Chapter 5: Effects of Climate Change on Native Fish and Other Aquatic Species

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Introduction

The diverse landscapes of the Intermountain Adaptation Partnership (IAP) region contain a broad range of aquatic habitats and biological communities. A number of aquatic species are regional endemics, several are threatened or endangered under the U.S. Endangered Species Act (ESA), and many have declined because of the introduction of nonnative aquatic species, habitat fragmentation, and human development. Environmental trends associated with human-caused climate change have been altering the habitats of these species for several decades (Barnett et al. 2008; Hamlet and Lettenmaier 2007; Luce and Holden 2009; Mote et al. 2005), and more significant changes are expected during the 21st century (chapters 3, 4). For animals that live in or near aquatic environments such as fishes, amphibians, crayfish, mussels, and aquatic macroinvertebrates, changes in habitat and hydrological regimes are expected to shift their abundance and distribution (Ficke et al. 2007; Hauer et al. 1997; Poff et al. 2002; Rieman and Isaak 2010; Schindler et al. 2008). This is primarily because many of these species are ectothermic; thus, thermal conditions dictate their metabolic rates and most aspects of their life stages—how fast they grow and mature, whether and when they migrate, when and how often they reproduce, and when they die (Magnuson et al. 1979). Buffering these changes are the topographic diversity and steep environmental gradients of many landscapes throughout the IAP region, which contribute to slow climate velocities (sensu Loarie et al. 2009) and often create climate refugia where populations of many species can persist under all but the most extreme climatic changes (Isaak et al. 2016a; Morelli et al. 2016).

A large literature exists that describes the many interactions among climate change, aquatic environments, and cold-water fishes such as trout, salmon, and char (Hauer et al. 1997; Isaak et al. 2012a,b; ISAB 2007; Mantua and Raymond 2014; Mantua et al. 2010; Mote et al. 2003; Rieman and Isaak 2010; Young et al. 2018). Rather than revisiting those sources, we focus on providing information specific to the IAP region. First, we describe the ecology, status, and climate vulnerabilities of species of concern. Through discussions with scientists and U.S. Department of Agriculture Forest Service (USFS) Intermountain Region staff and national forest resource managers, species were chosen for their perceived vulnerability to climate change and their societal prominence as ESA-listed threatened and endangered species. The species are Rocky Mountain tailed frog (*Ascaphus montanus*), Idaho giant salamander (*Dicamptodon aterrimus*), western pearlshell mussel (*Margaritifera falcata*), springsnails (*Pyrgulopsis* spp.), Yosemite toad (*Anaxyrus canorus*), Sierra Nevada yellow-legged frog (*Rana sierrae*), bull trout (*Salvelinus confluentus*), and cutthroat trout (*Oncorhynchus clarkii*). Second, we develop spatially explicit model projections and geospatial datasets showing where bull trout and cutthroat trout are most likely to occur in current and future climates. These projections are used to assess the potential and future distribution of suitable habitats for these species, but could also be used to design and implement long-term conservation strategies or monitoring programs. Although the availability of biological datasets and models for stream networks restricted our projections to trout in streams, the approach used here is easily extended to other aquatic species as more geographic data on these taxa are gathered and models are extended to standing waters. The preceding information was used to develop climate adaptation options for species of concern, including how new technologies and ongoing development of better information could enable strategic implementation of those options to maximize their effectiveness.

Analysis Area Network and Stream Climate Scenarios

This assessment encompasses all streams in the USFS Intermountain Region that flow through its 12 national forests and lands downstream of those forests. To delineate a network that represented streams within this area, geospatial data for the 1:100,000-scale National Hydrography Dataset (NHD)-Plus Version 2 were downloaded from the Horizons Systems website (Horizon Systems Corp. http://ww.horizon-systems.com/nhdplus/; McKay et al. 2012) and filtered by minimum flow and maximum stream slope criteria. Summer flow values predicted by the Variable Infiltration Capacity (VIC) hydrological model (Wenger et al. 2010) were obtained from the Western U.S. Flow Metrics website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml) and

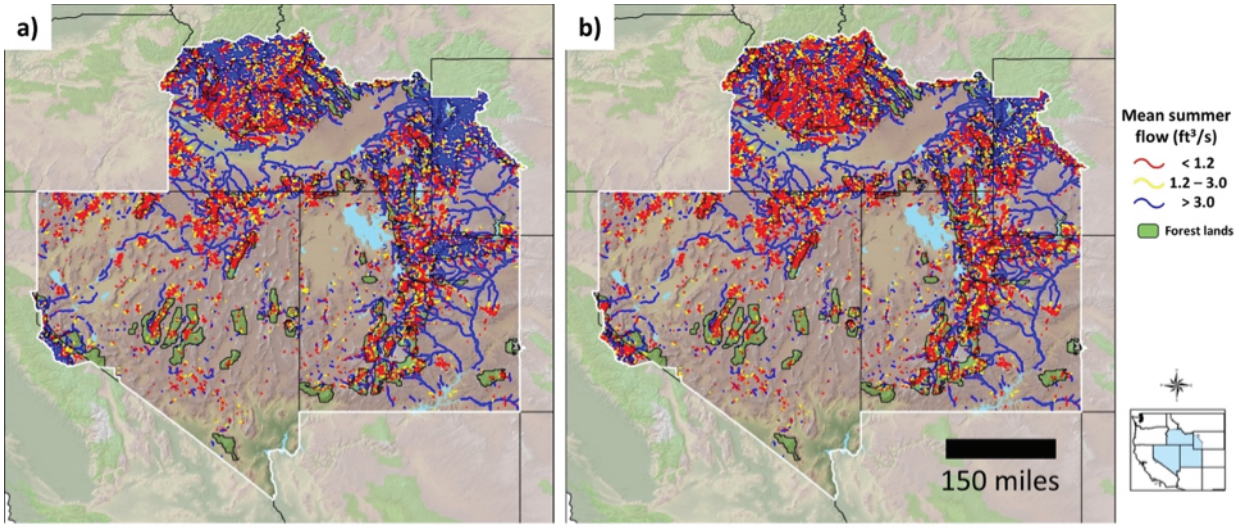

Figure 5.1—Stream network showing channels with perennial flows for (a) the baseline period and (b) 2080s based on the A1B emissions trajectory. Increasing prevalence of red stream reaches late in the century indicates a trend towards lower summer flows as winter snow accumulations decrease and melt earlier in the spring.

linked to NHDPlus stream reaches. Reaches with summer flows less than 0.2 cubic feet per second, approximating a wetted width of 3 feet (based on an empirical relationship developed in Peterson et al. 2013b), or with slopes greater than 15 percent were trimmed from the network because they are rarely occupied by fish or other aquatic vertebrates (Isaak et al. 2017b). The steepest headwater reaches are also prone to frequent large disturbances (e.g., postfire debris torrents) that may cause local extirpations of fish populations (Bozek and Young 1994; Miller et al. 2003). To exclude dry channels throughout much of the Great Basin, reaches that were coded as intermittent in the NHDPlus network were also trimmed. Application of these criteria resulted in a final network extent of 55,700 miles (fig. 5.1), which was almost evenly split between USFS (48 percent) and non-USFS (52 percent) lands.

Scenarios representing mean August stream temperature were downloaded from the NorWeST website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html) and linked to reaches in the analysis network. NorWeST scenarios have a 0.6-mile resolution and were developed by applying spatial-stream network models (Isaak et al. 2016b, 2017b; Ver Hoef et al. 2006) to temperature records for 8,726 summers of measurement at 4,277 unique stream sites within the IAP region. The predictive accuracy of the NorWeST model (cross-validated $r^2 = 0.91$; cross-validated root mean square prediction error = 2.0 °F), combined with substantial empirical support, provided a consistent and spatially balanced rendering of temperature patterns and thermal habitat for all streams. To depict temperatures during a baseline period, we used the S1 scenario, which represented average conditions for 1993–2011 (hereafter 2000s). The mean August stream temperature during this period was 57 °F for all streams, although temperatures were significantly colder in streams flowing through national forests, where the average was 52 °F (table 5.1, fig. 5.2).

Future stream temperature scenarios were also downloaded from the NorWeST website (USDA FS n.d.b) and chosen for the same climate periods (2030–2059, hereafter 2040s; 2070–2099, hereafter 2080s) and emissions scenario (A1B) as those used for the streamflow analysis in the IAP hydrological assessment (Chapter 4). With respect to scenarios used in other chapters of the IAP assessment, the A1B scenario is similar to the RCP 6.0 scenario associated with CMIP5 simulations (Chapter 3). The future NorWeST scenarios used were S30 (2040s) and S32 (2080s), which accounted for differential sensitivity and slower warming rates of the coldest streams (Luce et al. 2014). Future stream temperature increases were projected to range from 1.4 to 2.3 °F by mid-21st century and from 2.3 to 4.0 °F by late century, with variation occurring within and among river basins (table 5.2, fig. 5.2). Future temperature increases imply warming rates similar to those observed in recent decades (Isaak et al. 2012b, 2017a) and shifts of stream temperature isotherms upstream at 1,000 to 1,600 feet per decade (Isaak and Rieman 2013; Isaak et al. 2016a).

Changes in several ecologically relevant streamflow characteristics were discussed in the hydrological assessment (Chapter 3), indicating that future snowmelt and spring runoff will occur earlier, summer flows will decrease, stream intermittency may increase in marginal areas, and more high-flow events will occur during the winter in transitional watersheds where air temperatures are near freezing (Hamlet and Lettenmaier 2007). Those projections concur with historical trends that show streams now run off 1 to 3 weeks earlier in the spring (Stewart et al. 2005), and that summer flows have decreased 10 to 30 percent in the last 50 years (Leppi et al. 2012; Luce and Holden 2009). Hydrological changes make it likely that mountain wetlands

Table 5.1—Projected changes in mean August air temperatures, streamflow, and stream temperatures for major river basins in the Intermountain Adaptation Partnership region. Projections are based on the A1B emissions scenario represented by an ensemble of 10 global climate models that best predicted historical climate conditions during the 20th century in the northwestern United States (Hamlet et al. 2013; Mote and Salathé 2010). Additional details about scenarios are provided elsewhere (Hamlet et al. 2013; Wenger et al. 2010). For more information on flow, see the western United States flow metrics website (USDA FS n.d.c) and the Pacific Northwest Hydroclimate Scenarios Project website (University of Washington, Climate Impacts Group 2010). For more information on stream temperatures, see Isaak et al. (2017b), Luce et al. (2014), and the NorWeST website (USDA FS n.d.b).

