (Keane et al. 2012).

Management potential

Reviewers and peer-reviewed literature suggest pursuing whitebark pine restoration activities to improve overall resistance and resilience to climate changes and disturbance regimes. These management strategies are further outlined below, but please note that they represent only general, preliminary ideas for this species. More specific management strategies that address climate and non-climate stressors will be developed and presented in future documents.

Proposed Management Strategies[103]:

- Continue to manage for natural fire occurrence in whitebark pine habitat.
    - Potential benefits: Increase forest heterogeneity, create caching habitat for nutcrackers, reduce fuel density, increase whitebark pine resilience to beetle, rust, and fire mortality by reducing competition and fuel density (Keane and Parsons 2010; Keane et al. 2011; Keane et al. 2012).
    - Potential challenges: Administrative and regulatory barriers.

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[103] These proposed management strategies were identified by reviewers, and supported by the peer-review literature. Reviewer-generated strategies were gathered from regional experts at the Nez Perce-Clearwater National Forest and USFS Northern Region during a peer-review period from January-March 2014. Literature-generated management strategies were developed based on information from peer-reviewed literature, which may or may not be directly applicable to the Nez Perce-Clearwater region.

- Spread genetic resistance to the blister rust pathogen by saving putative and screened blister-rust resistant trees, collecting seeds and growing seedlings from rust-resistant trees, and allowing natural disturbances (e.g., wildfires) to remove competitors (Keane et al. 2013).
    - Potential benefits: Create whitebark pine forests that are resistant and resilient to climate and disturbance regimes.
    - Potential challenges: “Agency/manager fatigue” (Keane et al. 2013), funding, administrative and regulatory barriers.
- Foster diversity of forest ages, compositions, and structures to better manage mountain pine beetle outbreaks (Keane et al. 2013).
    - Potential benefits: Create whitebark pine forests that are resistant and resilient to climate and disturbance regimes.
    - Potential challenges: “Agency/manager fatigue” (Keane et al. 2013), funding, administrative and regulatory barriers.
- Conduct additional research to identify treatments, activities, and protocols that will improve restoration efficacy and efficiency (Keane et al. 2013).
    - Potential challenges: Funding, administrative and regulatory barriers, may require increased institutional capacity.

7. Conclusions

The results of this vulnerability assessment are intended to help guide and support a manager or planner in identifying which resources are likely to be most affected by changing climate conditions, and improving understanding as to why those resources are likely to be vulnerable. The results of this assessment are a new toolset among many that can be used in managing natural resources for climate change. The vulnerability assessment findings from this process are intended to be a living resource that new information can be added to as it becomes available. One way to keep apprised of new information for a particular resource is through the use of TACCIMO (Template for Assessing Climate Change Impacts and Management Options), a web-based tool that connects forest planning to current climate change science. Box 1 describes several ways to find information on a specific resource using TACCIMO.

Box 1. Using TACCIMO to find the most current climate change science for a resource

TACCIMO provides access to the most current climate change projections and science for forest resources, including dynamically linked peer-reviewed scientific statements described climate change effects on resources. To find current information on a resource (e.g., ecosystem, species), enter the TACCIMO site (http://www.taccimo.sgcp.ncsu.edu/) and click the tab “Generate a Report”. From the drop-down menu, select “Custom Reports-Beta Version”. There are two options for climate change effects on a resource: Effects By Source Report and Effects By Source Report – Keywords.

For coarse filter resources (e.g., ecosystems, habitats): select "Effects By Source Report" and, from the drop-down menu, select the relevant Factor (e.g., for aquatic systems, select Freshwater Ecosystems; for alpine ecosystems, select Plant Communities). Use the drop-down menu for Region to find literature for a specific region such as the Northern Region. Use the Category drop-down menu to select more specific components within a factor (e.g., riparian areas within freshwater ecosystems). Click “Run Report” once the Factor, Region, and Category have been identified.

For fine filter resources (e.g., species): select "Effects By Source Report - Keywords". Under Factor and Category drop-down menus click “<select all>”. Use the drop-down menu for Region to find literature on the resource for a specific place, such as the Northern Region. Type the scientific name of the species into the Keywords in quotations (e.g., "Pinus ponderosa"). Click “Run Report” and click on the factors in the left column to see current literature on the species.

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