	2040s (2030–2059)			2080s (2070–2099)		
NorWeST unit[a]	Air temperature change[b]	Streamflow change	Stream temperature change[c]	Air temperature change	Streamflow change	Stream temperature change
	°F	*Percent*	*°F*	*°F*	*Percent*	*°F*
Salmon River basin	5.9	-22.3	2.3	9.9	-31.4	3.7
Upper Snake River, Bear River basins	5.8	-7.6	1.5	9.5	-9.5	2.4
Middle Snake River	5.8	-19.5	2.2	9.8	-26.7	3.7
Utah basins	4.7	+2.3	2.2	10.4	12.6	4.0
Lahontan basin	4.8	+2.6	1.4	10.7	+6.5	2.4

[a] Boundaries of NorWeST production units as described in USDA FS (n.d.c).
[b] Changes in air temperature and stream flow are expressed relative to the 1980s (1970–1999) baseline climate period.
[c] Changes in stream temperatures account for differential sensitivity to climate forcing within and among river basins as described in Luce et al. (2014) and USDA FS (n.d.c).

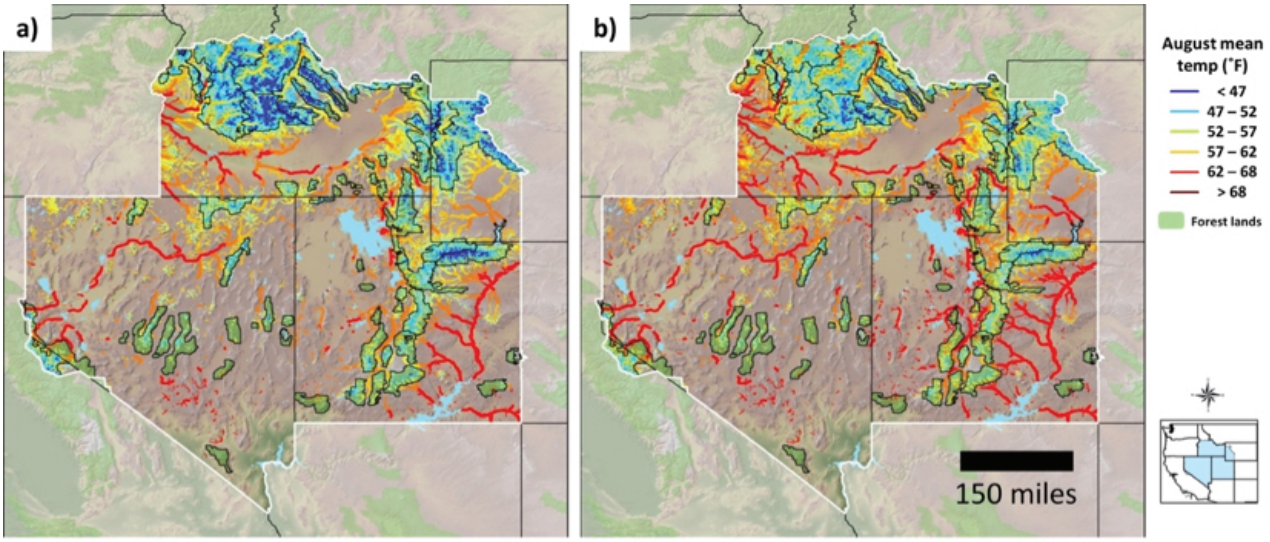

Figure 5.2—NorWeST August mean stream temperature maps interpolated from 8,726 summers of monitoring data at 4,277 unique stream sites across the 55,700 miles of streams in the analysis area. Map panels show conditions for (a) the baseline period (2000s) and (b) late-century scenario (2080s). Networks were trimmed to represent potential fish-bearing streams by excluding intermittent reaches and those with slopes greater than 15 percent and summer flows less than 0.2 cubic feet per second. High-resolution digital images of these maps and ArcGIS databases with reach-scale predictions are available at the NorWeST website (USDA FS n.d.b).

Table 5.2—Lengths of streams in the Intermountain Adaptation Partnership region (slope less than 15 percent and Variable Infiltration Capacity model-predicted summer flows greater than 0.2 cubic feet per second) categorized by mean August stream temperatures during the baseline and two future climate periods and by land administrative status. Values in parentheses are percentages of the total in the last column.

	<46 °F	46–52 °F	52–57 °F	57–63 °F	63–68 °F	>68 °F	Total[a]
Forest Service lands	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*
2000s	3,872 (14.4)	11,061 (39.5)	8,799 (32.7)	3,014 (11.2)	559 (2.1)	37 (0.1)	27,305
2040s	1,644 (6.3)	8,692 (33.1)	9,994 (38.0)	4,790 (18.2)	1,016 (3.9)	141 (0.5)	26,277
2080s	835 (3.2)	6,752 (26.2)	10,371 (40.2)	6,058 (23.5)	1,424 (5.5)	343 (1.3)	25,783
Non-Forest Service lands							
2000s	48 (0.2)	924 (3.2)	4,655 (16.2)	11,490 (39.9)	8,027 (27.9)	3,639 (12.6)	28,783
2040s	6 (0.0)	348 (1.2)	2,896 (10.2)	9,242 (32.5)	9,767 (34.3)	6,185 (21.7)	28,444
2080s	3 (0.0)	173 (0.6)	1,990 (7.0)	7,853 (26.8)	10,552 (37.3)	7,966 (28.2)	28,537

[a] Reductions in network extent in future scenarios result from projected decreases in summer flows as described in Chapter 4.

and moist areas near streams will become drier during future summer periods (Lee et al. 2015). Increased frequency or severity of wildfires in portions of the IAP region could also cause more extensive debris flows and channel disturbances in headwater streams with steep channels (Luce et al. 2012) (Chapter 3). Those combined changes suggest that stream environments and habitats for aquatic species will become more variable, subject to more disturbances, and gradually warmer throughout the rest of the 21st century.

Focal Species Ecology and Climate Vulnerability

Rocky Mountain Tailed Frog

The Rocky Mountain tailed frog occurs throughout central and northern Idaho, western Montana, and northeastern Oregon, but occurs within the IAP region only in Boise, Payette, Salmon-Challis, and Sawtooth National Forests. Populations inhabit steep, cold headwater streams, their distributions often extending upstream past waterfalls and cascades that limit fish distributions (Dunham et al. 2007; Isaak et al. 2017b). After eggs hatch in late summer, tadpoles grow for 1 to 4 years before metamorphosing into adults, which reach sexual maturity after another 4 to 5 years; local densities may be high (Hayes and Quinn 2015; Pilliod et al. 2013). Larval frogs are strictly aquatic, but adults often exploit cool, moist riparian zones to forage. Adult body size is 1 to 2 inches, and dispersal is limited, so floods and fire-related channel disturbances may suppress populations for some time after an event (Hossack and Pilliod 2011; Hossack et al. 2006). Populations are patchily distributed among headwater streams and show evidence of genetic divergence (Metzger et al. 2015).

The Rocky Mountain tailed frog is of conservation concern but does not appear on the sensitive species list of the Intermountain Region. Land use practices that warm streams, increase sedimentation, and reduce interstitial spaces in substrates or reduce habitat moisture (through loss of stream and terrestrial canopy cover) are thought to reduce habitat quality (Hayes and Quinn 2015). Nonnative fish predators may increase mortality where distributions overlap, as has been documented for other amphibians (Pilliod and Peterson 2001; Pilliod et al. 2010). Although existing data for tailed frogs suggest that the species occurs in many streams throughout its range (Isaak et al. 2017b), monitoring data are not available to describe temporal trends in abundance.

Rocky Mountain tailed frogs require cold water, so increasing temperatures may decrease the suitability of warmer downstream habitats (Isaak et al. 2017b). Their long generation times and relatively low fecundity cause populations to rebound slowly from disturbances; thus, extreme floods or postfire debris flows in steep channels may threaten persistence of some populations as climate change causes these events to become more common (Hossack and Pilliod 2011; Hossack et al. 2006). Tailed frog populations may also be negatively affected by more extreme summer droughts or wildfires that open riparian canopies, making areas adjacent to streams warmer and drier (Hossack and Pilliod 2011; Hossack et al. 2006).

Idaho Giant Salamander

The Idaho giant salamander occurs in northern and west-central Idaho and a small portion of west-central Montana, but is found within the IAP region only in Boise and Payette National Forests. Populations are patchily distributed and often co-occur with native salmonids in headwater streams, although salamanders also occupy reaches farther upstream from which fish are excluded (Sepulveda and Lowe 2009).

Giant salamander may also use lakes and ponds. Neotony (maturation as a strictly aquatic form with larval characteristics) is common (Blaustein et al. 1995). Uncertainty exists about timing of reproduction, although some literature suggests both spring and fall spawning (Nussbaum 1969). Females guard egg masses until hatching occurs and larval stages last several years before metamorphosing into adults. Adults reach body sizes of 7 to 12 inches and prey on a variety of aquatic and terrestrial species, including tailed frog tadpoles, in and near streams (Blaustein et al. 1995). Dispersal is limited, and population genetic structure varies among basins (Mullen et al. 2010).

Idaho giant salamanders are of conservation concern but do not appear on the Intermoutain Region sensitive species list. There is some indication that land use practices may affect their occurrence, but their patchy distribution and sparse datasets limit inferences about habitat requirements. They are prey for both native and nonnative fish species, but fish species presence is not known to affect their population status (Sepulveda and Lowe 2011). Overall, their distribution is poorly described, and monitoring data are not available to evaluate temporal trends. Idaho giant salamander sensitivities are presumed to be similar to Rocky Mountain tailed frogs, although the salamander may be even more susceptible to disturbance of headwater natal areas, given nest guarding behavior by females and multiyear development of larval stages before maturity (Blaustein et al. 1995; Nussbaum 1969).

Western Pearlshell Mussel

The western pearlshell mussel is found throughout the Columbia River Basin, in a portion of the Missouri River headwaters in Montana, and in internally draining basins such as the Humboldt, Truckee, and Provo Rivers. It has been recorded in all national forests in the IAP region except the three southern Utah forests. This sedentary filter-feeder inhabits cool streams and rivers at depths of 1.5 to 3.0 feet, and tends to congregate in stable substrates amid boulders, gravel, and some sand, silt, and clay (Roscoe and Redelings 1964). The species has limited mobility and will not tolerate accumulation of fine sediment. Western pearlshell larvae are obligate parasites of an array of salmonid species (Chinook salmon [*O. tshawytscha*], cutthroat trout, and rainbow trout [*O. mykiss*]; utilization of bull trout is unknown) and rely on these hosts for recruitment and dispersal (Karna and Milleman 1978; Meyers and Milleman 1977; Murphy 1942). Female mussels generally release larvae (or glochidia) in spring or early summer, depending on water temperature. Glochidia attach to fish gills and develop for a period of weeks to months. Once metamorphosed, juvenile mussels drop from their host fishes and burrow into the substrate (Murphy 1942).

The western pearlshell mussel ranges from Alaska and British Columbia south to California and east to Nevada, Wyoming, Utah, and Montana. Many examples exist of pearlshell decline or extirpation from streams and rivers across its range, especially in arid areas (Hovingh 2004; Stone et al. 2004). Threats include impoundments, loss of host fishes, channel modification, dredging and mining, pollution, sedimentation, nutrient enrichment, water diversion, degradation of native riparian vegetation, and introduction of nonnative fishes that outcompete host species. Many of these impacts, especially reduction in streamflow and increased stream temperatures, can be exacerbated by climate change.

The western pearlshell mussel occupies streams with broad ranges of thermal regimes, but nevertheless prefers cold water and perennial flows. Its habitat must also be suitable for its trout and salmon hosts, and mussel sensitivity to climatic variability will closely parallel that of salmonids. Although potentially vulnerable to climate change, the western pearlshell mussel does not face an immediate risk of extinction because it occupies such a broad geographic range.

Springsnails

Springsnails are hydrobiid snails that occur in freshwater habitats throughout much of western North America. About 100 species inhabit the IAP region (Hershler et al. 2014). These tiny mollusks (shell length 0.04–0.31 inches) are widespread and locally abundant (often greater than 100 per square foot) in perennial, groundwater-dependent springs and brooks. Spring habitats may be either ambient temperature or thermal, and springsnails are often concentrated near sources of groundwater discharge with stable water chemistry (Mladenka and Minshall 2001). They typically live on emergent plants and hard substrates, grazing on attached algae and fungi (Hershler 1998; Mladenka and Minshall 2001). They are gill breathers and do not tolerate desiccation.

Distributed from southern Canada to northern Mexico, the springsnail exhibits habitat specificity and low dispersal ability, which contribute to a high degree of endemism; many species occurr only within a single spring or seep (Hershler et al. 2014). Springsnails have life history traits that make them vulnerable to extinction. First, they have specialized habitat requirements, typically occurring in pristine, cold-water or thermal springs close to the spring source, where dissolved carbon dioxide and calcium concentrations are high (Mladenka and Minshall 2001; O'Brien and Blinn 1999). Slight changes in water chemistry or warming temperatures could negatively affect local populations (Jyväsjärvi et al. 2015). Second, springsnails are poor dispersers, and suitable habitats are generally isolated from each other by arid uplands. Once a springsnail population has been extirpated, it is unlikely to be naturally refounded. Threats to springsnails include groundwater pumping and aquifer drawdown, surface flow diversion for agriculture, impoundments, channelization of outflows, springhead development, physical alteration of thermal springs for bathing, overgrazing, and nonnative species (e.g., New Zealand mudsnail [*Potamopyrgus antipodarum*]).

The limited ability of springsnails to disperse and their narrow environmental tolerances make them vulnerable to emerging threats associated with climate change. Because they require particular hydrological conditions, specific and stable temperature regimes, and perennial flows, some taxa (e.g., eight Nevada springsnail species) have been rated as "extremely vulnerable" using the Climate Change Vulnerability Index (Young et al. 2012).

Yosemite Toad

The Yosemite toad occurs in the Sierra Nevada, restricted primarily to publicly managed lands at high elevations (3,000–12,000 feet) (Brown et al. 2015). It inhabits ponds and wet meadows as well as drier upland sites. Adult toads emerge from their overwintering refuges in rodent burrows or underground cover from late April to late June, depending on elevation and year (Brown et al. 2012). Females lay approximately 1,000 eggs per clutch in shallow standing water amid emergent vegetation. After hatching, tadpoles require 4 to 6 weeks to reach metamorphosis and are not known to overwinter (Jennings and Hayes 1994; Kagarise Sherman 1980; Karlstrom 1962). Adults spend most of their lives in upland habitats adjacent to breeding sites, and are capable of moving and dispersing several hundred feet through dry forests.

Yosemite toad populations are in decline. The Yosemite toad, once common in high-elevation aquatic ecosystems of the Sierra Nevada, had disappeared from half its historical range by the 1990s (Jennings 1996). More recent surveys indicate an 87-percent decline from watersheds occupied before 1990, with scattered remaining populations and fewer individuals per site (Brown et al. 2015). Toads were most recently recorded at 470 sites in 5 national forests (17 sites in Humboldt-Toiyabe National Forest) and about 100 more sites in national parks (Brown et al. 2015). Several factors, such as disease, drought, airborne contaminants, habitat alteration, water diversions, nonnative fishes, and wildfire, may have contributed to the declines, but there is no clear evidence targeting any particular threat (Brown et al. 2015).

The dependence of Yosemite toad on shallow, ephemeral breeding ponds filled by melting snow makes the species susceptible to risks of climate change (Kagarise Sherman and Morton 1993). Models project that climate change will lead to higher average temperatures in all seasons, higher precipitation, and decreased spring and summer runoff due to decreased snowpack (Smith and Tirpak 1989) (Chapter 4). Less runoff could affect egg and tadpole survival by premature drying of breeding sites. Earlier snowmelt could lead to earlier breeding with possible positive effects on developmental time, but negative effects and mortality from uncertain weather patterns. For example, toads that emerge early risk starvation or death in late-spring snowstorms (Kagarise Sherman and Morton 1993).

Sierra Nevada Yellow-Legged Frog

The Sierra Nevada yellow-legged frog currently inhabits the Sierra Nevada, restricted primarily to publicly managed lands at high elevations (4,500–12,000 feet), including streams, lakes, ponds, and meadow wetlands in Humboldt-Toiyabe National Forest (CDFG 2011). The species is highly aquatic during all times of the year (Mullally and Cunningham 1956). At high elevation, both frogs and tadpoles overwinter under ice in lakes and streams, and because tadpoles require 1 to 4 years to metamorphose, successful breeding sites cannot dry out in summer and need to be deep enough to preclude complete freezing or deoxygenation (Bradford 1983). Although almost always found in or near water, the frog moves seasonally between breeding ponds, foraging, and overwintering habitats, usually along watercourses. However, individuals are capable of moving several hundred feet over dry land, which facilitates recolonization of sites that have lost populations (Pope and Matthews 2001).

The Sierra Nevada yellow-legged frog was listed as endangered under the ESA after populations declined severely during the 20th century due to chytridiomycosis disease (caused by the chytrid fungus [*Batrachochytrium dendrobatidis*]), predation from nonnative fishes, livestock grazing, habitat loss and fragmentation, and perhaps airborne contaminants from the Central Valley (CDFG 2011). The species was estimated to be extirpated from 220 of 318 historical occurrence localities and most remaining populations were thought to have fewer than 100 post-metamorphic individuals (CDFG 2011). During the last 20 years, however, yellow-legged frog populations have increased significantly in Yosemite (Knapp et al. 2016). The disappearance of nonnative fish from numerous water bodies after cessation of stocking, combined with reduced susceptibility to chytridiomycosis, are thought to have stimulated the recovery (Knapp et al. 2016).

Sierra Nevada yellow-legged frogs may be vulnerable to climate change because the species relies on perennial waterbodies and needs several years to metamorphose and mature (Mullally and Cunningham 1956). Climate change could result in greater interannual climatic variability or drier summers and cause lakes, ponds, and other standing waters fed by snowmelt or streams to dry more frequently, which would reduce available breeding habitat and lead to more frequent stranding and death of tadpoles (Lacan et al. 2008). On the other hand, projected earlier snowmelt is expected to cue breeding earlier in the year, which could allow additional time for tadpole growth and development. However, earlier breeding may also expose young tadpoles and eggs to mortality from early spring frosts (Corn 2005).

Bull Trout

Bull trout are broadly distributed across the northwestern United States but are restricted to the northwestern portion of the IAP region in Boise, Humboldt-Toiyabe, Payette, Salmon-Challis, and Sawtooth National Forests (Rieman et

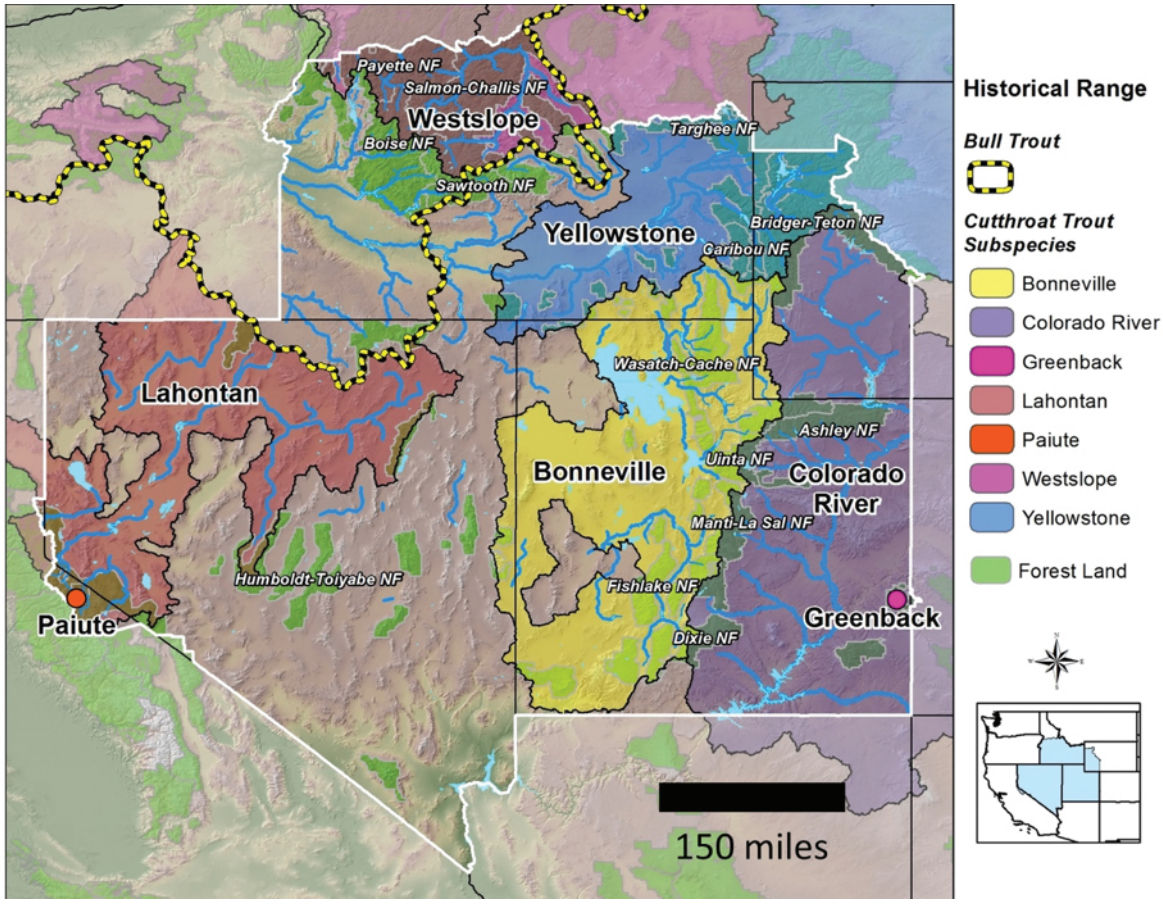

Figure 5.3—Native range distributions of (a) cutthroat trout and (b) bull trout in the Intermountain Adaptation Partnership region. The ranges of six cutthroat trout subspecies occur partly or wholly in this area.

al. 1997) (fig. 5.3). Populations may exhibit migratory or resident life histories. Migratory fish travel long distances as subadults to more productive habitats and achieve larger sizes and greater fecundity as adults before returning to cold natal headwater habitats to spawn (Howell et al. 2010; Rieman and McIntyre 1993). Fish exhibiting resident life histories remain in natal habitats and mature at smaller sizes, though often at the same age as migratory adults. Adults spawn primarily in September, and eggs incubate throughout the winter before juveniles hatch and emerge from stream substrates in late winter or early spring (Dunham et al. 2008). Reproduction and juvenile growth for the first 2 to 3 years is almost exclusively in streams with average August water temperatures less than 54 °F and flows greater than 1.2 cubic feet per second (Isaak et al. 2015). Bull trout populations are typically low density, even among strong populations in the best habitats (Isaak et al. 2017b; Rieman et al. 2006).

The bull trout is a sensitive species in the Intermountain Region and is ESA-listed as threatened (USFWS 2014). Their historical distribution has declined because of water development and habitat degradation (e.g., simplification of in-channel habitat complexity and fragmentation of some habitats), temperature increases, elimination of migratory life histories by anthropogenic barriers, harvest by anglers, and interactions with introduced nonnative fishes (Al-Chokhachy et al. 2010; Rieman et al. 1997, 2007). Encounters with nonnative fishes may involve wasted reproductive opportunities (e.g., interbreeding with brook trout [*Salvelinus fontinalis*]) (Kanda et al. 2002), competition, and predation (in streams, perhaps with brown trout [*Salmo trutta*]; in lakes, with lake trout [*Salvelinus namaycush*]) (Al-Chokhachy et al. 2016b; Martinez et al. 2009).

Bull trout evolved in western North America in interior and coastal streams that exhibit a wide array of flow characteristics and natural disturbance at scales from reaches to riverscapes (Dunham et al. 2003, 2008). Nevertheless, large habitats satisfying their restrictively cold thermal requirements for spawning and early juvenile rearing are relatively rare, and little evidence exists for flexibility in habitat use (Rieman et al. 2007). The length of connected cold-water habitats needed to support a bull trout population varies with local conditions, but current estimates suggest 10 to 30 miles are needed to ensure a high probability (>0.9) of habitat occupancy, with specifics contingent on water temperature, prevalence of brook trout, and stream slope (Isaak et al. 2015). Migratory life histories probably conferred greater resistance to extirpation under historical conditions

(Dunham et al. 2008; Rieman and Dunham 2000), but may no longer do so in some areas. Bull trout may also be sensitive to larger or more frequent winter high flows because eggs incubate in stream substrates throughout the winter where they are susceptible to bed-scour (Goode et al. 2013; Wenger et al. 2011a).

Cutthroat Trout

Cutthroat trout are represented by six subspecies in the IAP region: westslope cutthroat trout (*O. clarkii lewisi*), Yellowstone cutthroat trout (*O. c. bouvieri*), Lahontan cutthroat trout (*O. c. henshawi*), Paiute cutthroat trout (*O. c. seleniris*), Bonneville cutthroat trout (*O. c. utah*), and Colorado River cutthroat trout (*O. c. pleuriticus*) (Behnke 2002) (fig. 5.3). Although there was no historical overlap in the distribution of these subspecies, one or more were distributed throughout all national forests in the IAP region except where perennial streams are lacking (e.g., southern Nevada). These subspecies have a complex evolutionary history with two major clades (Crête-Lafrenière et al. 2012; Loxterman and Keeley 2012). One consists of westslope, Lahontan (including Paiute), and coastal cutthroat trout, and the other includes the rest of the interior subspecies. Phylogeographic structure in the latter group suggests that another one to four taxa may be present (Loxterman and Keeley 2012; Metcalf et al. 2012), but we confine our discussion to the prevailing taxonomy (Behnke 2002).

Cutthroat trout exhibit resident and migratory life history strategies similar to bull trout, but are spring spawners that reproduce in stream reaches where August temperatures are slightly warmer (up to 57 °F) (Isaak et al. 2015). Cold stream reaches where average August temperatures are less than 48 °F are suboptimal for cutthroat trout because of frequent recruitment failures associated with small numbers of growing degree days (Coleman and Fausch 2007). Cutthroat trout populations are generally found at higher densities than are bull trout (Isaak et al. 2017b).

Among cutthroat trout, all subspecies are either ESA-listed as threatened (Lahontan cutthroat trout [USFWS 1995] and Paiute cutthroat trout [USFWS 1985]) or have been petitioned for listing and found not warranted. Those not listed are on the Intermountain Region sensitive species list (Bonneville, Colorado River, westslope, and Yellowstone cutthroat trout). Distributions of these subspecies have declined more than 50 percent in response to the same stressors affecting bull trout (Behnke 2002; Shepard et al. 2005). Declines in response to nonnative species can be more severe than in bull trout, probably because cutthroat trout historically used slightly warmer habitats and overlap with more nonnative species. Brook trout have replaced cutthroat trout in many waters in the IAP region (Benjamin and Baxter 2012; Dunham et al. 2002a). These invasions seem to be influenced by the distribution of low-gradient alluvial valleys that may serve as nurseries for brook trout (Benjamin et al. 2007; Wenger et al. 2011a). Introduced rainbow trout have introgressively hybridized with cutthroat trout at lower elevations and in warmer streams (>50 °F) across their historical ranges (McKelvey et al. 2016), although genetically pure populations often persist in cold headwaters where climatic conditions limit the expansion of hybrid zones from downstream areas (Young et al. 2016, 2017).

Cutthroat trout occupy a broader thermal and stream size niche than do bull trout and can persist in small habitat patches for extended periods (Peterson et al. 2013a; Whitely et al. 2010). However, they still require cold-water natal habitat patches exceeding 2 to 6 miles to have a high probability of persistence (Isaak et al. 2015), with the habitat size depending on the prevalence of brook trout, water temperatures, and stream slope. Temperatures at the upstream extent of cutthroat trout populations in extremely cold streams will become more suitable from climate warming, but that trend may be countered by decreasing flows as snowmelt and runoff occur earlier (Chapter 3).

Climate Vulnerability and Adaptive Capacity of Focal Species

Warmer temperatures and declining summer streamflows will have broadly similar effects on aquatic species in the IAP region by reducing habitat volumes in perennial streams, fragmenting large habitat patches into smaller areas of suitable habitats (Isaak et al. 2012a; Rieman and Isaak 2010; Rieman et al. 2007), and shifting thermally suitable habitats upstream (Isaak et al. 2016a). Nonnative trout species more tolerant of warmer temperatures—brook trout, rainbow trout, and brown trout—will expand their distributions upstream and further constrain, replace, or prey on native trout and amphibians in some stream reaches. The relatively warm thermal niches of most nonnative species other than brook trout will restrict them from colonizing the coldest headwater streams, so refugial habitats, mostly at higher elevations, will continue to persist in some mountainous areas for the foreseeable future.

Wildfires may cause more extensive geomorphic disturbances and debris flows into streams, especially the smallest and steepest channels at the upstream extent of the drainage network (Miller et al. 2003; Sedell et al. 2015). Less water, more variable environments, and declining fluvial connectivity (e.g., from water development or interactions with road culverts) may favor resident life histories, as would greater separation between spawning and adult growth habitats. Smaller, more isolated populations will be more susceptible to extirpation from local environmental disturbances and during years of extreme drought and high summer water temperatures.

Species-specific vulnerabilities to these changes depend on the nexus among species ecological attributes, rates at which habitats are changing, and extent of current distributions (table 5.3). Bull trout and some cutthroat trout subspecies (e.g., westslope cutthroat trout, Yellowstone cutthroat trout) are moderately vulnerable because they are widely distributed, have good dispersal abilities, and

Table 5.3—Summary of anticipated vulnerability of selected aquatic species to climate change in the Intermountain Adaptation Partnership (IAP) region.

Species or subspecies	Taxonomy and phylogeography	Range extent[a]	Population trend	Climate vulnerability	Comment
Bull trout	Resolved	Locally common in north and elsewhere	Stable	Moderate	ESA listed[b]
Cutthroat trout subspecies					
Paiute	Resolved	Narrow endemic in west	Stable	High	ESA listed
Yellowstone	Resolved	Widespread in northeast and elsewhere	Stable	Moderate	
Westslope	Resolved	Widespread in north and elsewhere	Stable	Moderate	
Colorado River	Pending	Widespread in east and elsewhere	Stable	Moderate	
Bonneville	Pending	Restricted distribution in east	Stable	Moderate	
Lahontan	Resolved	Restricted distribution in west	Stable	High	ESA listed
Western pearlshell mussel	Resolved	Widespread in north and elsewhere	Unknown	Moderate	
Springsnails	Partially resolved	Widespread	Unknown	High	
Idaho giant salamander	Resolved	Restricted distribution in north and northern Idaho	Unknown	Moderate	
Rocky Mountain tailed frog	Resolved	Restricted distribution in north but more common elsewhere	Unknown	Moderate	
Yosemite toad	Resolved	Restricted distribution in west	Declining	High	Warranted but precluded
Sierra Nevada yellow-legged frog	Resolved	Restricted distribution in west	Increasing	High	ESA listed

[a] Geographic location (north, west, etc.) refers to IAP region only.
[b] ESA refers to U.S. Endangered Species Act.

occupy headwater habitats that are relatively resistant to thermal changes. Other subspecies of cutthroat (Bonneville, Lahontan, and Paiute cutthroat trout) are more vulnerable because distributions are limited to small numbers of isolated stream habitats.

Vulnerability of western pearlshell mussel will track that of their native trout hosts, but summer flow declines may be especially problematic because adult mollusks are immobile. Debris flows triggered by increased wildfire frequency could extirpate local mussel populations, although the species usually inhabits low-gradient stream reaches, where those events are rare (Stagliano 2005). However, fine sediments from debris flows could propagate downstream and smother mussel beds. Rocky Mountain tailed frog and Idaho giant salamanders are poor dispersers occupying headwater habitats and could be vulnerable to debris flows and more frequent disturbances.

Reduced aquifer recharge caused by altered seasonal precipitation and runoff could adversely affect groundwater-dependent and lake ecosystems that support endemic taxa such as springsnails, Yosemite toad, and Sierra Nevada yellow-legged frog (Burns et al. 2017; Hershler et al. 2014; Jyväsjärvi et al. 2015). The extreme endemism of springsnails means that the drying of individual springs could result in extirpation or extinction of a species. Arid land springs, which provide habitat for most springsnail species, are usually isolated, and dispersal of snails may be impossible without assistance by humans or other animal vectors. Recent and abrupt declines in the number and extent of Yosemite toad puts this species at high risk regardless of climate-induced environmental change. Altered aquatic habitat conditions, especially the greater environmental stochasticity that is expected, are predicted to exacerbate existing stressors and further degrade the resilience of remaining populations.

Niche conservatism suggests there is little capacity for rapid evolutionary or physiological adaptations to warmer water temperatures or desiccation within the aquatic species considered here (McCullough et al. 2009; Narum et al. 2013; Wiens et al. 2010). However, species with good dispersal abilities may track shifting habitats or recolonize previously disturbed habitats or those that have been

recently restored as long as artificial barriers do not impede their movement (Fausch et al. 2009). Some species exhibit both migratory and resident life history strategies, and the relative frequencies of these strategies may evolve based on how climate change affects metabolic rates, water temperature, stream productivity, and connectivity. Development of disease resistance or other adaptive responses associated with phenology may also bolster population resilience in ways that allow species to persist in dynamic environments (Knapp et al. 2016; Kovach et al. 2012).

Climate Refugia for Native Trout

Trout Distribution Model and Scenarios

Species distribution models for bull trout and cutthroat trout were developed previously in the Coldwater Climate Shield (CS) project by compiling large species occurrence datasets (more than 4,000 sites in over 500 streams) from field biologists, peer-reviewed literature, and State and Federal agency reports (Isaak et al. 2015). The CS models use the high-resolution stream temperature and flow scenarios described earlier with stream slope and the prevalence of brook trout as predictor variables in logistic regression models that predict occurrence probabilities of juvenile bull trout and cutthroat trout in cold-water habitat (CWH) patches (fig. 5.4). Juvenile occurrence is used as an indicator of important natal habitat locations and serves as evidence of locally reproducing populations for salmonid fishes (Dunham et al. 2002b; Rieman and McIntyre 1995). The CS models are also designed to identify CWHs that are too cold (<52 °F mean August temperature) for invasions by most nonnative species other than brook trout and thus should require limited management interventions to support native trout populations.

Previously, Young et al. (2018) applied the CS models to describe bull trout and cutthroat trout distributions and refugia in the Northern Rockies Adaptation Partnership region (Halofsky et al. 2018). Here, we repeat that exercise and summarize CS model predictions for native trout populations across the IAP region. Information for these summaries was downloaded from the CS website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/

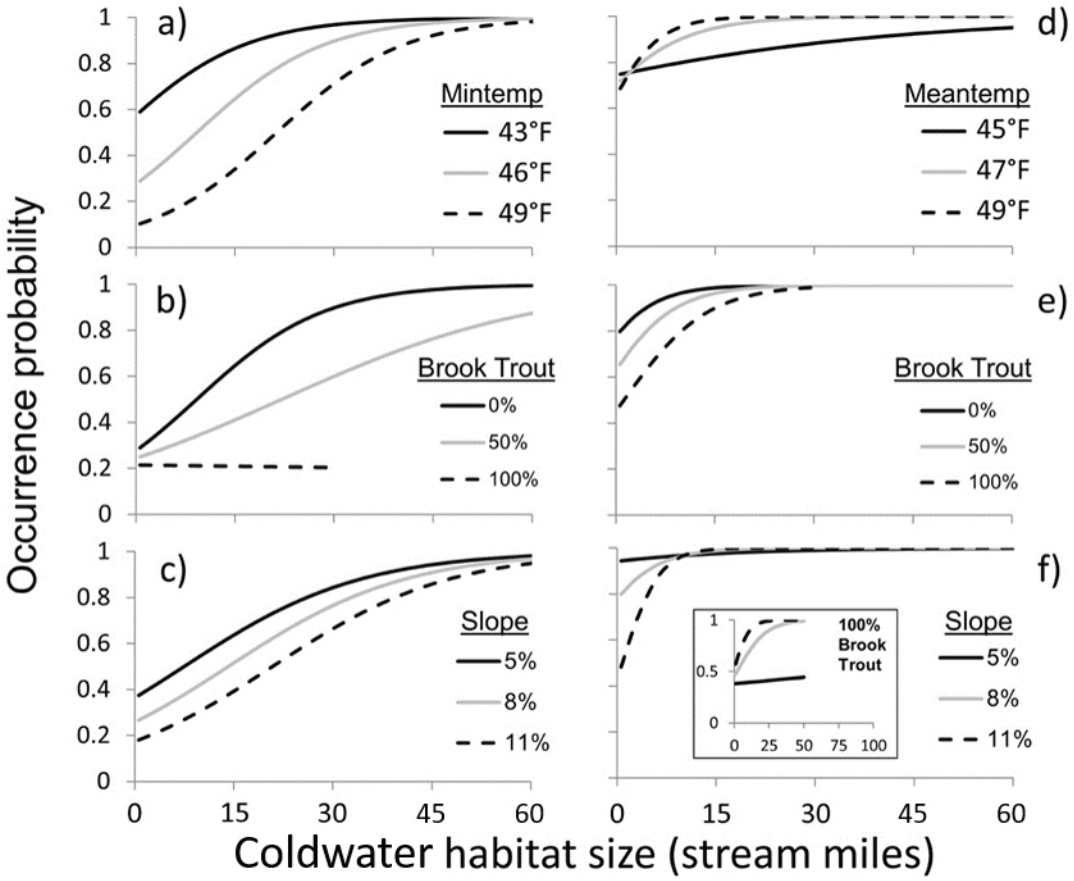

Figure 5.4—Relationships between predictor variables in the Climate Shield models and the probability of occupancy by juvenile native trout for (a–c) 512 bull trout and (d–f) 566 cutthroat trout streams characterized by cold-water habitats less than 52 °F (from Isaak et al. 2015). Relationships shown are conditioned on mean values of the two predictors not shown in a panel. An exception occurs for cutthroat trout with regard to stream slope (panel f), where brook trout values of 0 and 100 percent were used to highlight the interaction between these covariates.

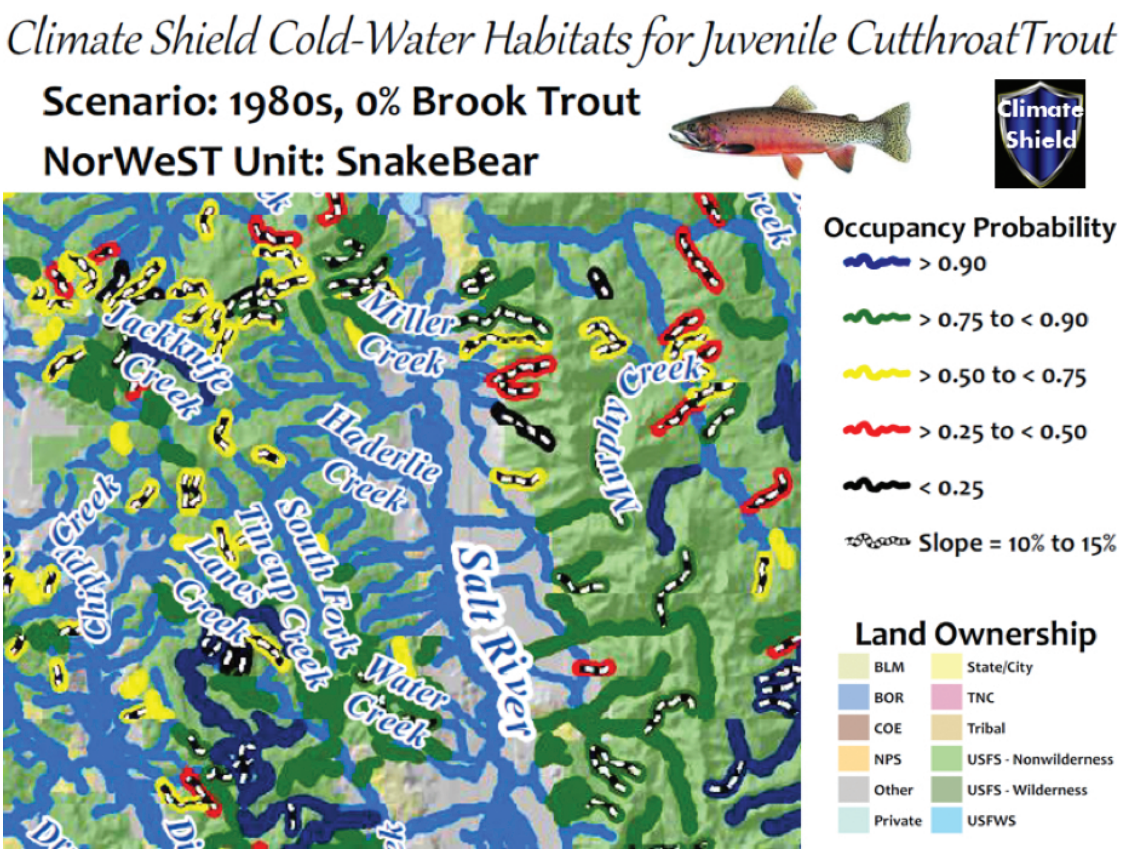

Figure 5.5—Example of a detailed Climate Shield map that shows probabilities of juvenile cutthroat trout occupancy in cold-water stream habitats. Information is available for three climate periods and five brook trout invasion scenarios for bull trout and cutthroat trout streams in the Intermountain Adaptation Partnership region at the Climate Shield website (USDA FS n.d.a).

ClimateShield.html), which also provides extensive metadata descriptions, a user-friendly digital map archive (fig. 5.5), and geospatial databases showing stream-specific model predictions for all streams in the USFS Northern, Rocky Mountain, Intermountain, and Pacific Northwest Regions in the western United States.

Scenarios used to assess native trout distributions were based on the same climate periods (2040s and 2080s) and A1B trajectory already described earlier for the NorWeST stream temperature and VIC scenarios. To account for uncertainties in brook trout distributions across the IAP region, native trout occupancy probabilities were also summarized for a pristine scenario (no brook trout) and a broad invasion scenario that assumed brook trout would be present at half the sites within each CWH (50-percent brook trout). For the IAP region, we did not summarize a scenario in which brook trout were present at all sites because their prevalence rarely exceeded 50 percent in the largest cold-water habitats (>25 miles), which are most likely to serve as strongholds for native trout (see Appendices S2 and S3 in Isaak et al. 2015), and because not all stream locations are suitable for brook trout (Isaak et al. 2017b; Wenger et al. 2011a). Brook trout prevalence may reach 100 percent in small, low-gradient streams, so native trout probabilities for a full range of invasion scenarios (0-, 25-, 50-, 75-, and 100-percent brook trout prevalence) have been calculated and integrated into the CS geospatial databases to facilitate stream-specific assessments (if local information is available on brook trout prevalence).

Cutthroat Trout Status and Projected Trends

Invasion-resistant streams with mean August temperatures less than 52 °F encompassed 15,000 miles of the 56,000-mile network draining the IAP region during the baseline period; 94 percent of those cold streams are on national forests (table 5.2). The number of discrete CWHs that were potentially occupied by juvenile cutthroat trout during the baseline period was estimated to be 2,600 and encompassed nearly 13,000 stream miles across all subspecies (table 5.4, fig. 5.6). Three subspecies—Lahontan cutthroat trout, Bonneville cutthroat trout, and Paiute cutthroat trout—had restricted ranges and are summarized separately (table 5.5). Bonneville and Lahontan cutthroat had about 620 miles of CWHs during the baseline period, whereas habitat for the Paiute cutthroat trout was less than 12 miles because it occurred in only a few streams.

Across all cutthroat trout subspecies, 89 percent of CWHs were predicted to have probabilities of occupancy exceeding 50 percent in the current period if brook trout

Table 5.4—Number and length of cold-water habitats for all subspecies of juvenile cutthroat trout by probability of occurrence for three climate periods and two brook trout invasion scenarios in the Intermountain Adaptation Partnership region.

		Probability of occurrence (percent)					Total cold-water habitats
	Period	<25	25–50	50–75	75–90	>90	
Cold-water habitat number				*Number*			
0% brook trout prevalence	2000s	73	206	540	872	909	2,600
	2040s	49	170	544	791	680	2,234
	2080s	66	252	479	572	476	1,845
50% brook trout prevalence	2000s	80	261	1,296	736	227	2,600
	2040s	50	193	1,152	606	233	2,234
	2080s	66	260	975	440	104	1,845
Cold-water habitat length				*Miles*			
0% brook trout prevalence	2000s	215	388	1,272	2,322	8,794	12,776
	2040s	68	288	951	1,858	5,871	9,036
	2080s	93	398	931	1,402	3,665	6,489
50% brook trout prevalence	2000s	578	864	2,894	4,447	4,208	12,991
	2040s	140	386	2,202	3,047	3,260	9,035
	2080s	93	454	2,008	2,290	1,643	6,488

were absent (table 5.4), largely because of the relatively small stream networks that a cutthroat trout population requires for persistence (e.g., 6 miles is associated with a 90-percent probability of occupancy) (fig. 5.4). Nonetheless, the largest CWHs accounted for a disproportionate amount of the habitat most likely to be occupied; 35 percent of CWHs had probabilities greater than 90 percent, but these accounted for 68 percent of the length of CWHs. As expected, the number and extent of CWHs were predicted to decrease substantially (14–50 percent) in future periods, but 1,845 potential habitats encompassing almost 6,500 miles were projected to remain in the extreme 2080 scenario.

Where headwater stream reaches are currently too cold for cutthroat trout, future warming may increase habitat suitability and the probability of occupancy in portions of the Uinta Mountains and other cold streams draining the upper Green, Salmon, and Snake Rivers. As expected, the brook trout invasion scenario did not affect the number or amount of CWHs because the habitats remained potentially suitable for cutthroat trout, but occupancy probabilities declined (table 5.4). The sensitivity of streams to brook trout invasions varies with local conditions, but impacts were most pronounced in small CWHs with low slopes (fig. 5.4).

Bull Trout Status and Projected Trends

The historical range of bull trout occupies a smaller portion of the IAP region than cutthroat trout, so the number of discrete CWHs for bull trout during the baseline climate period was estimated to be 984, encompassing 7,700 miles (table 5.6, fig. 5.7). In contrast to cutthroat trout, most CWHs for bull trout had probabilities of occupancy less than 50 percent because of the relatively large stream networks that bull trout require (30 miles is associated with a 90-percent probability of occupancy) (fig. 5.4). Although only 8 percent of CWHs had probabilities greater than 90 percent, they provided 36 percent of the total length of CWH, emphasizing the contribution of large CWHs to the amount of habitat predicted to be occupied. The requirement for larger CWHs caused projected decreases in the number (9–28 percent) and network extent (35–57 percent) of bull trout CWHs to be more substantial than those for cutthroat trout, particularly for the CWHs with the highest probabilities of occupancy. However, more than 700 CWHs representing 3,330 miles were projected to remain even in the extreme 2080 scenario.

Similar to the effect on cutthroat trout, the brook trout invasion scenario showed reduced bull trout occupancy rates (especially in CWHs with greater than 50-probability of occupancy), and as few as three to four CWHs with probabilities greater than 90 percent remained under the extreme warming scenario with a ubiquitous brook trout presence. However, many of the large bull trout habitats are less susceptible to broad-scale brook trout invasions given the preference by the latter species for small low-gradient streams (Dunham et al. 2002a; Isaak et al. 2015). Not surprisingly, CWHs with the highest bull trout occupancy probabilities in all scenarios coincided with river networks containing the largest number of cold streams (headwater portions of the Boise, Middle Fork Salmon, and Upper Salmon Rivers) (fig. 5.7).

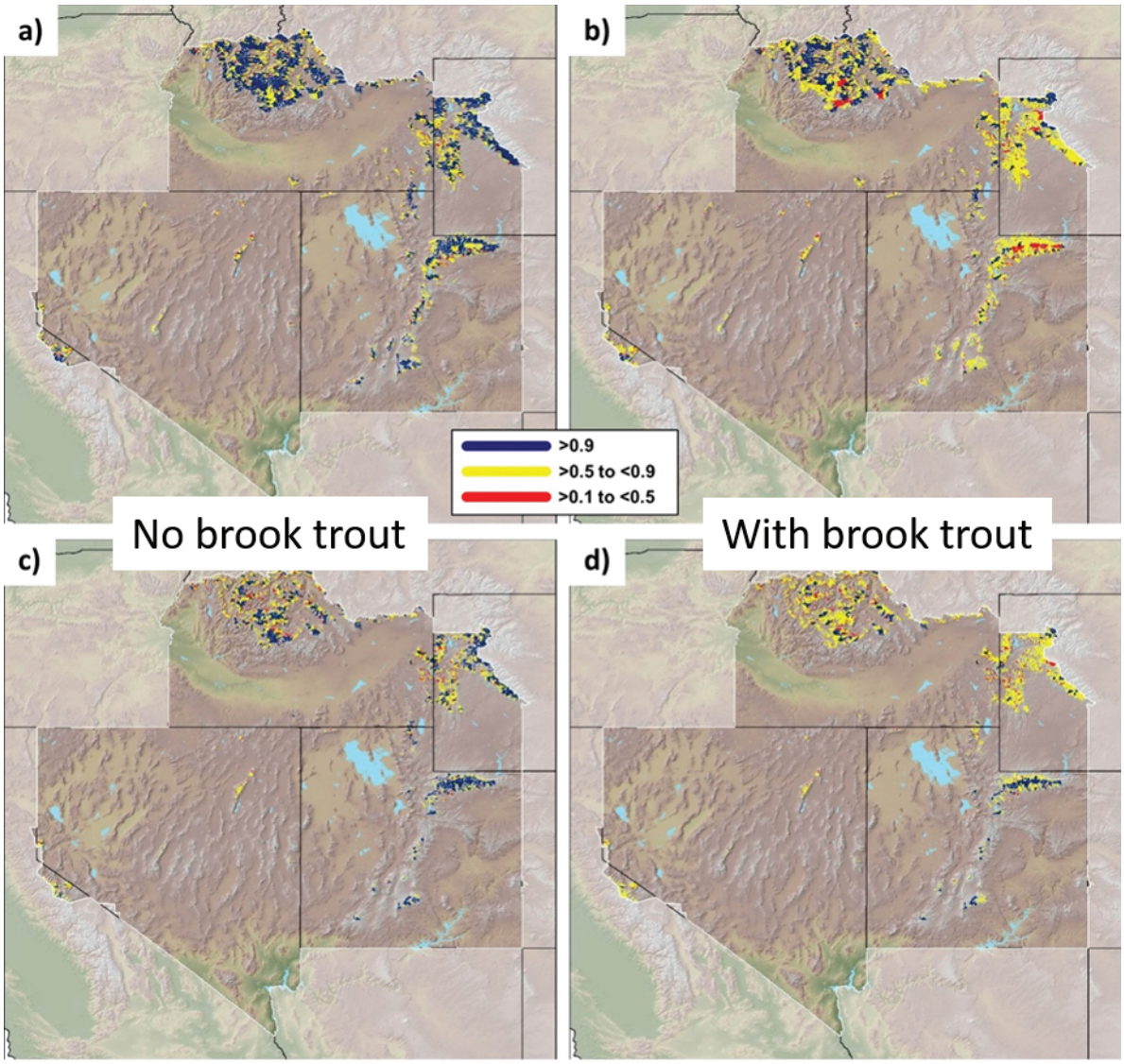

Figure 5.6—Distribution of cold-water habitats with probabilities of occupancy greater than 0.1 for juvenile cutthroat trout for (a–b) the baseline period (2000s) and (c–d) late-century scenario (2080s). Panels a and c illustrate occupancy when brook trout are absent; panels b and d illustrate occupancy when brook trout prevalence is 50 percent.

Conclusions and Implications

A changing climate has significant implications for distributions of aquatic habitats and species dependent on them in the IAP region. Vulnerability to habitat shifts or losses this century may be high, especially for taxa that have either restricted habitats or limited dispersal abilities. Yet the symptoms of rapid climate change have been manifest throughout the western United States for several decades (Barnett et al. 2008; Hamlet and Lettenmaier 2007; Luce and Holden 2009; Mote et al. 2005) without widespread losses of aquatic populations or species. Three factors help explain this apparent paradox. First, the steep topography in parts of the IAP region translates to slow climate velocities, which may enable species to track gradual shifts in their habitats (Isaak et al. 2016a). Second, the thermal and hydrological changes that have accumulated to date are relatively small compared to changes expected throughout the remainder of the 21st century (Chapter 3). Third, existing monitoring programs and analyses of available datasets may be inadequate for detecting the subtle distribution shifts or extirpation of small populations that are expected with climate change (Isaak and Rieman 2013).

In one of the few attempts to rigorously measure how distributions of stream organisms are responding to climate change, site revisits over a 20-year period revealed that juvenile bull trout distributions were contracting upstream within the Bitterroot River basin of Montana (Eby et al. 2014). In a larger study across western Montana, long-term monitoring indicated that brown trout populations were increasing in abundance and gradually expanding into streams

Table 5.5—Number and length of cold-water habitats for three cutthroat trout subspecies by probability of occurrence for three climate periods, assuming the absence of brook trout. Reduced occurrence probabilities associated with brook trout invasions would be similar to declines described in table 5.4.

Cutthroat trout subspecies	Period	Probability of occurrence (percent)					Total cold-water habitats
		<25	25–50	50–75	75–90	>90	
Bonneville				*Number*			
Cold-water habitat number	2000s	1	24	70	90	96	281
	2040s	0	12	34	62	54	162
	2080s	0	8	23	22	44	97
				Miles			
Cold-water habitat length	2000s	0	33	100	176	530	839
	2040s	0	12	39	124	310	485
	2080s	0	9	26	48	228	311
Lahontan				*Number*			
Cold-water habitat number	2000s	3	25	39	63	29	159
	2040s	2	16	40	50	25	133
	2080s	1	10	40	31	11	93
				Miles			
Cold-water habitat length	2000s	5	48	86	175	227	541
	2040s	3	31	78	137	150	399
	2080s	2	17	85	93	58	255
Paiute				*Number*			
Cold-water habitat number	2000s	0	0	0	1	2	3
	2040s	1	0	1	3	0	5
	2080s	1	0	0	2	0	3
				Miles			
Cold-water habitat length	2000s	0	0	0	4	8	12
	2040s	1	0	1	6	0	7
	2080s	1	0	0	5	0	6

that were previously too cold (Al-Chokhachy et al. 2016b). Similar monitoring efforts and site resurveys are needed at broad scales for many species to document the effects of climate change on aquatic life and to provide the basis for strategic planning and proactive conservation actions (Comte and Grenouillet 2013; Craine et al. 2007).

The first step in promulgating an informative aquatic biodiversity census is the aggregation and organization of historical datasets into functional databases, allowing species occurrence information to be linked with environmental covariates, then modeled and analyzed for trends. Development of the CS models involved organizing a small fraction of existing datasets for trout species (Isaak et al. 2015), but even this initial effort yielded significant improvements in the ability to predict where trout populations are most likely to persist, has assisted monitoring efforts, and is setting the stage for developing more precise models in the near future. For example, the spatially explicit CS model predictions are being used to guide an interagency crowd-sourcing campaign to collect environmental DNA (eDNA) samples from 7,000 locations throughout the native range of bull trout, which includes the northern portion of the IAP region (Young et al. 2017). Those samples will be paired with new spatial stream-network models (Isaak et al. 2014; Ver Hoef et al. 2006) to model bull trout occurrence at high resolution (<1 mile) across broad areas to provide accurate predictions of distribution and abundance and a better understanding of environmental constraints. This new generation of eDNA samples could be compared to historical occurrence datasets to provide broad-scale climate trend assessments. The bull trout eDNA samples also contain the DNA of many

Table 5.6—Number and length of cold-water habitats for juvenile bull trout by probability of occurrence for three climate periods and two brook trout invasion scenarios in the Intermountain Adaptation Partnership region.

		Probability of occurrence (percent)					
	Period	<25	25–50	50–75	75–90	>90	Total
Cold-water habitat number		----------*Number*----------					
0% brook trout prevalence	2000s	406	289	141	70	78	984
	2040s	538	216	90	31	23	898
	2080s	387	215	76	22	12	712
50% brook trout prevalence	2000s	474	301	132	52	25	984
	2040s	608	211	56	19	4	898
	2080s	456	197	43	13	3	712
Cold-water habitat length		----------*Miles*----------					
0% brook trout prevalence	2000s	1,158	1,480	1,245	1,003	2,806	7,692
	2040s	1,630	1,232	874	464	766	4,967
	2080s	1,059	999	645	289	344	3,336
50% brook trout prevalence	2000s	1,452	1,950	1,651	1,097	1,542	7,692
	2040s	1,994	1,531	723	529	190	4,967
	2080s	1,337	1,180	463	216	140	3,336

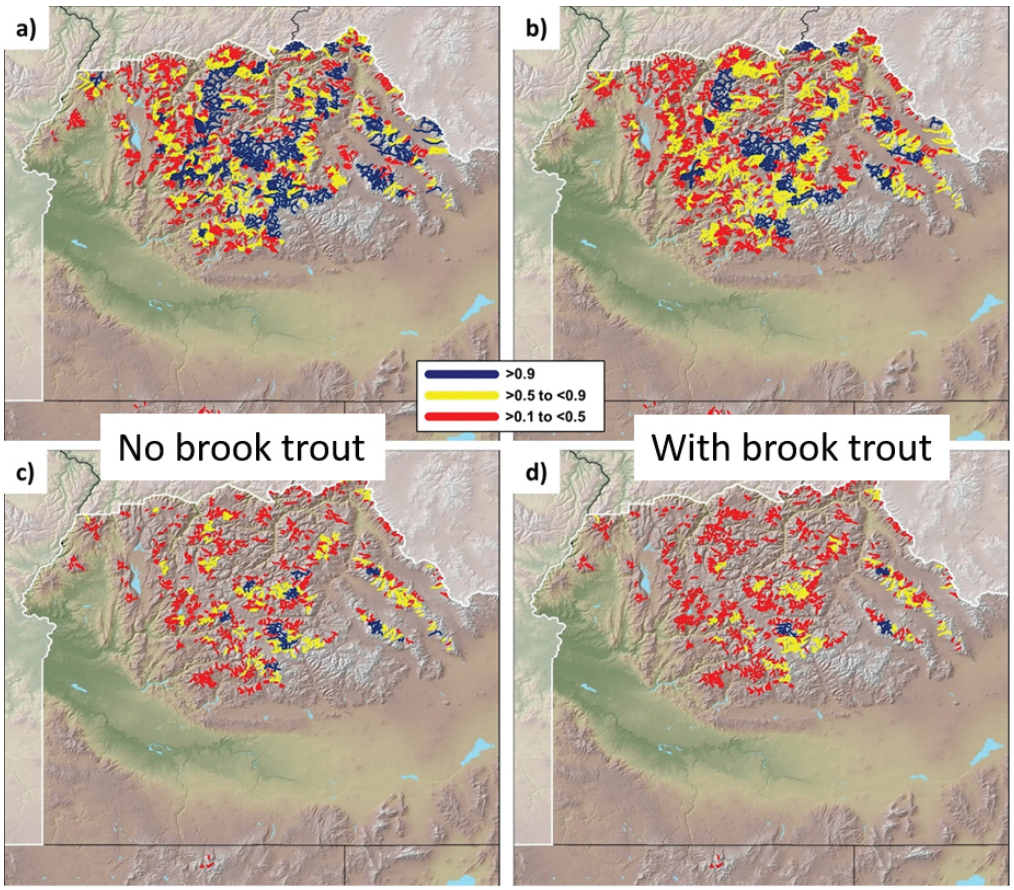

Figure 5.7—Distribution of cold-water habitats with probabilities of occupancy greater than 0.1 for juvenile bull trout for (a–b) the baseline period (2000s) and (c–d) late-century scenario (2080s). Panels a and c illustrate occupancy when brook trout are absent; panels b and d illustrate occupancy when brook trout prevalence is 50 percent.

other aquatic taxa, so will help establish baseline conditions for many species that lack data. Moreover, thousands of additional eDNA samples are now being collected annually across the western United States through partnerships with the National Genomics Center for Wildlife and Fish Conservation (http://www.fs.fed.us/research/genomics-center), so a taxonomically diverse and geographically comprehensive monitoring system for aquatic species is emerging.

For native bull trout and cutthroat trout, the CS models provided accurate, spatially explicit predictions of habitat occupancy throughout the IAP region with respect to current conditions. Assuming that species responses are related to the effects of climate on stream ecosystems—and the accuracy of the models supports this contention—the models also provide reasonably robust projections of habitat occupancy in light of anticipated climate change. Although both native trout species are regarded as cold-water taxa, their exact responses to a changing climate are expected to differ. Bull trout, and most members of the genus *Salvelinus*, are adapted to some of the coldest freshwater environments in the Northern Hemisphere (Klemetsen et al. 2003). These species also tend to inhabit highly variable environments, often with strong gradients in productivity that appear to favor migration as a life history tactic (Klemetsen 2010). It is unsurprising, therefore, that a species with those attributes near the southern end of its distribution would be susceptible to range contractions as temperatures warm. In the IAP region, we anticipate declines in bull trout distributions, but the species is unlikely to be extirpated from the region because many climate refuge habitats will persist. As climate change proceeds this century, it is possible that conditions favoring migratory or resident life histories will change, although it is unclear how these conditions would be accommodated or exploited by bull trout. As we learn more about the extent and prevalence of populations occupying CWHs with varying probabilities of occupancy, an understanding of the environmental drivers of bull trout life history may emerge.

Cutthroat trout, in contrast, can accommodate a wider range of thermal environments, consistent with their broad latitudinal distribution and an evolutionary history. Since the late Miocene or early Pliocene, this species was exposed to intervals cycling between warm/dry and cool/moist periods in western North America (McPhail and Lindsey 1986; Minckley et al. 1986). Cutthroat trout are relatively plastic with respect to life history strategies, ranging from highly migratory populations dependent on large rivers or lakes to promote growth and fecundity, to resident populations that move little and have been isolated for decades to centuries (Northcote 1992; Peterson et al. 2013b). Although we anticipate net losses in cutthroat distribution in the IAP region, they are not expected to be as severe as for bull trout, and some basins that are currently too cold to support cutthroat trout may become high-quality climate refugia in the future (Al-Chokhachy et al. 2013; Coleman and Fausch 2007; Cooney et al. 2005). Of greater importance may be how nonnative salmonids, which often displace or replace cutthroat trout, respond to warming conditions (Wenger et al. 2011a). These factors do not represent similar risks to the six cutthroat trout subspecies, primarily because of the large differences in quality and quantity of habitats currently occupied. For example, Yellowstone and westslope cutthroat trout are widely distributed, occupy many streams throughout their ranges, and exhibit a broad array of life histories. However, remaining subspecies often persist in small, isolated headwater habitats that may be especially vulnerable to brook trout invasions (Benjamin and Baxter 2012; Dunham et al. 2002a).

The presence of brook trout is problematic for both bull trout and cutthroat trout. The tolerance of brook trout to cold temperature is nearly equivalent to that of cutthroat trout, and brook trout favor the low-gradient environments preferred by cutthroat trout and bull trout (Wenger et al. 2011a). However, very large habitats appear less likely to be invaded by brook trout, possibly because this species prefers small streams but also because large systems usually contain other salmonid species, such as rainbow trout or brown trout, that compete with brook trout (Fausch et al. 2009). Rainbow trout and brown trout are expected to shift their distribution upstream as temperature isotherms optimal for these species move in that direction (Isaak and Rieman 2013; Wenger et al. 2011b), but may be constrained by unsuitable stream steepness because these species are rare where slopes exceed 4 percent (Isaak et al. 2017b). Where overlap occurs, both species appear to have negative effects on cutthroat trout, but cold, steep headwater streams that resist their invasions are expected to remain widespread.

Most current and future CWHs occur on public lands, mostly national forests, emphasizing the critical role of the USFS in conservation of native fish. Exploring an array of conservation strategies will be important because most of the CWHs are outside designated wilderness areas or national parks so are subject to various land management activities. Conservation options will vary by location because current and future CWHs are expected to be more abundant and persistent in some river basins than others across the IAP region. Where CWHs are abundant, maintaining those conditions and avoiding significant new impairments may be all that is necessary to ensure the persistence of native fish populations. In contrast, very few habitats that function as climate refugia may occur in other basins or where current habitats for some species or subspecies are very limited. Those circumstances favor strategic, active management to promote population persistence, whether by manipulating habitat or fish populations, or both. And because many CWHs are in landscapes that have multiple resource values, balancing among competing interests will remain an underlying theme of public land management (Rieman et al. 2010). Retaining native populations of aquatic organisms in many of these areas may require conservation investments that are unsustainable or

could prove ineffective if climate warming accelerates appreciably this century. In these circumstances, reallocating investment resources to areas where native populations are more likely to persist may be preferable.

Climate Adaptation Options

Many things can be done to adapt to climate change and improve the resilience of aquatic species in the IAP region (Chapter 14). Climate change adaptation options for aquatic species have been the subject of a number of comprehensive reviews for the IAP region (Rieman and Isaak 2010) and the Pacific Northwest (Beechie et al. 2013; Isaak et al. 2012a; ISAB 2007; Luce et al. 2012; Williams et al. 2007, 2015; Young et al., in press b). Several key themes emerge from these reviews: (1) be strategic; (2) implement monitoring programs; (3) restore and protect natural flow, sediment, and temperature regimes; (4) manage fluvial connectivity; (5) remove or suppress nonnative species, and (6) consider assisted migration.

Be strategic—Prioritize watershed restoration such that the most important work is done in the most important places because the funds, labor, and time available for management of native fish populations are limited (Peterson et al. 2013a). Efforts are directed at only a few of the locations and problems that could be addressed. For example, climate refugia for native trout that are in wilderness areas may neither require nor be amenable to habitat modification to ensure the persistence of those populations. Similar refugia outside protected areas could be targeted to improve habitat conditions or remove or reduce nonnative species presence, particularly if doing so increases the probability of occupancy of such habitats by native species.

Implement monitoring programs—Being strategic means reducing current and future uncertainties for decisionmaking. More data are needed for streamflow (more sites), stream temperature (annual data from sensors maintained over many years), and species distributions. These data could be used for better status and trend descriptions, and to develop robust (or more accurate) models for species to understand their response to climate change, natural variation, and land management. The feasibility of monitoring at small to broad scales is increasing with the advent of rapid, reliable eDNA inventories of aquatic organisms (McKelvey et al. 2016; Thomsen et al. 2012) and the availability of inexpensive, reliable temperature and flow sensors (USEPA 2014).

Restore and maintain natural flow, sediment, and thermal regimes—Persistence of native species can be enhanced using a variety of habitat techniques to improve stream shade, narrow unnaturally widened channels, minimize flow diversions, and improve stream substrate conditions. Actions may include decommissioning or relocating roads away from streams (Al-Chokhachy et al. 2016a; Zurstadt 2015), limiting seasonal grazing in some areas, and managing streamside riparian forest buffer zones to maintain effective shade and cool, moist riparian microclimates (Nusslé et al. 2015). Tactics may also involve directly managing water, such as increasing water storage in headwater reservoirs, restoring populations of American beaver (*Castor canadensis*) (Bouwes et al. 2016; Pollock et al. 2014), or acquiring water rights to maintain or enhance summer streamflows (Elmore et al. 2015). Such actions obviously have implications and consequences far beyond enhancing the persistence of native fish populations, but being open to opportunities is part of strategic thinking.

Manage connectivity—Obstacles to fish migration may be removed in hopes of enhancing the success of migratory life history forms, or permitting native species to reoccupy former habitat or supplement existing populations. This also presents a dilemma: Accessible waters can be invaded by nonnative species that sometimes replace native species (Fausch et al. 2009). The alternative is the installation of barriers to prevent these invasions. Native populations above barriers may be secure if they can adopt resident life histories, but could be susceptible to loss from catastrophic events in small habitats and will require human intervention for refounding or supplementation. Barriers are also temporary, and eventually will require reconstruction if nonnative species still remain downstream. Barriers may also be associated with small headwater diversion structures that sometimes route fish out of streams, where they are susceptible to stranding when water ditches are seasonally dewatered (Roberts and Rahel 2008; Walters et al. 2013). Headwater diversions are numerous and may be difficult to locate, so tools for locating them may be useful. For example, the Trout Unlimited Water Transaction Tool (McFall 2017) shows the locations of all diversion points in the Idaho Department of Water Resources database relative to the CS native trout refuge streams within Idaho. The tool enables users to identify and visit points of diversions in critical trout habitats to determine their potential impact to fish populations and design mitigation strategies.

Remove nonnative species—Removal of nonnative fish species will be essential to maintain or restore some populations. These efforts typically consist of chemical treatments or electrofishing, and both tend to be feasible only in smaller, simpler habitats (Shepard et al. 2002). Both are also costly, in part because they need to be conducted on multiple occasions to be effective (Peterson et al. 2008). Chemical treatments are controversial because of their perceived effects on water quality. Furthermore, success with either method is obtained only if the source of nonnative species is removed, often by the installation of a migration barrier. Unauthorized introductions are also common, and can undermine conservation efforts. Finally, using control measures to manage the abundance of nonnative species rather than removing them has been applied in some areas (e.g., the removal of lake trout to promote bull trout persistence or regular electrofishing to depress brook trout and favor cutthroat trout). Such activities are likely to be successful only if conducted at regular intervals for the

foreseeable future, which assumes funding and enthusiasm for such ventures will be available indefinitely.

Consider assisted migration—Moving native fish species from one location to another, a historically common activity in fish management, has typically been used to found populations in previously fishless waters. This tactic may be further pursued in the IAP region where a few basins are currently fishless (or have only limited populations of nonnative species) because of natural barriers such as waterfalls, and may create high-quality climate refugia in the near future. Moving native fish to such areas is feasible but controversial because other at-risk native taxa may be vulnerable to predation or competition with native fish species. Reintroductions of native species (Dunham et al. 2011, 2016) may also be performed when natural refounding is not an option, such as where populations are isolated and periodically fail, or suffer population bottlenecks. Management intervention at this level will require an understanding of genetic principles and broodstock establishment.

In conclusion, responding to the environmental trends associated with climate change will require a diverse portfolio that includes many of the actions described earlier. We need to adapt our mindsets and administrative processes to a new paradigm of dynamic disequilibrium. Under this paradigm, stream habitats will become more variable, undergo gradual shifts through time, and sometimes decline. Many populations are resilient enough to persist in, or track, suitable habitats, but others could be overwhelmed by future changes. It is unlikely that we will be able to preserve all populations of aquatic species as they currently exist this century. But as better information continues to be developed in the future, managers will have ever more precise tools at their disposal to know when and where resource commitments are best made to enhance the resilience of existing populations or to benefit other species for which management was previously not a priority. There is much to do as climate change adaptation continues in future years (Chapter 14), and USFS lands will play an increasingly important role in providing refuge habitats for aquatic biodiversity.

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